World Electric Vehicle Journal

MDPI

Article

Comparative Life Cycle Assessment of Battery and Fuel Cell Electric Cars, Trucks, and Buses

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Abstract: Addressing the pressing challenge of global warming, reducing greenhouse gas emissions in the transportation sector is a critical imperative. Battery and fuel cell electric vehicles have emerged as promising solutions for curbing emissions in this sector. In this study, we conducted a comprehensive life cycle assessment (LCA) for typical passenger vehicles, heavy-duty trucks, and city buses using either proton-exchange membrane fuel cells or Li-ion batteries with different cell chemistries. To ensure accuracy, we supplemented existing studies with data from the literature, particularly for the recycling phase, as database limitations were encountered. Our results highlight that fuel cell and battery systems exhibit large emissions in the production phase. Recycling can significantly offset some of these emissions, but a comparison of the technologies examined revealed considerable differences. Overall, battery electric vehicles consistently outperform fuel cell electric vehicles regarding absolute greenhouse gas emissions. Hence, we recommend prioritizing battery electric over fuel cell vehicles. However, deploying fuel cell electric vehicles could become attractive in a hydrogen economy scenario where other factors, e.g., the conversion and storage of surplus renewable electricity via electrolysis, become important.

Keywords: life cycle assessment; electric vehicles; Li-ion battery; fuel cell; car; truck; bus; decarbonisation

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Citation: Syré, A.M.; Shyposha, P.; Freisem, L.; Pollak, A.; Göhlich, D. Comparative Life Cycle Assessment of Battery and Fuel Cell Electric Cars, Trucks, and Buses. *World Electr. Veh. J.* **2024**, *15*, 114. https://doi.org/10.3390/wevj15030114

Academic Editor: Joeri Van Mierlo

Received: 14 February 2024
Revised: 8 March 2024
Accepted: 8 March 2024
Published: 15 March 2024

1. Introduction

The transportation industry currently contributes to over 20% of global greenhouse gas (GHG) emissions, mainly due to its reliance on oil-based products [1]. However, there is a growing trend towards adopting alternative powertrain systems, such as battery electric and hydrogen fuel cell (FC) systems. To maximize the ecological benefits of this shift, it is crucial to minimize GHG emissions through measures like recycling used components and utilizing renewable energy sources. While internal combustion engines (ICEs) are still the most common powertrain technology, there has been a significant increase in the number of battery electric vehicles (BEVs) and this growth is expected to continue [2]. FC electric vehicles (FCEVs) are currently in use in small numbers but are also gaining popularity [3]. Both BEVs and FCEVs are often considered "zero emission" technologies. However, this is only true if the scope is limited to the tank-to-wheel phase and upstream processes, production, and end-of-life (EoL) effects are disregarded.

When comparing both technologies, a comprehensive life cycle assessment (LCA) is necessary to determine which option is more advantageous in terms of reducing GHG emissions. BEVs often require larger battery packs, leading to higher weight, while FCEVs with smaller battery packs typically have higher payload capacity. However, FCEVs have a well-to-wheel efficiency of only 30% in passenger cars, significantly lower than the efficiency of around 70% of BEVs [4]. Currently, the ecological aspects of BEVs and FCEVs are a subject of frequent discussions in both the scientific and engineering communities. Most LCAs of BEVs and FCEVs deal with their manufacturing and use phases. While LCA studies and inventories on recycling Li-ion batteries are widely available, LCA studies and inventories on recycling FC powertrains are scarce [5].

Hydrometallurgical and pyrometallurgical recycling of Li-ion batteries are the two most widely discussed options that have received attention in recent years [6]. Due to the Lithium Nickel Manganese Cobalt Oxide (NMC)-111 cell chemistry's commercial success, it became prevalent in research [7–10]. However, other cell chemistries, such as Lithium Iron Phosphate (LFP), Lithium Nickel Cobalt Aluminium Oxide (NCA), and different proportions of NMC chemistries are arousing interest in the industry and scientific community [11–13]. Table 1 summarizes the global warming potential (GWP) associated with the production and recycling of Li-ion batteries with different cell chemistries. Dai et al. conducted an LCA for NMC111 batteries in automotive applications, which is included in the EverBatt [14] and Ecoinvent [15] databases [7,14,15]. Rajaeifar et al. performed an LCA for different methods of pyrometallurgical recycling [8]. The results varied between −0.77 and −1.22 kg CO_2-eq./kg in NMC111 Li-ion batteries for the closed-loop scenario. Kallitsis et al. [9] evaluated the recycling of NMC111 batteries based on the recycling methods described by Mohr et al. [11]. They supplemented the latter study with different pre-treatment steps, such as sorting, transporting, and dismantling, and modelled the state-of-the-art recycling processes for copper and aluminium. For the case of battery production and recycling in Europe, the results showed a decrease in GWP of 27.4% in the case of pyrometallurgical recycling and 29.9% in the case of hydrometallurgical recycling [9]. Sun et al. assessed the production and hydrometallurgical recycling of NMC622 batteries in China [10]. They discovered that production emits about 124.5 kg CO_2 eq./kWh, while hydrometallurgical recycling reduces the total emissions to 93.6 kg CO_2 eq./kWh. Mohr et al. compared hydrometallurgical, pyrometallurgical, and advanced hydrometallurgical (Duesenfeld) recycling for LFP, NMC111, and NCA batteries based on information from the industry and the GREET [16] database [11]. The NCA cell chemistry has the most substantial recycling benefit, while NMC111 has the lowest GWP from the production phase, resulting in the lowest total GWP value among all cell chemistries. Recycling LFP batteries has the smallest decrease in GWP due to the lack of cobalt, nickel, or manganese in its cathode chemistry. Ciez and Whitacre also compared NMC, NCA, and LFP cell chemistries, but included only the benefits for cell materials [12]. Thus, in contrast to other research, the results showed significantly fewer benefits of recycling. Despite variations in the cell chemistries evaluated, functional units, and system boundaries among these studies, they consistently demonstrate certain similarities. One notable commonality is the superior efficiency of hydrometallurgical compared to pyrometallurgical recycling in lowering the life cycle GWP of batteries. This shared trend can be attributed to the various materials that can be recovered through both recycling methods. Moreover, the studies that included LFP batteries indicated that recycling LFP batteries does not necessarily benefit the overall GWP.

Table 1. GWP of Li-ion battery production and recycling in different studies.

Reference	Cell/Pack Level	Region	Cathode Chemistry	Energy Density (Wh/kg)	GWP (kg CO_2 eq./kWh)		
					Production	Recycling	
						Hydro.	Pyro.
Dai et al. (2019, [7])	Pack	USA	NMC111	143	72.9	-	-
Rajaeifar et al. (2021, [8])	Pack	UK	NMC111	-	-	-	−0.77 (/kg)
Kallitsis et al. (2022, [9])	Pack	China NA Europe	NMC111	105	168.8 133.8 124.1	−65.1 −45.1 −37.1	−59 −40.6 −34.1
Sun et al. (2020, [10])	Pack	China	NMC622	115	124.5	−30.9	-
Mohr et al. (2020, [11])	Cell	-	NMC111 NCA LFP	170 174 108	75.5 85.6 101	−16.4 −18.3 −3.52	−13.8 −15.9 0.45
Ciez and Whitacre (2019, [12])	Cell	USA	NMC622 NCA LFP	210 190 100	42 49 45	−5 −3.5 8	3 1.5 10

Figure 1 presents the LCA results of proton exchange membrane (PEM) FC{ XE "FC" \t "Fuel cell" } systems. Miotti et al., Evangelisti et al. and Usai et al. analysed an 80 kW fuel cell system, while Benitez et al. and Lotrič et al. considered 100 kW and 5 kW systems [17–21]. Miotti et al. analysed a modern fuel cell technology with future scenarios using different materials [17]. Usai et al. updated this study with new data on FC components [19]. Benitez et al. modified the study from Miotti et al. with a focus on the hydrogen tank [20]. Evangelisti et al. (2017) assessed an equivalent system but with variations in data regarding component production [18]. Lotrič et al. generally investigate fuel cells and hydrogen without focusing on fuel cells as an automotive application [21]. They considered a 5 kW fuel cell system, excluded the hydrogen tank, and focused on EoL strategies. All analysed studies differ in the total GWP value per kW FC, primarily due to the different platinum contents and sizes of hydrogen tanks. However, a common trend across all studies is the substantial impact of hydrogen tank production and FC stacks, as these components require platinum for catalyst production and carbon fibre for hydrogen tank production.

