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Journal

Advanced Energy Materials, 11(33)

ISSN

1614-6832

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Publication Date

2021-09-01

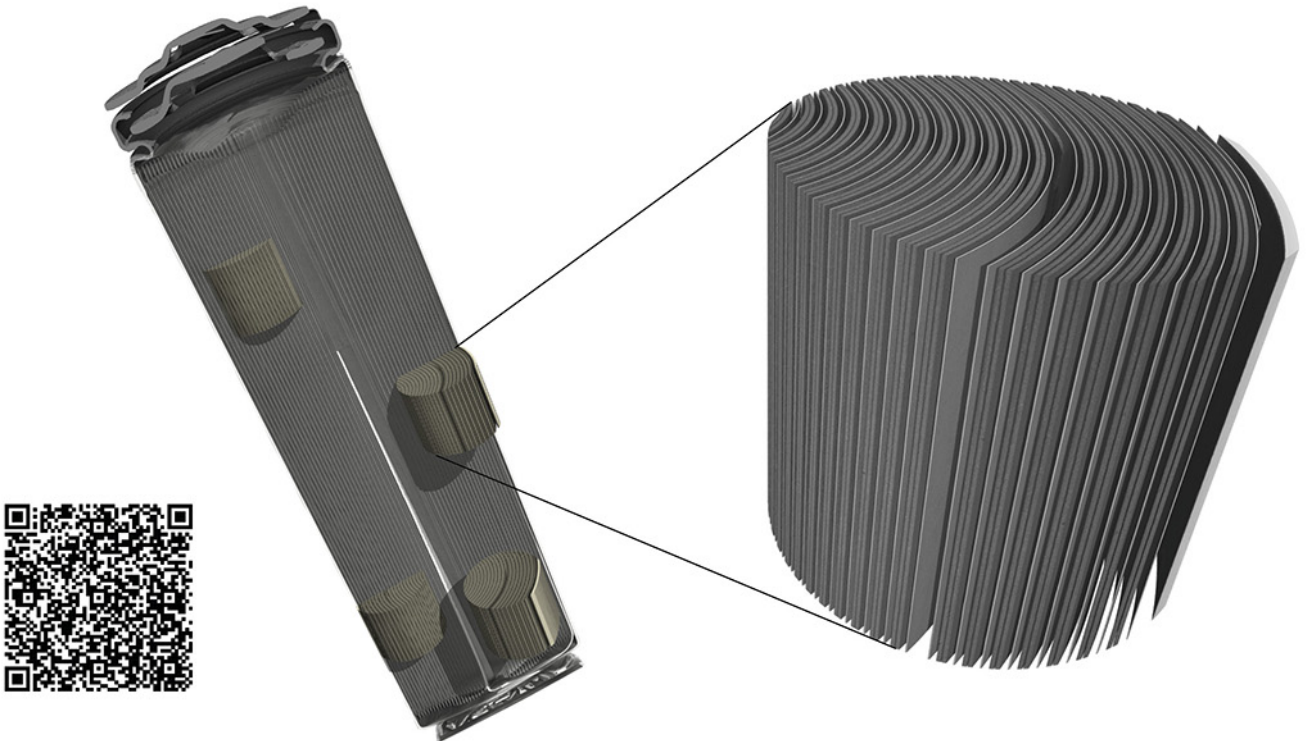
DOI

10.1002/aenm.202100771

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Life-Cycle Assessment Considerations for Batteries and Battery Materials

Jason Porzio and Corinne D. Scown*

Rechargeable batteries are necessary for the decarbonization of the energy systems, but life-cycle environmental impact assessments have not achieved consensus on the environmental impacts of producing these batteries. Nonetheless, life cycle assessment (LCA) is a powerful tool to inform the development of better-performing batteries with reduced environmental burden. This review explores common practices in lithium-ion battery LCAs and makes recommendations for how future studies can be more interpretable, representative, and impactful. First, LCAs should focus analyses of resource depletion on long-term trends toward more energy and resource-intensive material extraction and processing rather than treating known reserves as a fixed quantity being depleted. Second, future studies should account for extraction and processing operations that deviate from industry best-practices and may be responsible for an outsized share of sector-wide impacts, such as artisanal cobalt mining. Third, LCAs should explore at least 2–3 battery manufacturing facility scales to capture size- and throughput-dependent impacts such as dry room conditioning and solvent recovery. Finally, future LCAs must transition away from kg of battery mass as a functional unit and instead make use of kWh of storage capacity and kWh of lifetime energy throughput.

1. Introduction

Energy storage is essential to the rapid decarbonization of the electric grid and transportation sector.^[1,2] Batteries are likely to play an important role in satisfying the need for short-term electricity storage on the grid and enabling electric vehicles (EVs) to store and use energy on-demand.^[3] However, critical material use and upstream environmental impacts from manufacturing are often cited as a drawback to widespread use of rechargeable batteries.^[4,5] Life-cycle assessment (LCA) is a widely used approach for examining the potential impacts of large-scale battery production, use, and disposal and/or recycling. At its core, LCA is a methodology for quantifying the direct and indirect environmental burdens associated with a product or service.^[6] It is also a useful framework to explore environmental tradeoffs between different technologies that provide a comparable service. However, applying LCA to batteries is chal-

lenging for a variety of reasons ranging from methodological choices to scarcity of primary data on battery manufacturing.

To date, there has not been consensus in the field of LCA as to how the environmental impact of batteries should be analyzed, nor how the results should be reported. Studies use a wide variety of system boundaries, functional units, primary data sources (which in turn report data at different levels of granularity), and life-cycle inventory, midpoint, and impact categories. This makes cross-comparisons of different technologies challenging and limits the ability for LCA to provide a feedback loop to early scientific research and technology development. It can also limit our ability to detect and correct errors in the literature; it is common for life-cycle inventory results to vary by one or more orders of magnitude across the literature and most reviews are unable to explain the underlying cause of the differences.

Prior review papers on the LCA of lithium-ion batteries (LIBs) can be categorized into three main groups dependent on their goals: identifying and reducing sources or uncertainty/variability;^[7–9] synthesizing results and determining key drivers to inform further research;^[10,11] and critical review of literature to improve LCA practices.^[12] Sullivan and Gaines^[9] reviewed life-cycle inventory estimates for lead-acid, nickel–cadmium, nickel-metal hydride, sodium-sulfur, and Li-ion batteries and calculated their own estimates for comparison; the conclusions focused on the need to fill key data gaps. Ellingsen et al.^[7]

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DOI: 10.1002/aenm.202100771

focused on life-cycle greenhouse gas (GHG) emissions and noted that previously published results differed by an order of magnitude, with differences driven by direct energy demand for cell manufacturing and pack assembly. Pellow et al.^[8] focused on gaps in the range of use cases evaluated and the need for additional studies on end-of-life management. Nealer and Hendrickson^[10] focused on EVs and summarized prior studies' findings on the energy and GHG advantages of EVs. Nordelöf et al.^[12] reviewed 79 LCAs for hybrid, plug-in hybrid, and pure EVs and the study focused more broadly on sources of uncertainty related to light-duty vehicles and their use, as opposed to battery technologies specifically. Peters et al.^[11] reviewed a wide array of battery LCAs and offered valuable insights into which studies used primary data and which relied on secondary data sources; the review also provides an in-depth discussion of battery cycle life and round-trip efficiency. Peters et al.^[11] did note, as other reviews have, that manufacturing energy use estimates vary across the literature by at least an order of magnitude. However, they did not attempt to offer a detailed explanation and ultimately relied on calculating averages of results from prior studies. Using these calculated averages, combined with 1995 European normalization factors for each life-cycle impact, Peters et al.^[11] suggested that global warming potential (GWP) may not be the most important environmental metric, as abiotic depletion, acidification potential, and human toxicity potential all resulted in larger normalized impacts.

In this paper, we will not revisit all prior LCA studies on life-cycle energy and environmental impacts of batteries. Instead, we will focus on three key issues that have not been adequately explored in the literature to-date: 1) selecting relevant environmental performance metrics and acknowledging their limitations and data requirements, 2) understanding discrepancies in reported battery manufacturing impacts, and 3) defining appropriate functional

units. With these key issues in mind, we provide a critical review of recent LCA studies applied to rechargeable batteries produced for grid- and vehicle-based applications and suggest some best practices for the field. We draw from previously published work with a focus on LIBs, although most of the insights in this article can apply to a wide variety of battery technologies.

