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Review article

Global warming potential of lithium-ion battery energy storage systems: A review

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ABSTRACT

Decentralised lithium-ion battery energy storage systems (BESS) can address some of the electricity storage challenges of a low-carbon power sector by increasing the share of self-consumption for photovoltaic systems of residential households. Understanding the greenhouse gas emissions (GHG) associated with BESSs through a life cycle assessment (LCA) is important. This review is the first review to look at life cycle assessments of residential BESSs. Our analysis reveals that GHG emissions associated with 1 kWh lifetime electricity stored (kWh_d) in the BESS are between 9 and 135 g CO_2eq/kWh_d. Surprisingly, BESSs using NMC were consistently reported with lower emissions for 1 kWh_d than BESSs using LFP. Expanding the system boundary to include the photovoltaic system used for charging the BESS, the photovoltaic system contributed 40–70 % to total GHG emissions. Only two out of 13 LCA studies provided own primary data for the BESS. Therefore, additional sources for primary data were identified. GHG emissions associated with LFP and NMC lithium-ion battery production showed mixed results, depending on the data source. Employing most up-to-date primary data we find LFP with 8 g CO_2eq/kWh_d and NMC with 12–14 g CO_2eq/kWh_d, challenging some results of reviewed studies.

1. Introduction

The reduction of annual greenhouse gas (GHG) emissions, among which carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are the most prominent, is a fundamental issue [1–3]. Estimates put the remaining carbon budget to limit global warming to 1.5 °C at around 500 $GtCO_2$. This contrasts with emissions of 38.0 $GtCO_2$ in 2019, slightly declining in 2020 but on track to reach similar levels in 2021. If the global warming potential (GWP) of other greenhouse gases such as N_2O and CH_4 is included, emissions in 2019 were >50 $GtCO_2$ equivalents (CO_2eq) [4,5]. To address climate change, substantial investments in renewable energy systems and low-carbon technologies are necessary [6–8]. The power sector accounts for a considerable proportion of all GHG emissions [4,9]. In 2021, 40 % of electricity has been generated by low-carbon sources, slightly surpassing the 35 % share of coal. Still, wind and photovoltaic (PV) generate only 10 % of global electricity [6]. The current rate of adding renewable energy systems is substantially short of the 2030 International Energy Agency (IEA) annual target rate of 390 GW new wind power and 630 GW new solar photovoltaic [6].

One inherent problem of wind power and photovoltaic systems is intermittency. In consequence, a low-carbon world would require sufficiently large energy storage capacities for both short (hours, days) and long (weeks, months) term [10,11]. Different electricity storage technologies exist, such as pumped hydro storages, compressed air energy storage or battery energy storage systems (BESSs) [11–13]. Lithium-ion batteries (LIBs) have become the dominant technology for BESSs, in particular for short term storage [14–17]. Residential BESSs are employed to increase self-consumption of photovoltaic systems, sometimes referred to as energy time shift. Trends indicate that a significant share of residential photovoltaic systems come with BESSs in economically feasible locations [16,18,19]. Large-scale battery storage systems are used for a wider range of applications such as frequency regulation,

Abbreviations: ANL, Argonne National Laboratory; BESS, battery energy storage system; EoL, end of life; GHG, greenhouse gas; GREET, Greenhouse gases, Regulated Emissions, and Energy use in Technologies model; IEA, International Energy Agency; IPCC, Intergovernmental Panel on Climate Change; kWh, kilowatt hour; LCA, life cycle assessment; LCI, life cycle inventory; LCIA, life cycle impact assessment; LIB, lithium-ion battery; LFP, lithium iron phosphate; LMO, lithium manganese oxide; NCA, lithium nickel cobalt aluminum oxide; NMC, lithium nickel manganese cobalt oxide.

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black start, and voltage support but also to increase self-consumption of renewable energy sources [14,15,20,21]. Storage capacity of battery systems typically ranges from residential systems with 2–25 kWh to industrial battery systems on a MWh scale [14–16].

Demand for BESSs continues to grow and forecasts expect that almost 3000 GWh of stationary storage capacity will be needed by 2040, providing substantial market opportunities [22]. Investments in battery energy storage systems were more than $5 billion in 2020. $2 billion were allocated to small-scale BESS and $3.5 billion to grid-scale BESSs [23]. This might seem small in comparison to $118 billion invested in electric vehicles in 2020, or the $290 billion investment in wind and solar energy systems. However, the growth rate throughout the last decade is substantial (investment in 2010 was at $0.1 billion) [24].

Important factors for strategic decisions about suitable energy storage systems include environmental considerations. Here, life cycle assessments (LCAs) provide a standardized framework for assessing environmental impacts of products or services [25,26]. LCAs assess up to 20 different environmental impact categories – from global warming potential to ozone depletion and human toxicity [27–29]. Global warming potential (GWP), commonly expressed in $kgCO_2eq$, is the environmental impact most looked at in energy systems [30–33]. In fact, reading publications from the Intergovernmental Panel on Climate Change (IPCC) or IEA, which frequently serve policy makers in their decision-making puts heavy emphasis on the global warming potential of different energy transition scenarios and renewable energy systems [1,2,6,34]. While focusing too heavily on climate change poses the risk of omitting other factors of the United Nations Sustainable Development Goals [3], climate change has become a central element in any political debate. Expected increases in the taxation of GHG emissions alongside requirements to provide more information about life cycle CO_2 emissions of products require a thorough understanding of GHG emissions from a LCA perspective [34,35].

To the best of our knowledge, no has study has yet reviewed life cycle assessments of residential battery energy storage systems.[1] Therefore, the present work closes this gap and provides guidance for future research on BESSs from an environmental perspective. Emphasis is placed on residential BESSs to boost self-consumption. Also, particular attention is paid to addressing different LIB chemistries. Global warming potential has, although criticized, remained the most central environmental impact category of many LCAs conducted for lithium-ion batteries [33,36,37]. As the data basis for GWP remains the strongest and most accessible it has been chosen as the reference impact category in the present work.

The structure of this paper is as follows: Section 2 provides technical detail about BESSs and outlines the search strategy for relevant publications. In Section 3 reported findings of the studies are presented, visualized, and compared. Depending on the design of the publications, results are presented with and without GWP of the photovoltaic electricity used for charging the BESS. Few LCA studies of BESSs provided own primary data, arguably one of the most reliable data sources [25,26], for the production of the battery system but relied on earlier work instead. Therefore, Section 4 examines some often-referenced primary data studies. The section also looks at additional sources of primary data, not yet used as reference in LCAs on BESSs. Section 5 summarizes key findings and addresses opportunities for future residential BESSs with low global warming potential.

2. Technical details of BESSs and literature search

2.1. Technical description of BESSs

Increasing the self-consumption of photovoltaic electricity is the main application for residential BESSs and one of many applications for large-scale BESSs [14,15]. Fig. 1 recaps the working principle of a residential BESS used for increasing self-consumption. PV-systems generate electricity when the sun shines. Thus, during cloudy periods and at night PV-systems do not generate electricity. Charging a BESS during the day with excess photovoltaic electricity allows that some electricity demand is met in the evening by discharging the battery [43]. Several studies have looked at sizing criteria for the PV-system and the BESS as a function of the annual electricity consumption of the household and its geographic location [43,44]. For an average residential household in northern Europe, a self-sufficiency of around 30 % is reached with a 1 kW_p/1 $MWh_{consumption}$ PV-system. This means that 30 % of the annual electricity consumed by the household is provided by the photovoltaic system. Without a BESS, this leaves the household drawing 70 % of its annual electricity consumption from the grid [43,44]. The optimal size of the photovoltaic system and BESS are often expressed in relation to the total annual electricity consumption. If a household has an annual electricity demand of 4 MWh, this leads to a 4 kW_p photovoltaic system according to the above-mentioned sizing criteria. Adding a BESS with a battery capacity of 1 kWh_c/1 $MWh_{consumption}$ to the 1 kW_p/1 $MWh_{consumption}$ photovoltaic system results in a self-sufficiency rate of 60 % [44]. The household in Fig. 1 can draw electricity from three potential sources with different availability profiles: I. Electricity consumption directly from the photovoltaic system, II. Electricity consumption from the BESS which has been charged with PV-electricity at an earlier time, III. Electricity consumption from the grid. The example above leads to I = 30 %, II = 30 % and III = 40 % so that 30 % of the electricity is provided directly by the photovoltaic system, another 30 % by the BESS which was charged with photovoltaic electricity earlier, and the remaining 40 % from the grid.