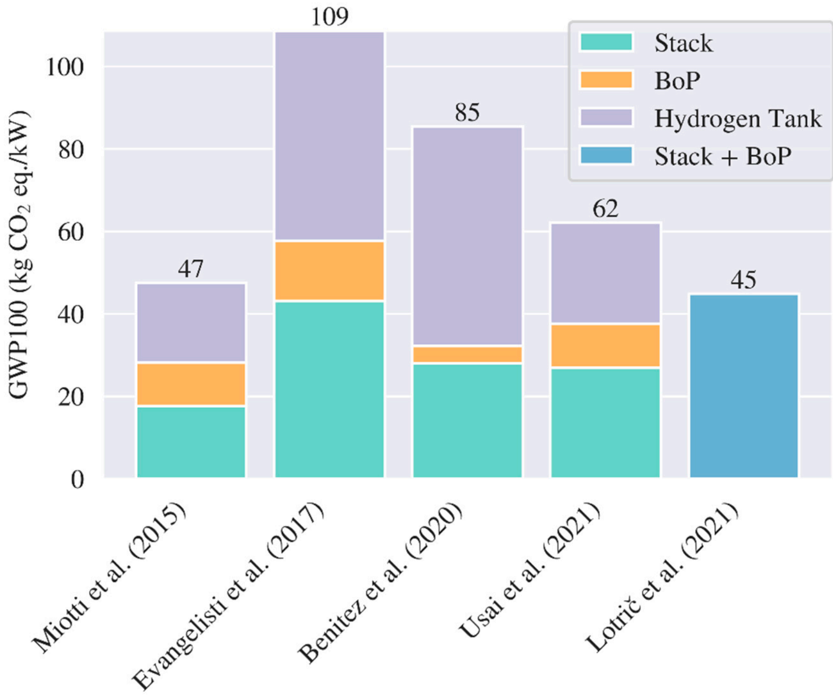


Figure 1. Comparison of the GWP of the production phase for a fuel cell system [17–21].

For evaluating the EoL{ XE "EoL" \t "End of life" } of FC systems, two studies conducted detailed assessments regarding the recycling of platinum catalysts in PEM FC systems [22,23]. These investigations involve a comparison of two distinct hydrometallurgical recycling methods, specifically ion exchange resin and solvent extraction. Duclos et al. found that both recycling approaches yield a comparable reduction in environmental impacts, emphasizing the significance of platinum recycling. Stropnik et al. analyse critical materials in PEM FC systems, including the EoL [24]. Their EoL approach is based on the data by Duclos et al. [23] complemented by industry data for the stack and the balance of plant (BoP{ XE "BoP" \t "Balance of Plant" }) components. Their findings indicate a 12.3% reduction in GWP when accounting for the EoL of the PEM FC system. Notably, their study excludes the hydrogen tank from consideration. Lombardi et al. compare the impacts of various powertrain technologies, including the EoL of the vehicles [25]. However, their approach is also based on the data by Duclos et al. [23]. For the hydrogen tank, they assume recycling for steel and aluminium and disposal to landfill for the remaining materials. Their results indicate a modest reduction in GWP during the EoL phase. It is essential to note that their assessment pertains to a plug-in hybrid FC vehicle, potentially featuring smaller

FC power and tank sizes compared to a pure FCEV. Additionally, they used a 200-bar glass-fibre tank, which may result in fewer environmental impacts than the carbon-fibre tanks considered by the other studies.

Most studies which compare the environmental impacts of BEVs and FCEVs deal with passenger cars [25–28]. Rüdisüli et al. emphasise the importance of the energy source in the use phase while excluding the vehicle production and EoL phases [28]. They underscore that the GHG emissions from electricity and hydrogen are highly dependent on the energy sources (renewable or carbon-intensive). Idris and Koestoer arrive at a similar conclusion and emphasise the significance of vehicle recycling [26]. While the aforementioned studies highlight the importance of the energy sources during the use phase, Joshi et al. extend this perspective by underlining its influence on the production phase [27]. Lombardi et al. stand out by incorporating the EoL phase of vehicles [25]. Their findings reveal a noteworthy reduction in GWP through recycling the plug-in hybrid FCEV and an increase in GWP for the EoL of BEV. It should be noted that they exclusively considered LFP batteries. Moving beyond passenger cars, Sacchi et al. examined the influence of size, payload, and range of alternative truck powertrains on GHG emissions [29]. Their results indicate that for long-range vehicles, the FCEVs exhibit lower GHG emissions per tkm, while BEVs emit fewer absolute emissions. This discrepancy primarily arises from FCEVs' higher payload capacity. Contrarily, Booto et al. report a 20% higher reduction in GWP for battery electric trucks (BETs) compared to FC electric trucks (FCETs) [30]. Shifting focus to electric buses, Grazieschi et al. and Munoz et al. found that with a similar energy source, battery electric buses (BEB) consistently outperform FC electric buses (FCEBs) in terms of lowering the GWP [31,32]. However, it is crucial to note that both studies do not consider passenger kilometres, which might yield different results. This is particularly relevant as BEBs and FCEBs with similar ranges and sizes often exhibit differences in passenger capacity.

The state of the art sheds light on how LCA is applied to batteries and FC systems, particularly in the context of vehicles. However, there is a notable gap in current studies: to the best of our knowledge, no work has been conducted that covers a thorough evaluation combining the production and EoL phases of batteries and FC systems, along with their use in various vehicle types, within a single comprehensive framework. To address this gap, the goal of our study is to compare the climate change potential of PEM FC and Li-ion batteries. However, difficulties arise when comparing both products to a suitable unit. FCs can be compared with the help of power (kW), and batteries by their capacity (kWh). To reasonably compare FCs and batteries with each other, it is necessary to consider them in the context of their application. Complementarily, the analysed literature underlines the need for an LCA study that combines and compares all life phases of BEV and FCEV powertrains, using comparable vehicles, functional units, and nominalization factors. This approach makes it necessary to analyse the pack level of the batteries with different battery chemistries in BEVs, as well as the FC stack, BoP, and hydrogen tank in FCEVs. Considering the electricity and hydrogen mixes in various years allows us to estimate the EoL and use phase for the vehicles. Therefore, we present the GWP of the individual powertrain systems for electric vehicles. We compare and evaluate their impacts in different applications while excluding all similar parts of the vehicle. Therefore, we first describe the LCA approach in our study, as well as the inventories of batteries, FC systems, grid mixes, hydrogen, and typical vehicles for the use phase. Following this, we present the results of the production and EoL of the batteries and the FC system and of the use phase in different vehicles. Thereafter, we compare the production and EoL GWP of batteries and the FC system to other studies. Then, we discuss the results for the use phase of batteries and FC system in different vehicles, and the limitations of our study. Lastly, we present the conclusion of our study.

2. Materials and Methods

The LCA of battery and FC electric powertrains follows DIN EN ISO 14044 [33], using Ecoinvent 3.9.1 as the primary data source [15]. External data sources were consulted when specific components or processes were not available in Ecoinvent. To ensure a comprehensive comparison between these technologies, this study encompasses the entire life cycle of the powertrains, including the production, use, and end-of-life phases. Recycling is considered as the EoL process for both technologies, except for the hydrogen tank (Figure 2).