1.1. Lithium-Ion Battery Technologies

LIBs are the most commonly used battery chemistry and, although this paper is not focused on the details of the technologies, it is worthwhile to briefly describe the most common types of LIBs explored in the current literature. Research has continued on the development of non-LIB battery technologies, including sodium-ion batteries, potassium-ion batteries, solid-state batteries (Li-metal, Li-sulfur, and rechargeable zinc alkaline), flow batteries, and multivalent batteries,^[13,14] but LIBs are likely to continue to dominate the market in the near-term. LIBs are typically differentiated based on their cathode material: lithium manganese oxide (LMO), lithium nickel manganese cobalt oxide (NMC), lithium iron phosphate (LFP), and lithium nickel cobalt aluminum oxide (NCA). Most batteries explored in prior LCA studies use a graphite carbon anode. As shown in **Table 1**, NMC, NCA, LFP, and LMO batteries with graphite anodes are typically estimated to last for 1000–3000 cycles or more.^[15–21] These batteries have specific energy at the cell level ranging from 90 to 250 Wh kg⁻¹.^[15] Researchers are exploring other anodes, such as lithium titanate (LTO) and we have included LFP-LTO battery data in Table 1 as well; the LFP-LTO battery offers longer cycle life (5000+) at the expense of specific energy, which is lower than all other types of LIBs in Table 1.^[15,16,19,20,22] NCA-graphite batteries achieve the highest specific energy, but stand out for

Table 1. Battery characteristics by common lithium-ion battery chemistries. Price per kWh for each battery chemistry is qualitatively described relative to other battery chemistries.^[14,23] Safety rating for each battery chemistry is qualitatively described, primarily dependent on battery stability, thermal behavior, and resiliency to abuse.^[15] Data sources: Bloch et al., Mitchel and Waters, Ralon et al., Gantenbein et al., Ecker et al., Battaglia, Srinavasan, Zubi et al., Stewart et al., Buchmann, Grolleau et al., Nelson et al., and Nitta et al.^[14–23,25–27]

Li-ion battery chemistry	Cell-level specific energy [Wh kg ⁻¹]	Nominal voltage [V]	Cycle life [Cycles]	Shelf life [Years]	Operating temperature [°C]	Thermal runaway [°C]	Price per kWh rating	Price per kWh rating	Primary use cases	Representative manufacturers	Common cathode compositions
NMC ^{a)} –Graphite	140–200	3.7	2000+	8–10	0–55	210	Med-high	Med-high	Power tools, EV	CATL, Sanyo, Panasonic, Samsung, LG Chem, SK Innovation	NMC-111 ^{a)} , NMC-532 ^{b)} , NMC-622 ^{c)} , NMC-811 ^{d)}
NCA ^{e)} –Graphite	200–250	3.6	2000+	8–10	0–55	150	Med	Med	EV	Tesla/Panasonic	–
LFP ^{f)} –Graphite	90–140	3.2	3000+	8–12	0–55	270	Low	Low	EV, Grid-Scale Stationary	BYD, K2, Lishen, Saft, GS Yuasa, A123, Valence, BAK	–
LFP–LTO ^{g)}	≤80	2.7	5000+	10+	–40–55	>270	Very high	Very high	Personal electronics, some EVs	Altairnano, Toshiba, Yabo	–
LMO ^{h)} –Graphite	100–140	3.7	1000–2000	6–10	0–55	250	Low	Low	Power tools, EV's (Typ. with NMC blend)	Hitachi, Samsung, LG Chem, Toshiba, NEC	Layered spinal

^{a)}Li_{1.05}(Ni_{0.33}Mn_{0.33}Co_{0.33})_{0.95}O₂; ^{b)}Li_{1.05}(Ni_{0.5}Mn_{0.3}Co_{0.2})_{0.95}O₂; ^{c)}Li_{1.05}(Ni_{0.6}Mn_{0.2}Co_{0.2})_{0.95}O₂; ^{d)}Li_{1.05}(Ni_{0.8}Mn_{0.1}Co_{0.1})_{0.95}O₂; ^{e)}LiNi_{0.8}Co_{0.15}Al_{0.05}O₂; ^{f)}LiFePO₄; ^{g)}Li₄Ti₅O₁₂; ^{h)}LiMn₂O₃.

their relatively poor safety rating, with a far lower thermal runaway temperature than its competitors.^[15,23] Other LIB chemistries, such as LCO were intentionally omitted due to their decreasing relevance in vehicle- and grid-scale energy storage systems. “Anode free” configurations, such as Zinc MnO₂ batteries, are in the early stages of development and have the potential to improve energy density relative to batteries with graphite anodes.^[24] The remainder of this review will focus on the LIB chemistries outlined in Table 1.

1.2. Life-Cycle Assessment Overview

To compare the environmental impacts of competing battery technologies, or simply understand the full impact of increased battery production and use, the LCA must be designed to answer a well-defined question. LCAs are commonly defined by four key phases, all of which are essential to completing a meaningful study: a) the goal and scope definition phase, b) the inventory analysis phase, c) the impact assessment phase, and d) the interpretation phase.^[6] It is during the goal and scope definition phase that researchers must decide what question they seek to answer and let that question guide the definition of system boundaries, environmental metrics, and one or more functional units. In the context of batteries, LCA results can be used to inform battery research and development (R&D) efforts aimed at reducing adverse environmental impacts,^[28–30] compare competing battery technology options for a particular use case,^[31–39] or estimate the environmental implications of large-scale adoption in grid or vehicle applications.^[40]

LCA is most straightforward to apply to a well-defined functional unit; in other words, any environmental impact is simple to normalize per unit of a product or service that is being provided (e.g., g CO₂ emitted per kWh of electricity generated or liters of water withdrawn per bushel of corn produced). However, batteries pose a particular challenge for LCA as it has historically been applied. Batteries are simply storing energy for later use, and how batteries are cycled will impact their longevity and the value of the service they provide in ways that are not straightforward to predict. If a study is comparing multiple battery technologies applied to the same use case, it makes most sense to normalize results in the basis of the service provided. The downside of this approach is that it is very difficult to compare multiple studies, as they inevitably use different assumptions about how the battery is used over the course of its lifetime. In contrast, an assessment of one or more early-stage battery technologies intended to inform further research may not have reliable use-phase performance data. In this case, it may be appropriate to draw a system boundary ending at the factory gate, and simply note any potential difference in cycle life that may ultimately impact the batteries' longevity.

Studies that define system boundaries excluding the use phase and end-of-life are commonly referred to as cradle-to-gate (where “gate” refers to a factory gate). Studies including the use phase and end-of-life are referred to as cradle-to-grave; this can include reuse, recycling, and ultimately disposal.^[6] The term cradle-to-cradle has been used to refer to systems that include recycling, but is generally meant to suggest a zero-waste

process^[41] and thus is not commonly used to refer to battery life cycles, even if they include recycling.

2. The Life Cycle of Stationary and Vehicle Li-Ion Batteries

Figure 1 shows the typical life cycle for LIBs in EV and grid-scale storage applications, beginning with raw material extraction, followed by materials processing, component manufacturing, cell manufacturing, and module assembly.^[14] Finished modules may be assembled into packs and placed in vehicles, assembled into racks on-site for shipment to stationary storage facilities, or shipped directly as modules for off-site rack assembly at energy storage sites.^[42–45] All LCAs must begin with raw material extraction, regardless of scope. Researchers then decide whether to tie their analysis to a particular use case and, if so, whether to extend the system boundaries through the use phase and end-of-life (EOL). Even in cradle-to-gate studies, researchers must be careful to indicate the form batteries take at the factory gate; module, pack, or fully assembled rack (in the case of stationary storage).

The majority of battery LCAs are tied to a particular use case, such as EVs, hybrid solar and battery systems, or standalone grid-connected storage.^[46–52] This is the preferred approach, where feasible, because performance differences (e.g., round-trip efficiency and cycle life) are important to the definition of a common functional unit across which different alternatives can be compared. However, not all studies include battery use phase for a specific application, nor is this always feasible for more advanced, pre-commercialization battery technologies. Wang et al. (2019)^[53] conducted a use-agnostic analysis to compare the environmental impacts of different cathode materials and Wang et al. (2018)^[54] conducted a cradle-to-gate analysis of lead acid, LMO, and LFP batteries. For a use-agnostic cradle-to-gate analysis of an LIB, researchers must still select a pack or rack configuration that is tied to a stationary or EV application. In a truly use-agnostic LCA, the system boundary may need to be set at the module assembly stage, since the assembly of the pack or rack (including such components as thermal management and electrical control) will differ substantially depending on how the battery will be used (see Figure 1). Cradle-to-grave LCAs consider how batteries will be used and treated at their end of life including collection, recycling and/or disposal. There are multiple options for recycling, ranging from specialized, chemistry-specific direct recycling to hydrometallurgical recycling and pyrometallurgical recycling, which are more flexible and aim to recover only valuable metals from the batteries.^[55,56]

2.1. Raw Material Extraction and Delivery

LIBs' reliance on finite resources, combined with dramatic growth in production (approximately doubling every 5 years)^[21] and uncertain future recycling practices has generated concern over material constraints. Olivetti et al.^[57] explored the potential bottlenecks in critical material supplies for LIB manufacturing. The breakdown of material comprising batteries, from active material through individual cells, modules, and packs, is well

