The environmental impact of Li-Ion batteries and the role of key parameters – A review

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Abstract

The increasing presence of Li-Ion batteries (LIB) in mobile and stationary energy storage applications has triggered a growing interest in the environmental impacts associated with their production. Numerous studies on the potential environmental impacts of LIB production and LIB-based electric mobility are available, but these are very heterogeneous and the results are therefore difficult to compare. Furthermore, the source of inventory data, which is key to the outcome of any study, is often difficult to trace back. This paper provides a review of LCA studies on Li-Ion batteries, with a focus on the battery production process. All available original studies that explicitly assess LIB production are summarized, the sources of inventory data are traced back and the main assumptions are extracted in order to provide a quick overview of the technical key parameters used in each study. These key parameters are then compared with actual battery data from industry and research institutions. Based on the results from the reviewed studies, average values for the environmental impacts of LIB production are calculated and the relevance of different assumptions for the outcomes of the different studies is pointed out. On average, producing 1 Wh of storage capacity is associated with a cumulative energy demand of 328 Wh and causes greenhouse gas (GHG) emissions of 110 gCO_2eq. Although the majority of existing studies focus on GHG emissions or energy demand, it can be shown that impacts in other categories such as toxicity might be even more important. Taking into account the importance of key parameters for the environmental performance of Li-Ion batteries, research efforts should not only focus on energy density but also on maximizing cycle life and charge-discharge efficiency.

Keywords:

Life cycle assessment, Li-Ion battery, battery production, environmental impact, GHG emissions

1 Introduction

The electrification of the transport sector and the buffering of fluctuating electricity generation in the grid are considered to be key elements for a future low-carbon economy based mainly on renewable energies [1], [2]. Lithium-Ion batteries (LIBs) have made significant progress in the last decade and are now a mature and reliable technology with still significant improvement potential [3]–[5]. For mobile applications, they are already the dominating technology and their share in stationary energy systems is steadily increasing [6]. Several different types of LIB chemistries are widely established and broadly available, each with its own advantages and drawbacks [7]. Their increasing presence in daily life has also focused the attention on potential environmental concerns related to their production and disposal [8]. This issue has been repeatedly addressed by researchers, and numerous studies on the potential environmental impacts of LIB production and LIB based electric mobility are available [9]–[11]. For the quantification of the potential environmental benefits, these studies apply life cycle assessment (LCA). This is a standardized methodology for quantifying environmental impacts of products or processes, taking into account the whole life cycle [12]–[14]. The vast majority of existing studies focuses only on one or two types of batteries~~,~~ and all apply their own impact assessment methodology. Furthermore, studies often rely on the inventory data of previous publications, differ significantly in scope and system boundaries, and use fundamentally different assumptions for certain key parameters like battery cycle life or efficiency. Thus, the LCA results differ significantly due to these high uncertainties, and it is difficult to get a clear picture of the environmental performance of each LIB chemistry. Several reviews have been published in this regard but these are either comparably old [15] or focus primarily on electric mobility [9]–[11], rather than on battery production. In fact, there is currently no recent review about life cycle assessments of LIB. This paper reviews existing studies on the environmental impact of Li-Ion battery production. It provides a detailed overview of all relevant studies in the field and the key parameters of the LIBs assessed by them. By comparing the results and the assumptions made in the different studies, key drivers of uncertainty and thus of discrepancies among existing studies can be identified, providing recommendations for future LCA studies on LIB.

2 Review methodology

An extensive literature review is conducted in order to identify all available studies published on the environmental impacts of LIB production. The literature search is done in Science Direct, Scopus and Google Scholar using the search strings ‘LCA battery, “assessment battery production”, “assessment Li-Ion battery”, “analysis battery production”, and “battery impact environment”. All publications on life cycle assessment of batteries or battery production from 2000 to 2016 are considered. Those studies on e-mobility and stationary battery storage systems are also taken into account whenever the battery production phase is included and assessed as a separate process step. Furthermore, studies on new LIB technologies like all-solid-state cells are also taken into consideration and listed in the corresponding tables, since they show the potentials of future developments in LIB technology. Nevertheless, they are excluded when it comes to calculating average values from the reviewed studies, since they are still in a very early development phase and their technical properties are too different for being directly compared with conventional LIB. Studies focusing only on cathode materials or laboratory cells are generally excluded in order to maintain a sound basis for comparison. For all studies, the key assumptions and the obtained results are extracted and recalculated for 1 Wh of energy storage capacity. This allows for comparing studies that use different functional units and for calculating the mean value from all corresponding results as generic average. Whenever value ranges are given in the studies, the average value is used for calculations. Furthermore, the key sources of original Life Cycle Inventory (LCI) data are traced back thoroughly for each study to identify possible interdependencies and common data sources, thus providing valuable information for future works. For all reviewed studies, the key parameters used for modelling the battery production process but also for characterizing the battery performance are extracted and contrasted, and their relevance for the life cycle environmental impact is determined.

Finally, the key assumptions regarding battery performance parameters are compared to the current state of the art in battery technology in order to assess their robustness. For this purpose, a specific technology database for electrochemical storage systems is used (Batt-DB) [16], [17]. It is based on a permanent review of battery specifications available from manufacturers and research articles, providing an all-embracing picture of the current state of the technology. The Batt-DB currently contains 563 datasets from 49 scientific publications and 39 industry data sources (battery manufacturers) from 1999 to 2016. This allows for a statistical technology assessment. The sources included in the Batt-DB mainly consist of peer-reviewed articles from renowned scientific databases (Scopus, Science Direct and IEEEXplore) as well as reports from research institutes (e.g., Sandia Laboratories, Fraunhofer etc.). Manufacturer data is mainly obtained from publicly available technical data sheets and web pages. The database search is limited to include only lithium-based chemistries and publications not older than 2009; the same applies to the existing LCA studies, where the vast majority and, above all, the most relevant publications were released after 2009. This limitation provides a still sufficient amount of up-to-date datasets from scientific publications [18]–[60] and industry data sources [61]–[83].

Since the review focuses primarily on the impact of battery production, recycling of batteries is not considered, although this might have a considerable influence on the results. Especially the impacts associated with mining and resource extraction for the battery active materials can be reduced by recycling, since the demand for new virgin materials is decreased [10], [84]. Nevertheless, the recycling of batteries can also be associated with high efforts (temperature treatment, chemical treatment), which might even outweigh the positive environmental effects for some environmental indicators [85], [86]. Since no recycling technology is yet established on a larger industrial scale [87] and the environmental benefits vary strongly ~~between~~ in different technologies and different battery types. Including recycling technologies in the review would introduce additional uncertainties and therefore not contribute to the principal aim of this study.

3 Literature review results

3.1 Available studies

The literature search identifies an overall of 79 available LCA studies on LIBs and 34 on electric mobility. After a thorough review of all of these 113 publications, a total of 36 LCA studies are identified, that fulfil the selection criteria (e.g. that provide detailed results for LIB production and disclose sufficient information as to re-calculate these results on a per kg or per Wh of storage capacity basis). From these 36 studies, the most relevant parameters used and the main sources of inventory data are extracted and resumed in Table 1. As can be observed, the studies assess different battery chemistries, which are based on different fundamental assumptions, and use different electricity mixes or system boundaries. Furthermore, varying life cycle impact assessment (LCIA) methods are used, even for the same impact category (e.g. human toxicity; HTP), making a direct comparison of these studies difficult. Finally, it is found that the amount of original life cycle inventories (LCI) is limited and that numerous studies use or recompile LCI from other works, often in little transparent ways.

Of the 36 studies resumed in Table 1, six assess advanced LIB technologies: three include the use of nanomaterials for battery electrodes [88]–[90], two evaluate all-solid-state (SS) batteries [91], [92], and one a LIB with lithium metal anode [93]. While all these are listed in Table 1, the results reported by two of them (Li et al. [88] and Troy et al. [91]) are not taken into account for the calculation of the generic average results out of all studies. They report extreme values for environmental impacts due to the highly energy intensive production of specific materials (nanomaterials / all solid state electrolyte) and are thus considered outliers. Nevertheless, nanomaterials are increasingly used in electrode preparation for achieving higher capacities or cycle stability and are actually very energy intense in their preparation. The limited amount of studies assessing this aspect in detail indicates a demand for further research on the environmental trade-off between increased energy demand for nanomaterial production and the improved battery performance due the application of these materials [94].

Two additional publications - not included in Table 1 - are worth mentioning: (i) the recent assessment of electric vehicles by Bauer et al. [95], excluded from the table since it does not provide data regarding the impacts of battery production on a per Wh of storage capacity basis and (ii) the study by Gallagher et al. [96] about a Li-air battery, excluded because Li-Air is a technology considered to be too different from Li-Ion. Nevertheless, the study by Bauer et al. is taken into account for discussion and inventory data source analysis since it provides some interesting information in this regard.