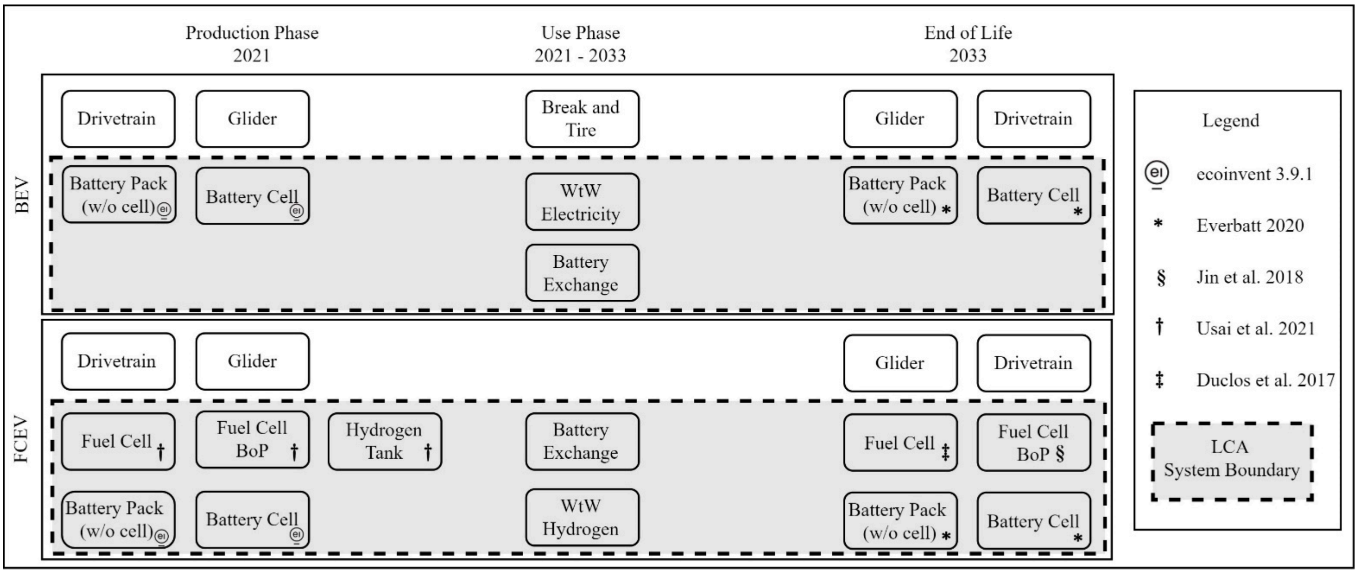


Figure 2. Comparative life cycle assessment approach (product system) [14,15,19,23,34].

The impact assessment of this LCA is based on the ReCiPe 2016, the state-of-the-art impact assessment method [35]. While our primary focus is on environmental impacts related to GWP, we have also included assessments of all ReCiPe impact categories in the supplementary materials (Figures S1 and S2 and Tables S1–S6). This study uses the consequential system model in Ecoinvent. The consequential system model incorporates various assumptions to analyse the effects of a change in a system (e.g., change in demand). In contrast to attributional LCA, marginal increase in supply chains is included, and substitution is employed by crediting processes with avoided burdens from supply chains replaced by their generated by-products. This study uses the consequential approach, and recycled materials are classified as avoided burdens and credited with benefits [36].

2.1. Inventory Analysis of Batteries, Fuel Cells, and Hydrogen Tank

This study combined inventories for several life cycle processes for batteries and FCs, hydrogen production, and electricity generation in Germany and China.

For the BEVs, we compare four different cell chemistries: NCA, LFP, and NMC with varying proportions of nickel, cobalt, and manganese (NMC111 and NMC 811). The Ecoinvent 3.9.1 database provides the production processes of all four cell chemistries [15]. These cell chemistries have a relatively high energy density, which makes them well-suited for automotive applications (Table 2). However, Ecoinvent 3.9.1 lacks data on the recycling processes for these batteries [15]. To address this gap, we incorporated data from EverBatt, which considers hydrometallurgical, pyrometallurgical, and direct cathode recycling, as well as the transportation and dismantling of battery packs [14]. The whole recycling process is divided into three steps. The first step involves transporting the battery pack to the recycling facility. The second step entails disassembling the battery pack, separating the battery modules from each other and the battery pack itself. This provides access to the initial recyclable components. The third step varies depending on the applied method. For hydrometallurgical recycling, the batteries need to undergo a mechanical recycling process, which separates the cathode from the rest of the battery. In this step, the disassembled battery modules are shredded into smaller pieces. Subsequently, the binder and electrolyte are burnt in a calcination process. The shredded battery pieces are then sorted to recover materials such as aluminium, steel, copper, graphite, and plastics. Eventually, the sorted materials undergo a leaching process and solvent extraction to obtain cobalt, nickel, manganese, and lithium compounds. In the case of pyrometallurgical recycling, batteries are fed into a smelter, where the electrolytes and plastics are burnt to generate heat for the thermic reaction. Some materials (aluminium, lithium, and carbon { XE "C" \t "Carbon" }) are oxidized and end up in the slag, while other materials (iron, copper, nickel, cobalt, and manganese { XE "Mn" \t "Managanese" }) become part of the matte, which can be recycled to separate the materials. Both slag and matte can undergo further treatment to recover materials, but EverBatt only provides matte treatment data [14]. Further recycling steps for the matte are analogue to the final steps of hydrometallurgical recycling: the matte undergoes the processes of acid leaching, precipitation, and solvent extraction. All steps after dismantling are summarized in EverBatt [14] and therefore modelled in the same way in Ecoinvent [15]. The considered FC is a low-temperature PEM FC based on the inventories by [19]. To compare the vehicles, we scale the PEM FC, hydrogen tank, and BoP with the respective FC power. In addition, the powertrain contains hydrogen tanks and the BoP. The weight of one tank and the BoP for the PEM FC is based on Usai et al. [19]. Each tank is considered to be a 5 kg hydrogen tank with a weight of 105 kg per tank. For the BoP, we assume a weight of 85.3 kg for an 80 kW FC stack. For the PEM FC, the production process was modelled with background processes provided by Ecoinvent 3.9.1, based on the data from Usai et al., which reflects current state-of-the-art FC technologies [19]. This data set includes the production of the FC stack, auxiliary systems, and hydrogen tank.

Table 2. Energy densities of battery cells and packs.

Battery Chemistry	Energy Density, Cell (Wh/kg)	Energy Density, Pack (Wh/kg)
NMC111 [7]	197	143
NMC811 [37]	209	149
NCA [37]	224	158
LFP [37]	159	116

The FC recycling inventory follows the production process from Usai et al. [19] and covers three steps: transport of the FC to the recycling facility, disassembly, and recycling of the materials. According to the used production data, the FC powertrain is split into different components (Supplementary Materials). We assumed recycling for the FC catalyst based on Duclos et al. [23]. For other components of the FC stack and the BoP, it was assumed that plastic and electronic components are treated as waste materials. At the same time, metals such as copper, steel, and aluminium are modelled as materials for recycling. Hydrogen tanks have a lifetime of up to 30 years [38]. However, insufficient information on the second life of hydrogen tanks was available and the authors of the reviewed studies suggested only one vehicle life for the hydrogen tank use phase [18,20,39]. Additionally, insufficient information on the EoL of hydrogen tanks was available. Therefore, no EoL process was modelled. The air management system's production is modelled using permanent magnets, so a recycling process for these magnets was added based on data from Jin et al. [34].

2.2. Inventory Analysis Grid Mix and Hydrogen

Electricity mixes from Germany { XE "EU" \t "European Union" }for the years 2021, 2030, and 2050 were compared, and for the years in between, data was interpolated (Figure 3). The year 2021 was chosen as the representative last year instead of 2022, which included significant changes in the European energy market that led to increased consumption of fossil energy sources and is treated as an outlier. Analysing future electricity mixes is important for the consideration of the whole use phase of the analysed vehicles, which also includes the foreseeable future. Regarding the production phase, we assume that battery production takes place in China, so the Chinese electricity mix from 2021 is used. For the end-of-life phase, recycling possibilities in Germany are considered. To simulate the recycling of batteries for vehicles that are produced now, electricity mixes for 2030 are applied.

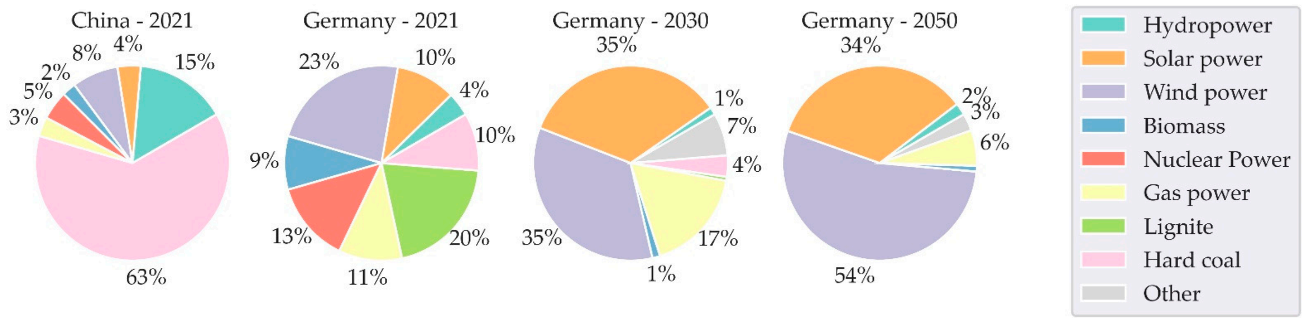


Figure 3. Energy sources for different electricity mixes in different regions, based on [40–42].