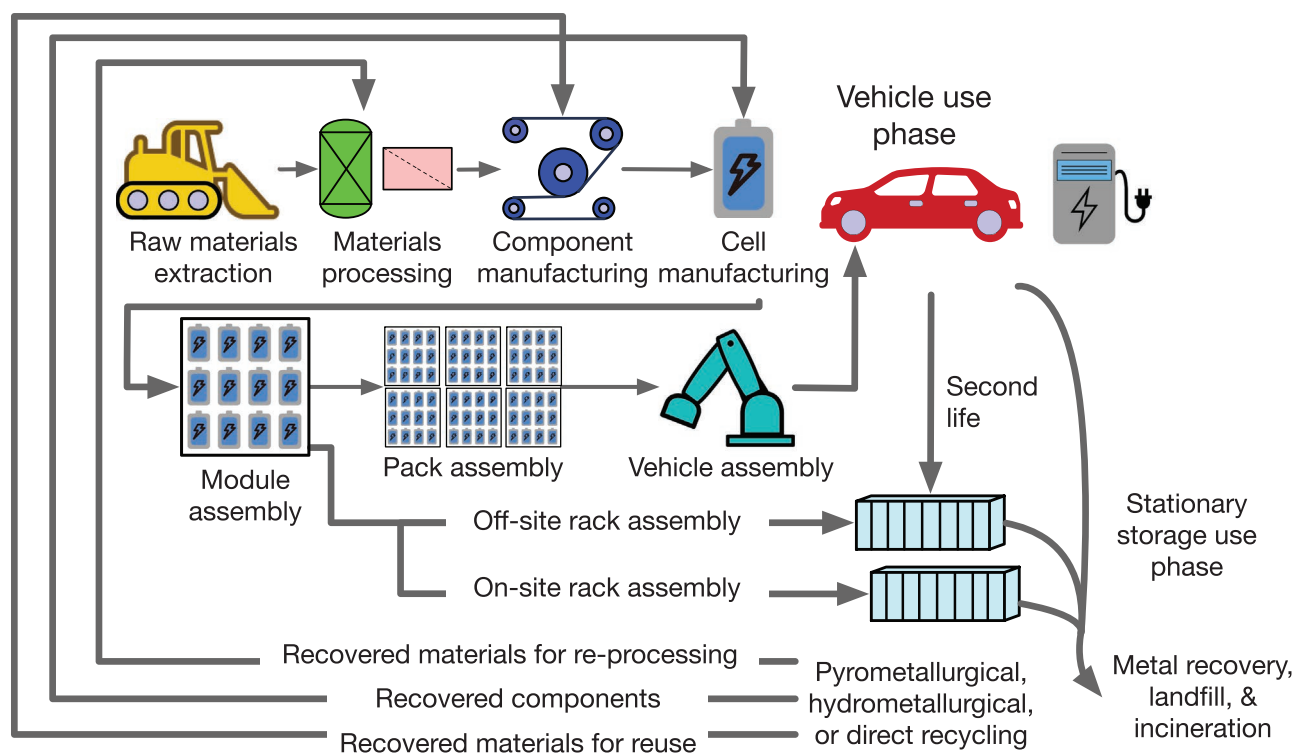


Figure 1. Major life-cycle stages for vehicle and stationary batteries.

documented in the literature; breakdowns of elements present in each type of cathode active material are shown in **Table 2** and mass breakdowns per kWh for modules are shown in **Table 3**. Although the use of critical materials is often discussed as a single challenge, there are three separate issues worth discussing. First, there is the question of resource availability relative to consumption and whether scaling up battery manufacturing will deplete critical material reserves and/or drive up prices. Second, there is the geopolitical risk associated with highly concentrated production, which can lead to conflict, price instability, and artificial shortages. Here, we characterize this concern as supply chain risk, and argue that researchers too often conflate supply chain risk with resource depletion. Traditional LCA methods are not well equipped to capture supply chain risk. Third, there are environmental and social impacts associated with mining operations, which is well within the purview of LCA.

2.1.1. Resource Depletion and Conflation with Supply Chain Risk

Olivetti et al.^[57] synthesized the available data on consumption rates relative to available reserves for nickel (Ni), manganese (Mn), cobalt (Co), lithium, (Li), and natural graphite. They found that the ratio of known reserves to primary mine production (also known as the static depletion index) has increased for Co, Li, and natural graphite, suggesting that continued demand has resulted in additional exploration and extraction. Mn and Ni did not show an upward or downward trend, indicating that the ratio of production to known reserves has remained relatively constant. This highlights the challenge of attempting to place a single number on finite resource depletion as part of an LCA; society's understanding of available resources is not static. Increased demand drives advancements in exploration and recovery technologies. Material recovery potential through

Table 2. Element mass ratio per cathode active material. Data sources: Nelson et al. and Nitta et al.^[26,27]

Element	NMC-111 [% mass]	NMC-532 [% mass]	NMC-622 [% mass]	NMC-811 [% mass]	NCA [% mass]	LFP [% mass]	LMO [% mass]
Li	0.078	0.022	0.077	0.077	0.072	0.044	0.038
Ni	0.197	0.083	0.354	0.471	0.489	–	–
Mn	0.184	0.466	0.111	0.055	–	–	0.608
Co	0.198	0.334	0.119	0.059	0.092	–	–
Al	–	–	–	–	0.014	–	–
Fe	–	–	–	–	–	0.354	–
P	–	–	–	–	–	0.196	–
O	0.343	0.095	0.339	0.338	0.333	0.406	0.354

Table 3. Module material inventory per kWh for a module with a 10 kWh energy capacity. Data source: Nelson et al.^[26]

Material	Cell component	NMC-111	NMC-532	NMC-622	NMC-811	NCA	LFP	LMO
Cathode active material [g kWh ⁻¹]	Cathode	1757.45	1288.79	1481.48	1257.53	1358.70	2031.28	2341.74
Graphite [g kWh ⁻¹]	Anode	858.02	863.13	840.45	841.18	857.71	956.61	793.59
Carbon black [g kWh ⁻¹]	Cathode additive	36.61	26.85	30.86	26.20	28.31	42.32	48.79
PVFD [g kWh ⁻¹]	Cathode binder	36.61	26.85	30.86	26.20	28.31	42.32	48.79
	Anode binder	17.51	17.61	17.15	17.17	17.50	19.52	16.20
Aluminum [g kWh ⁻¹]	Cathode current collector	104.32	77.24	88.18	75.08	79.12	156.85	149.66
	Positive terminal assembly	29.57	27.28	27.06	25.75	27.05	35.88	32.08
	Cell container	57.55	50.93	51.93	48.85	51.48	73.53	66.51
	Module heat conductors	103.34	90.39	93.27	87.34	91.67	133.48	121.61
	Module enclosure	69.03	64.43	64.03	61.14	63.57	82.09	74.86
Copper [g kWh ⁻¹]	Anode current collector	244.18	182.03	206.82	176.73	186.31	364.82	347.55
	Negative terminal assembly	98.13	90.52	89.81	85.46	89.75	119.06	106.47
	Cell interconnection	29.45	27.17	27.01	25.65	26.85	35.77	31.98
LiPF ₆ [g kWh ⁻¹]	Electrolyte salt	2.79	2.37	2.52	2.31	2.40	3.75	3.42
Ethylene Carbonate [g kWh ⁻¹]	Electrolyte fluid	221.14	188.14	199.40	183.43	190.14	297.29	270.79
Dimethyl Carbonate [g kWh ⁻¹]	Electrolyte fluid	179.26	152.51	161.63	148.69	154.13	240.98	219.50
Polypropylene [g kWh ⁻¹]	Separator	16.80	12.40	14.18	12.07	12.73	25.33	24.19
	Cell container	8.83	7.81	7.96	7.49	7.89	11.28	10.20
Polyethylene [g kWh ⁻¹]	Separator	16.80	12.40	14.18	12.07	12.73	25.33	24.19
Polyethylene Terephthalate [g kWh ⁻¹]	Cell container	3.83	3.39	3.46	3.25	3.43	4.90	4.43
Misc electronics [g kWh ⁻¹]	BMS	11.08	11.32	10.28	10.26	11.05	11.81	10.34
Total [kg kWh ⁻¹]		3.90	3.22	3.46	3.13	3.30	4.71	4.75

recycling adds another layer of complexity. As Olivetti et al.^[57] rightly point out, there is a time lag between when batteries are manufactured and when they reach the end of their life, so regardless of what can be recovered, recycling is unlikely to address any near-term (10–20 year) material supply constraints. If battery technologies do not evolve away from reliance on these critical materials in coming decades, recycling can be an important long-term strategy. Pyrometallurgical recycling facilities will recover Ni, Co, Mn, and Copper (Cu),^[58] while hydrometallurgical recycling facilities will recover all of the aforementioned metals, as well as Li and aluminum (Al). Direct recycling will recover an even larger range of materials, many of which can be reused without further processing. Given this context, it is worth revisiting the assertion by Peters et al.^[11] that abiotic depletion (an impact category representing depletion of non-renewable resources, such as minerals and fossil fuels) is the most important impact of batteries on a normalized basis, exceeding the importance of GWP. Many of the nuances in critical material use and supply outlooks are lost in typical LCA practices and reducing these impacts to a single score is more likely to create confusion than generate useful insights, particularly when fossil fuel and critical material depletion are

combined in a single score. This is particularly true for materials like Co and Li, for which demand is growing rapidly. In fact, a greater cause for concern is the geographic diversification, or lack thereof, in reserves and supply for some of these materials.