>>> **Table 1.** <<<

Author	Year	Impact Cat & LCIA Method	BattChem	BattSize	SpecEnerg [$Wh \cdot kg^{-1}$]	LT [cycles]	LTSE [$kWh \cdot kg^{-1}$]	LCI data source	E-Mix	MA	SB	Eff [%]
Zackrisson [93]	**2016**	GWP, ADP (ILCD)	LFP-Li	149.7g (only cell)	107	4,000	342.4	Cell: own laboratory data, amended by Zackrisson [97] and Dunn [98] Materials: Ecoinvent [99] Assembly: Zackrisson [97]	EU, SE	T-D	WTW	90%
Ellingsen [100]	**2016**	GWP	NCM-C	177kg 253kg 393kg 553kg	100.0 105.1 107.1 108.3	180,000km / 12 years		Cell: Ellingsen [101] Materials: Ellingsen [101] Assembly: own estimation, based on Ellingsen [101]	EU	T-D	WTW	95%
Troy [91]	**2016**	ILCD Midpoint, CED	LCO-Li (SS)	4.2g (only cell)	58			Cell: own laboratory data Materials: GaBi [102] Assembly: own laboratory values	DE	n/a	CTG	--
Ambrose & Kendall [103]	**2016**	GWP	NCA-C NCM-C LMO-C LFP-C LFP-LTO			1,000 1,700 685 3,200 5,000		Cell: Calculated with BatPaC [104] Materials: GREET [105], [106] Assembly: average from literature	US	T-D	CTG	--
Sakti et al. [107]	**2015**	Cost	NCM-C	varied	n/a			Cell: Calculated with BatPaC [104] Material: not modelled (no LCA) Assembly: Dunn et al. [98]	US (n/a)	B-U	WTW	--
Latoskie & Dai [92]	**2015**	CED, GWP, HTP, PMF, POF, FE, MDP (ReCiPe Midpt.)	LCO-C LMO-C NCM-C LCO-C (SS) LMO-C (SS) NCM-C (SS) NCA-C (SS)	40 kWh	150 115 135 300 230 270 220	1,000 1,000 1,300	120 92 140	Cell: Calculated with BatPaC [104] Material: Hischier [108] Assembly: Dunn et al. [98]	US (2004)	B-U	CTG	--
Hammond & Hazeldine [109]	**2015**	GWP, AP; Particulates, Cost	LCO-C LCO-C (polymer)	30 kWh	120 140	1,500 400	144 44.8	Cell: mainly Rydh & Sandén [110] LCIA: Own methodology Very simple, e.g. disregard different electrolytes in Li-Polymer and Li-Ion and assembly	n/a	n/a	CTG	90%
Dunn et al. [111]	**2015**	CED	NCM-C LNCM-SiC LNCM-C LCO-C (SS) LCO-C LFP-C (SS) LFP-C LMO-C	180 kg 140 kg 160 kg 170 kg 170 kg 230 kg 230 kg 210 kg all: 28 kWh	155.6 200.0 175.0 164.7 164.7 121.7 121.7 133.3	n/a		Cell: Dunn et al. [112] Materials: Dunn et al. [112], GREET [105], [106] Assembly: Dunn et al. [112]	US (n/a)	B-U	CTG	--
Ellingsen et al. [101]	**2014**	ReCiPe Midpoint	NCM-C	253 kg 23.6 kWh	93.3 (pack) 174 (cell)	2,000	149.2	Cell: Majeau-Bettez [113] ; own primary data Materials: Majeau-Bettez [113]; Hischier [108] Assembly: battery producer (primary data)	own mix (similar US avg.)	T-D	CTG	95%

Faria et al. [114]	**2014**	ADP, AP, EP, GWP (CML)	LMO-C	300 kg 24 kWh	114	1,070@0.4C 1,260@0.6C 1,300@0.8C	118.6	Cell and assembly: Notter [115] Materials: Hischier [108]	PT (2011)	B-U	WTW	86%
Dunn et al. [112]	**2014**	CED	LNCM-SiC NCM-C LCO-C(SS) LCO-C LFP-C(SS) LFP-C LMO-C	28 kWh	191.8 151.3 164.7 164.7 119.1 119.1 130.2	n/a		Cell: Own data; calculated with BatPaC [104] Materials: Own LCI; GREET [106], [116], Majeau-Bettez [113] Assembly: BatPaC [104]	US, Chile (2009)	B-U	CTG	--
Li et al. [88]	**2014**	GREET Midpoint, CED	NCM-Si(n)	120 kg 43.2 KWh	360	200,000km 1,000 cycles at 80% DoD	274.5	Cell: Own data, US-EPA 2013 [90] Materials: Own data (nanomaterials), GaBi [102] (other) Manufacturing: GaBi [102]	US (2010)	n/a	CTG	90%
Hamut et al. [117]	**2014**	EI99 Endpoint	LFP-C	197 kg / 17.3 kWh	88	n/a		Cell and materials: Majeau- Bettez [113] Assembly: not considered	EU (2004)	T-D	WTW	--
US-EPA [90]	**2013**	own LCIA CED, ADP,AP, EP,GWP, ODP,POF ETP,HTP, Cancer	LMO-C NCM-C LFP-C	40 kWh (BEV) / 11.6 kWh (PHEV)	80-100 (not given for each chemistry)	10 years or 193,120 km -> 1,053 cycles	84.2 assumed for all types	Cell: Notter [115], Majeau-Bettez [113], add. data from primary sources. Material: Notter [115], Majeau-Bettez [113], GaBi [102] Assembly: Notter [115] (LMO) and Majeau- Bettez [113] (NCM and LFP)	US (2010)	B-U	CTG	85%
Simon & Weil [118]	**2013**	CED	LFP-C NCM-C	195 kg 175 kg /20 kWh	102.6 114.3	n/a		Cell: Notter [115], Zackrisson [97], Majeau-Bettez [113], Matheys [119] Materials: Hischier [108] Assembly: Notter [115], Zackrisson [97], Hischier [108]	n/a	T-D	CTG	--
Hawkins et al. [120]	**2013**	GWP	LFP-C NCM-C	273 kg 214 kg / 24 kWh	87.9 112.1	1,350	105.7 121.1	Cell & assembly: Majeau-Bettez [113] Materials: Majeau-Bettez [113], Hischier [108]	EU (n/a)	T-D	WTW	--
Mc Manus [121]	**2012**	ReCiPe Midpoint, CED	LFP-C (water and solvent based)		128-200	600	78.7	Cell and assembly: Zackrisson [97]; Rydht, Sanden [110]; Samaras & Meisterling [122] Materials: Hischier [108]	n/a	n/a	CTGr	--
Dunn et al. [98]	**2012**	CED, GWP	LMO-C	210 kg 28 kWh	130			Cell: Own data; based on BatPaC [104] Materials: own calculations, GREET [106] Assembly: Own estimation (process level)	US (n/a)	B-U	CTG	--
Gerssen-Gondelach & Faaij [30]	**2012**	CED, GWP, cost	NCM-C LFP-C		110 110	1,000 / 8 years	88 88	LCI based on Campanari et al. [123], who do not provide battery LCI. Up-stream LCI not modelled; only energy demand/emissions due to operation.	EU (2004)	n/a	WTW	90%
Aguirre et al. [124]	**2012**	CED, GWP	NCA-C	300 kg (BEV), 50 kg (HEV)	100	180,000 mi 1.5 batteries - >1,690cycle s	135.2	Cell and assembly: Sullivan & Gaines [116], Rydh & Sanden [110] Materials: Sullivan [116]	US-Calif. (2007)	T-D	WTW	--*
Majeau-Bettez et al. [113]	**2011**	ReCiPe Midpoint, CED	NCM-C LFP-C		112 88	3,000 6,000	269.2 422.2	Cell: own; based on Gaines & Cuenca [125], Schexnayder [126] Materials: Own data; Hischier [108] Assembly: Rydh & Sandén [110]	EU (2004)	T-D	WTW	90%
Gaines et al. [127]	**2011**	CED	NCA-C	75.9 kg	n/a	160,000 miles		Cell: Gaines and Nelson [128] Materials and assembly: not given	US (n/a)	n/a	CTG	--
Kushnir & Sandén [89]	**2011**	CED	LCO-C LCN-C LFP-C(n) LCN-LTO(n) LFP-LTO(n)	n/a	114-145 155 100 76 55	500-1,400 500-1,400 2,000-4,000 5,000-15,000 2,500-15,000	98.4	Cell: Gaines & Cuenca [125], Gaines and Nelson [128] Materials: Not modelled Assembly: Not given	EU (2010)	n/a	CTG	90%
Frisch-knecht [129]	**2011**	GWP, CED, ecopts	generic	312 kg	130	75,000 km		not indicated	n/a	n/a	WTW	--
Held [130]	**2011**	GWP, AP (CML)	NCM-C	40 kWh		8 years / 114,400 km		not indicated. No LCI data source given	DE (2010)	n/a	WTW	--
Notter et al. [115]	**2010**	EI 99 Endpoint CED, GWP, ADP	LMO-C	300 kg; 34 kWh	113.3	1,000	90.7	Cell: Primary data (reference cell) Material: Own calculations; ecoinvent [99] for secondary inputs Assembly: Own estimations (process level)	CH (2004)	B-U	WTW	80%*
Zackrisson et al. [97]	**2010**	GWP, AP, EP, ODP, POF (CML)	LFP-C (water- and solvent-based)	107 kg 10 kWh	93	3,000	223.2	Cell: Gaines & Cuenca [125] Materials: Hischier [108] Assembly: approximated from manufacturer's annual report [131]	EU (2004)	T-D	WTW	90%
Sullivan et al. [116]	**2010**	CED, GWP	NCA-C	139 kg (BEV)	100			Cell: Rydh & Sandén [110] Materials: GREET [106] Assembly: GREET [106], Rydh &	US (n/a)	T-D	CTG	--