Each electricity stream comes with different environmental impacts, for example, lifecycle GHG emissions associated with 1 kWh of electricity delivered[2] (kWh_{pv}, kWh_{d+pv}, kWh_{grid}, see Fig. 1). While life cycle GHG emissions of the photovoltaic system are somewhat dependent on the geographic location of the system, substantial variance exists for the GHG emissions associated with 1 kWh of electricity drawn from the grid. For example, emissions are <30 gCO_2eq/kWh_{grid} in Norway and >700 gCO_2eq/kWh_{grid} in Russia [45]. If the location is known, GHG emissions associated with stream III (grid) are reasonably well studied, as are GHG emissions associated with stream I (photovoltaic system). For instance, Ecoinvent 3.7 provides large quantities of data on GHG emissions of 1 kWh electricity generated by photovoltaic systems [45]. Data is available at a regionalized level, thereby allowing for comparison of

[1] Two reviews have recently been published on life cycle assessments of stationary energy storage systems. First, Pellow et al. [36] reviewed LCAs for large, grid-scale battery storage systems, employed in different applications such as frequency regulation and energy time shift. Rahman et al. [37] looked at LCAs for a variety of energy storage systems including mechanical, chemical, and electrochemical technologies. Their study took a high-level perspective on lithium-ion batteries and did not differentiate between cathode chemistries, such as LFP, NMC, LMO and NCA which are known to determine the electrochemical properties, such as energy density and lifespan [38,39]. Earlier reviews have looked at life cycle impacts of lithium-ion batteries with focusing on electric vehicle applications [40,41] or without any specific battery application [33,42]. Peters et al. [33] reported that on average 110 $kgCO_2eq$ emissions were associated with the cradle-to-gate production of 1kWh_c lithium-ion battery capacity. Ellingsen et al. [41] reported a substantial variety between 38 $kgCO_2eq$ and 356 $kgCO_2eq$ as results for 1kWh_c of lithium-ion battery capacity. For stationary storage applications and particularly small BESSs used for increased self-consumption of renewable energy, emissions associated with 1kWh_d of electricity stored and delivered throughout the entire lifetime of the battery should be more relevant.

[2] Units and description of electricity flows: kWh_d (1kWh lifetime electricity stored in BESS); kWh_{pv} (1kWh electricity generated by photovoltaic system); kWh_{d+pv} (1kWh lifetime electricity stored in BESS and charged with photovoltaic electricity); kWh_{grid} (1kWh electricity drawn from grid); kWh_c (1kWh of battery capacity).

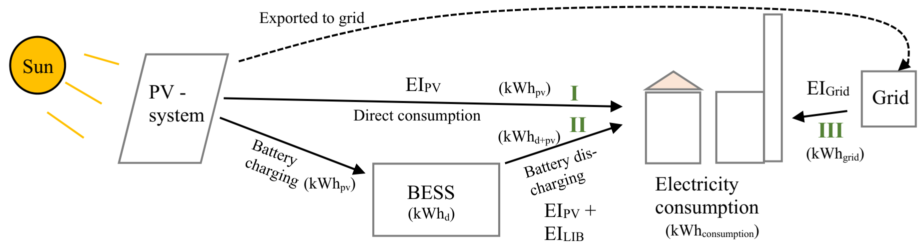

Fig. 1. Working principle of a residential photovoltaic system with added battery energy storage system.

photovoltaic systems in different countries, different sizes and technologies. A similar level of detail is available for environmental impacts of grid electricity in different countries [45]. On the other hand, little to no information is present in Ecoinvent 3.7 for stream II, that is GHG emissions per 1 kWh_d of electricity stored by BESSs [45,46].

Properties of LIBs are influenced by anode and cathode material. Commonly, graphite is applied as anode material. However, more advanced anode materials include small amounts of silicon to increase energy density [38,39]. Commercial materials frequently used as cathode material include LMO, LFP, NMC and NCA, see Table 1. Lithium cobalt oxide (LCO) has been used in consumer electronic applications, but its market share is declining due to the high cobalt content [47]. As summarized in Table 1, cathode materials with high nickel content, such as NMC and NCA have a comparatively high energy density. Efforts are underway to increase the nickel content in NMC cathodes to achieve higher energy density and less need for cobalt [38,39]. For residential BESSs, NMC and LFP have become the most popular choice, with a number of well-known manufactures selling these systems. Still, as research has also been conducted for residential BESSs with LMO and NCA cathodes [21,48] these are included in the present work to provide a more comprehensive overview.

While high specific energy of LIBs is of particular importance for electric vehicle application to achieve good performance levels, restrictions for stationary applications are lower. For residential BESSs high cycle and calendar life are crucial. Cycle life and calendar life are the two main factors involved in battery degradation [49]. Temperature, depth of discharge (DOD) and charge-discharge rates have an impact on the battery life [50]. Regularly, capacity fade to 80 % is seen as end of life (EoL) for a battery system [38], although Thomas et al. [48] have argued that LIBS in BESSs should be used until the capacity has decreased to 60 % of the initial capacity. Research shows that different battery chemistries show different aging profiles. For NMC cathodes, increasing the nickel content generally leads to a reduced lifespan so that NMC111 shows higher cycle life than NMC811 [38]. On average, LFP achieves the highest cycle numbers, followed by NMC, NCA and LMO, see Table 1. To provide some perspective on the cycle life in the context of BESSs, a system sized according to the sizing criteria of 1 kWh_c/1 kW_p/1 MWh (ratio of BESSs capacity to photovoltaic power to annual electricity consumption of the household) absolves around 300 full cycle equivalents per year [43,44]. Thus, a cycle life of 5000 places the LFP BESSs at 17 years of operation before EoL, a time reasonable from a calendar aging perspective.

2.2. Literature search

The search strategy applied in the present review through Scopus identified >800 publications.[3] Of those publications, more than half were concerned with electric vehicle applications (see for example, [52–60]). Another substantial part looked at lead-acid or next-generation battery technologies (for example, lithium-air [61–63], sodium-ion [64–66] or zinc-air [67]) and the manufacturing of lithium-ion cells [68]. Around 50 studies addressed energy storage integration into renewable energy systems but did not address BESSs in detail. Another 50 studies addressed recycling of LIBs. Some LCAs of residential BESSs were found but had to be excluded either because data was provided only on an endpoint-indicator level or with too little detail to separate impacts from the battery and photovoltaic system from impacts associated with the electricity grid [69–71]. Thus, only 13 studies provided enough level of detail to be included for further analysis of life cycle assessments for residential BESSs.

3. Global warming potential of lifetime electricity stored (kWh_d)

3.1. Reported impacts for BESS, PV-system or both

Nine studies looked at residential BESSs with <25 kWh of battery capacity, see Table 2. In most studies the BESS was employed to boost self-consumption of a residential photovoltaic system [21,48,72–78]. Three studies looked at grid-scale BESSs, also employed to boost self-consumption [79–81]. Another study evaluated both residential and grid BESSs [82].

Further, publications in Table 2 can be divided into three groups. The first group (black in Table 2) consists of studies which provided GHG emissions for the PV-system (kWh_{pv}) and the BESS (kWh_d) separately [21,48,73,74,80]. The second group (blue in in Table 2) consists of those which did not disclose the contributions of the PV-system and the BESS separately but provided only aggregated results for 1 kWh electricity generated by the photovoltaic system and stored in the BESS (kWh_{d+pv}) [75,76,82]. The third group (green in in Table 2) is formed by publications which provided data only for the BESS (kWh_d). This was either because charging of the BESS was outside the scope of the study [77,79] or because the source of electricity used for charging the system was something other than photovoltaic electricity [72,78,81].

3.2. Functional unit, life cycle inventory data, and cycle life

In life cycle assessments, selecting an appropriate functional unit is a central element of the goal and scope definition [26]. As can be seen in Table 2, most studies selected some form of “1 kWh electricity supplied by the battery storage systems throughout its entire life” as the functional unit of the LCA. Still, further alignment of the functional units would benefit comparability between studies. For example, Stougie et al. [77] chose “storage capacity of 10 kWh for 20 years lifetime” as the functional unit. While this functional unit might be appropriate for addressing their research question, it makes comparison with other studies difficult.[4]

In the context of battery electric vehicles, most LCAs have agreed on “1 km driven by the electric vehicle” as standard functional unit, leaving little room for misinterpretation, especially if the study is sufficiently transparent on system boundaries [33,40,41]. In comparison, functional

[3] Keywords used: “batter*” AND (“life cycle assessment” OR “LCA”); Field of search: Title, abstract, keywords; Focus: January 2010–September 2021; Number of studies: 845.