For hydrogen production, the mix of hydrogen produced by different sources was modelled (Figure 4). Data from 2020 and forecasts for 2030 and 2050 were interpolated to estimate changes in hydrogen production for each year. The dataset for 2020 also includes the production of hydrogen with fossil fuels with carbon capture as 0.7% of the generated hydrogen [1]. Ecoinvent 3.9.1 includes processes for methane steam reforming and petroleum refinery operations. However, the processes for hydrogen production with coal gasification, fossil fuels with carbon capture, and by-product production were absent in Ecoinvent 3.9.1. Therefore, the coal gasification process data were taken from Burchart et al. [43]. Hydrogen production with carbon capture was deemed negligible and subsequently excluded from our study owing to the challenges associated with the implementation of processes involving carbon capture. It is assumed that hydrogen as a by-product of other processes does not impact the GWP of hydrogen, as its emissions are included in the processes of the main product. We do not factor in any rise in by-product generation; instead, we assume a yearly growth in the quantity of hydrogen generated through electrolyzers. For 2050, it is assumed that 100% of all hydrogen used in automotive applications will be produced by electrolysis. While high-temperature steam electrolysis in combination with nuclear power plants can utilize the exhaust heat from nuclear reaction, in combination with renewable electricity, the heat demand must be served by natural gas [44]. Therefore, hydrogen from renewable electricity is assumed to be produced with PEM electrolyzers. The electrolysis of hydrogen is not present in Ecoinvent 3.9.1 and is therefore modelled according to [45] with the German electricity mix of 2030 and 2050. The process includes the upstream emissions for the construction of a 1 MW PEM stack and BoP. The production of 1 kg hydrogen requires 55 kWh of electricity for the electrolysis (efficiency 60%). Hydrogen production in 2030 is assumed to be a mix of 50% of electrolysis and 50% of the hydrogen mix of 2020. This conjecture is comparable to the IEA's APS scenario for 2030, where 51% of hydrogen is generated by electrolysis and the rest by a mix of fossil fuels, as described in [1].

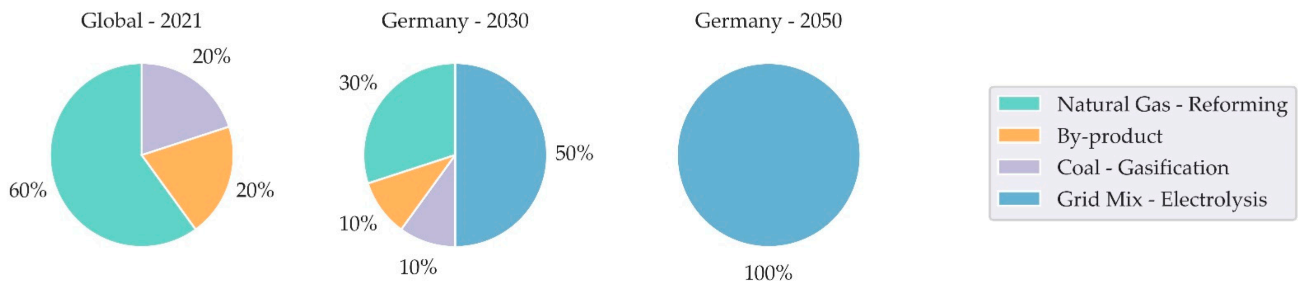


Figure 4. Hydrogen production mix, based on [1].

Table 3 depicts the GWP of electricity and hydrogen production. As of 2030, the emission-free production from by-products diminishes proportionally, and the use of non-renewable electricity for electrolysis leads to higher emissions compared to the conventional mix. Consequently, there is an observable initial increase in emissions per kilogram of hydrogen by 2030. However, by 2050, there will be a significant decline in emissions per kilogram of hydrogen, aligning with the emissions trend of the grid mix. Until 2050, the GWP of electricity production in Germany will decrease by 81%, and the GWP of hydrogen production by 60%. When considering hydrogen's lower heating value of 33.3 kWh/kg, the emissions per kWh are 0.393 kg CO_2 eq./kWh for 2020 and 0.156 kg CO_2 eq./kWh for 2050.

Table 3. GWP of medium voltage electricity and hydrogen production.

Process	Electricity Production (kg CO_2 eq./kWh)				Hydrogen Production (kg CO_2 eq./kg)		
Year	2021	2021	2030	2050	2020	2030	2050
Region	China	Germany	Germany	Germany	Global	Germany	Germany
GWP	0.740	0.490	0.251	0.094	13.103	13.224	5.184

2.3. Selection of Typical Vehicles

In this study, we selected typical vehicles in each category. We assume a lifetime of 12 years for the investigated vehicle types [46,47]. For buses and trucks, we included a battery change after six years [46,48]. The following three subsections define the details of modelled trucks, buses, and passenger cars used to demonstrate application scenarios of energy storage systems. Since the considered vehicles are comparable, we omit modelling the production and end-of-life of all similar parts (vehicle body, chassis, and electronics are not in the scope of the study). For all vehicles, we consider the nominal battery capacity. While every analysed vehicle includes a Li-ion battery (with varying capacity), FCEVs contain, in addition, a fuel cell system with auxiliaries (BoP) and a hydrogen tank.

2.3.1. Passenger Cars

Table 4 shows the vehicle properties for different passenger cars available on the market. The vehicles were chosen to be comparable regarding their weight and vehicle class (coupés and SUVs). Since the FCEV's range is an advantage, we added the Mercedes EQS 580 4Matic to represent long vehicle ranges.

Table 4. Vehicle specifications passenger cars (SFC: specific fuel consumption).

Type	FCEV	FCEV	BEV	BEV	BEV
Reference Vehicle	Toyota Mirai 2 [38]	Hyundai Nexo [39]	VW ID.3 [34]	VW ID.4 [40]	MB EQS 580 4Matic [41]
Range (km)	650	666	426	346	672
SFC ((kg or kWh)/100 km)	0.84	0.95	15.5	16.7	17.7
Battery size (kWh)	0.95	1.56	62	55	120
Fuel Cell size (kW)	128	95	0	0	0
Hydrogen Tank (kg)	6.2	6.33	0	0	0
Passenger Capacity (P)	4.5	4.5	4.5	4.5	4.5
Vehicle weight (kg)	1950	1948	1805	1966	2585
Vehicle Class	Coupé	SUV	Coupé	SUV	Coupé

The consumption data is based on the worldwide harmonised light vehicles test procedure (WLTP) cycle with low test energy to use standardized BEV values. The SUVs consume more energy than a coupé independent of the weight, which usually is caused by larger surface areas, leading to higher aerodynamic drag. The considered lifetime mileage is 180,000 km over 12 years [49], resulting in an average yearly mileage of 15,000 km/a.

2.3.2. Heavy-Duty Trucks

For trucks, we compare four heavy-duty trucks with a gross vehicle weight of 40 t (Table 5). For heavy-duty vehicles, energy consumption is dependent on payload, among other factors. For regional freight traffic, the respective payload is normally around 50% and most manufacturers publish consumption values for 50% payload. However, in long-distance freight transport, vehicles often use 90–100% of their payload [50]. Therefore, we compare the Daimler eActros 300 with a trailer with 50% payload and the same vehicle with 100% payload. Additionally, we compare the Daimler eActros LongHaul (eLongHaul), which represents a heavy-duty BET with a very large battery capacity and 100% payload, and the Daimler GenH2 with 100% payload, as an example of a heavy-duty FCET. The considered yearly mileage amounts to 67,415 km/a [47].

Table 5. Vehicle specification FCET and BET (* calculated with range and battery/tank size, SFC: specific fuel consumption).