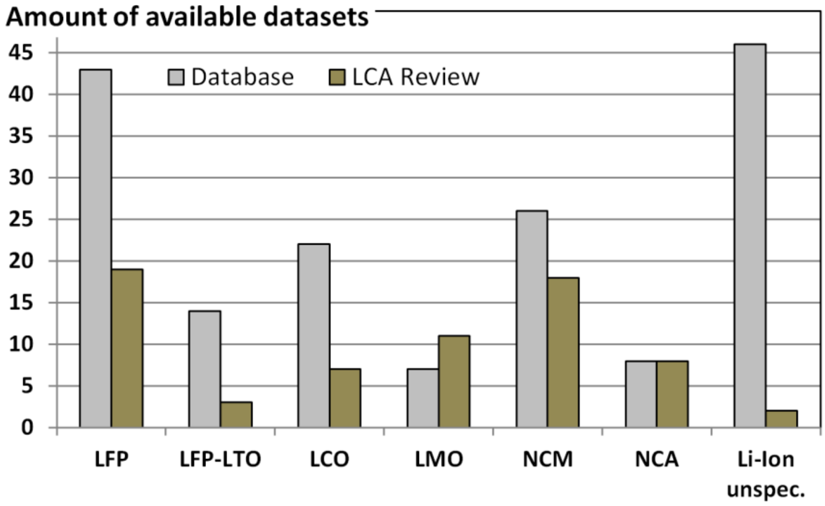
								Sandén [110]				
Bauer [132]	**2010**	GWP, HTP (CML); AP,EP, ETP (EI99)	NCA-C LFP-LTO	142 kg 482 kg / 25 kWh	132 52	5,000 10,000	528 416	Cell: Own data Materials: Hischier et al. [108] Assembly: Hischier et al. [108]	JP (2004)	T-D	CTG	--
Van Mierlo et al. [133]	**2009**	GWP	generic	408kg	125	160,934 km		LCI directly from Matheys [119]	n/a	n/a	WTW	--
Samaras & Meisterling [122]	**2008**	CED, GWP	NCA-C	75/250kg	100	2,500	200.0	Cell, materials and assembly: Rydh & Sandén [110]	US (2004)	T-D	WTW	--
Hischier et al. [108]	**2007**	n/a	LMO-C	301 kg 43,2 kWh	143.5	n/a		Cell: Own calculations based on Linden & Reddy [134] Materials: Own LCI Assembly: Estimated based on Industry data [135]	EU (2004)	T-D	CTG	--
Matheys et al. [119]	**2006**	EI99 Endpoint	generic	92 kg / 11.5 kWh	125	1000	100.0	n/a (no references given, no LCI data source and no LCI data)	n/a	n/a	WTW	90%
Rydh & Sandén [110]	**2005**	CED	NCA-C	4-6t	80-120	3,000-5,000	320.0	Cell: Primary data (battery manufacturer) Materials and assembly: Own data; Almemark et al. [136]	n/a	T-D	CTGr	85-95%
Ishihara et al. [137]	**2002**	CED, GWP	LCO-C LMO-C	2-4 kWh 2-4 kWh	n/a	n/a		Cell and assembly: Primary data (battery manufacturer) Materials and LCIA: not given	JP (n/a)	T-D	CTG	--
Gaines & Cuenca [125]	**2000**	Cost	LMO-C	100 Ah / 35 kWh		1,000		Cell: Own data; based on various literature sources and statistic data Material: not modelled (no LCA) Assembly: based on an existing plant, with adaptations according to author´s engineering judgement	US (n/a)	T-D	CTG	--

3.2 LCA framework in existing studies

3.2.1 Goals and scopes

16 of the 36 studies contained in Table 1 assess e-mobility on a well-to-wheel (WTW) basis with the battery production being only part of the assessed system. The remaining studies focus explicitly on battery production. Studies for stationary energy storage that include the production phase as an individual process are rare [110], [121], and classified as cradle-to-grave (CTGr) studies in Table 1. Assessed cathode chemistries include lithium iron phosphate (LFP), lithium cobalt oxide (LCO), manganese spinel oxide (LMO), and composite oxides (LCN, NCM and NCA) (including nickel (N), cobalt (C), aluminium (A) or manganese (M)). Two studies do not mention the type of battery chemistry at all and only show results for a generic Li-Ion battery (defined as "Li-Ion unspecific"). Li-polymer batteries, while of certain relevance for small mobile devices [138], are not considered as a separate battery type, but classified according to their electrode chemistry. The most assessed battery chemistries are LFP (assessed in 19 studies) and NCM (18 studies), while only few studies deal with LCN and NCA type batteries (2 and 8, respectively). As anode material almost exclusively carbon (C), normally in the form of graphite, is considered. Only three studies also assess anodes based on the lithium salt of titanium oxide (lithium titanate; LTO-type); two in combination with LFP and one with an LCN cathode. Another three studies deal with a silicone-graphite anode, all in combination with NCM cathodes. Finally, one single study focuses explicitly on a lithium-metal anode.

The amount of data sets used in the battery database (Batt-DB) and obtained from the LCA-review regarding the different LIB chemistries is given in Figure 1. It can be seen that the relative amount of LCA studies published on each of the different battery chemistries corresponds fairly well with their distribution within the Batt-DB, i.e. the relevance of the different battery types is reflected within the LCA studies. The highest number of datasets is available for LFP type batteries, and significantly less for LCN and NCA. LFP is an established technology, while LCN and NCA are still under development, thus decreasing the reliability of technical data for these chemistries [15].