[4] It is possible to get results for 1 kWh_d with some calculations.

Table 1
Properties of lithium-ion batteries with graphite anode and different cathode materials. Sources: [21,38,50,51].

Cathode material	Stoichiometry	Cycle life	Specific capacity [mAh/g]	Voltage potential [V vs. Li^+/Li]	Manufacturer of BESS using this cathode material
LFP	$LiFePO_4$	5000 [21] 2500–9000 [50] 2000 [12]	165	3.45	BYD, SonnenBatterie
LMO	$LiMn_2O_4$	1500 [21]	110	4.10	
NMC_{111}	$LiNi_{0.33}Mn_{0.33}Co_{0.33}O_2$	4000 [21] 1000–2000 [12]	160	3.70	Tesla, Samsung SDI, LG Chem
NMC_{811}	$LiNi_{0.8}Mn_{0.1}Co_{0.1}O_2$	200–2500 [50]	190	3.70	Tesla, Samsung SDI, LG Chem
NCA	$LiNi_{0.8}Co_{0.15}Al_{0.05}O_2$	3000 [21] 250–1500 [50] 1000–1500 [12]	188	3.70	

units chosen for BESSs seem more ambiguous and have, seemingly, not yet agreed on one standardized functional unit. As a solution, we recommend using 1 kWh_d of lifetime electricity stored, as provided in Thomas et al. [48] and Baumann et al. [21] as standard functional unit in LCAs of BESSs. This will allow for greater transparency and comparability across LCA studies. A similar standard has established for production of LIBs. Here, 1 kWh_c of battery capacity and 1 kg of battery weight are frequently provided even if the main functional unit of the study is something different [58,86].

Gathering life cycle inventory (LCI) data is a key element of the second phase of a life cycle assessment [25,26]. As demonstrated in Table 2, most of the identified life cycle assessments rely on life cycle inventory data of earlier LCA studies which provided some primary data about the production of LIBs. In this context, Oliveira et al. [74] and Carvalho et al. [78] stand out as they provided a significant amount of own primary data. Primary data came from a cooperation with a Japanese LMO battery manufacturer in Oliveira et al. [74]. Carvalho et al. [78] cooperated with an Italian battery producer. The remaining studies relied on primary data obtained by Bauer [84], Notter et al. [60] (sometimes through Ecoinvent [45]), Zackrisson et al. [59], Majeau-Bettez et al. [83], Ellingsen et al. [58], and Li et al. [85]. Thomas et al. [48] used standardized LCI data from Peters & Weil [86], which in turn is based on the aforementioned studies except Li et al. [85].

Even though five studies included different battery chemistries within their analysis, only two studies considered the impact of battery chemistry on the cycle life of the BESSs. Lifespan estimates in Table 1 suggest the following cycle life ranking: LFP > NMC > NCA > LMO, which is in line with Baumann et al. [21]. However, Thomas et al. [48] estimated cycle life in a different order: NCA > NMC > LFP > LMO. This ranking contrasts with Table 1. Thomas et al. [48] stated that technical data about lifespan of the LIBs was taken from warranties and data sheets of manufacturers. But, having both NMC442 and NCA performing more full cycle equivalents than the LFP system seems inaccurate. Thus, a discussion about the appropriateness of warranty data for cycle life inference might be necessary.

3.3. Reported GHG emissions for BESSs between 9 and 135 gCO_2eq/kWh_d

Visualization in Fig. 2 is based on a cradle-to-gate perspective. This means that potential recycling impacts are outside the scope and only the production and use phase of BESSs are included. Large-scale recycling of LIBs is just starting and LCA data for recycling is limited. Research into recycling of LIBs states that advanced hydrometallurgical treatment is generally the most environmentally friendly recycling option [87–89]. Further, results in Fig. 2 for GHG emissions per kWh of electricity stored in the BESS throughout its lifetime (kWh_d) do not include CO_2eq emissions associated with the generation of electricity stored in the BESS. This makes results more comparable since they are not impacted by source of the electricity generation. Some studies (blue in Table 2) did not allow to distinguish between impacts associated with the BESS and those associated with the generation of photovoltaic electricity and had to be omitted at this point.

Average GHG emissions are 55 gCO_2eq/kWh_d with a range of 9–135 gCO_2eq/kWh_d. Low GHG emissions between 25 and 35 gCO_2eq/kWh_d were found in Vandepaer et al. [81] and Stougie et al. [77]. Interestingly, these two studies have reported almost the same GHG emissions per kWh_d even though Vandepaer et al.'s [81] LFP BESS endures almost twice as many cycles (5000 cycles) as Stougie et al.'s [77] NMC BESS (3000 cycles). Lowest GHG emissions of <25 gCO_2eq/kWh_d were reported in Hiremath et al. [73], Mostert et al. [79] and Carvalho et al. [78]. Remarkably, Hiremath et al. [73], whose system was modelled on an LMO-NMC blend cathode assumed their BESS to last 20 years or 10,250 full cycles. Those cycle numbers are high compared to all remaining studies which typically assume up to 5000 cycles till EoL. Mostert et al. [79] assumed 3650 cycles for a LMO BESS built from remanufactured electric vehicle batteries and 4300 cycles for a BESS with brand-new LIBs, a lifespan comparable to the other studies. Their results are the lowest reported in any study, with 9 gCO_2eq/kWh_d for second-use cells and 11 gCO_2eq/kWh_d for new cells. LCI data for the production of the BESS is based largely on Notter et al. [60] which, as will be addressed in Section 4, provides fairly low GHG emissions associated with the production of 1 kWh_c LMO battery capacity. Carvalho et al. [78] had access to primary data for the production of LIBs, assumed 5000 cycles for the LFP and both NMC (532/622) BESSs and reported life cycle emissions below 25 gCO_2eq/kWh_d as well.

High GHG emissions of >90 gCO_2eq/kWh_d are found in Oliveira et al. [74] (LFP home storage), Ahmadi et al. [72] (LFP home storage), Baumann et al. [21] (LMO home storage), and Raugei et al. [80] (both LFP and NMC grid storage). Oliveira et al. [74] have taken an unusual approach to the lifespan of the battery storage as it was assumed that the system would remain in operation until battery capacity reached 30 %. Ahmadi et al. [72] considered a scenario similar to Mostert et al. [79] in which a repurposed LIB from a BEV was subsequently utilized for up to 10 years or 3650 cycles in a stationary storage system.

3.4. No systematic decline in reported emissions during last five years

Reported GHG emissions appear not to have systematically fallen between 2015 and 2021. This is in contrast to the expectation that increases in battery production efficiency and increased lifespan of LIBs should have led to lower GHG emissions associated with storing 1 kWh_d of electricity, all other things being equal. One explanation for the lack of a systematic decrease is that LCI data for the BESSs are largely based

Table 2
Life cycle assessments of battery energy storage systems. Colour: Green, study provided environmental impacts for kWh_d but not kWh_{pv}; Black, study provided environmental impacts for kWh_d and kWh_{pv} separately; Blue, study provided environmental impacts for kWh_{d+pv}.

