Environmental Science Advances

ROYAL SOCIETY OF CHEMISTRY

PAPER

View Article Online
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Cite this: DOI: 10.1039/d5va00190k

Comparative analysis of systematic variations in life cycle analysis of lithium-ion batteries for automotive applications

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Lithium-ion batteries can have considerable environmental impacts due to the multiple emission-intensive materials and complex manufacturing processes involved in their production. These environmental impact hotspots need to be identified and managed to reduce emissions from battery manufacturing. Researchers have quantified these environmental impacts and emissions for various cathode chemistries in the past; however, the life cycle impacts reported in previous studies differ considerably. To analyze the reasons for these differences, a detailed comparative analysis of life-cycle studies of lithium-ion batteries over the last decade is conducted. The data inventory and results of these previous studies are compared with the state-of-the-art inventory and results from the current version of the R&D GREET® model. The analysis indicates that the primary reason for the divergence in life-cycle results across studies is the difference in approaches to sourcing secondary life-cycle inventory data, as well as the use of outdated or non-relevant inventory data. The non-standardized definition of battery assembly, a key energy- and emission-intensive step in lithium-ion battery production, is identified as another potential source of uncertainty and variation. The results indicate that controlling emissions from materials and processes used in cathode production can be a key factor in reducing the battery's greenhouse gas, particulate matter, sulfur oxide, and nitrogen oxide emissions. Battery assembly is also identified as a key source of greenhouse gas emissions with clear opportunities for reduction, while nickel sulfate is identified as the primary cause of sulfur oxide emissions in nickel-based cathode chemistries.

Received 27th June 2025
Accepted 29th May 2026

DOI: 10.1039/d5va00190k

rsc.li/esadvances

Environmental significance

Life cycle emissions from lithium-ion batteries are a key consideration when evaluating electric alternatives to current gasoline-powered vehicles. However, life cycle analysis (LCA) studies on lithium-ion batteries over the past decade have shown significant variability in greenhouse gas emission results. This variability can create substantial uncertainty in comparing electric and conventional mobility solutions. To address this, we provide a comprehensive investigation into lithium-ion battery LCA literature from the last decade, identifying the causes of these discrepancies and exploring ways to reconcile them. Additionally, we present the latest findings on life cycle energy demand, greenhouse gas emissions, and sulfur and nitrogen oxide emissions, highlighting the challenges and opportunities for reducing these emissions.

1. Introduction

Lithium-ion batteries (LIBs) have experienced a significant increase in demand over the past decade, primarily driven by their use in automotive and stationary energy storage applications.[1,2] This rising LIB demand is the outcome of several factors, including their high energy density and round-trip efficiency, long service life (5–15 years),[3,4] and their ability to reduce both local (tailpipe) emissions and global (greenhouse gas or GHG) emissions. However, LIBs also involve substantial use of critical materials, the production of which is often greenhouse gas (GHG) intensive, in addition to the considerable energy required to manufacture and assemble LIB components.[5,6] The upstream emission impacts of these stages, *i.e.*, LIB material extraction and processing, battery component manufacturing, and battery pack assembly, must be tracked to ensure comprehensive and robust accounting.

Multiple life-cycle analysis (LCA) studies have been published over the last decade, seeking to quantify the life-cycle emissions associated with producing LIBs with different cathode chemistries.[5–10] Additionally, multiple LCA models and inventories, such as GaBi,[11] SimaPro,[12] and R&D GREET®[13] (Research and Development Greenhouse gases, Regulated Emissions, and Energy use in Technologies), are now available to quantify the life cycle emissions of LIBs. However, LIB life-cycle emissions reported in different studies vary considerably for the same battery chemistry, especially due to significant

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differences in LCA assumptions and methodologies.[14] This makes it challenging to compare these life-cycle emission results across different papers, with only a few studies having undertaken such comparisons to date.[14–17]

In this study, we provide novel insights into how the LCA of LIBs has evolved over the past decade and how these updates affect their key life-cycle environmental burdens. We conduct a detailed comparative LCA that benchmarks GHG emissions for six prevalent lithium-ion chemistries (only with graphite anodes) in the latest R&D GREET release (R&D GREET 2024) against results reported over the past decade (2015–2024). The R&D GREET model is selected as the benchmark because it annually updates the bill of materials for LIBs with different cathode chemistries based on the latest technological advancements. By systematically harmonizing the functional unit and tracing the data lineage of each study's LCI, we identify the principal sources of divergence among the published results. We further analyze how updates in the R&D GREET LCI shift inventory flows and emissions, thereby clarifying the origins of variance and subsequently reconciling discrepancies with the broader literature. Together, these analyses deliver the state-of-the-art cradle-to-gate GHG intensities for six prominent LIB chemistries in R&D GREET 2024 and provide an evidence-based explanation for why published LCA results have a high variance and eventually suggest ways to reduce this variance.

2. Previous LCA studies

LIB pack manufacturing is an intricate process that involves complex procedures (ranging from mixing to final pack sealing) and an extensive array of components (from electrodes to the battery management system). LCA researchers have devoted considerable efforts to quantifying the life cycle energy use and environmental impacts of LIBs. Peters *et al.*[18] summarizes 36 LCA studies on automotive LIBs over 2000–2016, while Zhao *et al.*[19] identifies 76 such publications over 2005–2020. However, only a small subset of these studies has contributed novel LCI data for LIB pack production. The LCIs from most studies can be attributed to three categories, as noted in previous literature reviews on the LCA of LIBs.[5,20,21] The first category includes seminal papers on this topic, such as Notter *et al.*,[22] Majeau-Bettez *et al.*,[8] Dunn *et al.*,[7] Ellingsen *et al.*,[9] and Kim *et al.*[10] Most of these studies were conducted over 10 years ago and may no longer reflect the present-day LIB pack inventory. The second inventory resource category includes LCA databases, both commercial (*e.g.*, Ecoinvent) and open-source (*e.g.*, Argonne National Laboratory's R&D GREET). Most studies published over the past few years rely on this category as they contain more recent, up-to-date data on the bill of materials and energy inputs for LIB pack manufacturing. The third category comprises experimental and/or industrial data, which sometimes informs the first and second categories of inventory resources.

Our literature review across ScienceDirect, Google Scholar, and Scopus yielded 52 LCA publications on LIBs published between 2015 and 2024, restricting the search to six prevalent lithium-ion chemistries with graphite anodes only. Table S1 in the SI includes all the studies considered in this review, while Fig. 1 illustrates the life-cycle GHG emissions of the battery pack from these studies on a per-kilowatt-hour (kWh) basis. The GHG emissions per kWh from the 52 studies is further divided based on the year of publication and cathode chemistry in Fig. 1 which indicates that higher-nickel-content NMC chemistries like NMC532, NMC622, and NMC811 have been the subject of lesser LCA research as compared to older and more widely adopted chemistries like NMC111 and LFP.

We consider only GHG emissions for inter-study comparison in this article, as the typical objective of advancing LIBs is to

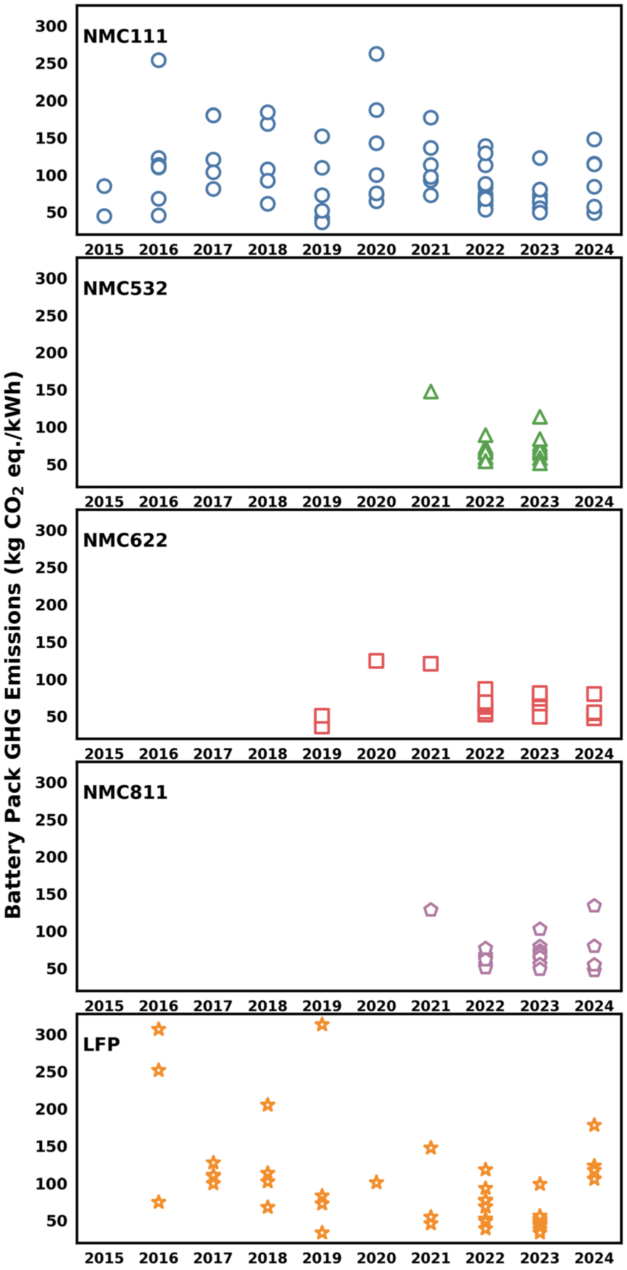

Fig. 1 Greenhouse gas emissions per kWh for different battery chemistries over the years, as reported in various battery life cycle analysis (LCA) literature over the last decade.

reduce these emissions across different technologies. Various functional units are used across these studies, including per kWh of battery pack, per kWh of battery cell, per kg of battery pack, per km of vehicle range, and per battery pack. To ensure consistency when comparing GHG results across studies, we convert all results to a single functional unit of per kWh of battery pack.