Type	BEV	BEV	FCEV
Reference Vehicles	Daimler eActros 300 + Trailer [51]	Daimler eLongHaul [52]	Daimler GenH2 [53]
Assumed Payload (%)	50 (100)	100	100
Max. Payload (t)	23	22	25
Range (km)	220 (170)	500	1000
SFC ((kg or kWh)/100 km)	153 * (198 *)	120 *	8 *
Battery size (kWh)	336	600	70
Fuel Cell Power (kW)	-	-	300
Hydrogen Tank (kg)	-	-	80

2.3.3. City Buses

Extensive initiatives to introduce fully electric city buses are underway in many regions of the world. FCEV city buses have also been commercially available for several years. In this study, we limit ourselves to buses in urban areas. City buses are classified in different lengths and as single-deck or double-deck buses, the latter with a very small market share. We compare two 12 m and 18 m single-deck FCEBs{ XE “FCEB” \t “Fuel Cell Electric Bus” } and BE{ XE “BEB” \t “Battery Electric Bus” }Bs, and adopt the vehicle specifications of the Solaris Urbino Electric and Solaris Urbino Hydrogen buses (Table 6). The 18m BEB has a significantly smaller battery and must be combined with opportunity charging at terminal stops. This infrastructure contributes additional emissions, which are not in the scope of this study. The application of depot or “opportunity charging” is based on operational and cost aspects, as discussed in [46].

Table 6. Vehicle specification of FCEBs and BEBs (* calculated with range and tank size, SFC: specific fuel consumption).

Type	FCEV	FCEV	BEV	BEV
Reference Vehicles	Solaris Urbino Hydrogen [54–57]		Solaris Urbino Electric [58–60]	
Vehicle Length (m)	12	18	12	18
Range (km)	350	350	300	60
SFC ((kg or kWh)/100 km)	10.7 *	14.6 *	158	218
Battery size (kWh)	60	60	658	193
Fuel Cell Power (kW)	70	100	-	-
Hydrogen Tank (kg)	37	51	-	-
Passenger Capacity (P)	87	140	60	138

The energy consumption of electric buses in particular is highly dependent on the ambient temperature, as a very large passenger compartment has to be heated or cooled [61]. In this study, we used the results of a simulation which includes traction and auxiliary consumption for the BEB [60]. For the FCEVs, the SFC is based on manufacturer information (Table 6). Finally, the GHG emissions per passenger kilometre are calculated. The considered yearly range is 60,000 km with an average passenger occupancy of 65%. We assume 3.8 driving days per week to compensate for maintenance days and holidays with less occupancy of the bus fleet.

3. Results

3.1. LCA Batteries—Production and End of Life

Figure 5 presents the GWP per kWh battery. Despite the relatively high energy density of NMC111, this cell chemistry has the highest GWP of all, which can be attributed to its high cobalt content, significantly influencing the battery's GWP. NMC811 and NCA have lower GWPs due to their reduced cobalt content. Both compensate for the lower amount of cobalt with a higher amount of nickel, which results in a higher GWP for the nickel content. Although the cobalt content of the NCA is higher than that of the NMC811, it still has a lower GWP as it has the highest energy density among all examined cell chemistries. The LFP cell chemistry has the lowest GWP of all cell chemistries. This can be attributed to the absence of cobalt and nickel. However, due to its low energy density, the GWP per kWh is still close to other chemistries. Another implication of the low energy density of the LFP chemistry is the exceptionally high impact of aluminium on its GWP. Aluminum plays a significant role in the GWP of all cell chemistries, as it is used as a casing material for prismatic cells in Ecoinvent 3.9.1.

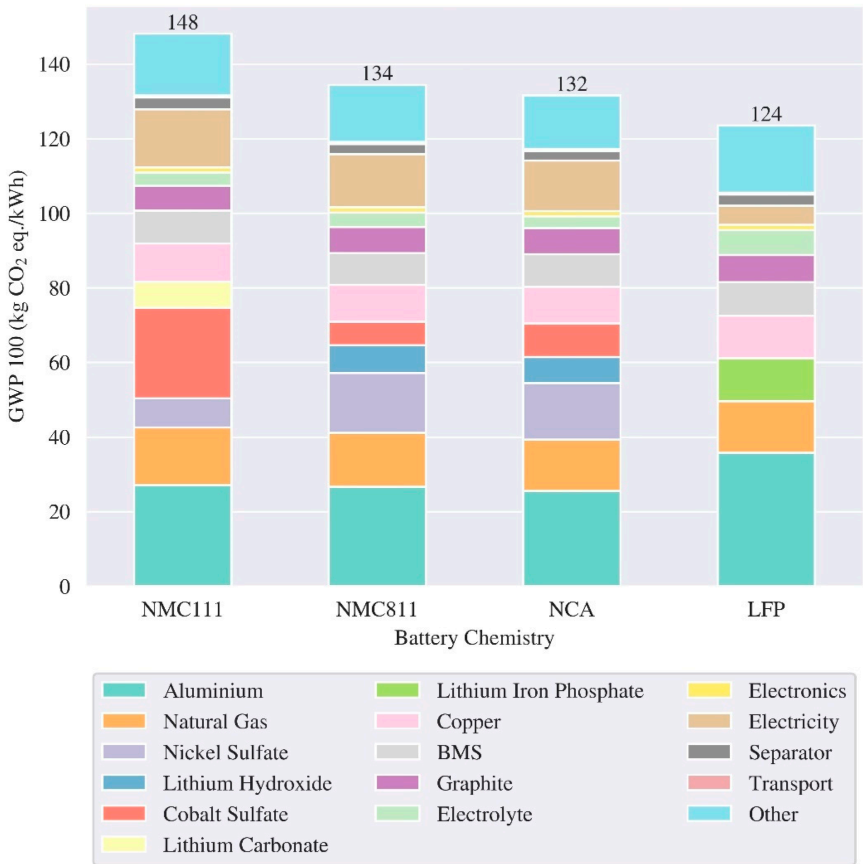


Figure 5. Battery pack materials and processes for different cell chemistries—Production.

Figure 6 depicts the results of the LCA for different kinds of battery recycling applied to batteries with varying cell chemistries. The positive values show the processes marked as the effort of the recycling, and the negative processes depict the avoided GWP emissions due to the decreased need for primary materials. The results demonstrate the significant advantage of hydrometallurgical recycling over pyrometallurgical in all cases. Nickel and cobalt are the materials with the most substantial influence on the overall decrease in GWP, followed by lithium carbonate. It is also noticeable that recycling LFP batteries has the smallest decrease in GWP since this cathode chemistry lacks nickel or cobalt, which have higher benefits regarding the GWP than other battery materials.

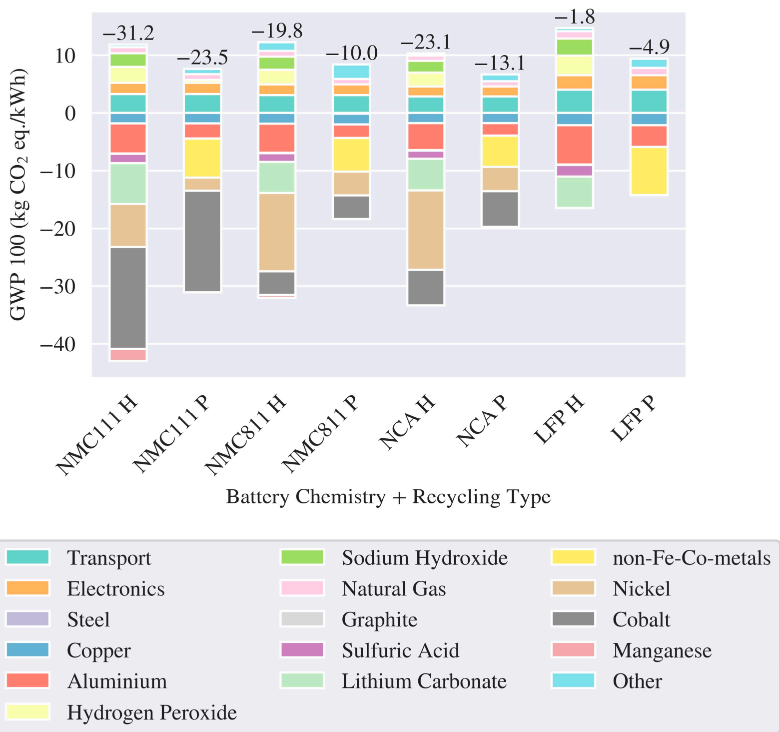


Figure 6. Battery pack materials and processes for different cell chemistries—Recycling (H—hydrometallurgical; P—pyrometallurgical).