No	Name, year	Chemistry	Application	Data sources	LCIA model	Electricity source for charging	Cycles till EoL	Functional unit
1	Ahmadi et al. [72], 2015	LFP	EV (first life) + Home storage (second life) (10kWh)	[83]	ReCiPe	Ontario grid	8 years first life (2920 cycles) - 10 years second life (3650 cycles)	1kWh of electricity delivered by the battery pack over its full life
2	Hiremath, Derendorf & Vogt [73], 2015	LMO-NCA-NMC average	Home storage	[60, 83]	ReCiPe	PV-electricity	10250 cycles - 20 years	1MWh of electricity delivered
3	Oliveira et al. [74], 2015	LFP, LMO	Home storage (1kWh)	Primary data (LMO), [83] (LFP)	ReCiPe	PV-electricity	4000 cycles till 30% EoL	Delivery of 1kWh of energy previously storage in battery
4	Baumann, Peters, Weil & Grunwald [21], 2017	LMO, LFP, NMC, NCA	Home storage (8kWh)	[59, 83] (LFP), [60] (LMO), [84] (NCA), [58, 83] (NMC)	GWP IPCC	PV-electricity	LFP: 5000 LMO: 1500 NCA: 3000 NMC: 4000	1kWh stored throughout battery life
5	Jones, Peshev, Gilbert, and Mander [75], 2017	LFP	Home storage (10kWh)	[42, 83]	no information	PV-electricity + UK grid	5000	1kWh electricity consumption by the building
6	Vandepaer, Cloutier, and Amor [81], 2017	LFP	Grid storage (75kWh & 6MWh)	[83]	IMPACT 2002+	Wind power	5000	1MWh of electricity delivery by the battery
7	Mostert, Ostrander, Bringezu, and Kneiske [79], 2018	LMO	EV (first life) + grid storage (second life), grid storage with new LIB (2MWh)	Ecoinvent, [60]	GWP IPCC	No charging	8 years first life (2920 cycles) - 10 years second life (3650cycles), 11,8 years (4300 cycles) new LIB	Amount of usable electricity which can be provided by EES based on a unified energy fed-in
8	Üçtuğ and Azapagic [76], 2018	LMO	Home storage (2.1kWh)	Ecoinvent, [60]	CML	PV-electricity	10 years	1kWh lifetime electricity supplied by PV+BESS system
9	Stougie et al. [77], 2019	NMC111	Home storage (10kWh)	[85]	ReCiPe	No charging	3000	Storage capacity of 10kWh for 20 years lifetime
10	Aberilla, Gallego-Schmid, Stamford, and Azapagic [82], 2020	LMO	Home storage (10kWh), grid storage (448kWh)	[60]	ReCiPe	PV-electricity	15 years	Generation and supply of 1kWh electricity
11	Raugei, Leccisi, and Fthenakis [80], 2020	LMO, NMC111, LFP	Grid storage (240MWh)	[60] (LMO), [58] (NMC), [83] (LFP)	CML	PV-electricity	2550	1kWh electricity delivered by PV + BESS system
12	Thomas, Schmidt, Gambhir, Few, and Staffell [48], 2020	LFP, LMO, NCA, NMC442	Home storage (8.1kWh)	[86] (LFP, NCA, NMC, LMO)	ReCiPe	PV-electricity	LFP: 7016 LMO: 5840 NCA: 9281 NMC: 7043 All till 60% EoL	1kWh lifetime energy delivered
13	Carvalho, Temporelli and Girardi [78], 2021	LFP, NMC532, NMC622	Home storage (26.6kWh)	Primary data, [58, 83]	ICLD 2011	Grid or PV + Wind	5000	1kWh of energy released

on the same few prior studies (see also Table 2), which do not represent current state-of-the-art LIB manufacturing [90,91] and have not been updated sufficiently. Thomas et al. [48] have provided a good example of how a better understanding of the LCI of LIBs should be incorporated into the assessment of residential BESSs by using the updated and standardized LCI data from Peters & Weil [86] instead of relying on the original LCI data. Conclusions from Peters & Weil [86] suggest that studies which use Ellingsen et al. [58], Majeau-Bettez et al. [83], Bauer [84], Zackrisson et al. [59] for NCA, NMC or LFP batteries overestimate GHG emissions while LMO data based on Notter et al. [60] tends to underestimate GHG emissions. This explains in part the very low GHG emissions obtained by Moster et al. [79] who used standard Ecoinvent 3 data for the LMO battery storage, which assumed the low energy demand for cell manufacturing stated by Notter et al. [60].

Despite the benefits of using standardized and updated LCI, Aberilla et al. [82], Raugei et al. [80] and in part also Carvalho et al. [78], whose work took place after the publication of Peters & Weil [86] continued to rely on the original, not-standardized LCI data. Therefore, some of the data pitfalls addressed in Peters et al. [33] and Peters & Weil [86] were not avoided. The lack of a clear trend of GHG emissions associated with the storage of 1 kWh_d electricity in BESSs is problematic if those results are used for comparison with other means of energy storage technologies, such as pumped hydro storage, compressed air storages or power-to-X-to-power technologies with more up to date data.

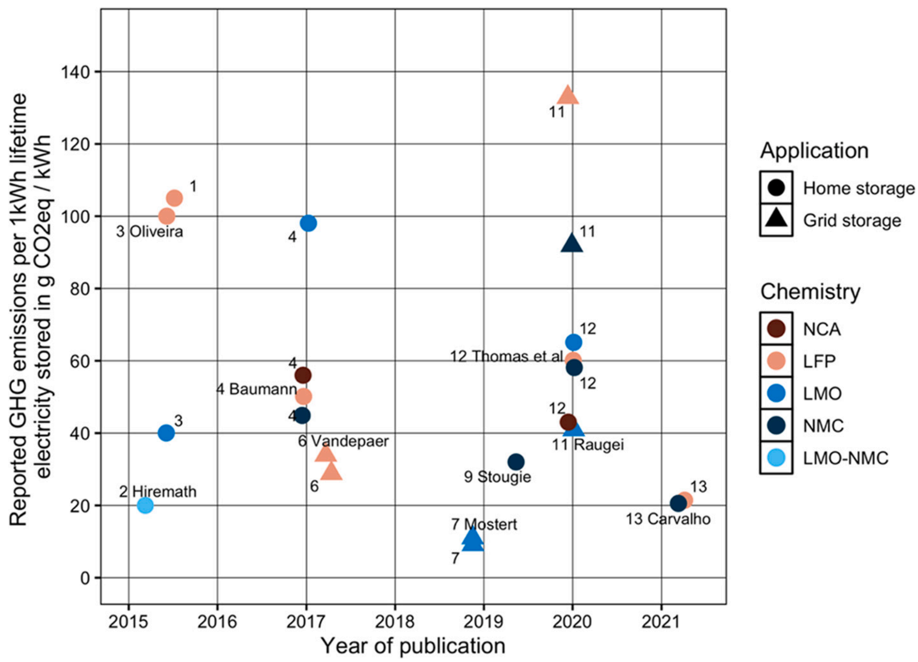

Fig. 2. Reported GHG emissions for 1 kWh lifetime electricity stored in the BESS (kWh_d); impacts are associated with production and operation of the BESSs but do not include impacts of the electricity used for charging the battery during operation.

3.5. Impact of cathode material

In Baumann et al. [21], a BESS with NMC had the lowest GHG emissions for 1 kWh_d, closely followed by LFP and NCA. The LMO BESS had substantially higher GHG emissions (more 40 gCO_2eq/kWh_d difference) than NCA, NMC and LFP BESSs. In Thomas et al. [48], reported differences between battery chemistries were smaller. The BESSs with NMC, LFP or NCA all came with around 60 gCO_2eq/kWh_d. The NCA BESS, interestingly, was associated with low GHG emissions of 40 gCO_2eq/kWh_d. GHG emissions of a LFP BESS in Raugei et al. [80] are more than three time higher than for one using LMO (135 g CO_2eq/kWh_d vs. 40 g CO_2eq/kWh_d). The BESS with NMC111 showed GHG emissions which were 30 % higher than for LFP. GHG emissions for the LFP BESS used in Oliveira et al. [74] were more than two times higher than for the LMO system (100 gCO_2eq/kWh_d vs. 40 gCO_2eq/kWh_d). Carvalho et al.'s [78] is the study with the lowest difference between battery chemistries. Reported differences between NMC622, NMC532 and LFP hardly exist. NMC532 was associated with slightly lower GHG emissions than NMC622, which is not surprising given the increase of nickel content at the expense of the manganese content leads to higher environmental burdens of the battery raw material [66]. Commercial manufacturers of BESSs have advertised the superior environmental benefits of LFP BESSs compared to BESSs with NMC [92,93]. Surprisingly, our findings show that at least from a global warming perspective the reviewed studied suggest quite the opposite.

3.6. Relative contribution of PV system and BESS

Whether the PV-system or the BESS is responsible for the larger share of the GHG emissions per 1 kWh_{pv+d} of the combined system is not immediately obvious. As can be seen in Fig. 3, GHG emissions associated with the generation and storage of 1 kWh_{pv+d} electricity range from 43 gCO_2eq/kWh_{pv+d} to 195 gCO_2eq/kWh_{pv+d}. The 43 gCO_2eq/kWh_{pv+d} from Üçtuğ and Azapagic [76] are by far the lowest emissions reported. In fact, Üçtuğ and Azapagic [76] is the only study in which the combined GHG emissions of the BESS and the PV-system (kWh_{pv+d}) are lower than the average GHG lifecycle emissions of photovoltaic electricity (kWh_{pv}).

In Raugei et al. [80] the PV-system accounted for <30 % of GHG emissions while the LFP and NMC BESS were responsible for >70 %. Interestingly, the same photovoltaic system was responsible for 47 % of total GHG emissions in combination with the LMO BESS in Raugei et al. [80]. Hiremath et al. [73] report absolute GHG emissions of the photovoltaic system (kWh_{pv}) similar to the 100 gCO_2eq/kWh_{pv} in Baumann et al. [21]. Still, in combination with very low impacts of the battery system of just 20 gCO_2eq/kWh_d this led to a 83 % contribution of the photovoltaic system to the total emissions (kWh_{pv+d}). Generally, studies seem to agree that the contribution of the photovoltaic system is between 40 and 70 % and the contribution of the BESS between 30 and 60 %.