Data from each study used to perform this conversion into a common functional unit are provided in the Excel sheet included in the SI. To address potential inconsistencies in these conversions, we prioritize study-reported conversion parameters wherever available (*e.g.*, pack energy capacity, pack mass, nominal capacity, cell count, vehicle battery size, *etc.*). When a required parameter is not explicitly reported, we apply a transparent assumption hierarchy: first using values provided in the main text or SI, then values implied by other reported specifications within the same study, and only if needed, using a clearly flagged proxy based on closely matched chemistry. Importantly, we do not impose a uniform battery pack boundary (*e.g.*, adding BMS or cooling system) across studies when these are not reported, as doing so would require speculative adjustments. Instead, such boundaries are documented in the supplementary Excel file so that researchers can trace the basis of each datapoint in the future and interpret any remaining scatter in the literature comparison accordingly.

Battery packs studied in the reviewed literature (Fig. 1) exhibit a wide range of GHG emission impacts. Notably, nickel-manganese-cobalt (NMC) LIBs show a decrease in life-cycle GHG impacts with increased nickel (Ni) content, due to higher energy density (tabulated in Section 3 of this paper). Most of the reviewed LCA studies identify cathode and aluminum as prominent contributors to the GHG impacts of LIB packs. The GHG impacts of lithium iron phosphate (LFP) batteries are generally lower than those of NMC batteries in most of the reviewed studies, due to the absence of GHG-intensive critical materials, such as nickel sulfate ($NiSO_4$), and cobalt sulfate ($CoSO_4$). Such critical materials are also prone to supply chain risks because their extraction and refining processes are concentrated in limited geographic locations worldwide.[23,24]

The complete list of studies, along with the chemistry for which each study was used, the study region, and the reference year for LCA calculations, is given in Table S1 of the SI. Over the past decade, LIB LCAs have generally shifted from assuming high and uncertain cell-manufacturing energy demand to lower and more empirically grounded values, with partial convergence as more studies draw on similar industrial data sources and process models. However, the results still diverge because the authors include different unit processes (such as dry-room operation and solvent recovery) and use different regional electricity mixes and years. In parallel, the concern for key burdens has moved from manufacturing energy toward materials, which makes the evolution of background databases more consequential: successive Ecoinvent, GaBi, and GREET releases have updated electricity grids, regionalized mining and refining routes, changed co-product allocation and recycling modeling, and introduced more specific datasets for battery-relevant chemicals and precursors. Those changes can shift the impacts of the supply chains of Ni, Co, Al, and graphite. Differences between databases still exist because their modeling conventions remain distinct. Finally, inventory assumptions at the foreground level have evolved toward higher cell energy density and more detailed bills of materials, which tend to lower impacts per kWh of rated capacity while simultaneously increasing sensitivity to chemistry-specific material choices (*e.g.*, higher-Ni cathodes). Overall, these trends we observed motivate us to conduct this study, which can transparently update LIB manufacturing boundaries, performance, and bill of materials (BOMs). It can also show the impact difference caused by the choice of data source.

Fig. 2 shows the interconnections among the data sources for the reviewed publications, with bubble sizes and the height of the corresponding bars indicating the number of citations among the 52 battery LCA publications investigated in this study (list provided in Table S1 of the SI). Therefore, Fig. 2 essentially represents how each seemingly standalone battery inventory source is itself sourced from different inventories, models, or publications. Since some data sources cite multiple other data sources in their inventories, some bubbles are placed at the heads of multiple arrows.

Fig. 2 indicates that the GREET model is the single-most frequently cited source among the reviewed studies, followed by Ecoinvent, both of which were cited more than any single peer-reviewed study. Most seminal studies, such as Notter *et al.*,[22] Majeau-Bettez *et al.*,[8] and Dunn *et al.*[7], have been cited less frequently in recent years. However, some influential older studies, such as Ellingsen *et al.*,[9] continue to be cited in recently published papers as they provide a complete industrial-scale LCI for LIB manufacturing. As a group, LCA databases such as GREET and Ecoinvent are cited more frequently than the combined citations across all independent peer-reviewed literature. Therefore, while LCA researchers are likely to draw on independent literature for part of their battery inventory or BOMs, most of their sources come from popular LCA databases such as R&D GREET and Ecoinvent.

Fig. 1 illustrates substantial discrepancies in reported life cycle GHG results across different battery chemistries, driven by several key factors beyond differences in the electricity grid, supply chain, and location. The differences in GHG results across studies, as shown in Fig. 1, are more than artifacts of varied LCA assumptions, as evidenced by considerable intra-chemistry variation that exceeds the variation possible due to grid mixes or material supply chain differences among studies, as discussed later in Sections 4 and 5 of this paper.

The first source of discrepancies across studies is the mismatch between the described LCA methodology and the inventory data used. Among the reviewed literature, some studies employ a "bottom-up" LCA approach, using process modeling or secondary data from industries to account for the inputs and outputs of materials and energy for a battery cell or pack. On the other hand, some key studies adopt the "top-down" LCA approach, which begins by estimating the total impacts of a facility or region and then allocating them to different products and materials.

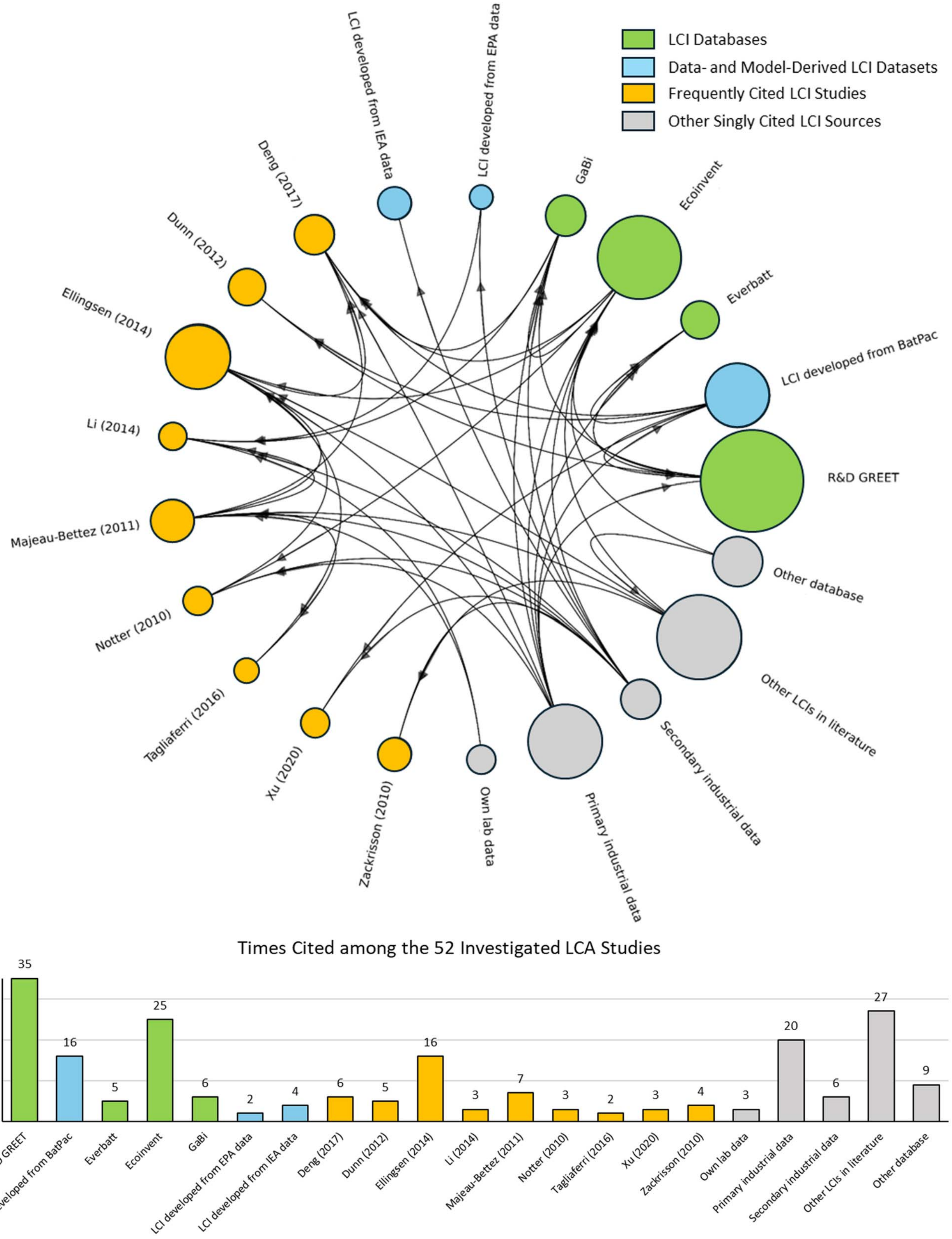

Fig. 2 The web diagram shows the interconnection among the different life cycle inventory sources of the 52 investigated studies over the last decade. The graph below the web diagram shows how often these inventory sources were cited across the 52 studies.