Olivetti et al.^[57] note a consensus that Co, Li, and to a lesser extent, natural graphite pose the greatest supply risks. These risks are driven by the concentration of known reserves and current production in a small number of countries. **Table 4** shows the countries with largest reserves and current production for raw materials used in LIB production. Co, which is required for batteries with NMC and NCA cathodes, is generally regarded as posing the greatest risk because of its geographically-concentrated supply. Batteries are responsible for around 60% of global Co demand, and total demand is expected to double by 2025.^[59,60] Over 60% of current supply, and half of estimated reserves are located in the Democratic Republic of the Congo (DRC), where there have been serious environmental and social consequences ranging from child labor to human exposure to heavy metals, particularly from unregulated artisanal and small-scale mining operations.^[60–64] Co is largely a co-product of Ni and Cu mining, with 55% of supply coming

Table 4. 2019 global reserves for materials relevant to Li-ion battery production, 2019 mining production, and distribution of resources. Data source: USGS.^[61]

Element	Li-ion battery component	Global reserves [tons]	Country with largest reserves	Share of largest reserves [%]	Global mining production [tons]	Country with largest mine production	Share of largest mine production [%]
Aluminum (Al)	Cathode current collector, NCA cathode	30 000 000 ^{a)}	Australia ^{a)}	17 ^{a)}	63 200 ^{b)}	China ^{b)}	55 ^{b)}
Iron (Fe)	LFP cathode	84 000 ^{c)}	Australia ^{c)}	29 ^{c)}	1 520 000 ^{c)}	Australia ^{c)}	37 ^{c)}
Phosphorus (P)	LFP cathode	71 000 000	MAR and EH ^{d)}	70	227 000	China	42
Manganese (Mn)	LMO, NMC cathode	1 300 000	South Africa	40	19 600	South Africa	30
Carbon (C) ^{e)}	Graphite anode	320 000 000	Turkey	28	1 100 000	China	64
Nickel (Ni)	NMC, NCA cathode	94 000 000	Indonesia	22	2 610 000	Indonesia	33
Copper (Cu)	Anode current collector	870 000	Chile	23	20 400	Chile	28
Cobalt (Co)	NMC, NCA cathode	7 100 000	DRC	51	144 000	DRC	69
Lithium (Li)	All cathodes, electrolyte	21 000 000	Chile	43	86 000	Australia	52

^{a)}Representative of bauxite reserves; ^{b)}Representative of smelter production; ^{c)}Representative of iron content in ore; ^{d)}Morocco and Western Sahara; ^{e)}Natural Graphite.

from Ni mining, while <10% comes from dedicated Co extraction.^[57,65] Co processing capacity is even more concentrated; 95% of all Co refining occurs in China.^[65]

Li production is concentrated in Australia, Chile, and Argentina. Combined, Australia, China, Argentina, and Chile make up 90% of global Li supply^[59–61] and demand is continuing to increase.^[66] Australia produces lithium concentrate from a lithium aluminum inosilicate called spodumene, while Chile and Argentina produce Li₂CO₃ from brine. As Olivetti et al.^[57] note, Li supply for battery manufacturing is a controversial topic, discussed at length in the literature, but there are many potential avenues for increasing Li supply. Potential lithium resources exist in the US, DRC, Bolivia, and regions throughout Europe, but these sites are not yet developed for commercial production of Li.^[59,61] Two forms of lithium deposits are viable for extraction: hard rock deposits and brine lake deposits.^[59,67,68] A majority of reserves are located in brine lakes of South America, while much of the remaining deposits are located in hard rock, predominantly located in Australia.^[67,69,70] As of 2016, Li-ion batteries accounted for 34% of global lithium demand.^[60] The most likely bottleneck for Li supply will be the ability to ramp up Li production quickly enough to avoid short-term supply constraints and price spikes.^[57]

As noted earlier, LCA is ill-equipped to capture the dynamics of demand, reserves, and annual trends in production and distill these factors into a single impact value. LCA is even less well-suited to capturing the supply chain risks associated with geographically-concentrated extraction and processing, as these are all dependent on the location of demand and the perceived political stability and/or friendliness of the region from the perspective of the study's researchers. A study centered in the US may view the China-dominated co processing industry as a source of supply risk, while a study in China would likely not share that perspective. Although it is entirely appropriate to raise these issues as part of an LCA, and perform quantitative analysis to elucidate potential short- and long-term resource depletion, we advise against assigning a single impact value that may be reliant on outdated data or irrelevant assumptions. One approach in LCA that may offer some more useful

insights is the notion of average versus marginal and incremental production impacts. For example, if the environmental burden of Li extraction will increase as a result of a shift toward hard rock deposits and brine lake deposits in the future, sensitivity analyses could contrast the impacts of current average Li production with those of future sources that will supply the next ton (or million tons) of material. Such an approach draws on the strengths of LCA, as a method for quantifying the environmental impact of a product or service, to capture one dimension of the broader concerns around resource depletion.

2.1.2. Environmental Impacts of Raw Material Extraction and Processing

For perspective, battery materials are estimated to comprise approximately one third of total primary energy demand to produce an LMO-graphite battery pack, with the remaining energy demand almost entirely owed to battery manufacturing.^[46,52] As discussed later in this article, however, this ratio can vary considerably based on the specifics of the manufacturing process and facility scale. Al, Cu, and graphite comprise the largest shares of LIB packs (using graphite anodes) by mass^[71] and, in an LMO battery, Al is estimated to be the largest contributor to the materials energy footprint by a fairly wide margin, followed by the cathode, battery management system (BMS), Cu, and graphite.^[72] A challenge for conducting the life-cycle inventory for material inputs to LIB manufacturing is that there are not one or two components that dominate the energy use or emissions; the impacts are spread across a wide array of components; Al, for example, is used in the cathode current collector, positive terminal assembly, cell container, module heat conductors, module enclosure.^[73–75] Battery designs continue to evolve and detailed material breakdowns are, in many cases, proprietary.

Energy used for raw material mining and processing is typically some combination of diesel fuel to operate mining equipment and transport material, electricity to run mechanical processes, and natural gas for thermal energy during processing. As a general rule, the processing/refining stage is usually what

distinguishes materials as being more or less energy intensive. For example, the energy footprint and resulting impacts of Al production are dominated by smelting and refining, which requires large amounts of electricity.^[76] The majority of global Al production and refining occurs in China,^[61,76] which further increases the impact because China's national average grid mix continues to be coal-dominated.^[77] The energy footprint of Cu is dominated by solvent extraction and electrowinning (also known as electroextraction).^[78] In the case of Co, researchers must exercise caution in sourcing data from prior studies, as Co is a co-product of Ni or Cu production, meaning some form of allocation is required and the ratios of production will differ by location. In other words, extraction and processing impacts must be attributed appropriately to multiple outputs, which can be done using system expansion (to avoid formal allocation) or mass- or market-value based allocation.

Li, although discussed extensively from a resource depletion standpoint, has so far not proved to be a dominant contributor to the energy and environmental footprint of LIBs. Two thirds of Li is extracted from brine,^[68] and the energy footprint of Li_2CO_3 can vary by a factor of two depending on the concentration in the brine.^[72] As indicated by Yuan et al.^[52] and Dunn et al.^[72] for LMO batteries, the LMO cathode itself is the second-largest contributor to energy use (a distant second to Al). However, the single largest driver of this energy footprint is the LiMn_2O_4 production, which includes a roasting process.^[72] Rock-based Li production has been shown to be far more energy-intensive brine extraction; Jiang et al.^[79] estimated that it is an alarming 48 times more GHG-intensive than brine extraction. For most environmental impacts, leaching and roasting seem to be larger contributors.^[79] These results strongly suggest that LCA studies must thoroughly explore the average, marginal, and incremental Li sources and conduct sensitivity analyses to account for different mining and processing alternatives.

Like Li, graphite is an input required across nearly all LIBs as it is the most common anode material. The graphite market is supplied by about half mined natural graphite and half synthetic.^[80] China mines roughly 65% of commercial natural graphite globally, followed by Mozambique and Brazil both producing under 10% of natural graphite supply.^[61,81] While natural graphite is cheaper and thus the typically preferred option for battery anodes, synthetic graphite is produced through byproducts of fossil fuel industries and is produced in high quantities in the US.^[57,82,83] The impacts of graphite are driven by the high thermal energy inputs during processing.^[82]

Although energy use can sometimes be the most expeditious proxy for environmental impacts, particularly with some knowledge of what fuels are combusted and in what types of equipment (internal combustion engines, boilers, etc.), it is far from a perfect metric. Nearly all of the articles cited so far have reported energy use without noting what fraction is in the form of primary fuels for thermal energy (e.g., natural gas or coal), how much is satisfied by electricity, and how much is used for operating liquid fuel-powered vehicle and equipment (e.g., diesel). Even if energy use can serve as a proxy for GHG emissions, there are other environmental impacts associated with raw material extraction. As noted by Peters et al.,^[11] acidification potential, eutrophication potential, human toxicity, and ozone depletion potential have all been incorporated into prior battery

LCAs. The question, however, is what impacts are not directly tied to combustion and how much confidence do we have in the published estimates? Acidification potential, most likely, is tied to SO_2 emissions from combustion sources.