Compared to the production phase (Figure 7), hydrometallurgical recycling of NMC111 batteries shows a 21% reduction in GWP, which is the highest among all cell chemistries and recycling methods. However, the NCA battery has the lowest GWP after the recycling phase—108 kg CO_2 eq/kWh. The LFP battery has the highest GWP after the recycling phase—118 kg CO_2 eq/kWh.

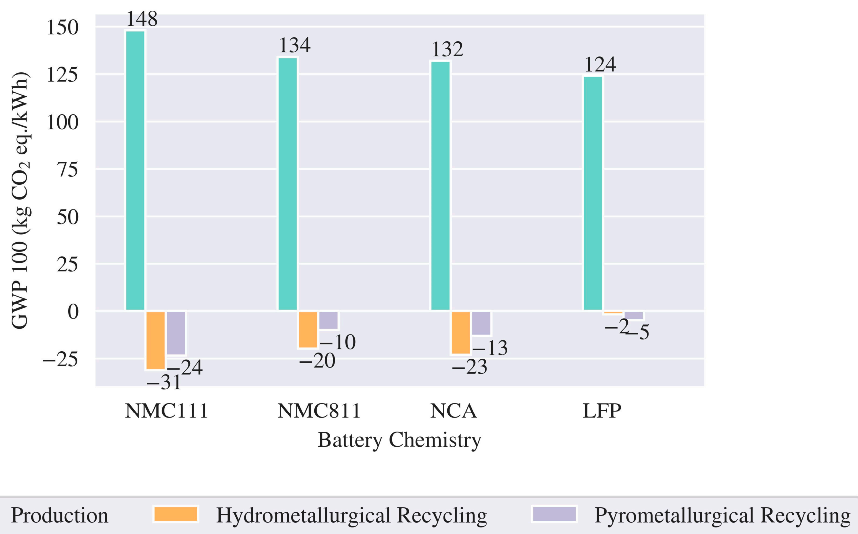


Figure 7. GWP of battery pack recycling and production.

3.2. LCA Fuel Cell and Hydrogen Tank—Production and End of Life

Figure 8 depicts the GWP per kW for recycling and producing an FC tank and BoP. The tank and BoP emissions refer to an 80 kW fuel cell and are therefore divided by 80 for the representation. The recycling process includes all components of the FC. The largest contributor to the production GWP is the hydrogen tank, followed by the PEM production, bipolar plates, and other BoP production. Recycling the PEM with the catalyst also leads to a significant reduction in GWP, followed by the recycling of the bipolar plates. As a rare metal, platinum has an exceptionally high GWP in the production phase, which can be reduced through recycling. The recycling of the BoP reduces the GWP while the collection, transportation, and recycling of other stack components increases the total GWP.

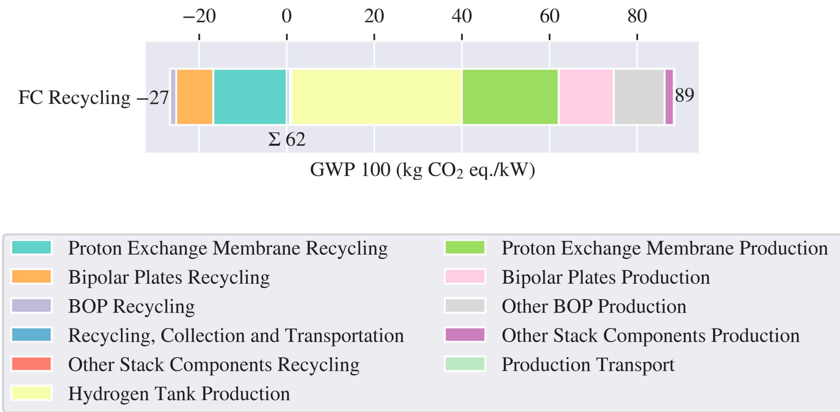


Figure 8. GWP of fuel cell system—Production and recycling.

3.3. LCA Vehicles (w/o Glider, Motor, Brake, and Tire)

3.3.1. Passenger Cars

The results for passenger cars show the benefits of BEVs in the private-use sector (Figure 9). The VW ID.3 has the lowest overall GWP impact, which is slightly lower than that of the VW ID.4. VW ID.3 has a bigger battery than VW ID.4. However, its overall vehicle mass is lower compared to the ID.4, which reduces the energy consumption in the use phase. The FCEVs have a higher GWP impact overall due to higher use-phase emissions. Only the MB EQS 580 4Matic's GWP is close to the GWP of the FCEV, caused by its large battery, high vehicle weight, and therefore considerable energy consumption. It should be noted that GWP impact results only include the emissions from energy use in the use phase and the life cycle emissions of batteries and FC systems.

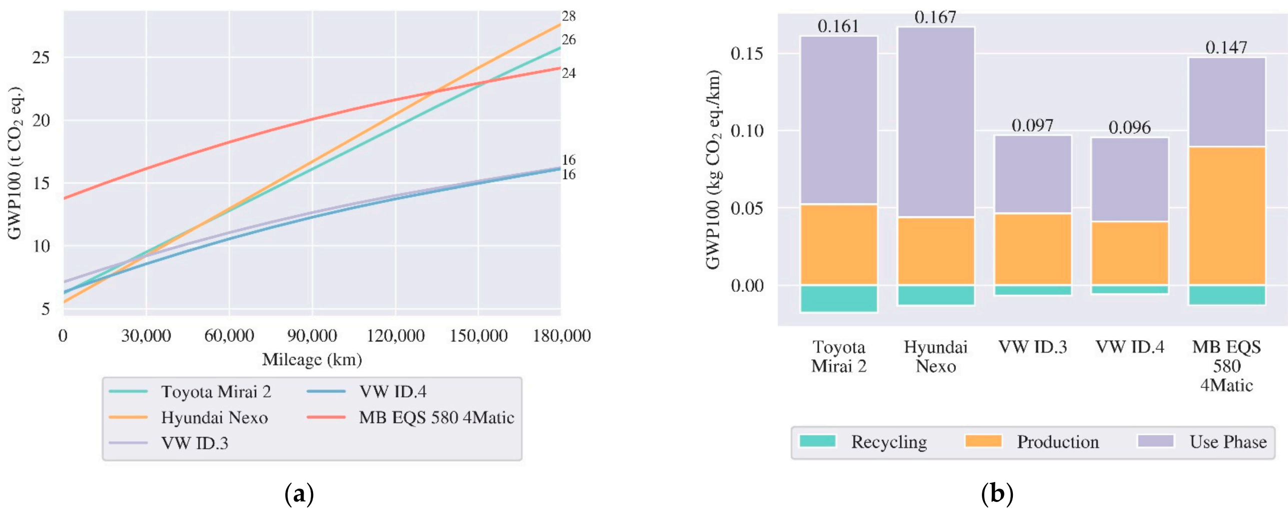


Figure 9. GWP Passenger Cars: (**a**) absolute GWP; (**b**) GWP per km.

3.3.2. Heavy-Duty Trucks

The BETs with 100% payload have a lower absolute GWP than the FCETs (Figure 10). The eLongHaul exhibits the highest production share in absolute GWP; however, the low use-phase emissions result in the lowest absolute GWP. While the eActros 300 with 50% payload exhibits the second lowest absolute emissions, the same truck with 100% payload shows an approximately 30% higher absolute GWP than the eLongHaul. However, regarding the emissions per tkm, the eLonghaul with 100% payload shows the lowest GWP and the eActros 300 with 100% payload performs better than the GenH2 per tkm. The eActros 300 with 50% payload has the highest GWP per tkm. The GenH2 reveals the lowest production and highest use phase share in GWP.