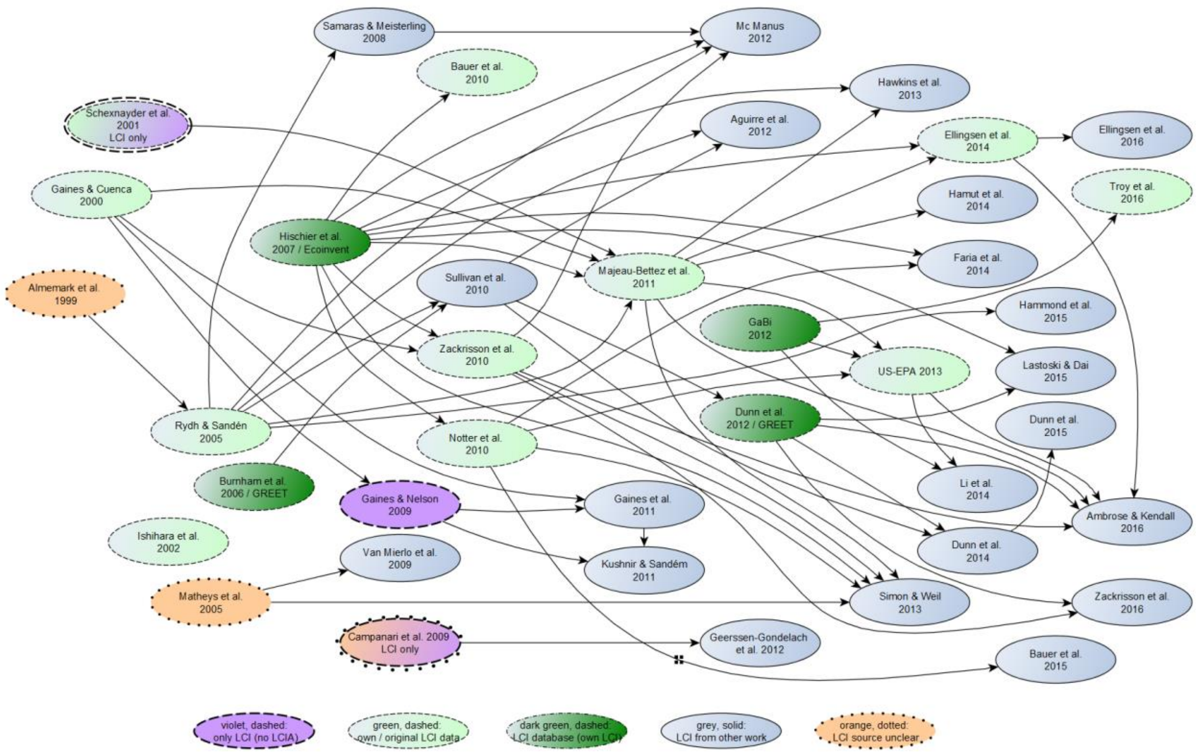


>>> **Figure 1.** <<<

3.2.2 Sources of inventory data

The quality of the inventory data is one of the keys to reliable results. In this sense, the limited amount of original life cycle inventory (LCI) data underlying the reviewed studies is noteworthy. Literature data are often re-used and new studies are based on previously published results or inventories. We identify a total of 15 studies that use own LCI data. Of those, seven studies rely exclusively on own primary LCI, while another eight re-use these LCI partially, amending them with own original data. The remaining 22 studies (including the one by Bauer et al. [95] not contained in Table 1) are based completely on the LCI of previous studies. Figure 2 gives an overview on the interdependencies of the LCI data sources for every reviewed study (as far as provided). The corresponding references can be retrieved from Table 1.

As can be observed in Figure 2, the principal LCI data sources for most LCA studies on LIB are the following eight publications: Gaines and Cuenca (2000) [125], Rydh and Sandén (2005) [110], Hischier et al. (2007; ecoinvent) [108], Zackrisson et al. (2010) [97], Notter et al. (2010) [115], Majeau-Bettez et al. (2011) [113], Dunn et al. (2012) [112] and US-EPA (2013) [90]. The vast majority of the remaining studies do not provide own inventories, but base their assessments on one or several of these studies. Although their LCI might be recompiled and acquired from several other studies and thus give new LCA results, they nevertheless depend on the primary LCI. Among the more recent studies, only Ellingsen et al. [101] and Troy et al. [91] provide own original LCI, and especially Ellingsen et al. in a very detailed way, why their study can be expected to become another reference source for LCI data in future.



>>> **Figure 2**. <<<

3.2.3 Modelling of manufacturing energy demand

Among the reviewed studies, a major difference in modelling the energy demand of the battery manufacturing process is identified. Basically, in literature two different approaches are used: (i) The top-down approach, which uses data from industry for a complete manufacturing plant (often not only producing batteries) and then divides the gross energy demand of this plant by the output of the plant (or allocates it according to economic value of the products in case of plants with multiple products) [97], [101], [113], [132], and (ii) the bottom-up approach, which uses data from industry or from theoretical considerations for certain key processes within the manufacturing line (which are assumed to represent a determined share of the total plants energy demand) and extrapolates the whole plant energy consumption on this basis [90], [98], [104], [115]. These two modelling approaches are found to impact the calculated energy demand of the battery manufacturing process by as much as an order of magnitude, and propagate into the studies that rely principally on the corresponding LCI data.

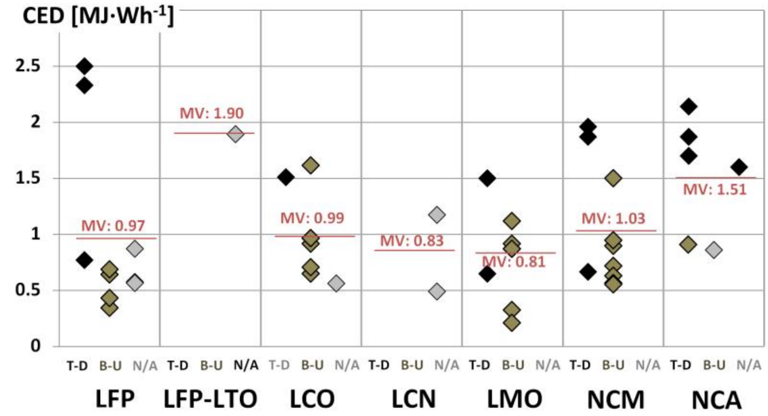
3.2.4 Applied impact assessment methodology

The majority of the reviewed studies focus on energy demand and GHG emissions. Global warming potential (GWP) is the most frequently assessed category (24 studies), followed by cumulative energy demand (CED; 19 studies). Other environmental impacts, such as toxicity or acidification, are considered less often. 16 studies quantify impacts in additional categories, mainly abiotic depletion (ADP), acidification (AP), eutrophication (EP), human toxicity (HTP) and ozone depletion (ODP). Other impact categories are used only occasionally. For these, only a few data points are available. Often data is only available for the most common battery chemistries, making a comparison between battery types difficult and in some cases even impossible. The impact assessment methodologies used for quantifying these impacts are ReCiPe [139] (four studies), CML [140] (three studies), EI99 [141] (three studies) and ILCD [142], while four other studies use own LCIA methods, and one study combines CML and EI99 [132]. Almost all reviewed studies use midpoint indicators, and only these three that use EI99 for the impact assessment calculate an endpoint result (EI99 single score). The impact assessment methodology used by each study and the assessed categories are contained in Table 1.

3.3 LCA results from existing studies

3.3.1 Energy demand of battery production

Figure 3 shows the CED results as published in the reviewed studies, broken down to battery chemistries and manufacturing modelling approach. The overall mean CED for producing 1 Wh of storage capacity is 1.182 MJ (or 328 Wh), although the CED obtained from different studies varies up to one order of magnitude. This is mainly the result of the high uncertainties associated with the discussed modelling approaches of the battery cell manufacturing process (top-down vs. bottom-up) essentially splitting the results into two groups. Figure 3 illustrates how the top-down approach tends to result in higher CED values as compared to the bottom-up approach.