Peters et al.^[11] suggests that Ni and Co extraction have significant toxicity impacts. Farjana et al.^[84] evaluated Co mining and presented a comparison between Co, Ni, and Cu. Their results indicated that acidification (driven by blasting-related emissions) and, to a lesser extent, particulate matter and non-cancer human toxicity were the most prominent impacts on a normalized basis and Ni mining is responsible for three to four times greater impacts than either Co or Cu on a per-kg output basis. However, that study relies on Australian data and is likely not representative of average global production. There is ample evidence to suggest that artisanal Co mining occurring in the Katanga Copperbelt of the Democratic Republic of Congo (DRC), estimated at 15–20% of total DRC production or 9–12% of global supply, has dramatically increased human exposure to heavy metals in the workers and surrounding population.^[63,64] This case serves as a reminder that local practices and regulations or lack thereof can be as influential to environmental impacts as the specific type of mining activity itself. Although artisanal Co mining makes up a minority of total global supply, its proximity to populated areas combined with unsafe practices are likely to drive an outsized share of the sector's overall health impacts. For this reason, conducting an LCA that relies only on estimated emissions and energy use in formal mining operations^[84] may underestimate the average environmental impacts. This concept is analogous to the notion of “super-emitters” in natural gas operations,^[85] which make up a small fraction of overall sector activity but, if ignored, lead to large underestimates in emissions and impacts. Accurately capturing these impacts, however, can be a challenge if the operations are informal, as most LCAs available on raw material extraction and processing rely on datasets provided by large mining companies.^[86]

Amarakoon^[87] provided a helpful discussion of the underlying drivers of other less commonly-reported impact categories. For example, they noted that their LFP-graphite battery was modeled based on Canadian production, and at the time of the report, the Canadian grid had substantially higher trichlorofluoromethane (CFC-11) emissions than the U.S. grid. This grid-related CFC-11 emission factor dominated the overall results. However, more recent research has suggested that CFC-11 emissions are steeply declining,^[88] which suggests that drawing from older studies that include ozone depletion potential may result in overestimates of this impact.

Estimates of eutrophication potential associated with raw material extraction and processing for LIBs can be particularly difficult to parse. Depending on the impact assessment method used, eutrophication may be separated into freshwater and marine eutrophication or combined into a single metric. These impacts are driven by emissions of ammonia, phosphate, other water-soluble nitrogen and phosphorus-containing compounds to water bodies, as well as biological and chemical oxygen demand. Eutrophication is an impact most commonly associated with agricultural systems or other facilities that result in nutrient runoff or discharge. Amarakoon^[87] found that any battery relying on steel for the pack/housing (in their case, LMO)

generated net negative eutrophication results because the US Life-Cycle Inventory (LCI) database entry for cold-rolled steel suggested that nutrient concentrations in incoming process water had higher concentrations of phosphate, ammonia, and other nutrients than the effluent, suggesting that the steel-making process was removing these nutrients from the water. In contrast, Amarakoon^[87] found that LFP-graphite batteries resulted in net positive eutrophication potential, presumably owed in part to the diammonium phosphate and phosphoric acid production required to produce the cathode material.^[87] Ellingsen et al.^[74] indicated that Cu mining, and specifically management/discharge of sulfidic mine tailings, is responsible for 62% of freshwater eutrophication potential and 65% of freshwater ecotoxicity potential. However, it is difficult to find any primary literature data to support the claim that copper mine tailings, which are acidic, are a significant cause of eutrophication. Ellingsen et al.^[74] also suggested that the *N*-methyl-2-pyrrolidone (NMP) solvent is a major contributor to marine eutrophication potential, driven by the upstream manufacturing of dimethylamine. Our survey of the literature on eutrophication impacts of raw material extraction and processing suggests that, for the most part, compounds containing nitrogen or phosphorus tend to appear as contributors. In some cases, data on how nutrient-containing wastewater is managed at processing facilities is sparse, making the results difficult to confirm. For perspective, the 8.0 kg P-equivalent for a battery pack provided by Ellingsen et al.^[74] would translate to the freshwater eutrophication associated with consuming around 300 kg of beef, based on our simple calculations and data from

de Vries et al.^[89] It is clear that eutrophication potential, while not entirely irrelevant, must be approached with caution and skepticism about the underlying data inputs.

2.2. Battery Manufacturing

Battery manufacturing comprises around two thirds of the cradle-to-gate energy demand for LIBs,^[52] although this ratio can vary considerably depending on manufacturing facility scale and utilization.^[46] Unlike raw material extraction and processing, most environmental impacts during the battery manufacturing process are directly linked to energy use (on-site combustion and off-site electricity generation),^[74] so this section will focus on energy use as the key driver of impacts. Despite the importance of understanding energy use at manufacturing facilities, prior studies are inconsistent in how individual processes within the manufacturing facility are reported, making harmonization or even basic comparisons difficult. Here, we attempt to demystify the manufacturing process and key drivers of energy use and environmental impacts. This will allow future researchers to focus data gathering and sensitivity analysis efforts on the largest contributors to the environmental impacts of battery manufacturing, and avoid becoming mired in the details of processes that consume minimal energy and do not have other appreciable environmental impacts.

Li-ion cell production is generally divided into three phases: electrode manufacturing, cell assembly, and cell finishing (see Figure 2). Electrode manufacturing is largely independent of

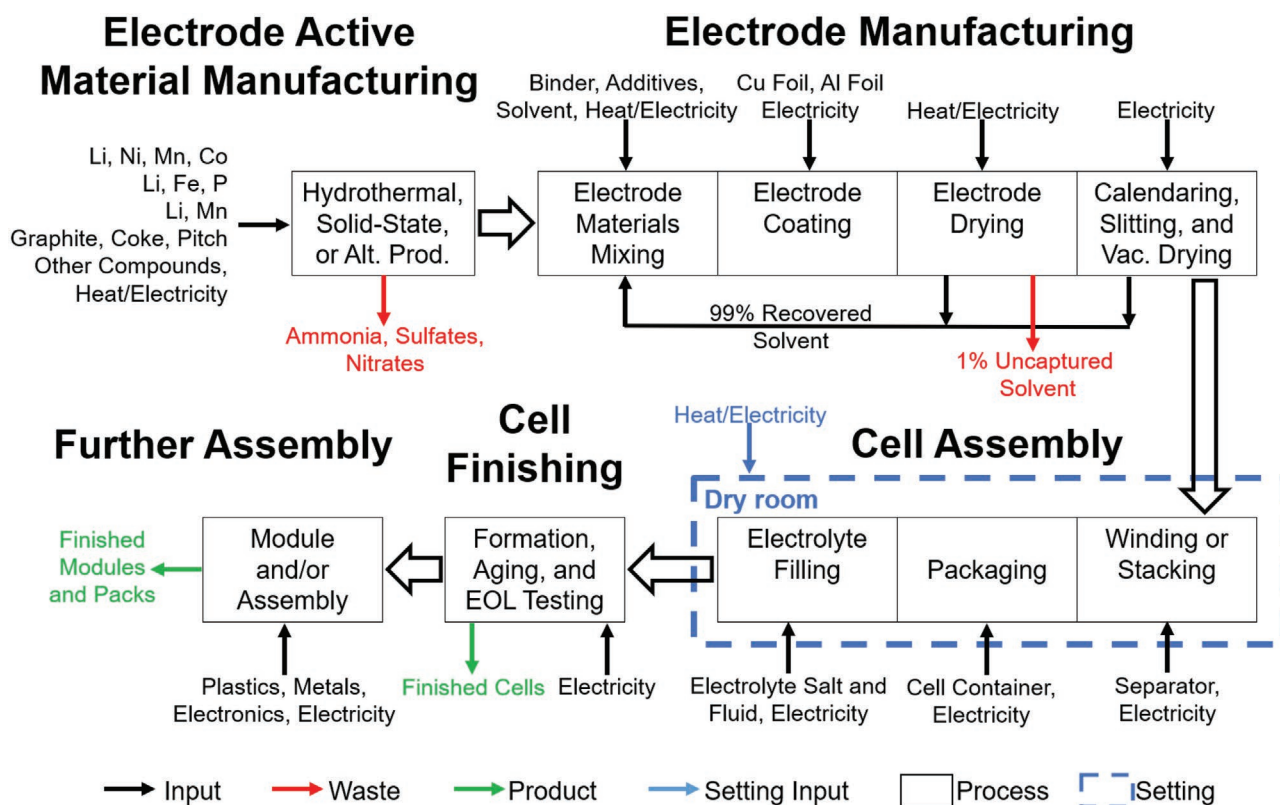


Figure 2. Li-ion battery production process flow diagram.^[26,82,90,92,93]

cell type but may vary by battery chemistry. Cell assembly and cell finishing are typically independent of battery chemistry, but vary by cell configuration. Cylindrical and prismatic cells follow the same manufacturing procedure with some minor differences, while pouch cells deviate from this procedure by requiring the stacking of electrodes and separators instead of winding.^[90] While some manufacturing steps do vary by the cell configuration, the main processes contributing to energy consumption and environmental impacts, NMP solvent evaporation and dry room conditioning, are common across all cell configurations. This is not to say that cell configuration is negligible with regards to the environmental impact of the Li-ion battery and its manufacturing. Ciez et al.^[91] compare pouch and cylindrical cells for NMC, NCA, and LFP batteries and the results suggest that pouch cells are consistently ≈10% less GHG-intensive than cylindrical cells, enabled by the lower ratio of cell hardware to energy stored per cell. LCAs should clearly identify which cell type(s) are being analyzed and potentially explore multiple cell types to capture this variation in material requirements and environmental impacts.