Both these approaches are relevant and useful within their respective contexts;[25] however, using LCI data from a top-down approach from one study to obtain LCA results using a bottom-up methodology in another paper may lead to considerable discrepancies and variances in results across literature. For instance, Wu *et al.*[26] and Hao *et al.*[27] use a bottom-up LCA approach and report considerably higher emissions for LFP battery packs because they use parts of LCI from Majeau-Bettez *et al.*,[8] which uses a top-down approach. Conversely, studies such as Shu *et al.*,[28] Quan *et al.*,[29] Jiang *et al.*,[30] and Llamas-Orozco *et al.*[31] report considerably lower GHG emissions for LFP LIBs as they follow a uniform bottom-up approach to data collection. This explains the high variance in GHG impacts of LFP batteries across various studies, as shown in Fig. 1.

Secondly, discrepancies also arise from reliance on outdated inventory data. Many studies likely miscalculate life-cycle burdens by relying on legacy datasets that may no longer be relevant or representative of recent battery manufacturing practices. Based on our literature review, it is evident that many LCIs continue to rely on emission impact data derived from older seminal studies, some of which are based on laboratory-scale experimentation.

Our conclusion regarding the use of outdated inventory data is also supported by the recent literature reviews. Following a detailed systematic review from 1996–2025, Bassani *et al.*[32] conclude that the latest study to present an original inventory or bill of materials for NMC chemistry was published in 2019. Since then, most studies have used at least part of their battery inventory data from older secondary studies or inventory databases. Similarly, Song *et al.*[33] indicate that no more than seven studies in the past 24 years have provided novel battery LCA inventories. Therefore, even the most recent studies base their life cycle results on only a handful of older seminal studies.

While these seminal studies have played a vital role in informing our understanding of battery impacts, they do not account for the technological advancements and process changes in industrial-level pack production over the past half-decade. As a result, the life-cycle emissions data from such studies may not accurately reflect current operational practices or the scale of LIB manufacturing, leading to potential discrepancies when compared with more recent, industry-specific data. This highlights the importance of updating emission factors, methodologies, and BOMs to better reflect modern industrial processes and practices, thereby enabling more accurate characterization of the life-cycle environmental performance of LIBs.

The third potential reason for the discrepancy among current studies is the inconsistent and ambiguously defined scope of the term "battery assembly" across studies, despite its significant contribution to energy demand during battery manufacturing. Most studies use terms like cell assembly[5,34–37] or cell production[38–40] to refer to a wide range of processes, including slurry mixing and coating, electrode drying, calendaring, notching and stacking, cell welding, electrolyte filling and wetting, and dry room conditioning processes. On the other hand, other studies refer to the same set of processes as battery assembly[31] or final assembly[41] (Definition 1 in Fig. 3).

Interestingly, some well-cited studies like Ellingsen *et al.*[9] use the term battery assembly to refer to a different set of processes, which include mounting cells onto a module base, connecting busbars, attaching battery management and cooling systems, and combining modules to make LIB packs (Definition 2 in Fig. 3). The energy required for this battery assembly is small, as assumed per Ellingsen *et al.*[9] and is manually operated; an assumption also shared by many other studies. In such literature, the high energy requirement for cell production is not ignored; it is referred to as cell manufacturing. Some studies use battery assembly to specifically refer to combining modules to make LIB packs (Definition 3 in Fig. 3). To add to the complexity, some studies, like Syre *et al.*[42] or Drachenfels *et al.*,[43] either include all the energy used for cell manufacturing into upstream materials or further disaggregate the energy consumption for cell manufacturing into sub-steps such as electrode production, cell assembly, and finishing (Definition 4 in Fig. 3). Therefore, our review suggests that cell and battery assembly terminology in battery LCA literature should be standardized to prevent miscalculation of the impacts of this environmentally impactful step in LIB manufacturing.

The inconsistent and ambiguous use of the term battery assembly complicates energy attribution, cross-study comparability, and interpretation of results even if all the energy is ultimately accounted for. As explained above, the same term is used to describe fundamentally different process sets, as shown in Fig. 3, ranging from electrode-to-cell finishing steps to low-energy module and pack integration activities.

This ambiguity does not change the total energy demand estimates in principle, but affects how energy is allocated across system components, and eventually compared across studies. Practically, this can lead to omissions or double counting of energy-intensive step, obscure the dominant contributors to manufacturing emissions, and hinder meaningful comparisons across chemistries, and manufacturing locations. Since electrode and cell finishing processes are consistently identified as the most energy- and emissions-intensive stages of battery manufacturing, a clear and consistent terminology is necessary to avoid misinterpretation and support transparent benchmarking and policy-relevant analysis.

With a clear understanding of the three key potential drivers of discrepancy in previous LCA studies, we define the assumptions and the LCA methodology employed in this study, and present the life-cycle impact results for different battery chemistries as provided by the R&D GREET 2024 model. Subsequently, we compare these results with the corresponding emission impacts reported in the literature to understand the effects of those drivers.

3. Methods

The R&D GREET model, developed at Argonne National Laboratory, provides a detailed LCI of LIB (cathode) chemistries, based on data from peer-reviewed literature and industry sources. The chemistries provided in the latest version of the R&D GREET model include lithium manganese oxide (LMO), nickel cobalt aluminum (NCA), lithium iron phosphate (LFP) processed *via* hydrothermal and/or solid-state routes, and five variations of nickel-manganese-cobalt (NMC) chemistries – namely, NMC111, NMC532, NMC622, NMC811, and NMC95.

For this study, we focus on only six of the nine chemistries that are either currently widely used or are likely to be used in the future based on market projections:[44] NMC111, NMC532,

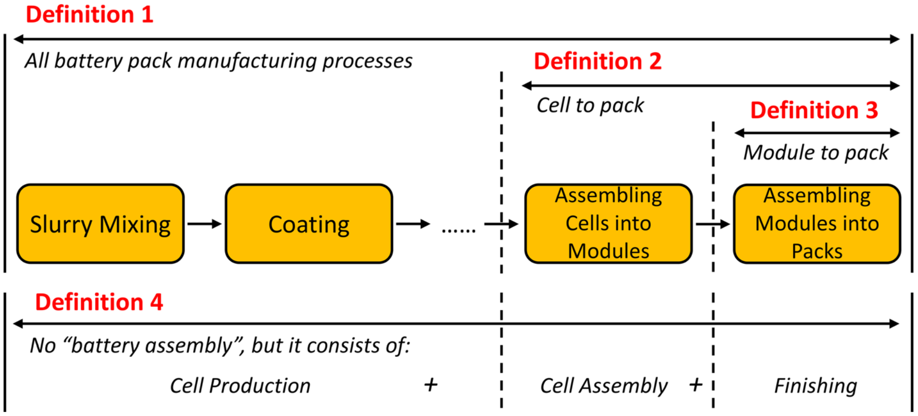

Fig. 3 Battery assembly definitions in the investigated studies.

NMC622, NMC811, NMC95, and LFP. All these cathode chemistries are assumed to be produced *via* solid-state processing. Solid-state processing refers to the process of mixing lithium salts with transition metals, followed by ball-milling and mixing, before undergoing high-temperature sintering.[45]

We obtained the BOMs for LIB packs based on these chemistries, which contain prismatic cells designed for a 300-mile-range U.S. vehicle, as provided by Argonne's Battery Performance and Cost (BatPaC) model.[46] These BOMs are provided in Table S2 of the SI. Table 1 presents the specific energy of the various battery chemistries examined in this study.