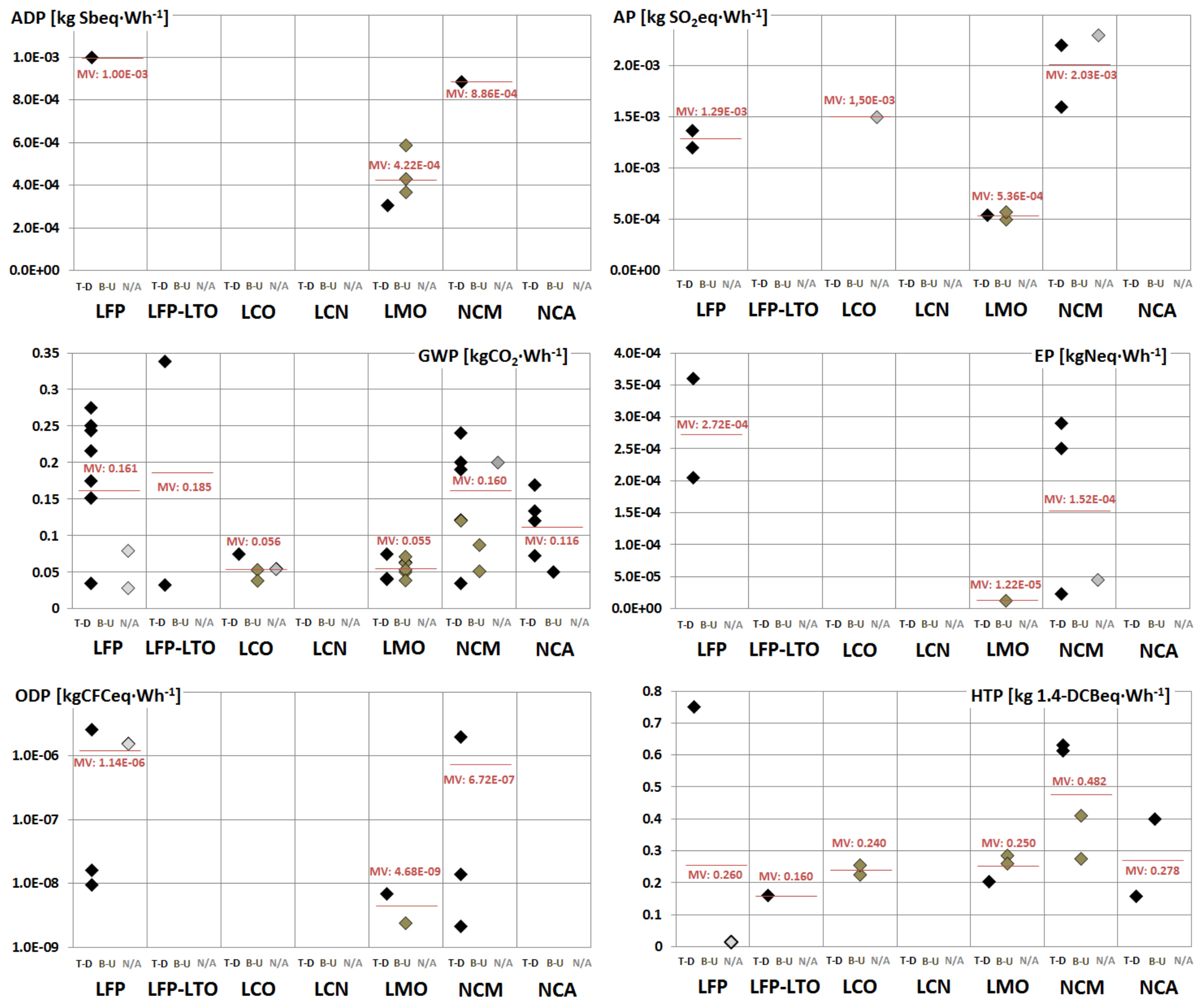


>>> **Figure 3.** <<<

Comparing the average values of the different battery chemistries, LFP-LTO shows the highest and LMO the lowest CED per Wh storage capacity. The high CED for LFP-LTO might be due to their low specific energy density, but partially also due to the use of nanomaterials in the electrode materials, which are associated with high energy expense for their production. Since the only study that quantifies the CED for LFP-LTO applies nanomaterials, this cannot be verified in comparison with other studies. Nevertheless, it has to be taken into account that many electrode materials often already contain "simple" materials on nanoscale like e.g. hard carbon. A clear distinction between nano- and conventional materials and thus the energy demand for their production is therefore often impossible. A high CED is also obtained for NCA, although NCA offers a comparably high specific energy density. Here, the high CED value obtained for this chemistry might at least partially be attributable to the modelling approach of the manufacturing process. Since only one study uses the bottom-up approach for the NCA. In this sense, the modelling approach of the manufacturing process might impact the results more severely than the choice of battery chemistry itself.

3.3.2 Environmental impacts of battery production

Figure 4 shows the results in the six most frequently assessed impact categories. Since various studies use different life cycle impact assessment (LCIA) methodologies, the results are provided in different units in certain impact categories and cannot be compared readily. Therefore, only those that report using the same unit as the majority of the studies are listed. However, it should be noted that although the same unit is used, different LCIA methodologies can use different characterization factors, further reducing the comparability of the results. Still, we consider the value of including an increased amount of datasets to compensate for the increased uncertainty due to comparing midpoint characterization results from different methodologies. For a summary of all values and the information about the LCIA methodology used in each study, see Table A1 in the Appendix and Table 1, respectively.



>>> **Figure 4** <<<

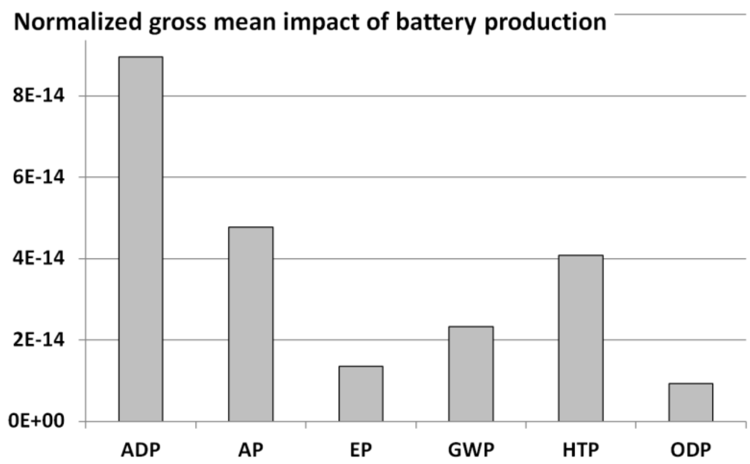
GWP is by far the most often assessed category, and when averaging the data of all existing studies, the total mean GHG emissions associated with the production of 1 Wh of storage capacity are found to be 110 g CO_2eq. For all other categories, only a few data points for certain battery chemistries are available. Nevertheless, the general picture obtained in the categories ADP and AP is similar to that for CED and GWP. Here, usually fossil energy demand is the main driver for environmental impacts. LFP and NCM type batteries cause comparably high impacts in these categories, while LMO scores significantly better. Although impacts in these categories depend heavily on the energy or electricity mix used in the assessments, in almost all studies the electricity mix shows a comparable share of fossil energy of between 50 and 70%. Details about the electricity mix used by each of the studies can be retrieved from Table 1.

Also the influence of the approach for modelling the manufacturing process has to be taken into account, with the distribution between bottom-up and top-down studies strongly varying between categories. For example, for the LFP- or NCA- type batteries, the studies that use top-down approaches clearly drive up the average results for CED and GWP, while studies using bottom-up approaches obtain significantly lower values (for the remaining categories, the amount of data points is too low as for drawing any sound conclusion in this regard). For LFP batteries seven of nine studies that assess the GWP use top-down approaches. This might be one of the reasons for the comparably high average GHG emissions for this chemistry. In any case, the influence of the approach for modelling manufacturing energy demand cannot be determined in an isolated way (e.g. independently from the influence of the used electricity mix), since no further details on the modelling of the electricity mixes is given in the corresponding studies.

For the toxicity categories, such as HTP, the manufacturing model approach (i.e., the energy demand for the manufacturing process) can be expected to be less relevant, since mining and resource production play a more significant role in this category [10]. Here, LFP performs best, probably attributable to the absence of materials such as nickel or cobalt, whose mining and production (but also end-of-life handling) cause significant toxicity impacts [143]. In general, few data points are available for the categories ADP, AP and EP. ODP offers a broader data basis, but its results vary by several orders of magnitude (note the logarithmic Y-axis in this category). Thus, the results in these categories are associated with very high uncertainties. In order to improve this situation, further research would be needed in this area.

3.3.3 Relevance of different impact categories

Normalization of LCA results can help to provide a rough idea of the relevance of the different categories for the overall environmental impact. For this purpose, the overall average impacts for battery production as obtained from the review are divided by the average annual impacts generated in Europe (Reference year 1995) [140]. Figure 5 displays the characterization results for battery production normalized in this way. Compared to the average annual impacts in Europe, battery production causes high relative impacts in ADP, AP and HTP, while GHG emissions, the most frequently assessed category, has a comparably low value. This underlines the importance of assessing additional environmental impacts apart from CED and GWP and indicates the need for further research on assessing these impacts. For some key materials like lithium or rare earth metals, no ADP characterization factors are implemented in common LCIA methods, so the impact in this category might be even higher [10], [139], [144].



>>> **Figure 5.** <<<

4 Discussion: Impact of the key assumptions on the results of the studies

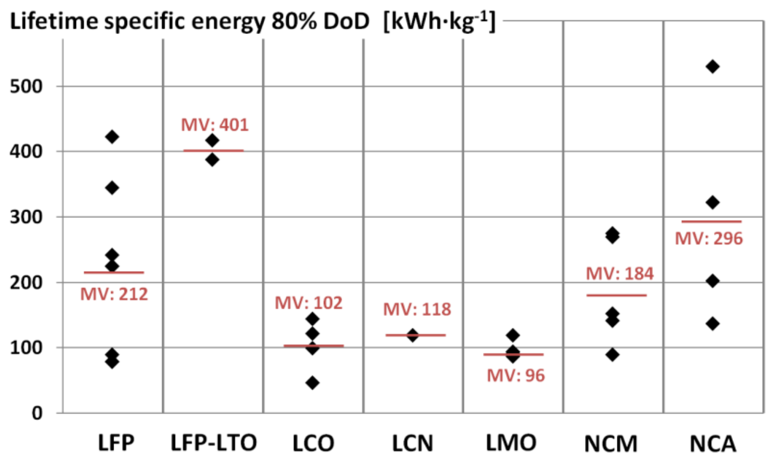
The assumptions used in the reviewed studies concerning key parameters like energy density, cycle life or internal efficiency vary significantly. In order to provide an idea of the relevance of these variations for the outcomes of the studies, the most critical parameters in the reviewed LCA studies are analysed in the following and compared with the corresponding actual battery data obtained from the battery database (Batt-DB). That way, possible correlations and discrepancies between the assumptions and actual battery specifications are identified, providing an idea of the corresponding uncertainties and the sensitivity of the final results on them. For this purpose, a cradle-to-gate perspective is used. The batteries are assumed to be used in electric vehicles, since this is also the battery application used in the vast majority of the reviewed studies. Including the use phase in the analysis allows for assessing the influence of electrochemical performance parameters on the total environmental impact of the studied LIB systems.