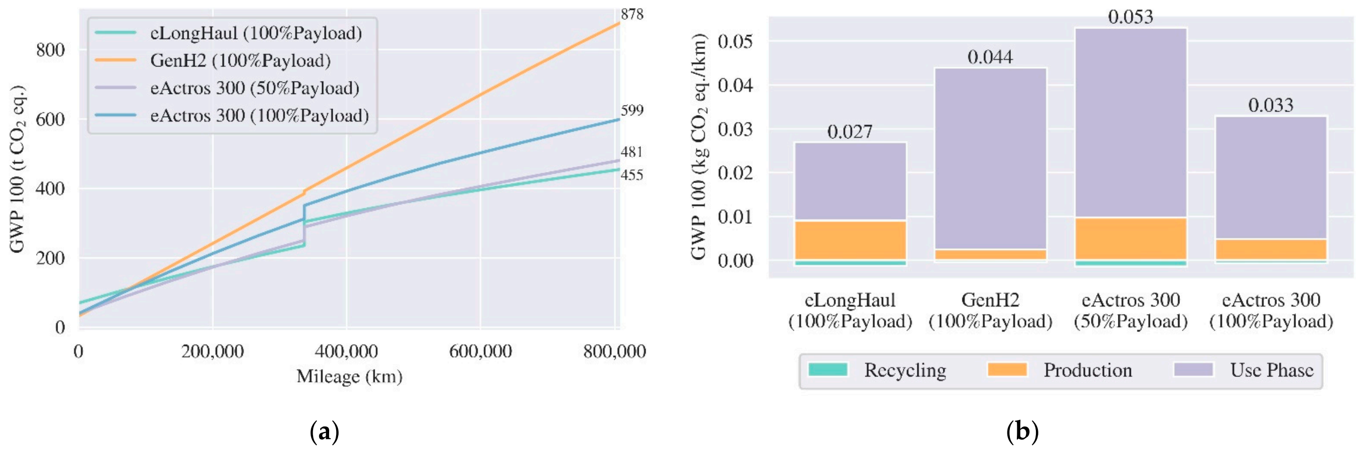


Figure 10. GWP Heavy-Duty Trucks: (**a**) absolute GWP; (**b**) GWP per tkm.

3.3.3. City Buses

The BEBs show lower values of GWP than the FCEBs (Figure 11). Despite its considerably smaller battery, the 18 m BEB has slightly higher overall GHG emissions than the 12 m BEB. However, regarding the emissions per passenger kilometre, the 18 m BEB has the lowest GWP. The 12 m BEB performs better than the 18 m FCEB and the 12 m FCEB has the highest emissions per passenger kilometre. The 12 m BEB has the highest share of production emissions, while the 12 m FCEB exhibits the highest use phase share in GWP.

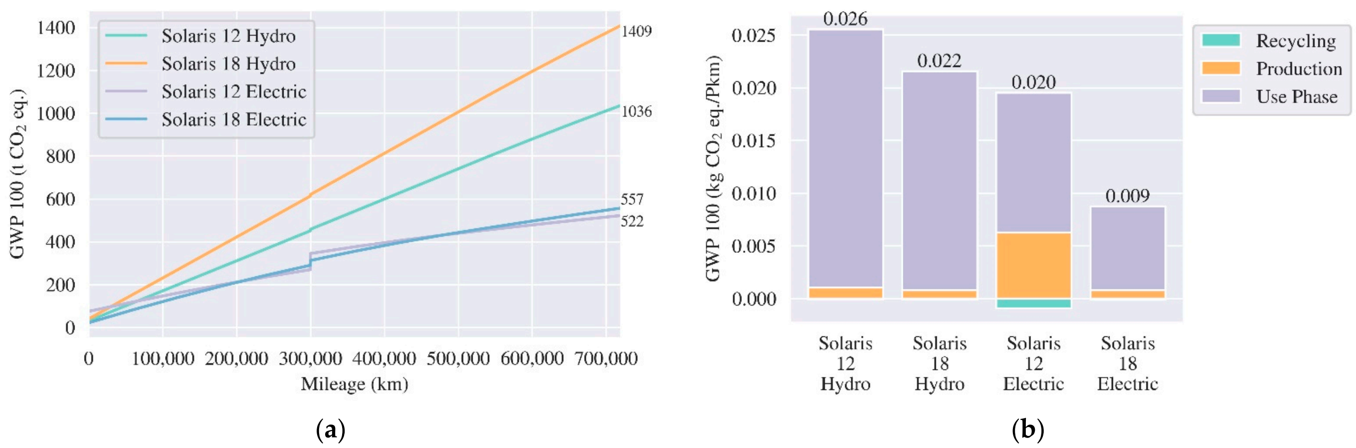
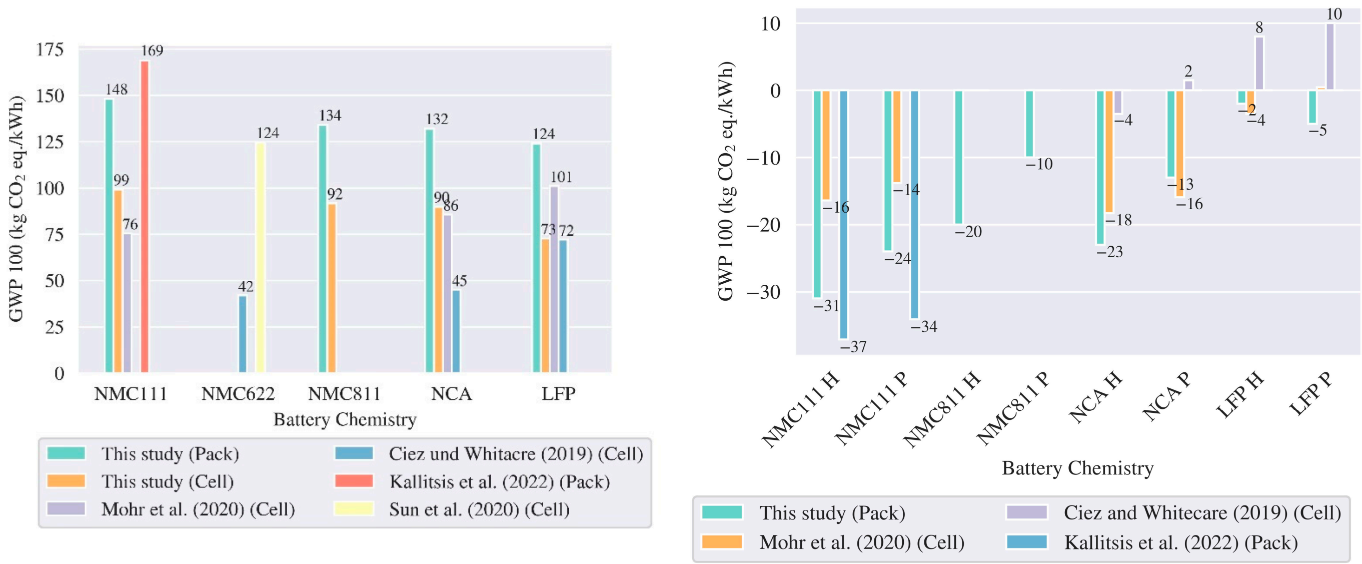


Figure 11. GWP City Buses: (**a**) absolute GWP; (**b**) GWP per Pkm.

4. Discussion

4.1. Comparison with other Studies

In contrast to the consequential LCA approach of this study, some of the compared studies used the attributional LCA approach [9,10,12,17]. Other studies did not explicitly specify their LCA approach [11,18–21] and were therefore considered as attributional LCA. We believe that a consequential LCA approach is better suited for a comprehensive comparative study which considers a substantial transformation of the transport sector. Figure 12 shows our results for cell and pack levels and the results from other studies. During the production phase, the LFP battery exhibited the lowest GWP among all the evaluated battery chemistries. In contrast to Mohr et al. and Ciez and Whitecare [11,12], the pyrometallurgical recycling of LFP batteries yielded benefits. However, it is noteworthy that recycling only slightly decreased the overall GWP for LFP batteries, which stands in contrast to the other chemistries where recycling led to a high reduction in GWP. Specifically, hydrometallurgical recycling of nickel-cobalt-based batteries yielded the most substantial benefits, while pyrometallurgical recycling resulted in intermediate gains.



(**a**) (**b**)

Figure 12. GWP of different cell chemistries, compared to other studies, cell and pack level [9–12] (**a**) Production (**b**) Recycling.

For the production phase of the FC, we find nearly identical results as Usai et al. [19]. The slight deviation in the production of the FC stack and hydrogen fuel tank may be attributed to the fact that we use a newer version of the Ecoinvent database. As for the EoL of the FC powertrain, the results show a significant potential of the recycling technology for the catalyst. Most of the recycling processes for the FC BoP and stack components other than the catalyst were simplified due to the lack of sufficient data on this topic. Therefore, electricity, heat and transportation emissions were not included, which would increase the overall GWP. Moreover, the analysed recycling approach of the FC catalyst does not depict recycling on an industrial scale, which can impact the results. Some of the process steps can be realized with improved efficiency on the industrial scale. Generally, the recycling process described by Duclos et al. [22] omits steps that are challenging to optimize for industrial applications.