Since battery material supply chains play an influential role in the life-cycle impacts of LIBs, we briefly describe the supply chains of different LIB constituents in the R&D GREET model. All cathode chemistries utilize one of two battery-grade lithium chemicals (Li-chemicals) to produce cathode precursors and/or cathodes: lithium carbonate (Li_2CO_3) and lithium hydroxide monohydrate (LiOH H_2O). R&D GREET assumes that Li_2CO_3 is used for cathode production in four LIB chemistries (NMC111, NMC532, NMC622, and LFP), while LiOH.H_2O is used for the same purpose in NMC811 and NMC95 batteries. According to the R&D GREET model, 55% of global Li_2CO_3 supply and 80% of global LiOH.H_2O supply originate from spodumene ores, with the remaining share coming from Salar brines.[47] NMC cathodes also use nickel sulfate ($NiSO_4$) and cobalt sulfate ($CoSO_4$) to produce cathode precursors. 40% of $NiSO_4$ supply is assumed to come from Class I nickel (Ni) produced from sulfidic ores,[45] while the remaining 60% is produced from mixed hydroxide precipitate (MHP) obtained through processing of lateritic Ni ores.[48,49] The $CoSO_4$ inventory is representative of cobalt mining in the Democratic Republic of Congo, and is shipped to China for processing into battery-grade $CoSO_4$.[48] We assume the cathode active material is manufactured within the United States.[48] The aluminum (Al) supply chain is based on the North American consumption mix, comprising 50.4% of aluminum sheet production from primary Al and the remaining 49.6% from secondary Al sources (including internal scrap).[50] The synthetic graphite anode is produced *via* calcination, carbonization, and graphitization of green petroleum coke obtained through petroleum refining.[51] Lithium hexafluorophosphate ($LiPF_6$) is the electrolyte used along with dimethyl carbonate and ethylene carbonate.[48] Finally, battery assembly is assumed to occur in the United States, primarily using natural gas (82%) as an energy source, with the remainder supplied by electricity.[48] It should be noted that the "battery assembly" process in this article includes mixing the cathode and graphite anode active material with a binder in a solvent, coating the current collectors, cutting electrodes to the appropriate size, adding electrolyte to the cell in the dry room, cell conditioning processes, and eventually assembling into packs.

For this LCA study, the system boundary covers the cradle-to-gate of LIB production, with the functional unit being 1 kWh of battery energy capacity. The system boundary includes all processes associated with material extraction from mines/ores, processing, transportation, LIB component manufacturing, and final assembly to produce LIB packs. We consider four life cycle impacts/emissions in this article: (a) primary energy use; (b) GHGs; (c) sulfur oxide (SO_x); and (d) nitrogen oxide (NO_x).

4. Results

Fig. 4 shows the total life-cycle energy consumption for LIB manufacturing across different battery chemistries based on the R&D GREET 2024 model, with dominant contributions from cathode-related materials and processes, marked in different shades of blue, as well as Li_2CO_3 and LiOH.H_2O shown in yellow and orange, respectively. While cathodes account for about half of the life-cycle energy consumption for NMC LIBs, they

Table 1 Specific energy of the different battery chemistries referenced in this study

Battery chemistry	NMC111	NMC532	NMC622	NMC811	NMC95	LFP
Specific energy (Wh kg^{-1})	177.3	185.8	187.0	197.7	204.6	142.9

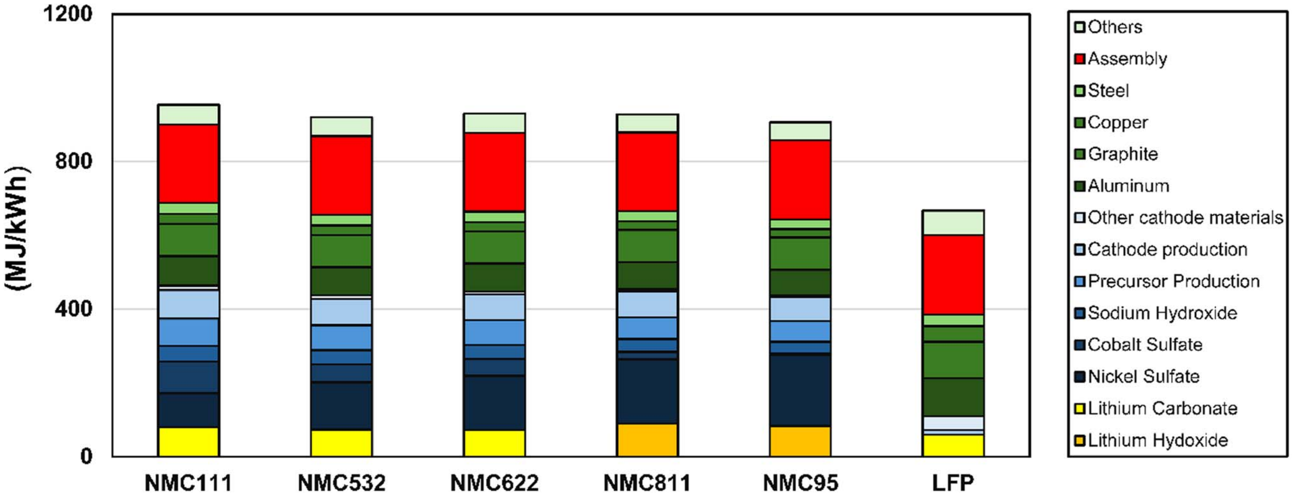

Fig. 4 Life cycle energy demand in megajoules to make 1 kWh of different Li-ion battery chemistries, broken down into key materials and processes. The yellow/orange parts, clubbed with the blue parts, represent the life-cycle energy requirements of the cathode materials and processes, while the green-shaded parts represent the non-cathode materials and processes. The energy required for the battery assembly process is shown separately in red.

contribute only 16% to the corresponding energy consumption for LFP LIBs. This divergence arises from the lack of energy-intensive materials, such as $NiSO_4$ and $CoSO_4$, in LFP cathode production.

Apart from individual cathode constituents and related processes, the battery assembly process (marked in red) contributes considerably to the life-cycle energy demand of LIB packs across all battery chemistries (22–24% for all NMC LIBs, depending on nickel content, and 32% for LFP LIB; Fig. 4). The primary reason is the high energy requirements of the dry room, where the required moisture content must be less than 100 ppm by volume to prevent the electrolyte from reacting with water.[5,36] Additionally, considerable energy is required to introduce hot air into the drying environment to recover *N*-methyl-2-pyrrolidine (NMP) from the electrode slurry while keeping the NMP vapor concentration below its flammability limit.[5] It should be noted that materials that collectively contribute <5% of the total energy (or subsequent emission impacts) are classified under the "Others" category in Fig. 4 (as well as other subsequent figures) for simplicity. The materials under the "Others" category include the battery management system, binder, electrolytes, plastics, rubber, and insulation.

For clear demarcation, all the subsequent figures classify the battery environmental impacts or emissions into four broad categories: (a) impacts from non-lithium cathode materials and associated processes, which are marked in different shades of blue (b) lithium chemical (Li_2CO_3 and $LiOH.H_2O$) impacts in shades of yellow/orange; (c) impacts from non-cathode materials and associated processes, which are marked in different shades of green; and (d) impacts of battery assembly process marked in red.

Battery assembly is classified as a separate process from cathode and non-cathode materials and processes because it extends to the entire battery, including both cathode and non-cathode components. Another reason for this demarcation is that, unlike battery assembly, all other materials and processing steps may occur across different geographical locations and contain varied activities that are difficult to control by a single entity.[52] On the other hand, the battery assembly process usually and increasingly occurs in nearby geographical locations and is controlled by a single entity or closely related entities.[53,54] Additionally, the energy required for the battery assembly process primarily includes electricity and/or natural gas for electrode drying and the dry room, which are relatively easier to control and modify than the energy requirements of upstream material processing.

Fig. 5 shows the cradle-to-gate GHG emissions (kg CO_2-eq. per kWh) for all the considered LIB chemistries. Cathode materials and processes (marked in shades of blue, along with lithium chemicals in yellow/orange) account for approximately half of the life-cycle GHG emissions for NMC-based LIBs. However, the contribution of cathode materials and process is smaller (19%) in LFP LIBs compared to NMC-based LIBs because of the absence of GHG-intensive materials like nickel and cobalt sulfate.

Battery assembly accounts for about 20% of the GHG emissions for NMC-based LIBs and about 30% for the LFP LIBs. The significance of battery assembly processes in the life-cycle GHG emissions of LIBs across all battery chemistries further underscores the need to define the term "battery assembly" uniformly across the battery LCA literature, as previously identified in Section 2 above.