4.1 Impact of calendric and cycle life

All reviewed studies that include the battery use phase find battery production to contribute a significant share to the environmental impact over lifetime. This share depends on the amount of charge-discharge cycles provided by the battery, which is therefore important for the overall environmental performance [101], [113], [145], [146]. The calendric and cyclic life time of an LIB is determined by different phenomena of degradation in the cell over time and cycles [39], and depend on the depth of discharge (DoD), charging-rate and operation temperature [55], [147]. An LIB is usually considered to be at its end of life when its usable energy capacity reaches 80 % of its initial value [55], [39]. While significant differences in cycle life exist between battery chemistries, almost all of the LCA studies that focus explicitly on battery production impacts assess the batteries on a storage capacity basis (normally 1 Wh), not accounting for the battery lifetime. This might give misleading conclusions when it comes to comparing battery chemistries. LFP chemistries for example, which show comparably low specific energy and increased GHG emissions per Wh of storage capacity, can achieve significantly higher cycle life than other established chemistries. The studies that include a well-to-wheel (WTW) perspective could take this into account, but they normally assume the battery to simply last one vehicle life. Still, some do consider cycle life limitations, but calculate the corresponding battery requirements by fractions (i.e. 1.5 batteries needed over one vehicle lifetime [124], [133]), while in reality a battery pack would most probably not be replaced partially. Others try to assess the remaining battery cycle life after the vehicle's end of life by giving credits for secondary use in stationary applications, but find very limited environmental benefit for this option [114]. Thus, a battery lifetime far above that of the corresponding vehicle glider and drivetrain might not provide significant environmental benefits either.

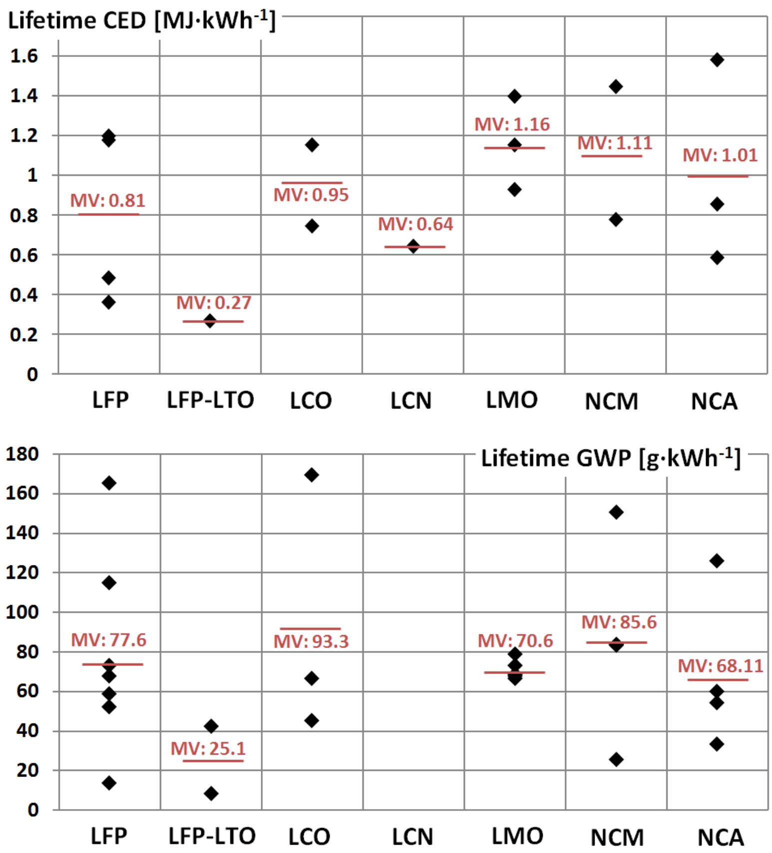
4.1.1 Life time environmental impacts

In order to account for the cycle lives of the different battery chemistries, the environmental impact per 1 kWh of storage capacity over the battery lifetime is calculated for all studies where information about the cycle life can be derived. An average 80% DoD for all battery types is assumed. Figure 6 shows the lifetime specific energy assumed by the studies that provide information in this regard, broken down to battery chemistry. The extraordinarily high cycle life of LFT-LTO batteries gives a high specific storage capacity when accumulated over lifetime.



>>> **Figure 6**. <<<

Based on the lifetime specific energy, Figure 7 shows the CED and GWP impacts per kWh of storage capacity over the whole battery lifetime. The high cycle life especially of the LFP-LTO type batteries leads to favourable results when assessing the lifetime impacts, making LFP-LTO type cells one of the most promising ones. LCN type batteries also achieve very good results, but again, data availability for this chemistry is low and the result is based on only one single publication. Averaged over all LIB chemistries, providing 1 kWh of electricity over battery lifetime requires 0.26 kWh of fossil energy and causes GHG emissions of 74 g only due to the production of the battery, i.e., without considering internal inefficiencies (Chapter 4.2) or end of life handling. Further research would also be needed regarding the impact of battery life on the vehicle lifetime. One could imagine that the need for a battery replacement in an older electric vehicle might be economically unfeasible and be considered a constructive total loss and thus decrease vehicle lifetime [148]. This could result in an even higher importance of battery lifetime.



>>> **Figure 7.** <<<

4.1.2 Life time assumptions compared to actual battery performance data

As mentioned before, only part of the reviewed LCA studies consider cycle life and those that do, assume fixed cycle life times at a DoD of 80 %. This is a strong simplification of reality as a traction battery will not be fully discharged every single time until the allowed minimum State of Charge (SoC) of 20 %.

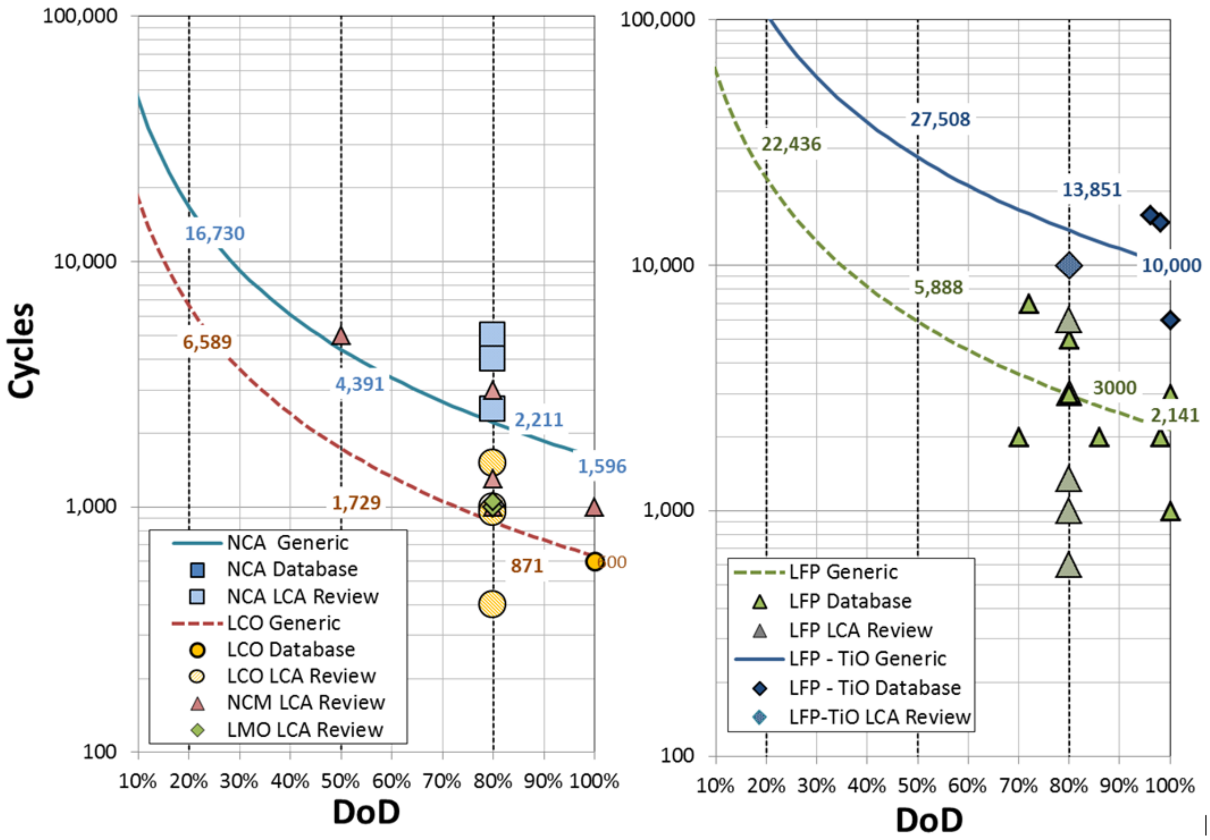
We use the available data in the battery database (Batt-DB) to calculate a simple approximation of cycle life time in dependence on DoD using Equation 1 [149]. To adopt it to different LIB types, a specific shape factor S_F is added, calculated according to Equation 2 based on an average amount of cycles at a certain DoD as given in the Batt-DB. Charging rates and temperature effects are not considered in this simplified calculation.