4.2. Comparative Life Cycle Assessment

The findings of the study clearly show that the recycling of FC systems results in a substantial decrease in GHG emissions and that the recycling of batteries exhibits a smaller but still significant reduction. However, even with higher production and end-of-life emissions, all BEVs exhibited a lower GWP over vehicle lifetime than their FC electric counterparts. This is attributed to lower emissions during the use phase. Hydrogen produced by fossil fuels without carbon capture or by electricity based on fossil fuels has high upstream emissions. Additionally, the overall tank-to-wheel efficiency of FCEVs is lower compared to BEVs. This means FCEVs would only have a smaller use phase GWP than their BEV counterparts in regions with grid mixes that are more carbon-intensive than the hydrogen mixes. This may be the case if, for example, green hydrogen is imported. At the same time, green hydrogen produced by a renewable electricity mix would always underperform direct electricity production.

The MB EQS 580 4Matic results in a higher GWP than the other BEVs, close to the GWP of FC electric passenger cars. However, if comparing the MB EQS 580 4Matic with other luxury FCEVs (for example the concept vehicles Audi h-tron quattro or Mercedes F-Cell), the higher consumption values of those vehicles would result in higher emissions. Consequently, for passenger cars in the same vehicle class, BEVs are most likely to have lower emissions than FCEVs.

The eActros 300 with 50% payload (100%) has 45% (32%) less emissions than the GenH2. The eLongHaul has the lowest GWP with 48% fewer emissions overall. However, it should be noted that the GenH2 and the eLongHaul are both prototype vehicles. Since the differences in payload for BET and FCET are noticeably small, the emissions per tonne-kilometre align with the overall results. At the same time, the eActros with 100% payload exhibits the second largest absolute emissions while having the second lowest GWP per tkm. It should be noted that the range of the vehicles drastically decreases with higher payloads. Thus, the trade-off between range, payload and consumption should be assessed in detail for individual applications.

Regarding their overall life cycle emissions, the BEB has a substantially lower GWP than the FCEB. Even when considering emissions per passenger-kilometre, the BEB show lower emissions, despite smaller passenger capacities. The 18m BEB had the lowest GWP per passenger kilometre with 0.009 kg CO_2 eq/Pkm.

4.3. Limitations

The primary challenge in modelling this LCA was the consistent integration of data from various sources. For instance, the battery recycling in Everbatt [14] did not account for the prismatic casing modelled in Ecoinvent [15]. The hydrogen mix from fossil fuels neglects the possibility of carbon capture completely. Hydrogen produced by reforming or gasification with carbon capture would decrease the GWP per kg of hydrogen accordingly.

Another challenge was finding comparable vehicles, as there are presently just a few vehicles on the market that can be sold in both BEV and FCEV versions. Comparing vehicles with similar properties is an appropriate approximation, although it introduces uncertainty.

The calculated impacts of FC powertrain recycling demonstrate decreases in GWP comparable to recycling Li-ion batteries. Nonetheless, the described recycling processes have yet to achieve industrial scale. To evaluate the recycling of fuel cell systems, our literature review and own assessment unmistakably underscore the need for industry data and scientific studies.

Manufacturers of hydrogen tanks state a service life of 30 years [38,62], which is longer than the service life of vehicles. We have not found any information on the possible reuse of hydrogen tanks in mobile applications. A reuse of the hydrogen tank would allow an allocation of the manufacturing and EoL emissions, and thus reduce the emissions related to a single vehicle life.

The losses of hydrogen due to transportation and compression are not included in our calculations. Considering these effects would increase the emissions of the FCEVs, therefore enlarging the gap between the FCEVs and BEVs.

The trucks in our study are comparable in size. However, the GenH2 offers twice the range of the eLongHaul, and four times that of the eActros 300. Therefore, the GenH2 can be used for more use cases. At the same time, regulations in most parts of the world require truck drivers to take a break after 3–5 h of driving. This break could be used to recharge/refuel the vehicle [50]. The question of which range is actually advantageous depends on numerous factors that cannot be analysed here.

Our assessment excluded all vehicle parts which are similar for FCEV and BEV. It is important to understand that our results do not display full life cycle emissions for the regarded vehicles and should only be used for comparing these two technologies used in comparable vehicles (regarding size, weight, payload, passenger capacity etc.). For comparing the vehicles with other technologies (e.g., ICEV), the differing components must be included. For a whole LCA of the vehicles, all parts of the vehicles should be considered.

This study focused on GWP. However, there are further environmental impact categories (e.g., acidification, resource depletion and air quality), but also social and economic categories which are impacted by alternative powertrains and not within the scope of this study.

5. Conclusions

The primary outcome of this study is a comparative cradle-to-grave LCA for batteries and fuel cell systems. With this work, we close a considerable gap regarding the LCA of zero-emission vehicles. We provide a detailed evaluation that combines the production and end-of-life of batteries and fuel cell systems with their use phase in typical vehicles. We expand the system boundary of the LCA for both technologies to integrate relevant differences for the vehicle types like mileage, payload, and passenger capacity. This way, we can present both the absolute emissions and the emissions relative to the transport capacities of both technologies. Our results highlight that fuel cell and battery systems exhibit large emissions in the production phase. Recycling can significantly offset some of these emissions, but a comparison of the examined technologies revealed substantial differences. The recycling of fuel cell systems offers significant potential for decarbonization primarily based on platinum recycling. The assessed recycling processes for NMC and NCA batteries primarily focus on nickel and cobalt, as the recovery of these materials is possible with high recovery rates and promising financial potential. The recycling of LFP batteries, which lack these materials, results in a smaller decarbonisation benefit. At the same time, it should be emphasized that decarbonisation is just one aspect of recycling, and material recovery can be of greater importance.

When comparing passenger cars, trucks and buses, battery electric vehicles consistently outperform fuel cell electric vehicles regarding absolute greenhouse gas emissions. Hence, we recommend prioritizing battery electric over fuel cell vehicles. However. deploying fuel cell vehicles could become attractive in a hydrogen economy scenario where other factors, e.g., the conversion and storage of surplus renewable electricity via electrolysis, become important.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/wevj15030114/s1, Figure S1: FC Production GWP; Table S1: FC Production GWP; Figure S2: FC Recycling GWP; Table S2: FC Recycling GWP; Table S3: FC ReCePi; Table S4: Batteries Production GWP; Table S5: Batteries Recycling GWP Infra; Table S6: Batteries ReCePi.

Author Contributions: Conceptualization, P.S. and A.M.S.; methodology, A.M.S., P.S. and L.F.; software, L.F., P.S. and A.P.; validation, A.M.S., P.S. and D.G.; formal analysis, P.S. and A.M.S.; investigation, P.S. and L.F.; resources, A.M.S. and D.G.; data curation, P.S., L.F., A.M.S. and A.P.; writing—original draft preparation, P.S., L.F. and A.M.S.; writing—review and editing, A.M.S., P.S., A.P. and D.G.; visualization, A.P., L.F., P.S. and A.M.S.; supervision, A.M.S. and D.G.; project administration, A.M.S. and D.G.; funding acquisition, D.G. and A.M.S. All authors have read and agreed to the published version of the manuscript.

Funding: Parts of this work were funded by the Deutsche Forschungsgemeinschaft (DFG, German Research Foundation), project number 398051144, and by the Federal Ministry of Education and Research of the Federal Republic of Germany as part of the Research Campus Mobility2Grid [www.mobility2grid.de (accessed on 7 March 2024)], funding code: 03SF0674A.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: Restrictions apply to the availability of inventory data. Data were obtained from Ecoinvent 3.9.1 and are available at [https://ecoinvent.org/ (accessed on 7 March 2024)] with the permission of Ecoinvent. All other data are publicly available or contained within the article or Supplementary Materials.

Acknowledgments: We acknowledge the support of the German Research Foundation and the Open Access Publication Fund of TU Berlin.

Conflicts of Interest: The authors declare no conflicts of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript; or in the decision to publish the results.

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