With the quantification of the effect of each constituent on overall GHG emissions per kWh, it is easier to see that the extent of variation in GHG emissions among a single chemistry (shown in Fig. 1) cannot be solely due to differences in electricity grid mix and material supply chain. Therefore, the breakdown of the GHG results in Fig. 5 further supports multiple factors, as outlined in Section 2, as the cause of GHG

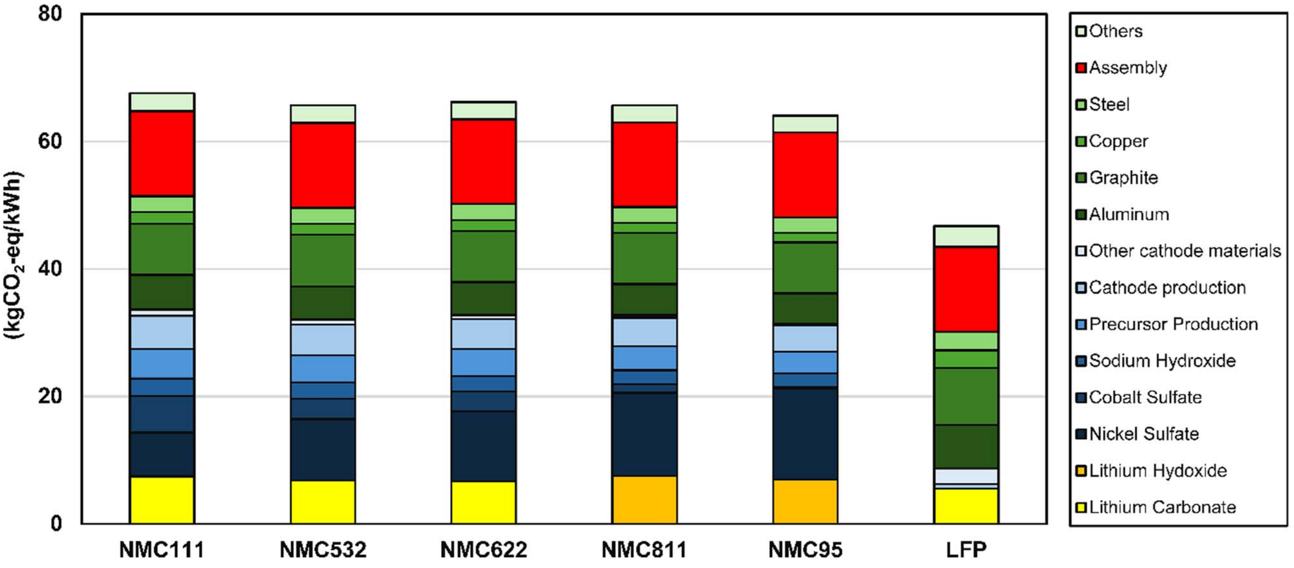

Fig. 5 Life cycle greenhouse gas emissions to make 1 kWh of different Li-ion battery chemistries are broken down into key materials and processes. The yellow/orange, along with the blue parts, represent the emissions due to cathode materials and processes, while the green parts represent the emissions from non-cathode materials and processes. The emissions from the battery assembly process are shown separately in red.

variation within a single chemistry, beyond simple locational assumptions.

Among non-cathode materials, graphite and aluminum accounted for 12% and 8% of the battery's GHG emissions, respectively. Most graphite GHG emissions come from energy-intensive processes of calcination, carbonization, and graphitization, which are in turn highly dependent (∼40%) on electricity emissions. Similarly, most aluminum-related emissions stem from the electricity required in smelters to produce primary aluminum and, to a lesser extent, from melting aluminum scrap. Therefore, switching to renewable electricity for graphite and aluminum production and processing can considerably reduce battery GHG emissions.

Apart from GHG emissions, we also estimate life-cycle SO_x emissions per kWh for all LIB chemistries, as these emissions contribute to environmental issues such as acid rain and ocean acidification, as well as public health issues like cardiovascular and pulmonary diseases.[55] Additionally, SO_x emissions cause disproportionately more severe environmental damage in locations where $NiSO_4$ is refined, leading to localized environmental concerns.[56] Fig. 6 shows the life cycle SO_x emissions per kWh battery for all LIB chemistries, indicating that NMC LIBs have considerably higher SO_x emissions than LFP LIBs. Further, $NiSO_4$ contributes the most (∼80–95%) to SO_x emission impacts for NMC LIBs, with the exact contribution determined by the specific NMC chemistry. These contributions are explained by the use of sulfur and sulfuric acid (H_2SO_4) to process Ni ore into $NiSO_4$. These heavy emissions may be controllable if captured and used to produce the required sulfuric acid, as done in Congo's cobalt sulfate manufacturing process.[57]

Fig. 7 shows the NO_x emissions per kWh for different LIB chemistries. As shown, NMC LIBs have higher life-cycle NO_x emissions than LFP LIBs (by ∼35–45%). These emissions stem from the use of heavy-fuel oil-based onsite electricity generation for Ni ore beneficiation, en route to producing $NiSO_4$ used in NMC cathodes.[58] The second-highest single contributor to these emissions is battery assembly, accounting for ∼18% of the NMC batteries' NO_x emissions. Conversely, diammonium phosphate, which is used as the phosphorus source to produce LFP cathodes and battery assembly process, accounts for ∼25% each of the life cycle NO_x emissions for LFP LIBs (Fig. 7).[59] NO_x emissions in battery assembly come from the assumed use of natural gas-based energy for battery assembly processes.[48] On the other hand, transporting diammonium phosphate's precursors using ocean tankers over large distances also contributes to high NO_x emissions in LFP LIBs.[60] The use of NO_x reduction technologies like exhaust gas recirculation systems, emission control systems, and scrubbers in ocean tankers can reduce these transport-based NO_x emissions associated with LFP LIBs.[61]

4.1. Life cycle GHG comparison with other studies

GHG emissions are the most-studied life cycle impact category across the battery LCA studies, as shown in our literature review and previously published review studies, such as Arshad *et al.*[15] Additionally, life cycle energy demand and GHG emissions are typically positively correlated in LCA studies, since high energy demand often involves substantial use of fossil fuels, resulting in higher GHG emissions. Hence, we use GHG emissions from this study as a point of comparison with other battery LCA studies.

Fig. 8 compares the GHG emissions from R&D GREET 2024 with those from the studies reviewed in Section 2, indicating that the GHG emissions estimates from R&D GREET 2024 are lower than most of the previous studies, especially for chemistries with more number of published legacy LCA papers *e.g.*

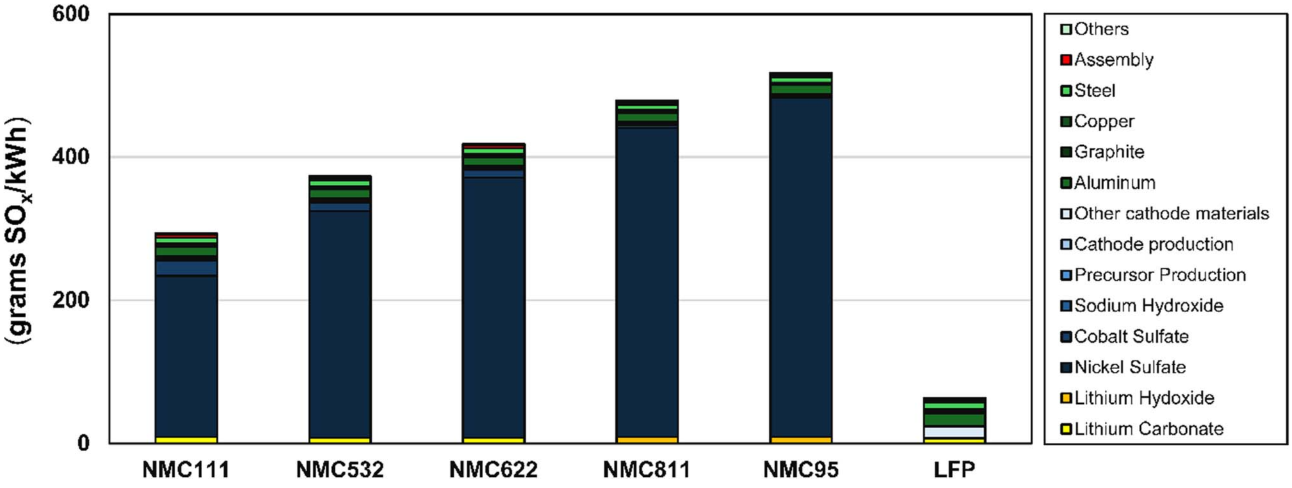

Fig. 6 Life cycle oxides of sulfur oxide (SO_x) emissions in grams to make 1 kWh of different Li-ion battery chemistries, broken down into key materials and processes. The yellow/orange along with the blue parts represent the emissions due to cathode materials and processes, while the green parts represent the non-cathode materials and processes. The emissions from the battery assembly process are shown in red.

NMC111 and LFP. A small portion of this deviation stems from the differences in energy consumption for battery assembly processes, as well as the chemical processes and energy sources used to extract and refine battery materials. A larger share of these deviations in GHG estimates is primarily due to the increased energy density of latest batteries in addition to the top-down or hybrid (top-down + bottom-up) emission estimation approach used in other literature *versus* a completely bottom-up approach adopted in this study, as discussed previously in Section 2. It is important that efforts to characterize current LIB LCA utilize models, methods, and findings that incorporate the latest understanding of battery materials and manufacturing processes; relying on older studies can lead to results that may not align with current practice.