3.7. Direct comparison with grid is problematic but can put results into perspective

To put the combined GHG emissions of the BESS and the photovoltaic system into some perspective, average life cycle GHG emissions for other means of electricity generation as well as average country specific GHG emissions (kWh_{grid}) for 2017 are taken from Ecoinvent 3.7.[5] Interestingly, all systems provide substantially lower GHG emissions than natural gas fired power plants (423 gCO_2eq/kWh) or even coal.

Also, Fig. 3 points to some countries which might benefit most from employment of BESSs charged with photovoltaic electricity. In Sweden and Norway for instance, widespread employment of BESSs in combination with photovoltaic seems to have less potential of reducing the GHG intensity of the grid any further. On the contrary, India and China, but also Russia, Germany and the United Kingdom show potential. While a comparison should ideally be made on a numerical basis for each use-case, results from Fig. 3 encourage future LCAs studies for locations in Russia, India, and China, which none of the reviewed studies has yet done.

[5] Consequentially, we compare electricity streams II and III from Fig. 1, taking the perspective of the household.

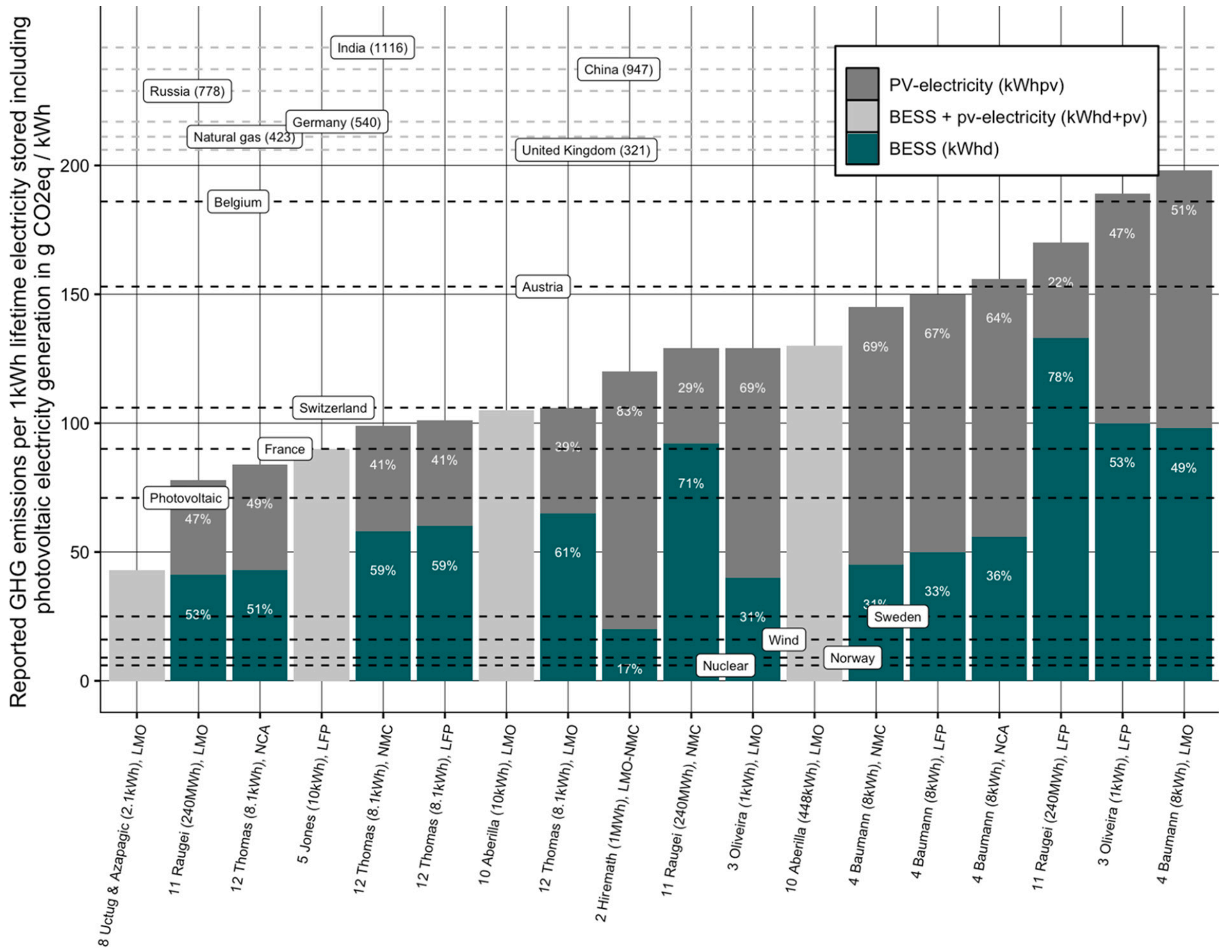

Fig. 3. Reported GHG emissions for 1 kWh lifetime electricity delivered by the BESS provided by photovoltaic system (kWh_{d+pv}). See Table 2 for references.

4. Primary data for global warming potential of 1 kWh battery capacity (kWh_c)

4.1. Understanding and expanding sources of primary data

The previous section has shown that only two out of 13 LCAs provided a substantial part of own primary LCI data for the BESS. As has been addressed before, this is problematic. Conceptually BESSs consist of lithium-ion battery packs and some electronic equipment for charging and discharging. In some photovoltaic + BESS combinations, the battery charging is done by the photovoltaic-hybrid inverter so that little additional equipment is necessary [93]. Therefore, we chose to take a closer look into sources of primary data for the production of LIBs. As a reasonable simplification, the environmental impacts associated with 1 kWh_d lifetime electricity stored in a BESS can be obtained by dividing the emissions for 1 kWh_c of battery pack production by the number of full cycle equivalents before the battery reaches end-of-life (total lifetime energy delivered). Thus, the focus here lies on the GHG emissions associated with the production of the lithium-ion battery based on 1 kWh_c battery capacity.

$$GWP(kWh_d) = \frac{GWP(kWh_c)}{Full\ cycle\ equivalents\ till\ EoL}$$

Table 3 shows life cycle assessments of LIB applications which were found during the literature search and provided a substantial amount of own primary data. Peters & Weil [86] have not provided own primary data but, as has been addressed in the previous section, put the primary data from earlier studies [58–60,83,84] on a standardized basis. Most studies had battery electric vehicles or plug-in hybrid electric vehicles as application of interest. In addition, most studies provided results for 1 kWh_c on a battery pack level. Only Chordia et al. [94] did not provide data for pack level but stopped their LCA study on cell level. However, energy requirements for cell manufacturing were obtained from the Northvolt Gigafactory in Sweden, which is a rare source of access to cell manufacturing on a GWh scale. In addition, the Greenhouse gases, Regulated Emissions, and Energy use in Technologies (GREET) model [95–98] from the Argonne National Laboratory (ANL) is not a typical LCA but researchers at the ANL have had some of the best access to primary data for the production of lithium-ion batteries and materials [95,97–102]. Generally, Table 3 reflects the situation that more interest has so far been paid to LCAs of electric vehicles applications. Only two of the studies with substantial LCI primary data were aimed at BESSs [74,78]. Most studies have looked at LFP and NMC batteries, with fewer work conducted for NCA and LMO battery systems.

4.2. Broadest primary data basis for LFP and NMC

GHG emissions associated with the production of 1 kWh_c LIB capacity are between 30 $kgCO_2eq/kWh_c$ and 270 $kgCO_2eq/kWh_c$, see Fig. 4. The two outliers with higher emissions have been subject to discussion because of a very high energy demand for cell manufacturing. Low results of around 50 $kgCO_2eq/kWh_c$ have consistently been reported for LMO batteries. The most recent publications (2020 onwards), have provided new primary data for NMC and LFP.

In direct comparison, USEAP [104] reported slightly higher GHG emissions for a NMC battery than for a LFP battery. Majeau-Bettez et al. [83] reported higher emissions for LFP than NMC. Standardization of Peters & Weil [86] had some effect on the overall results, in that

Table 3
Life cycle assessments of lithium-ion battery applications with substantial own LCI primary data.