Although battery manufacturing involves many different processes, the majority of energy used in most battery manufacturing plants operating today is dominated by two key activities, namely evaporating the NMP solvent and maintaining the facility's dry room.^[52,92] Each of these processes represents around 40% of the total energy use associated with cell manufacturing, together consuming around 80% of the total energy consumed.^[52] Evaporating NMP (for eventual condensation and recovery) is energy intensive because of the air flow needed to maintain a safe concentration of the flammable solvent; the result is that facilities use 45 times the minimum thermal energy expended to vaporize NMP for this drying process.^[93] This explains why some studies have suggested water-based cathode deposition can save energy and costs,^[94,95] although so far the industry continues to rely on NMP. The dominance of NMP evaporation and dry room conditioning also explains why most studies do not provide clear distinctions between electricity needs and primary fuel needs; dehumidification can be achieved through the use of varying amounts of electrical and thermal energy. Dunn et al.^[46] have suggested that electrode manufacturing is also an important contributor, and may comprise a larger fraction of energy use than dry room conditioning for large facilities. Beginning with these basic facts, we will discuss the underlying reasons why prior battery LCAs have produced manufacturing energy demand estimates that vary by multiple orders of magnitude, and what can be done to address these discrepancies in future studies.

2.2.1. Cathode Material Manufacturing

Dunn et al.^[46] suggested that cathode material production can be the largest or second largest contributor to energy use at battery manufacturing facilities under some conditions. Their results suggest that NMC cathode materials are more energy-intensive to make than LFP and LMO by a factor of two to three. For NMC cathode materials, production typically consists of 2 major phases: coprecipitation and calcination.^[92] Coprecipitation describes the process of reacting dissolved

metal nitrates, metal sulfates, or metal acetates with hydroxide (typically sodium hydroxide) in a solvent to form a mixed metal hydroxide, represented in this paper as $\text{Ni}_x\text{Mn}_y\text{Co}_z(\text{OH})_2$ (where the values of x , y , and z vary). This is performed in a continuously stirred tank reactor under a carefully controlled temperature, pH, and speed setting.^[46] The $\text{Ni}_x\text{Mn}_y\text{Co}_z(\text{OH})_2$ is then isolated and dried via the recovery of the solvent. As a result of coprecipitation, wastewater may be produced containing ammonia and sodium sulfate, which needs to be treated for proper disposal or reuse and could contribute to eutrophication potential. The majority of energy demand associated with coprecipitation is attributable to wastewater treatment (presumably for aeration), with the rest is attributable to the direct production and environmental control of the $\text{Ni}_x\text{Mn}_y\text{Co}_z(\text{OH})_2$ in the reactor as well as drying.^[46,92] Dai et al.^[92] reports that the coprecipitation step consumes 11.8 kWh of heat to produce 1 kg of $\text{Ni}_x\text{Mn}_y\text{Co}_z(\text{OH})_2$, but the quantity of energy consumed will vary by specific conditions.

Calcination describes the final production of the NMC active material through a high temperature sintering of the $\text{Ni}_x\text{Mn}_y\text{Co}_z(\text{OH})_2$ and a Li compound, typically a hydroxide or carbonate.^[46] In theory, this is a two-staged process with the first stage requiring temperatures of 400–500 °C for 4–5 h^[82] and the second stage requiring temperatures of 700–900 °C for 8–10 h.^[46] In practice, different manufacturers may set different temperatures and durations depending on the capabilities of their equipment (typically a heat roller kiln) and the requirements of their manufacturing process. Additionally, more stages of calcination may be required, as the priority of the manufacturers is maximizing yields as opposed to reducing energy consumption.^[92] The heat required to achieve the high temperatures for long durations represents a majority of the energy demand associated with both coprecipitation and calcination.^[46,92] The heat itself may be sourced through thermal fuel inputs or electricity. Dai et al.^[92] reports that the calcination step consumes 7.0 MJ of electricity to produce 1 kg of NMC active material, but the fuel type and quantity of energy consumed will vary by specific conditions.

LFP cathode materials can be produced via a hydrothermal production or a solid state production. The hydrothermal production of the LFP cathode material requires the input of lithium hydroxide, phosphoric acid, and iron sulfate.^[96] Iron sulfate is produced as a waste product of the steel industry and can be assumed to have minimal embedded energy or environmental burden if it is indeed sourced from the steel industry. Otherwise, its impacts must still be considered. These materials are coprecipitated to produce LFP with aqueous lithium sulfate as a byproduct to be removed.^[46] Dunn et al.^[82] reports that the hydrothermal preparation of the LFP cathode active material requires 10.0 kWh per kg of LFP. The solid state production of the LFP cathode material consists of the heating, cooling, and reheating of a Li, Fe, and P compounds, potentially lithium carbonate, iron oxide, and diammonium phosphate.^[96] The first heating stage reaches a temperature between 500 and 700 degrees Celsius and is then cooled to room temperature. The second heating stage reaches a temperature between 700 and 900 °C.^[82] Dunn et al.^[82] reports that the solid state preparation of the LFP cathode active material requires 0.82 kWh per kg of LFP. LFP cathode active material production

is not as well documented in the literature from an energy use standpoint, but the production is generally simpler than NMC cathode active material production. The energy demand for heating may be supplied by thermal fuel inputs or electricity depending on individual site conditions.

LMO cathode materials can be produced via multiple methods including solid state, sol-gel, hydrothermal, and combustion procedures. When referring to LMO cathode production, most literature discusses solid state production. This starts with the multi-stage washing of MnO_2 with an H_2SO_4 solution of which up to 98% may be recovered and reused.^[97] The cleaned MnO_2 is then mixed with lithium carbonate powder and water to be milled, producing a slurry which is then dried at around 150 °C. This dried, homogeneous mixture is then calcined at around 750 °C for 20 h, producing the LMO cathode material.^[97] Once again, the energy demand for heating may be supplied by thermal fuel inputs or electricity depending on the manufacturer. Susarla et al.^[97] describes the energy intensity of LMO production as 5.0 kWh per kg of cathode active material.

2.2.2. Electrode Manufacturing

During electrode manufacturing, the cathode material (usually Li metal oxide particles) is mixed in dry form with a small quantity of carbon black additive (e.g., acetylene black or graphite).^[93] The dry mixture is then combined with a polymeric binder such as polyvinylidene difluoride (PVDF) and the NMP solvent to form a homogenous slurry.^[92,93] This slurry is then intermittently applied to a Cu or Al current collector (Cu for anode, Al for cathode) creating a mother roll and the NMP solvent is evaporated and collected for reuse.^[90] NMP is typically used as the solvent for cathode manufacturing and may have a recovery rate of 99%.^[93] Water is generally used as a solvent for anode manufacturing, which is why NMP is only discussed in the context of the cathode. The mother roll of electrodes is then calendared to achieve a specific electrode porosity and improve cell performance.^[98] Finally the mother roll is split into several daughter rolls and vacuumed dried.^[90] Different battery chemistries may require different types and quantities of active materials, additives, and binders, resulting in unique impacts associated with this phase.^[26,96]

Yuan et al.^[52] reports that the energy use associated with electrode material mixing and coating are minor with their combined energy consumption representing $\approx 1\%$ of the energy consumption associated with battery manufacturing. Additionally, Yuan et al.^[52] reports that calendaring and slitting represent 2% and 4% of the energy use associated with battery manufacturing, respectively. The main driver of energy use and environmental impacts associated with electrode manufacturing is the NMP evaporation and recovery during cathode drying.

Ahmed et al.^[93] explored the process of cathode drying, and energy implications in detail by constructing a process model for a facility producing 100 000 packs per year of 60 kW, 10 kWh LIBs (this translates to 1 GWh per year of battery storage capacity output). In their model, the cathode is sent to a dryer where it is exposed to flowing hot air at 140 °C. NMP concentration in the air must never exceed 1150 parts per million (ppm).