The data points for different chemistries indicate that more LCA research has been conducted on chemistries like NMC111 and LFP compared to other chemistries, such as NMC532, NMC622, and NMC811, over the last decade. The review echoes the prominence of earlier scientific research and commercial adoption of NMC111 [62] along with a recent market interest in LFP LIBs.[63]

Apart from NMC811, the number of GHG emission data points across all chemistries from previous literature is sufficient to obtain a reliable estimate of the coefficient of variance, which measures the spread or variation in the data.[64] The coefficient of variance of the LFP chemistry was highest (0.623) among the studied chemistries, indicating a wide variability in results from the previous literature, as shown in Fig. 8. This moderate-to-high variance signals a correspondingly moderate-

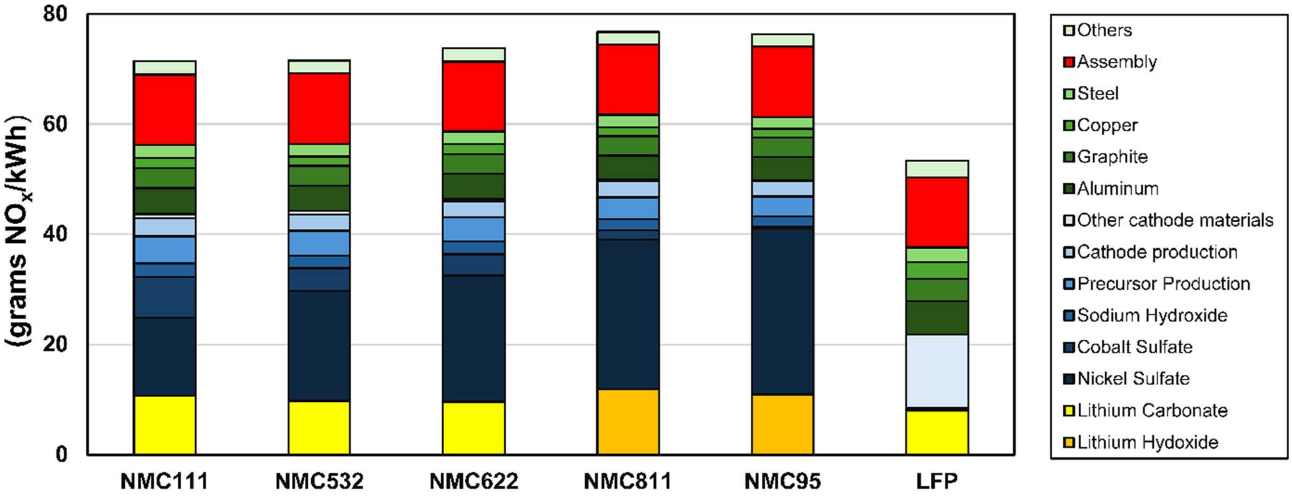

Fig. 7 Life cycle oxides of nitrogen (NO_x) emissions in grams to make 1 kWh of different Li-ion battery chemistries broken down into key materials and processes. The yellow/orange along with the blue parts represent the emissions due to cathode materials and processes, while the green parts represent the non-cathode materials and processes. The emissions from the battery assembly process are shown separately in red.

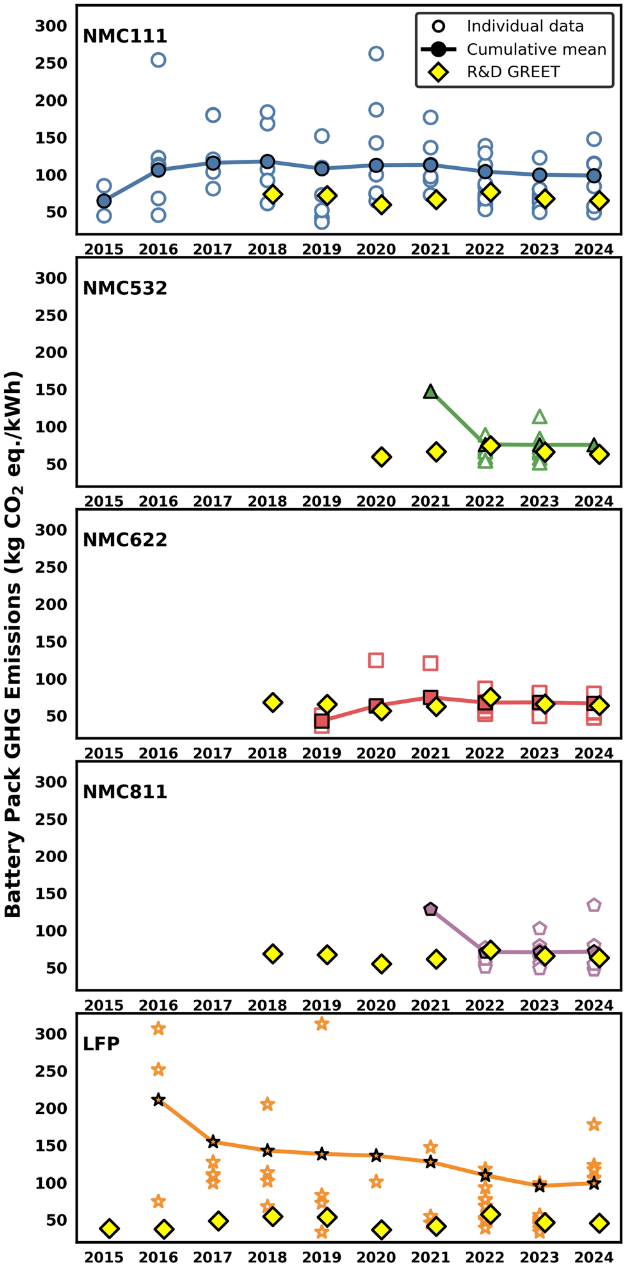

Fig. 8 The GHG emissions per kWh for various battery chemistries, as reported in R&D GREET 2024, are compared with emissions provided in other reviewed literature. The line in each graph represents the cumulative average of the GHG emissions of a battery chemistry for that year and the preceding years.

to-high uncertainty in the GHG emissions estimate for LFP LIBs. Therefore, more standardized LCA research on LFPs is needed, particularly given their growing market presence in automotive applications. The lack of sufficient data points for the NMC811 cathode chemistry suggests that additional LCA research is needed for this and other high-nickel cathode chemistries.

The connected dots in Fig. 8 represent the cumulative average GHG emissions per kWh for each chemistry, calculated from all studies published up to the preceding year. For example, the 2018 data point reflects the cumulative GHG emissions based on studies from 2015 to 2018. As indicated in the figure, the NMC111 graph shows a sharp increase in cumulative average values from 2015 to 2018, suggesting an LCA methodological artifact of incomplete inventory data or system boundaries in the early years, while later studies likely adopted more comprehensive LCA scopes and presented results based on updated inventory data. From 2018 onwards, the cumulative average somewhat stabilizes, suggesting convergence of the literature towards common values, while the high inter-year spread of data reflects persistent heterogeneity in assumed electricity grid mixes, system boundaries, and manufacturing locations.

High-nickel-content cathode chemistries like NMC532, NMC622, and NMC811 start from very different initial estimates than the later average-out estimates, because the average values are extremely sensitive to individual data points owing to the smaller sample size. This trend strongly indicates that more LCA studies are needed for these chemistries to provide more reliable estimates of GHG emissions across different grid mixes, co-product allocation, and regional factors.

For LFP chemistry, the cumulative average starts much higher than the NMC initial data points but then declines steeply and rapidly. It should be noted that LFP and NMC111 converge on a similar GHG intensity per kWh despite LFP's lower energy density. This decline reflects a combination of the rapid scale-up of LFP manufacturing,[65] improvements in LFP energy density over time,[65] the progressive decarbonization of electricity grids, especially in China, and the maturing of LCA methodology, which has led to more consistent system boundaries. The GHG emissions per kWh for NMC111 and LFP chemistries were in similar ranges, with high inter-study variation, indicating that manufacturing conditions, location, and LCA assumptions are likely more important factors than the choice of chemistry when selecting which of these two batteries will have lower GHG emissions per kWh.

4.2. Life cycle GHG variation over time within the GREET model

Battery life cycle GHG emissions can also evolve over time due to constantly changing battery material supply chain and evolving BOM, in addition to the changes in LCA methodology and data sources as described in the previous sections. Therefore, the changes shown in Fig. 8 may reflect inherent variation due to supply chain evolution, and it may be inaccurate to attribute the entire variation to misused or outdated life cycle inventory data.

To qualitatively isolate GHG variation due to such supply chain factors, we focus on one chemistry (NMC111) and carry out a decomposition analysis to identify the variation in the contribution of key materials/processes to per kWh GHG emissions over six years (2018–2024) using the same data source (GREET model). The reason for using 2018 as the baseline for comparison with 2024 is that the NMC111 chemistry was initially introduced in the GREET model in 2018 and has since been used as a benchmark by multiple peer-reviewed studies, as evidenced by the number of citations since its publication in 2019.[5] It should be noted that the name "R&D GREET" instead of "GREET" was adopted for the standard GREET model in the year 2023, so we adopt the same naming convention in this article.