No	Name, year	Chemistry	Application	Data sources	Production location	LCIA model
14	Bauer [84], 2010	NCA	EV	Primary data		CML
15	Notter et al. [60], 2010	LMO	EV	Primary data	Europe	GWP IPCC
16	Zackrisson, Avellán, and Orlenius [59], 2010	LFP	EV	Primary data (SAFT)	Sweden	Environmental Footprint Declaration (EPD)
17	Majeau-Bettez, Hawkins, and StrØmman [83], 2011	LFP, NMC442	EV	Primary data (SAFT)	Europe	ReCiPe
18	Dunn, Gaines, Sullivan, and Wang [103], 2012 and Dunn, Gaines, Barnes, Sullivan, and Wang [96], 2012	LMO	EV	Primary data	US	GREET
19	United States Environmental Protection Agency [104], 2013	LMO, LFP, NMC442	EV	Primary data	US	TRACI
20	Ellingsen et al. [58], 2014	NMC111	EV	Primary data		ReCiPe
21	Li, Gao, Li, and Yuan [85], 2014	NMC111	EV	Primary data	US	GaBi
3	Oliveira et al. [74], 2015	LFP, LMO	Home storage	Primary data (LMO), [83] (LFP)	Belgium	ReCiPe
22	Kim et al. [105], 2016	LMO-NMC	EV	Primary data (LG Chem), GREET2014	South Korea	
23	Deng, Li, Li, Gao, and Yuan [106], 2017	NMC	EV	Primary data, [58]	US	ReCiPe
24	Peters and Weil [86], 2018	LMO, LFP, NMC, NCA	Not specified	[59,83] (LFP), [60] (LMO), [58,83] (NMC), [84] (NCA)	Europe	ILCD
25	Wang et al. [107], 2018	LMO, LFP	Not specified	Primary data	China	ReCiPe
26	Cusenza, Bobba, Ardente, Cellura, and Di Persio [108], 2019	LMO-NMC	EV	Primary data	Japan	Product Environmental Footprint (PEF)
27	Dai, Kelly, Gaines, and Wang [109], 2019	NMC111	EV	Primary data, GREET2018	United States	GWP IPCC
28	Philippot, Alvarez, Ayerbe, Mierlo, and Messagie [57], 2019	NCA	EV	Primary data, [58], GREET2016	South Korea	GWP IPCC
29	Sun, Luo, Zhang, Meng, and Yang [52], 2020	NMC622	EV	Primary data, GREET2018	China	ReCiPe/CML
30	Winjobi, Dai, and Kelly [95], 2020 (GREET2020 model)	NMC811, NMC622, NMC111, LFP, NCA, LMO	EV	Primary data, previous GREET models	USA	GWP IPCC
13	Carvalho, Temporelli and Girardi [78], 2021	LFP, NMC532, NMC622	Home storage	Primary data, [58,83]	Italy	ILCD 2011
31	Chordia, Nordelöf and Ellingsen [94], 2021	NMC811 (cell level only)	Not specified	Primary data (Northvolt)	Sweden, South Korea	ReCiPe

differences between battery chemistries were smaller. Still, Peters & Weil [86] took the side of Majeau-Bettez et al. [83] in that NMC is associated with lower GHG emissions than LFP for 1 kWh_c battery pack capacity after standardization. Adding further to the discussion on LFP vs. NMC, the GREET2020 model shows substantially lower GHG emissions for LFP per 1 kWh_c than NCM. Carvalho et al. [78] reported hardly any difference between GHG emissions for 1 kWh_c of LFP and NMC, which explains the very small differences in Fig. 2.

4.3. Primary data shows lower decline for GHG emissions than costs during last decade

When looking at the overall picture, reported GHG emissions did not decline as fast as LIB costs during the last decade [110,111]. Also, a great number of studies have made predictions about the cost of LIBs up until 2050 [112], but none of the studies provided a prediction about future GHG emissions associated with the production of 1 kWh_c LIB capacity. Cost forecasts make use of the impact of different material and process parameters on overall battery costs [112,113]. Based on a similar approach future studies might apply detailed forecast technologies to address the lowest achievable GHG emissions of 1 kWh_c in the near or mid-term. Providing a target of, for example, 10 $kgCO_2eq/kWh_c$ should guide the development and improvement of future LIBs and help to achieve similar and sustained decreases in GHG emissions as has already been seen in LIB costs. If reductions in GHG emissions per kWh_c are not dealt with adequately, LIB manufacturers might risk being caught off guard once a substantial tax on CO_2 emissions is put in place (a carbon tax of \$100/t_{CO2} could imply additional costs of \$10/kWh_c for a LIB with 100 $kgCO_2eq/kWh_c$). As a result, reducing the associated GHG emissions of lithium-ion batteries should be an imminent priority.

4.4. Cathode material and energy for cell manufacturing responsible for high GWP

Reducing GHG emissions of LIB production benefits from understanding the contribution of different processes and materials during LIB production. Fig. 5 demonstrates that cathode, cell manufacturing and battery housing + BMS account for the majority of the GHG emissions associated with the production of 1 kWh_c LIB capacity. Energy requirement for cell manufacturing in Peters & Weil [86] is based on average values. GREET [95] meanwhile leverages access to battery manufactures to get insight into the energy requirement during cell manufacturing [95–98]. Still, even GREET [95] utilizes the assumption that energy requirements during cell manufacturing are not impacted by the battery chemistry, a proposition which can be contested since water-based electrode production of LFP-based cathodes has already become a viable technology [78]. Nevertheless, the overall tendency for energy requirements during cell manufacturing is that variance between different studies, for example GREET [95] vs. Peters & Weil [86] vs. Carvalho et al. [78] is significantly larger than any variance within the same study as a function of the battery chemistry.

Energy requirements for cell manufacturing usually consist of electricity and heat demand, with heat primarily being used for drying and operation of the dry-room and electricity for formation and operation of

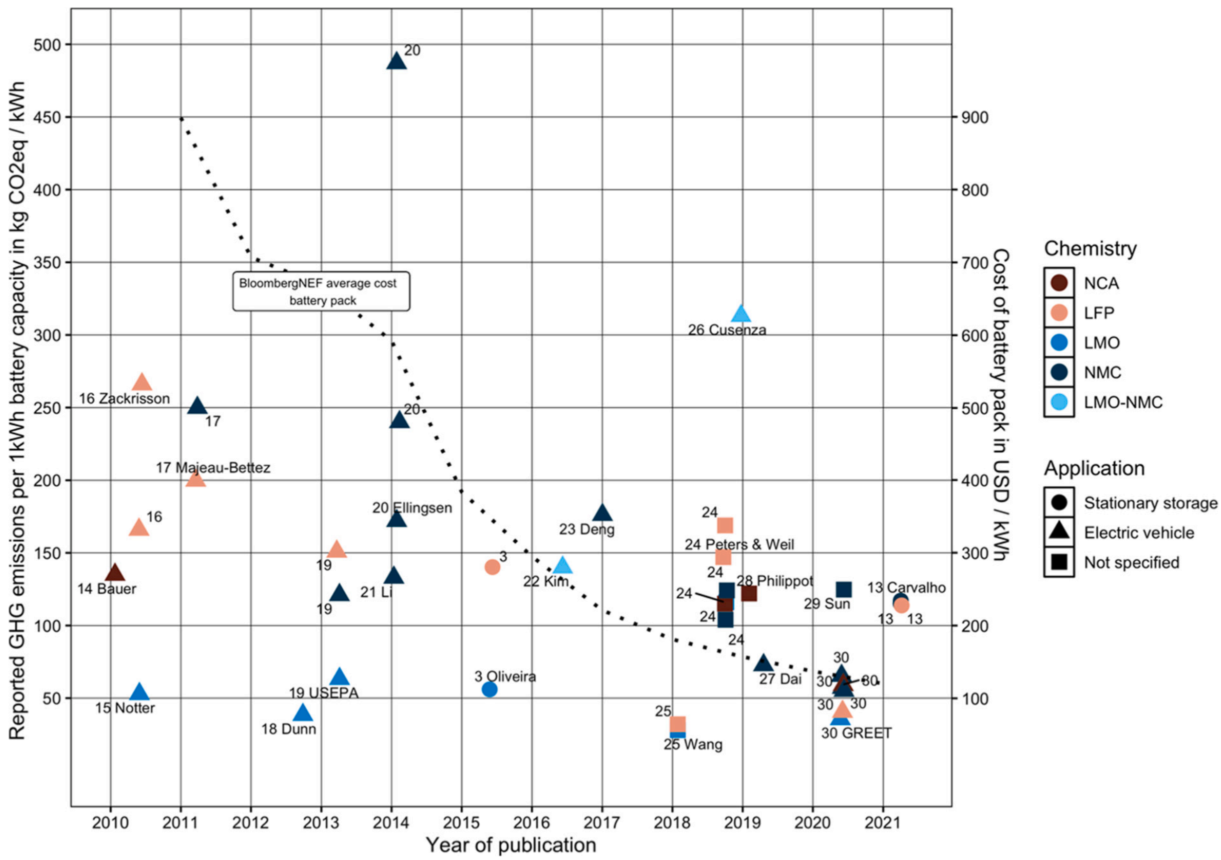

Fig. 4. Reported GHG emissions associated with the production of 1 kWh_c battery capacity from studies with substantial amount of new primary LCI data.