$$\boldsymbol{C_F} = \mathbf{exp}\left(\frac{-\boldsymbol{LN}(\boldsymbol{DoD})}{\mathbf{0.686}+\boldsymbol{S_F}}\right) \qquad \text{Equation 1}$$

$$\boldsymbol{S_F} = \boldsymbol{LN}(\boldsymbol{C_{av}}) + \frac{\boldsymbol{LN}(\boldsymbol{DoD_{av}})}{\mathbf{137}} \qquad \text{Equation 2}$$

With: C_F = Number of cycles in dependence of a specific DoD; S_F = curve shape factor; dependent of the assessed battery type (original value is 7.25); DoD_{av}= average DoD for given battery chemistry from Batt-DB; C_{av}=average cycle life from Batt-DB

The calculated correlation between cycle lifetime and DoD for different battery technologies is given in Figure 8. The average results for 80% DoD obtained in this way are compared with those used in the reviewed LCA studies in Table 2 for verifying the corresponding assumptions.



>>> **Figure 8**. <<<

>>> **Table 2.** <<<

	LFP	**LFP-LTO**	**LCO**	**LMO**	**NCM**	**NCA**
LCA studies - min	600	5,000	400	685	953	1,690
LCA studies - max	6,000	10,000	1,500	1,300	3,000	5,000
LCA studies – avg.	2,575	7,917	967	1,006	1,659	2,832
Batt-DB - avg	2,960	13,850	900	1,268	1,217	2,200

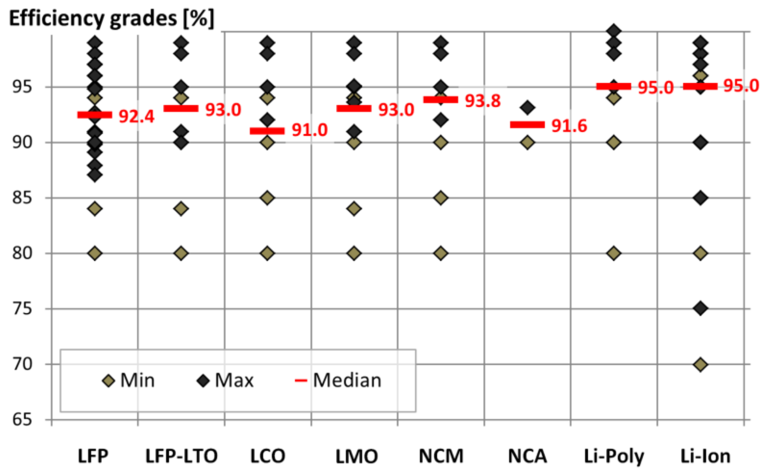
It seems that on average the cycle life assumptions made in the reviewed studies adequately reflect the current state of technology. Only the lifetime of LFP-LTO is underestimated significantly by the two studies that assess this chemistry. For NCM-C type batteries, the Batt-DB gives surprisingly low cycle life values, significantly below the value assumed in average by the LCA studies. In any case, data about the relation between DoD and cycle life is very scarce and usually not contained in technical datasheets or specifications, why a high variation can be observed both in the studies and in the Batt-DB for this parameter. Thus, special attention should be given to cycle life assumptions when assessing LIB, given its high impact on the environmental performance over lifetime.

The second ageing effect, calendric aging, is based on chemical side reactions which can occur over time and depends primarily on the cell’s storage temperature [17], [39]. Only a few of the LCA studies consider this type of battery degradation in a very simplified way [30], [88], [90]. Independent from battery chemistries, they all assume a calendric life of 10 years, and vary the lifetime in a sensitivity analysis by reducing / increasing this value by 30% or 50%. As a result of missing long-term experience and uncertainties in ageing models, data on calendric lifetime for different battery chemistries is very scarce [16], [17]. Nevertheless, especially for vehicles with a comparably low annual mileage and low average DoD, the calendric ageing could be a major cause of battery degradation and thus be potentially relevant.

4.2 Impact of battery efficiency

The battery´s internal efficiency determines the amount of energy lost in every charge / discharge cycle due to internal resistances. In general, LIBs have very high efficiency grades over 90 % under normal charging conditions [150]. There are several aspects that can influence LIB efficiency such as the charging rate, temperature and the used battery management system [39]. The majority of all LCA studies that take charge-discharge efficiency into account assume an average battery efficiency of 90% (the value used by each study can be retrieved from Table 1). For a charge-discharge efficiency of 90%, the CED_{nr} (nr= non-renewable) for storing 1 kWh of electricity caused by internal inefficiencies is about 0.3 kWh and the corresponding GWP 46.7 g CO_2eq (for an average European electricity mix (2012) with a CED_{nr} of 3 kWh and a GWP of 467 g CO_2eq per kWh [9]). Thus, the impacts of internal losses on CED and GWP over battery lifetime are in the same order of magnitude as those of the production of the battery itself. In consequence, the differences in internal efficiency between different battery technologies can have significant impacts and should not be neglected when assessing their environmental impacts.

Figure 9 shows the comparison of efficiency grades obtained from the battery database Batt-DB for different battery chemistries. “Li-Ion” represents the generic data sets obtained from the Batt-DB where information about the chemistry was not obtainable. It can be observed that the average charge / discharge efficiency greatly differs among the analysed chemistries, but is notably above 90% for all battery chemistries. In consequence, it seems that the existing LCA studies (if they consider this aspect at all) tend to underestimate the internal efficiency and thus overestimate the corresponding environmental impacts. However, the values from the Batt-DB are values for new batteries and efficiencies might decrease over lifetime, why over lifetime these discrepancies might actually be smaller.

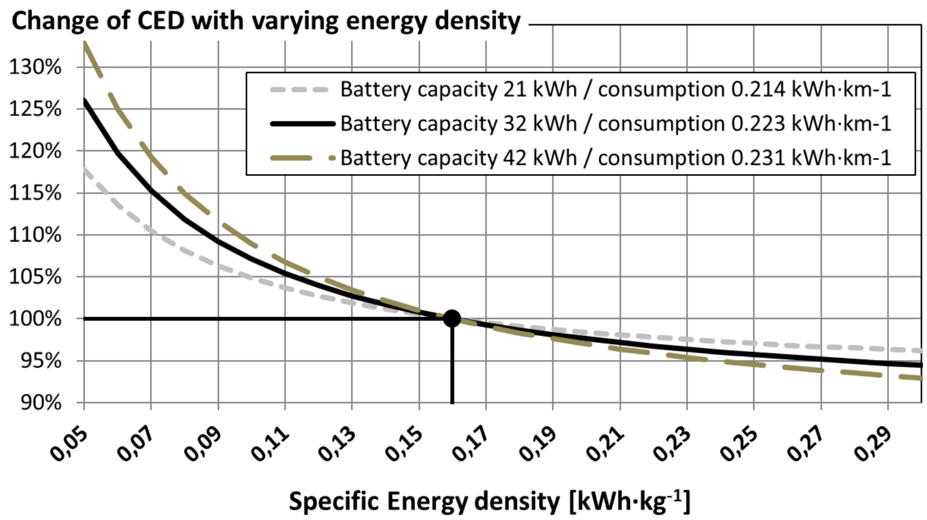


>>> **Figure 9** <<<

4.3 Impact of battery energy density

The energy density of Li-Ion batteries is determined by the capacity of active material and the amount of additional passive components (which are not storing energy but are necessary for functionality, e.g., the electrolyte) contained in the battery. Losses and internal inefficiencies and discharge limitations further reduce the available energy (deep-discharge of LIBs severely affects their lifetime; therefore the DoD usually does not surpass 80%) [94]. The energy density varies strongly between battery chemistries, with the more robust chemistries like LFP showing significantly lower energy densities than other high-energy types like LCO or NCM.