Fig. 9 shows key materials and processes' percentage contribution to 1 kWh of NMC111 battery as mentioned in GREET 2018 and R&D GREET 2024 based on the BOMs given in Table S3 of SI. Results indicate that percentage contribution of battery assembly process to 1 kWh battery remained almost similar across the years as opposed to other key materials like aluminum, graphite and lithium carbonate where considerable changes in the impact share were observed.

The overall battery GHG reduced by ∼12% per kWh despite considerable increase in the GHG intensity of materials like graphite and lithium carbonate. The graphite GHG intensity increased by 83% between GREET 2018 and R&D GREET 2024 due to availability of more industry relevant data.[51,66] Similarly, the GHG intensity of lithium carbonate increased by 3.7 times between 2018 and 2024 model versions due to a modeled shift in the supply chain from a 100% brine-based lithium to a 55% spodumene based lithium.[67]

On the other hand, the BOM used for LCA modeling changed significantly between 2018 and 2024 versions which was the primary driving force behind the reduced overall emissions. These changes reflect an increased energy density (20%), and considerable replacement of aluminum with steel and stainless steel within the battery pack (as shown in Table S3 of SI) informed by the prevalent industry practice.[46,68]

A detailed investigation of these variations across six years reveals two key insights. Firstly, data quality improvements often reveal variations in burdens for a material *e.g.* in case of graphite. Such variation in burden may create the perception that battery materials are becoming more GHG intensive with time but actually reflects an improvement in accounting of the process energy and material requirements.

Secondly, the technological improvements like replacing aluminum with steel, circuit miniaturization and system-on-chip integration have reduced the carbon intensities of pack production.

5. Discussion

As our results show, cathode-related constituents and processes have a significant influence on the various life cycle impacts of LIBs, be it life cycle energy use or various emissions (GHG, NO_x, and SO_x) (Fig. 4–7). Given the criticality of specific materials used in LIBs, such as lithium, nickel, cobalt, and graphite, there has been an increasing focus across multiple countries, including the United States, on developing options to secure robust supply chains of these materials from both virgin and recycled feedstocks. Apart from ensuring the reliability of battery material supply, these options can also significantly reduce the life-cycle impacts of these materials and, thereby, the final LIBs. We discuss two options here: (a) battery recycling, and (b) domestic critical material production.

Battery recycling is a logical decision from the perspective of life-cycle GHG reductions, as it can help lower the environmental impact from cathode-related materials and processes. However, the cost-effectiveness of such recycling may vary widely depending on the LIB chemistry and the level of final material that can be recovered (*i.e.*, elemental materials, chemical salts, or cathode materials themselves). For instance, NMC LIBs may offer greater recycling benefits than LFP LIBs, as they use supply-constrained critical materials such as nickel and cobalt, which can be recovered and reused in new batteries at the elemental or salt level, making recycling economically viable.[69,70] Conversely, the most valuable material that can be recovered industrially from LFP LIBs is lithium, but its recovery requires the use of considerable chemicals and energy through hydrometallurgical recycling.[71,72] Therefore, the economic viability of LFP recycling remains contested, and environmental

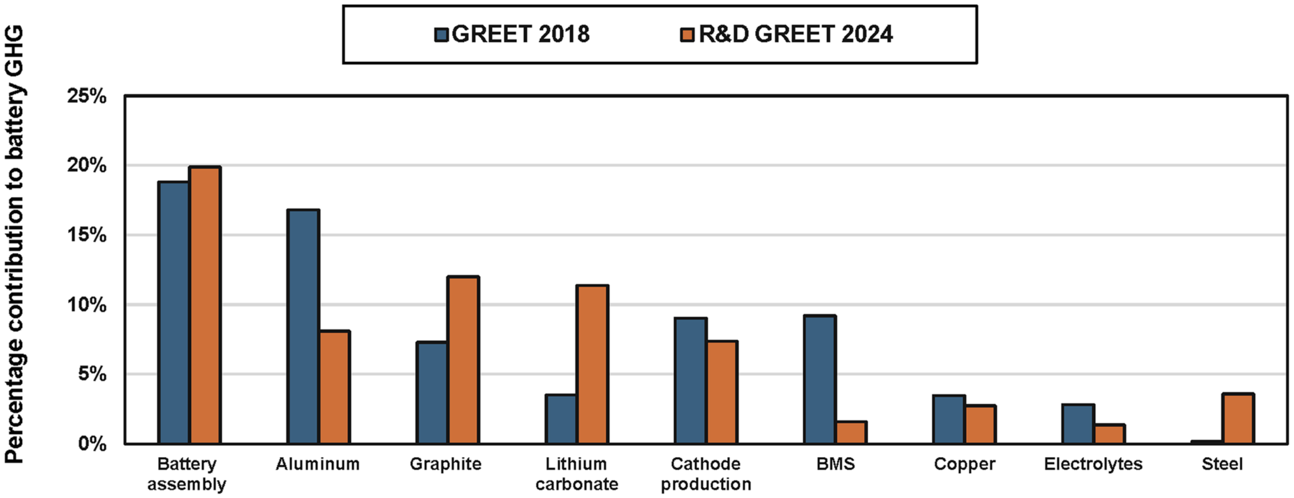

Fig. 9 The comparison of percentage contribution of key battery materials and processes to life cycle GHG emissions of 1 kWh of battery as per GREET 2018 and R&D GREET 2024.

benefits (if any) are highly route and context dependent to the point of being negative in case of conventional pyrometallurgical recycling approach.[73] A few recent studies show laboratory-scale feasibility of directly regenerating spent LFP cathode powder;[74,75] however, further research is necessary to scale the process to an industrial level, which is a prerequisite for an economically viable LFP recycling operation.[76]

Among the most GHG-intensive battery materials, lithium is also a critical material from a supply chain perspective, as nearly all lithium used in current batteries is produced outside the U.S.[38,77] Unlike other battery-related critical materials, the U.S. has historically been a significant producer of Li-chemicals, and it is endowed with sizable Li-containing clays and low Li-content brine resources that are being developed to mitigate the aforementioned supply chain issues. The literature indicates that, if successful, these projects will likely reduce unit GHG emissions from U.S.-made batteries, as Li-chemical production from these U.S.-based resources is expected to be less energy- and GHG-intensive than that from spodumene ore.[78]

Apart from battery material supply chains, the other major contributor to the life-cycle impacts of LIB packs is battery assembly (Fig. 4–7). Such prominent impact contributions stem from the large energy requirements of battery assembly processes and the considerable use of fossil fuels (natural gas) for these processes, as explained earlier. Multiple studies have focused on quantifying and reducing the impacts associated with the battery assembly process. For instance, Chordia *et al.*[79] show that battery production at scale in gigafactories is likely to be less energy- and emission-intensive per kWh than in small-scale factories, and can be further reduced through process optimization, such as recovering lost heat and recirculating air and heat in dry rooms. Studies have also investigated the impacts of using only electricity and a mix of electricity and natural gas for battery assembly,[80] in addition to varying the emission intensity of the assumed grid electricity.[80–82] All these studies conclude that overall battery environmental impacts can be considerably reduced by using more renewable-based electricity.

On a comparative basis, LFP LIBs exhibit lower life-cycle impacts than NMC batteries on a per-kWh basis, be it on life-cycle energy, GHG, and NO_x emissions (by ∼35–45%), or on SO_x emissions (by ∼360–715%) (Fig. 4–7). The significant SO_x emissions associated with $NiSO_4$ production from sulfidic Ni ores for NMC cathodes, which are absent in LFP cathodes, explain the drastic difference in SO_x impacts of LFP and NMC LIBs. This means that the increased use of LFP LIBs in vehicles is expected to result in a significant reduction in the life-cycle SO_x emissions of both LIBs and the resulting vehicles. Apart from this reduction, LFP LIBs also possess additional advantages over NMC LIBs, including lower cost, higher safety, and the avoidance of critical materials.[83] These advantages have led to a growing acceptance of LFP vis-à-vis NMC batteries in the electric automobile sector over the past few years.[83] However, the lower energy and volumetric density of LFP batteries can become a challenge for longer-range EVs (*i.e.*, larger kWh needs), especially in the United States, to address the range anxiety issue for consumers.[84] This lower energy density can influence both the life-cycle results (through increased material use per functional unit in high-range applications) and the suitability of LFP for certain vehicle segments, while potentially being less constrained for shorter-range vehicles or markets with different usage patterns.