roll-to-roll equipment. The GHG emissions associated with the cell manufacturing can be estimated by multiplying the electricity demand with the GHG emissions of the grid and multiplying heat demand with the GHG emission per MJ heat (usually provided by natural gas). Therefore, studies which estimate a high ratio of electricity demand to heat demand should see a higher impact of the geographic location than cell manufacturing estimates which are largely based on heat provided by natural gas. This is because GHG emissions per kWh electricity drawn from the grid differ substantially across countries whereas burning natural gas for the generation of heat results in similar levels of GHG emissions regardless of the battery manufacturing plant location. Carvalho et al. [78] stand out with no explicit heat demand but only electricity requirements. GREET2020 reports fairly low electricity demand for cell manufacturing of 7.2 kWh and heat demand of 142 MJ [95–98]. The electricity demand in Peters & Weil [86] varies between 64.7 kWh and 108.5 kWh. The variance in Peters & Weil [86] is based on the assumption that cell manufacturing energy requirement are the same for 1 kg of battery cells. Thus, the higher energy requirements for LPF compared to NMC are due to the lower energy density of LFP (more kg battery cells need to be manufactured to get 1 kWh of battery capacity). Meanwhile GREET assumes that energy requirements for cell manufacturing are constant for 1 kWh cell capacity. Chordia et al. [94] provide data only for NMC811 and some calculations are necessary to separate the energy requirement of the cell manufacturing from cathode material production. Based on the most recent publication of GREET [95], Chordia et al. [94] and Carvalho et al. [78] it seems appropriate to assume a total energy requirement for cell manufacturing of around 50 kWh (3.6 MJ = 1 kWh) per 1 kWh of battery capacity which can be divided between heat and electricity.

Not surprisingly, the different levels of energy requirements during cell manufacturing translate to a substantial variety in the contribution of cell manufacturing to the overall GHG emissions per kWh_c battery pack. Peters & Weil [86] report the overall highest impact of the cell manufacturing phase, resulting from an electricity demand substantially higher than reported in GREET. Impacts of the cell manufacturing in GREET are lower primarily due to the lower electricity demand, which is about 90 % lower than in Peters & Weil [86] while heat demand is on a comparable level. Data from Chordia et al. [94] seems to support the tendency that >50 kWh electricity is required for manufacturing of 1 kWh cell capacity. However, in combination with the fairly low-emission Swedish electricity supply overall GHG contribution of cell manufacturing is still fairly low.

All studies seem to agree that the absolute contribution of the anode, which is primarily graphite for anode active material and copper for the current collector, is around 5 $kgCO_2eq/kWh_c$. For the cathode, which consists primarily of an aluminum current collector and cathode active material, results for the contribution of layered-oxide cathodes (NMC, NCA) seem in better alignment than for LFP and LMO. The absolute contribution of nickel based cathodes is between 30 and 45 $kgCO_2eq/kWh_c$. Peters & Weil [86] report the NMC442 cathode with higher GHG emissions than the NMC111 one, which is surprising given that GREET sees decreases in the associated GHG emissions for increased nickel content. But it has to be considered that NMC111 and NMC442 cathodes in Peters & Weil [86] come from different studies, so that even after standardization some uncertainty about the comparability between both

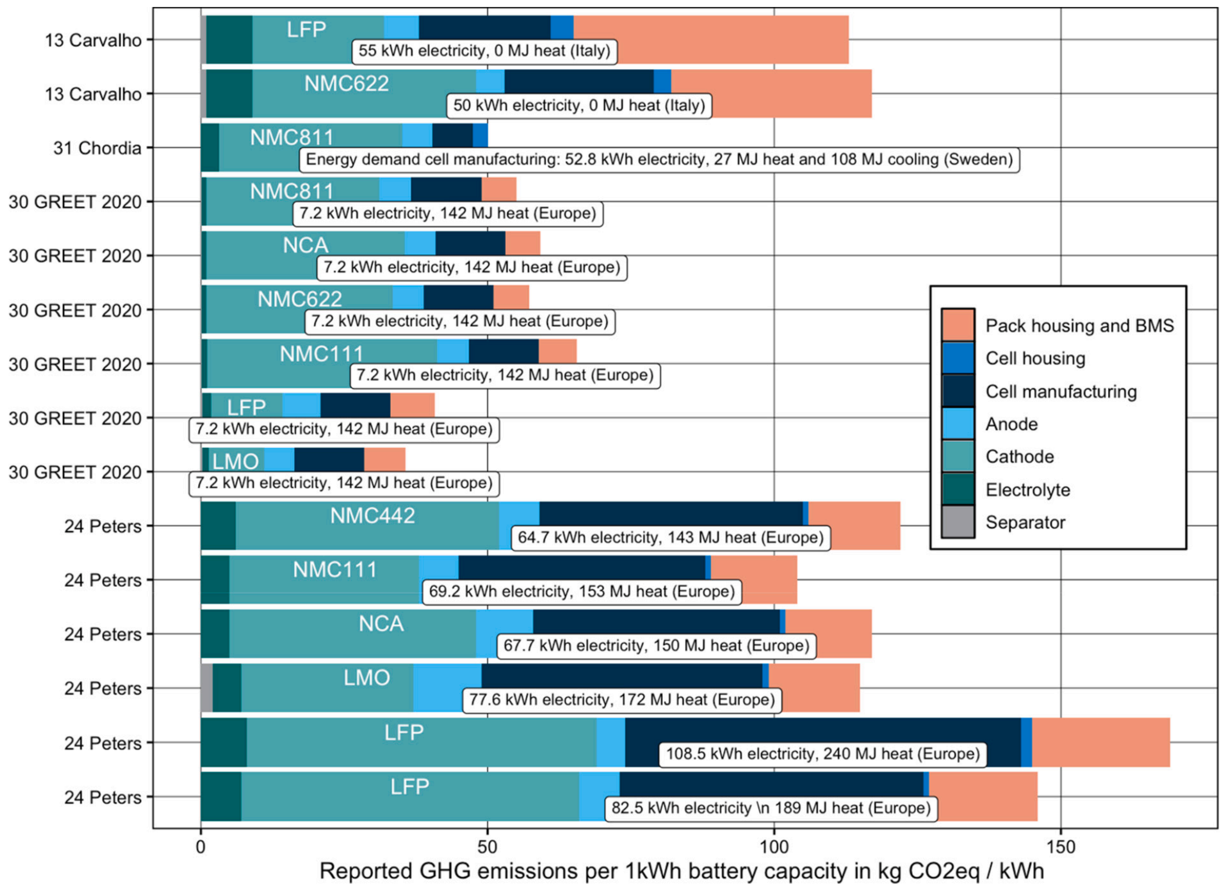

Fig. 5. Contribution materials and processes to combined GHG emissions for 1 kWh_c battery pack capacity. References in Table 3.

materials remains.[6] Looking at LMO and LFP cathodes, a great amount of uncertainty exists in Fig. 5. The LFP cathode in GREET contributes around 12 $kgCO_2eq/kWh_c$. On the other hand, the LFP cathode in Peters & Weil [86] is responsible for 60 $kgCO_2eq/kWh_c$. For LMO, the contribution of the cathode is <10 $kgCO_2eq/kWh_c$ in GREET and 30 $kgCO_2eq/kWh_c$ in Peters & Weil [86]. Also, we see that future studies should further investigate the impacts associated with the cathode active material since it contributes substantially to overall GHG emissions and also shows large uncertainty for LFP cathodes.

4.5. *Utilize primary data from GREET model for BESSs*

Peters & Weil [86] and the GREET2020 model [95] to-date provided the most comprehensive GHG emission data for the cradle-to-gate emissions associated with the production for a large selection of the current lithium-ion battery technologies. Therefore, Fig. 6 combines these two data sources with proper estimates about the cycle life, based on Table 1, of each cathode material to get an idea of the GHG emissions of 1 kWh_d lifetime electricity stored. Fig. 6 simplifies the BESS in that it assumes that the BESS consists only of battery packs. As addressed in Thomas et al. [48] this is usually not the case, as additional housing for the BESS as well as an inverter for charging the battery packs are also part of the system.