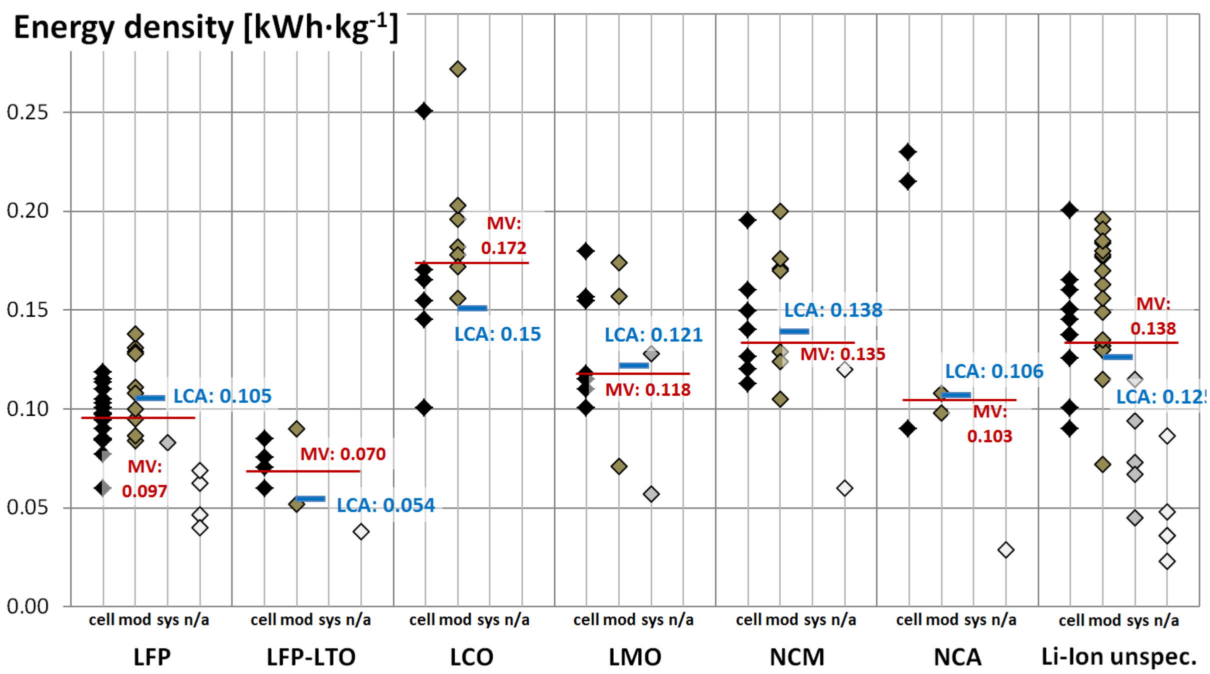
For the assumed use of the batteries in electric vehicles, the impact of battery storage capacity and energy density on electric vehicle fuel consumption can be calculated using the Common Artemis Driving Cycle (CADC)[151]. The relation of battery size and energy density to vehicle energy demand is given in Figure 10. Details on the calculation method can be found in the Appendix.



>>> **Figure 10.** <<<

Figure 10 gives a rough idea of the relevance of specific (mass based) energy density. If battery specific capacity is increased by e.g., 50% from 160 to 240 Wh·kg^{-1}, this would result in an increase in fuel economy of 2 to 5% [152], or a reduction of CED of 0.06 and 0.15 kWh per kWh of provided energy using the above assumptions. Thus, specific energy density (mass basis), usually one of the main aims of new battery developments, does not need to be more relevant than improving battery lifetime or charge-discharge efficiencies from an environmental point of view. The latter might even contribute more to the WTW performance than the elevated vehicle weight due to the traction battery [97].

The assumptions used in the reviewed studies regarding energy density can be contrasted with actual battery data from the battery database (Batt-DB). Figure 11 displays the energy densities obtained from the Batt-DB for the different battery chemistries in comparison with the average value obtained from the reviewed LCA studies. The values from the Batt-DB are given separately for cell, module, and system, according to the technical datasheet. Surprisingly, for several battery chemistries, higher values are obtained for battery modules than for cells, what seems to be due to the very different origins of the comparably heterogeneous datasheets contained in the database.



>>> **Figure 11.** <<<

It can be seen that the average values from the Batt-DB are comparable to those from the reviewed LCA studies. LFP-LTO type batteries show the lowest, and LCO the highest specific energy density.

While on average the assumptions made in the LCA studies represent the actual technical state of the art fairly, the high variation of results both in the Batt-DB and in the LCA studies has to be considered, underlining the importance of sensitivity analysis and a careful selection of the baseline assumptions for any assessment.

5 Conclusion

The review identified an overall of 79 studies that assess the environmental impact of Li-Ion battery production. Of those, 36 studies provide sufficient information as to extract the environmental impacts obtained per kg of battery mass or per Wh of storage capacity, respectively. The majority of the reviewed studies do not provide own original inventory data, but rely on those of previous works. Thus, the basis of original LCI data is comparable weak, with only a few publications providing the inventory data for all existing studies. Still, the variation in results is very high, what can be explained with the different assumptions made in the studies regarding key parameters like lifetime or energy density, but also manufacturing energy demand. The average CED and GHG emissions for battery production across all chemistries are 328 kWh and 110 kg CO_2eq per kWh of storage capacity, respectively. The majority of the identified studies focus on GHG emissions or energy demand, while potential impacts in other categories are quantified less often, in spite of the high relative importance especially of toxicity and acidification, but also resource depletion aspects.

The assumptions made by the reviewed studies concerning performance parameters like cycle life, internal efficiency and energy density are found to be equally relevant for the environmental life cycle performance of the batteries, while often modelled in a very simplified way or even disregarded. Especially a high cycle life is a key for a good environmental performance, converting the LFP-LTO type batteries into the most favourable battery chemistry in this regard. Averaged over all chemistries, providing storage capacity for 1 kWh of electricity over the entire life cycle of a battery is associated with a CED of 0.26 kWh and GHG emissions of 74 g CO_2eq. Interestingly, the approach for modelling the energy demand for battery manufacturing seems to influence the final environmental performance of the battery production more than the choice of the battery chemistry itself. Consequently, future LCA studies on LIB production should consider modelling energy demand during battery manufacturing, but also internal battery efficiency and battery lifetime more thoroughly. It can be assumed that the next generation of batteries, e.g. Li-S or $Li\text{-}O_2$, which are based on chemical conversion rather than intercalation, will potentially suffer from poor cycle efficiency. In such a case, their advantage in energy density might be outweighed by energy loss and / or lower lifetime. The explicit consideration of these parameters in future environmental assessments could thus help to significantly increase the quality and robustness of the results.

Glossary

Battery chemistries

C	Carbon (usually graphite for battery electrodes / anodes)
LCN	Lithium Cobalt Nickel Oxide
LCO	Lithium Cobalt Oxide
LFP	Lithium Iron Phosphate
LMO	Lithium Manganese Oxide
LTO	Titanate
NCA	Lithium Nickel Cobalt Aluminium Oxide
NCM	Lithium Cobalt Manganese Oxide
n	Nano (indicates the use of nanomaterials in the battery)
SS	Solid state (battery technology with solid electrolyte)

Environmental impact categories

ADP	Abiotic depletion
AP	Acidification potential
EP	Eutrophication potential
CED	Cumulative Energy Demand
GHG	Greenhouse Gas
HTP	Human Toxicity
ODP	Ozone depletion
PMF	Particulate matter formation
POF	Photochemical ozone formation

Others

BESS	Battery energy storage system
BEV	Battery electric vehicle
B-U	Bottom-up (approach for modelling energy demand for battery production)
CADC	Common Artemis Driving Cycle
CTG	Cradle-to-gate (use phase excluded in assessment)
DoD	Depth of discharge
E-Mix	Electricity mix used for an LCA study
EI99	Ecoindicator 99 (impact assessment methodology)

EV	Electric vehicle
GREET	Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation Model (impact assessment methodology)
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life Cycle Impact Assessment
LIB	Lithium-Ion battery
LT	Lifetime (charge-discharge cycles)
LTSE	Lifetime specific energy density (kWh·kg^{-1})
MA	Manufacturing approach (for modelling production energy demand)
nr	Non-renewable (subscript for CED)
PHEV	Plug-in hybrid electric vehicle
SB	System boundaries
SpecEnerg	Specific Energy density (Wh·kg^{-1})
T-D	Top-down (approach for modelling energy demand for battery production)
WTW	Well-to-wheels (use phase included in assessment)

Appendix

>>> **Table A1**. <<<

>>> **Table A2** <<<

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