The GHG emissions from a battery LCA study may depend on the regional natural gas and electricity emissions used for energy-intensive processes, such as battery assembly. The life-cycle GHG emissions of U.S.-based natural gas can vary by 49% to 152% from the average U.S. value, due to differences in recovery basins and delivery regions.[85] Similarly, life-cycle GHG impacts of electricity can vary from the U.S. average grid mix by −42% to +106%, depending on regional electricity mixes across the nation, according to R&D GREET 2024.[13] Considering this, varying domestic sourcing of natural gas and electricity can result in a −9% to +30% difference in NMC battery GHG impacts relative to the baseline value reported here, while for LFP batteries, the corresponding uncertainty range is −14% to +41%. This range assumes that the LIBs are assembled within the U.S.

Besides variation in impacts from energy sourcing, changes in $NiSO_4$ sourcing can also lead to significant variance in life cycle GHG impacts of NMC LIBs, as $NiSO_4$ contributes ∼10–25% of NMC LIB impacts (depending on NMC chemistry) (Fig. 4). The results presented in this paper, based on R&D GREET 2024, consider 100% virgin material sourcing for $NiSO_4$. However, this contribution and, consequently, the battery impacts can be reduced considerably if $NiSO_4$ is sourced from recycled LIBs. Conversely, a greater use of lateritic Ni ores for $NiSO_4$ is expected to increase the GHG intensity of its production and, thereby, the final LIB produced. Therefore, the sourcing of $NiSO_4$ can introduce considerable variability in life-cycle environmental impact results for NMC LIBs.

Finally, increasing the nickel content of the NMC chemistry (from NMC111 to NMC95) results in a 5% reduction in GHG emissions per kWh, primarily due to a 15% increase in energy density. However, the reduced GHG emissions come at the cost of 77% higher SO_x and 7% higher NO_x emissions per kWh, which may pose significant health risks and environmental concerns for specific localized communities. Therefore, further research is needed to develop technical solutions that can effectively capture SO_x and NO_x emissions during the production and manufacturing processes. It should be noted that the above-mentioned materials are not the only battery-related materials relevant from a supply chain standpoint; however, a detailed discussion of the supply chain implications of each battery material is beyond the scope of this study.

During the course of our analysis, we identified three key failure points that can lead to high variance in future battery LCA studies (apart from locational factors) based on insights gained from the 52 reviewed studies, which can be summarized as follows:

1. Mismatch between the stated LCA methodology of the study and the methodology by which the key inventory data is collected.

2. Reliance of the newer state-of-the-art studies on outdated inventory data.

3. Inconsistency and ambiguity in the scope of the battery assembly process.

The lack of transparency in battery LCA studies regarding the detailed BOMs and inventory is a common denominator across the three points described above. For instance, even some well-cited studies recently published in reputable journals, such as Degen *et al.*,[86] provide statements that their battery BOM was obtained from product teardowns, battery databases, and meta studies without clearly mentioning which part of the BOM is taken from each of these sources.

Therefore, providing clear, traceable sources for each part of the battery inventory data is the first step towards avoiding inconsistencies arising from the above points. Authors recognize that it may not always be possible to identify the source of each part of the battery inventory due to the industry's reluctance to share such data with researchers. We suggest clearly highlighting these inventory gaps and performing a sensitivity analysis of the impact of changes to these portions of the inventory. The standardization of the term "battery assembly" is another urgent need in battery LCA studies, as this component is often loosely defined or even mislabeled in different studies, despite having the highest primary energy demand and carbon emissions. Formalizing and standardizing the parts of the "battery assembly" process is likely to be rigorously followed by LCA researchers once the battery industry also formalizes this definition and reaches consensus on its specifics for the most widely used battery chemistries in the present and in the near future.

From an LCA perspective, we recommend the term "battery assembly" for the electrode-to-cell process chain, as this stage constitutes the physical assembly of a functional battery. Processes, including electrode stacking or winding, welding, electrolyte filling, formation, aging, and dry-room operation, are not auxiliary steps but define the transition from a component to a working electrochemical device. Using "battery assembly" to describe this stage provides a functionally accurate and environmentally meaningful system boundary, whereas alternative terms such as "cell assembly" are used inconsistently and less frequently, obscuring where the dominant energy and emissions burden arises in the value chain.

We also conducted a decomposition analysis of the key material/process contributors to battery GHG over six years for NMC111 chemistry using the GREET model as a benchmark. This analysis indicates that overall battery emissions per kWh decreased between the GREET 2018 and R&D GREET 2024 model versions, despite a considerably increased GHG intensity of key materials such as graphite and lithium carbonate. The reduced GHG emissions per kWh can be primarily attributed to increased battery energy density, the replacement of aluminum with steel in the battery pack, and improvements in the battery management system circuitry. Thus, technological improvements in batteries offset the increased GHG intensity of battery materials, at least according to the GREET model. The decomposition analysis also suggested that the increase in the GHG intensity of battery materials over the years can be attributed to the availability of more granular LCA inventory data.

One caveat to the results presented in this study relates to the design of LIBs considered here. Traditionally, LIBs are based on the cell-to-module-to-pack design, meaning that multiple cells are enclosed in modules, which in turn are enclosed in a single battery pack. However, any future changes in this design, such as a switch to a cell-to-pack design, may affect the LIB LCA results due to the reduced use of materials associated with module components.

It should be noted that the BOMs for each chemistry reflect not only the difference in active material composition but also chemistry-dependent variations in pack architecture, including cell count, module configuration, and thermal management requirements, because BatPaC (used for obtaining BOMs in our study) is a system design tool. This coupling is inherent to any functionally equivalent comparison anchored to a common vehicle range and is consistent with ISO14044 guidelines for comparative LCA, which suggest that systems to be compared need to deliver equivalent function.

6. Conclusions

This paper presents a comparative analysis of differences in life cycle GHG emissions results across Li-ion battery LCA studies published over the last decade. In this analysis, we identify three possible reasons for variance in LCA results and suggest ways to ensure that the variance across future LCA studies is not due to incomplete inventory data or poorly defined processes. A decomposition analysis, using the GREET model as a benchmark, indicated that technological improvements in batteries have offset the increase in battery GHG emissions due to the rising GHG intensity of key battery materials.

We present state-of-the-art LCA emission impacts for six widely used lithium-ion battery chemistries (with graphite anode) taken from Argonne's R&D GREET 2024 model and compare them with LCA results from studies over the last decade. We also investigated the number of citations and the interdependence of life-cycle inventories used in previously published battery LCA studies. Our literature review highlights the dominance of commercial and/or institutionally funded inventories, like Ecoinvent and R&D GREET, over peer-reviewed scientific studies. This is likely because the assumptions, results, and conclusions from peer-reviewed literature are subject to limited conditions, while the inventories from commercial/institutional LCIs are frequently updated and may be more easily accessed or utilized in an LCA. We also note the need for more LCA research on higher-nickel cathode chemistries such as NMC532, NMC622, and NMC811, given the current lack of sufficient data points to draw broad conclusions and to compare life-cycle emissions across studies, beyond differences attributable to grid mixes and material supply chain variations.

Our results indicate that, across most chemistries, the key contributors to the battery's environmental impacts are cathode production and battery assembly. SO_x emissions are strongly influenced by $NiSO_4$ content and associated production practices, whereas all other impact categories are strongly influenced by the battery assembly phase. We observed significant variation in the literature on battery GHG impacts, stemming from differences in data sourcing and how the data is used, as well as the evolution of battery BOMs and material supply chains.

It is important that efforts to characterize current battery LCA use models, methods, and findings that incorporate the latest understanding of battery materials and manufacturing processes; relying on older studies can lead to results that may not align with current practice.

Overall, this study provides a snapshot of the life cycle inventory used in the battery LCA literature over the last decade and compares state-of-the-art life cycle greenhouse gas emission results from R&D GREET 2024 with existing literature. We found an urgent need to standardize battery LCA due to considerable variance in nomenclature and inventory data across the existing literature, enabling the identification and remediation of emission hotspots within the lithium-ion battery supply chain globally. This study provides important insights into current state-of-the-art LIB life-cycle impacts that can serve as useful benchmarks for evaluating the comparative environmental performance of upcoming alternatives to automotive LIBs, such as solid-state and sodium-ion batteries.

Conflicts of interest

There are no conflicts to declare.

Data availability

The data supporting this article have been included as part of the supplementary information (SI). Supplementary information is available. See DOI: https://doi.org/10.1039/d5va00190k.

Acknowledgements

The authors acknowledge the support from the Vehicle Technologies Office of the U.S. Department of Energy under contract DE-AC02-06CH11357 for Argonne National Laboratory. The views and opinions of the authors expressed herein do not necessarily state or reflect those of the U.S. Government or any agency thereof or of any commercial entity. Neither the U.S. Government nor any agency thereof, nor any of their employees or employees of contributing companies, makes any warranty, expressed or implied, or assumes any legal liability or responsibility for the accuracy, completeness, or usefulness of any information, apparatus, product, or process disclosed or represents that its use would not infringe privately owned rights. The U.S. Government retains and the publisher, by accepting the article for publication, acknowledges that the U.S. Government retains a nonexclusive, paid-up, irrevocable, worldwide license to publish or reproduce the published form of this work, or allow others to do so, for U.S. Government purposes.

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