After being cooled in a chilled water condenser, NMP, water, and hydrocarbons are condensed and NMP is recovered via distillation and any remaining NMP is recovered using a zeolite wheel to reach total solvent recovery rates around 99%. The resulting energy demand was estimated at 1470 kW of electricity and 4381 kW of thermal energy (5851 kW in total). Based on 300 operating days per year, this translates to 112 MJ of total energy per kWh of battery capacity produced. The question, however, is how sensitive the NMP drying energy is to uncertain parameters, such as facility size and specific chemistries. Ahmed et al.^[93] also explored the impact of allowable NMP concentration in the dryer outlet, which varies roughly linearly with the flow rate and total energy demand. However, they did not discuss whether different regulatory frameworks might dictate different caps on NMP concentration or what sorts of safety measures could be taken to enable, for example, a doubling of allowable NMP concentration (which would cut energy demand in half). Yuan et al.^[52] suggested that the concentration of the PVDF in the NMP is tied to energy use for NMP recovery, and that reducing PVDF concentration can reduce energy demand for solvent recovery. However, they did not offer a clear explanation as to why this occurs. Yuan et al.^[52] also noted that energy demand for NMP recovery at a commercial-scale facility is considerably lower than a pilot-scale facility.

Dunn et al.^[46] acknowledged the use of NMP as a solvent but do not mention any energy use associated with NMP recovery, which results in a very small battery assembly energy footprint on par with Notter et al.^[50] and these results are likely not representative of the current state of the industry. Dai et al.^[92] does an excellent job of noting that some of the energy use differences across prior battery LCAs is driven by solvent assumptions; Notter et al.^[50] assume water as the solvent for both cathode and anode, Majeau-Bettez et al.^[32] and Ellingsen et al.^[74] assume NMP as the cathode and anode solvent. GREET^[96] assumes NMP for the cathode and water for the anode, which seems to be the most reasonable choice given current industry practices. Given how significant the energy footprint of NMP recovery is, we suggest that any LIB LCA must devote effort to carefully choosing their solvent use assumptions, and conducting sensitivity analysis as appropriate. This is an area where some of the most widely-cited studies have not done an adequate job of exploring and highlighting the impacts of solvent recovery on energy and environmental impacts.

2.2.3. Cell Assembly

Cell assembly occurs in a dry room, which is essential to battery manufacturing. Per Dunn et al.,^[72] this is where electrodes and separators are stacked or wound, current collectors are welded, the cells are enclosed in a container, electrolyte is added, and the cells are closed. While the cell assembly that occurs in a dry room makes up around 5% of the battery manufacturing energy demand according to Yuan et al.,^[52] the conditioning of the dry room itself is a major energy consumer. As Dunn et al.^[46] point out, energy demand for dry room conditioning is throughput- and scale-dependent. Because primary data, particularly in older (>5 year-old)

studies, is most likely to be sourced from small-scale manufacturing facilities, this has resulted in very large energy use estimates. For example, the primary data provided in Yuan et al.^[52] indicates that dry room conditioning is the single largest energy consumer, at 43% of total cell manufacturing energy demand. Those results are based on a facility that is operating at full capacity, but only producing 400 cells per day (at 129 cells per pack). Ahmed et al.^[99] explore the energy implications of dry room conditioning for a much larger facility in detail using a process simulation approach similar to their approach for estimating NMP drying energy.^[99] The model in Ahmed et al.^[99] is based on a facility manufacturing 100 000 automotive battery packs annually with a dry room volume of 16 000 cubic meters. In contrast to Yuan et al.,^[52] the results from Ahmed et al.^[99] indicate energy use for dry room conditioning that is an order of magnitude smaller than the energy required for NMP recovery (400 kW for dry room conditioning, compared with 5851 kW for NMP recovery^[93] for a comparably-sized facility). Dunn et al.^[46] use a similarly small dry room conditioning estimate, meant to represent a very large facility with high throughput.^[99]

From the analysis by Ahmed et al.,^[99] it is clear that any battery manufacturing process requiring a dry room is likely to generate widely differing energy demand estimates, depending on scale, as well as local climate (which impacts humidity in the inlet air) and the technological choices that dictate the fraction of electrical versus thermal energy used in the facility. Assembling a small number of scenarios that represent different facility scales, and generating results for each scenario, would provide much-needed clarity in manufacturing energy use results.

2.2.4. Cell Finishing and Further Assembly

Cell finishing consists of a variety of processes required for the cell to be ready for use. While the processes included in cell finishing may vary by manufacturer, some form of cycling and precharging are generally required for the formation of the cell. This is associated with a small portion of energy consumption, with Yuan et al. reporting around 1% of the energy used for battery manufacturing is attributable to cell finishing. Additionally, while not a step included in cell manufacturing, the finished cells may be further assembled into modules or packs. This requires the interconnection of individual cells, balance of systems to support the cell in this application, and the addition of structural components. Yuan et al.^[52] reports that this additional assembly is associated with a negligible amount of energy consumption when compared to the energy use associated with cell manufacturing.

2.2.5. Variation in Manufacturing Energy Demand Estimates

With the LIB industry in a state of rapid growth and cost reductions, it is not surprising that battery manufacturing energy use estimates have also shifted over time. **Figure 3** shows a downward trend across most of the studies based on primary data, with Pettinger and Dong^[51] as a notable exception. Some of these improvements may be driven by technological advancements, but we hypothesize that much of reductions in energy demand are more likely to be the result of increased facility utilization and scale. If, for example, rapid subsidized growth in LIB manufacturing facilities in China resulted in numerous

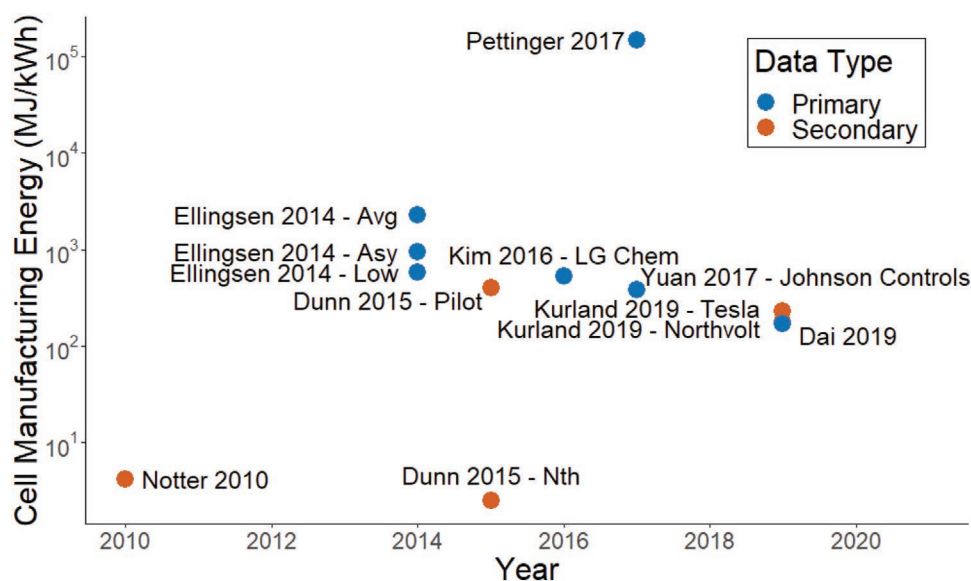


Figure 3. Cell assembly primary energy demand in literature over time. Kurland 2019—Tesla is the estimation of primary energy use at a Tesla, 35 GWh per year manufacturing plant. Kurland 2019—Northvolt is the estimation of primary energy use at a Northvolt, 8 GWh per year manufacturing plant.^[100] Dai 2019 is reporting primary energy at an unspecified 2 GWh year manufacturing plant.^[92] Ellingsen 2014—Low, Ellingsen 2014—Asy, Ellingsen 2014—Avg are the reported primary energy at an unspecified pilot plant representative of the lowest monthly consumption, stated asymptotic consumption, and average monthly consumption respectively.^[74] Notter 2010 is the modeled primary energy use at a pilot scale manufacturing plant.^[50] Dunn 2015—Pilot and Dunn 2015—Nth are the modeled primary energy use at a pilot scale and Nth scale plant respectively.^[46] Yuan 2017—Johnson Controls is the reported primary energy consumption at a Johnson Controls, 0.018 GWh per year manufacturing plant.^[52] Kim 2016—LG Chem is the reported primary energy use at a LG Chem 2 GWh per year manufacturing plant.^[49] Pettinger 2017 is the reported primary energy use at an unspecified pilot plant.^[51]