With the GREET data [95–98] for LFP, GHG emissions as low as 8 gCO_2eq/kWh_d are achieved, compared to 12–14 gCO_2eq/kWh_d for NMC. LCI data from Peters & Weil [86] leads to emissions of 30–33 gCO_2eq/kWh_d for LFP and 28–32 gCO_2eq/kWh_d for NMC. This underlines that the source of LCI data associated with the production of LIBs has a large impact on the GHG emissions and potential conclusion. Since emissions for LFP are the highest in Peters & Weil [86] it comes to no surprise that Thomas et al. [48], in combination with the cycle life assumption of NCA > NCM > LFP reports the LFP BESSs performing worse than the NMC based systems. This becomes problematic if results from prior LCA studies are read as an endorsement of NCM deployment in BESSs from a GHG perspective. Employing the more recent data from the GREET model with reasonable cycle life estimates strongly suggests that LFP based BESSs a preferable from a GHG perspective, challenging the conclusion of prior LCA studies.

5. Conclusion

The present work has provided insight into LCAs of residential BESSs. Employment of BESSs is increasing fast so that a sound understanding of environmental impacts is important. We contributed to a better understanding of GHG emissions associated with residential BESSs in several ways. Our analysis revealed GHG emissions for 1 kWh_d lifetime electricity stored between 9 and 135 gCO_2/kWh_d. During the last five years, reviewed studies reported no systematic decline in GHG emissions for 1 kWh_d lifetime electricity stored, in part explained by reliance on old life cycle inventory data for lithium-ion batteries. This presents a problem if results for residential BESSs are compared to

[6] From a theoretical perspective, increasing the nickel content at the expense of cobalt content should result in lower GHG emissions of the cathode due to two reasons. First of all, nickel as raw material has lower associated GHG emissions than cobalt [66,114] and secondly, increasing the nickel content leads to higher energy density on cell and pack level so that slightly less total material is required to reach 1 kWh_c [95]. However, some uncertainty exists around the fact that increasing the nickel content leads to decreased chemical stability and higher energy requirements for the dry-room and quality of lithium-source as nickel-rich cathodes tend to employ Li(OH) rather than $LiCO_3$ [115].

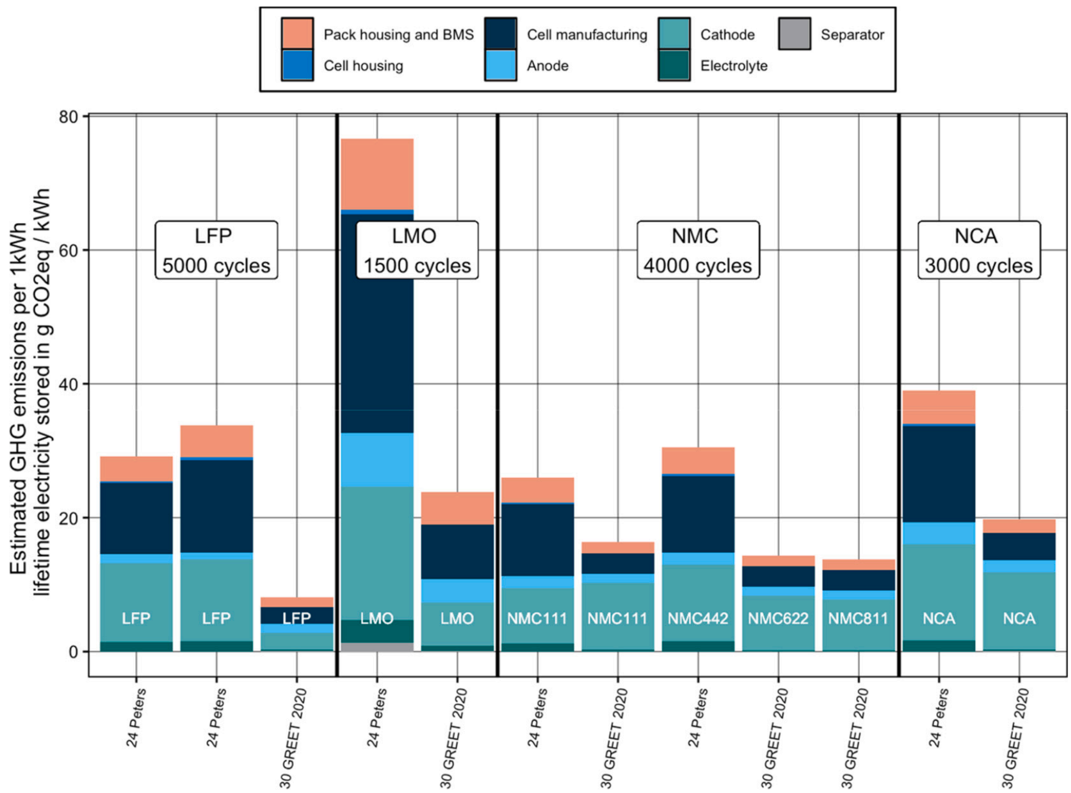

Fig. 6. Comparison of GHG emissions per 1 kWh_d lifetime electricity stored with standardized LCI from Peters & Weil [86] and GREET2020 [95] with consideration of cycle life of battery pack as function of chemistry.

environmental impacts of energy storage technologies with more up to data.

The review found that BESSs with LFP and NMC have been more frequently assessed than BESSs with NCA and LMO. Two studies found BESSs using LFP with substantially higher GHG emissions for 1 kWh_d lifetime electricity stored than BESSs with NMC. Another two studies reported BESSs using LFP with slightly higher GHG emissions for 1 kWh_d lifetime electricity stored than BESSs using NMC. Surprisingly, no study saw LFP based BESSs with lower emissions than NMC based ones, putting into question the marketing statements that BESSs using LFP are "greener" than those using NMC [92].

Expanding the system boundary to include the photovoltaic system used for charging the BESS showed GHG emissions between 43 and 195 gCO_2/kWh_{d+pv}. Generally, the PV-system was found to contribute a little more to combined emissions than the BESS (30–60 % BESS, 40–70 % PV), highlighting that attention should also be placed on environmental impacts of photovoltaic systems. Comparison with grid GHG emissions points to potentially interesting case studies for numerical comparison such as Russia, China, or India.

As only two studies provided own primary data in their LCAs of BESSs, the review expanded the search for primary data for LIB production. LIBs are the central element of BESSs, so that GHG emissions associated with the production of LIBs can be seen as a lower bound value for BESSs. The analysis found GHG emissions for the production of 1 kWh_c LIB capacity between 30 and 270 $kgCO_2/kWh_c$. Interestingly, the decline in reported GHG emissions during the last decade has not been at the same rate as the decline in LIB costs.

Cathode material and energy for cell manufacturing were seen as responsible for a large share of GHG emissions in LIB production. We found good alignment for the contribution of NMC cathode active material, but large uncertainty for LFP cathode active material. Comparing NMC with LFP, two studies reported NMC with lower GHG emissions than LFP for the production of 1 kWh_c LIB capacity. Primary data from USEAP [104] and GREET [95] reported LFP with lower GHG emissions than NMC. Problematically, none of the reviewed LCAs has used GREET or USEAP as data source for the BESS. Consequently, BESSs with LFP have been at a systematic disadvantage compared to BESSs with NMC. Using industrial LCI data from the GREET model showed lower bound emissions of 8 gCO_2/kWh_d for LFP. This is 30 % lower than the 12–14 gCO_2/kWh_d for NMC using GREET inventory data. These results challenge the conclusions of reviewed studies and indicate a need to better understand the environmental impacts of BESSs using LFP.

Finally, we provide three recommendations for future LCAs on residential BESSs.

- *Recommendation 1*: Use 1 kWh_d as standard functional unit. Even if research is interested in more complex functional units, providing results also for 1 kWh_d allows to put results and conclusions of different LCA studies into a more transparent perspective.
- *Recommendation 2*: Consider the impact of LIB battery chemistry on battery life.
- *Recommendation 3*: Use up-to-date, standardized LCI data for the BESS. Even recent studies have continued to use original LCI data from almost ten years ago even though more accurate LCI data has been available.

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CRediT authorship contribution statement

Moritz Gutsch: Writing – original draft, Conceptualization, Methodology, Investigation, Data curation, Formal analysis. **Jens Leker:** Supervision, Project administration.

Declaration of competing interest

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

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