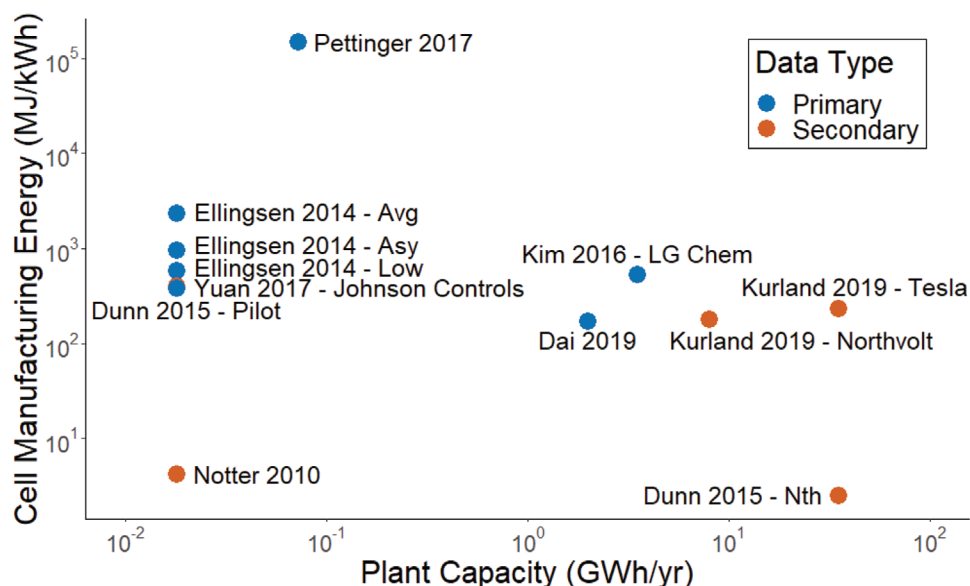


Figure 4. Cell assembly primary energy demand in literature versus plant capacity. Kurland 2019—Tesla is the estimation of primary energy use at a Tesla, 35 GWh per year manufacturing plant. Kurland 2019—Northvolt is the estimation of primary energy use at a Northvolt, 8 GWh per year manufacturing plant.^[100] Dai 2019 is the reporting of primary energy at an unspecified 2GWh per year manufacturing plant.^[92] Ellingsen 2014—Low, Ellingsen 2014—Asy, Ellingsen 2014—Avg are the reported primary energy at an unspecified pilot plant representative of the lowest monthly consumption, stated asymptotic consumption, and average monthly consumption respectively.^[74] Notter 2010 is the modeled primary energy use at a pilot scale manufacturing plant.^[50] Dunn 2015—Pilot and Dunn 2015—Nth are the modeled primary energy use at a pilot scale and Nth scale plant respectively.^[46] Yuan 2017—Johnson Controls is the reported primary energy consumption at a Johnson Controls, 0.018 GWh per year manufacturing plant.^[52] Kim 2016—LG Chem is the reported primary energy use at an LG Chem 2 GWh per year manufacturing plant.^[49] Pettinger 2017 is the reported primary energy use at an unspecified pilot plant.^[51]

facilities operating well below their capacity, this would translate to higher energy use estimates, as dry rooms must continue operating regardless of throughput. As larger facilities are built, and begin operating closer to full capacity, we expect that these estimates will stabilize. It is unclear whether they will achieve some of the more ambitious nth plant estimates documented in Notter et al.^[50] and Dunn et al.,^[46] both of which rely, or appear to rely, on unusual assumptions related to solvent use and recovery that do not reflect current industry practices.

Figure 4, inspired by a plot provided in Kurland^[100] illustrates the impact of facility scale on battery manufacturing energy use estimates. Where plant capacities were not deliberately stated or could not be determined, we used a simple method for approximating the scale: If a plant was described as “pilot,” “pioneer,” or otherwise novel and small, it was assigned the smallest yearly plant capacity observed in this study, 0.018 GWh yr⁻¹. If a plant was described as “nth,” it was assigned the largest yearly plant capacity observed in this study or 35 GWh yr⁻¹. The impacts of plant utilization and economies of scale can be observed with the Ellingsen et al.^[74] and Dai et al.^[92] data points. Ellingsen et al.^[74] examined the per unit energy consumption of a cell manufacturing plant for 18 months: their highest data point represents the average per unit energy consumption for this time, their lowest data point represents the lowest monthly per unit energy consumption, and their central point represents a set asymptotic value. The variation of this energy consumption over a relatively short period of time suggests that the cell manufacturer studied had inefficiencies in their cell manufacturing process, potentially arising from varying utilization levels of the plant. In contrast, the manufacturer that worked

with Dai et al.^[92] claimed to operate at a high energy efficiency, along with having a greater total capacity and operating during a period of greater cell demand.

Most energy demand estimates seem to have fallen in the range of a few hundred MJ per kWh of production. A thorough review of the literature from the perspective of manufacturing energy use and impacts suggests that nearly all LCAs published so far are plagued by unusual assumptions or outright omissions. The question of what a reasonable nth plant manufacturing energy footprint should be, and the savings to be had through very large scale production, remains unanswered in the literature. There is a pressing need for a thorough LCA that adequately addresses all the sources of uncertainty associated with battery manufacturing and its energy demands.

2.2.6. Geographic Representativeness in Battery Manufacturing

A final critique of battery LCAs is that, while most studies capture the appropriate locations and local grid mixes for raw materials extraction and processing, battery manufacturing has largely been modeled based on grid mixes and primary fuel choices appropriate for the location of the study rather than the most likely manufacturing location. Of the studies we reviewed that included the impacts associated with LIB manufacturing, over 30% used electricity mixes and fuel inputs representative of regions in the European Union, 25% used data representative of North America, and less than 15% used data representative of manufacturers in Asia. The remaining 30% did not provide or clearly state the manufacturing location

observed in the study. As of 2019, three quarters of global LIB manufacturing capacity was located in China.^[101] The distinction between battery manufacturing in China versus the US or Europe is important for assumptions about the grid mix as well as the primary fuel(s) likely to be combusted on-site. Coal remains the single largest source of primary energy in the Chinese industrial sector; 39% total coal demand comes from the industrial sector. Direct coal use (excluding coke and coal gas) for industrial facilities exceeds natural gas use by about a factor of four.^[77] The grid mix in China is coal-dominated, with 70% of total generation from fossil fuel-fired power plants, and of that fraction, 91% is coal. However, a consequential LCA may require a more nuanced look at the source of the marginal kWh across the country, as recent capacity investments in solar, wind, nuclear, and hydroelectricity comprise more than half of all new generating capacity.^[77]

The degree to which any given LCA should reflect current manufacturing practices and locations depends on the type of question it is seeking to answer. If the analysis is meant to provide insight into an early-stage battery technology, establishing clear, simple assumptions (e.g., natural gas as the sole source of primary fuel consumption and an average US grid mix) may be sufficient. Varying grid mixes and on-site primary fuels can always be explored through sensitivity analysis. However, for LCAs that seek to offer insights into how EVs broadly compare with other technologies or information on the net impact of widespread adoption of grid-connected stationary batteries, future studies must consider scenarios that reflect the supply chains currently in place. This means incorporating typical practices at Chinese facilities, at least as one of a set of scenarios.

2.3. Use Phase and End-of-Life

2.3.1. Use Phase

Many papers do not consider the use phase of an LIB when performing and LCA, often citing the uncertainty and complexity of battery performance and lifetime (see Table 5). However, accounting for different roundtrip efficiencies and lifetimes is essential when comparing different battery technologies.^[10,31,34,51,102–104] Other characteristics may be more or less relevant, depending on the specific application. For example, pack weight will impact vehicle efficiency in an electric car, truck, or aircraft, while weight is far less relevant for stationary applications. Table 1 presents several metrics used to describe an LIBs performance as it varies by battery chemistry, namely the battery's cycle life and shelf life.

Cycle life is defined as the number of charge/discharge cycles a battery can perform under defined conditions before its storage capacity degrades to a specified condition, typically 80% of its original capacity for EVs and 60% for stationary storage. A battery's actual cycle life will be impacted by its operating conditions, and when data is available, should be adjusted based on the expected use case before calculating lifetime energy throughput. Battery operations at extreme temperatures contribute to battery aging. Higher temperatures result in more efficient and faster reactions, but the aging reactions are also enhanced at high temperatures. Low operating temperatures may cause electrode materials to contract, reducing the available space for Li-ion insertion on the anode and potentially contributing to lithium plating, a major contributor to capacity fade.^[18,25,105,106] Battery operations at high or low state of charge

Table 5. Life-cycle assessment studies sorted by system boundary and application area.

		Scope					
Application	Sub-phase/ component	Cradle-to-gate	Cradle-to-gate and use	Cradle-to-grave	Cradle-to-grave (w/o use)	Review papers	
Automotive	Hendrickson et al. 2015	Wu and Kong 2018	Majeau-Bettez et al. 2011	Dunn et al. 2015	Sun et al. 2020	Nealer and Hendrickson 2015	
	Golroudbary et al. 2019	Jiang et al. 2020	Bekel and Pauliuk 2019	Notter et al. 2010	Wang and Yu 2020	Ellingsen et al. 2017	
		Ellingsen et al. 2014		Wang et al. 2020	Raugei and Winfield 2019	Nordelöf et al. 2014	
		Kim et al. 2016		Zhao and You 2019	Dunn et al. 2012		
		Cobas-Flores et al. 1996		Zackrisson et al. 2010	Ciez and Whitacre 2019		
				Deng et al. 2017*			
				Deng et al. 2019			
				Zackrisson et al. 2016*			
				Li et al. 2014			
				Longo et al. 2014*			
Stationary		Le Varlet et al. 2020	Hiremath et al 2015	Ryan et al. 2018		Pellow et al. 2020	
				Jenu et al. 2020			
				Weber et al. 2018*			
				Vandepaera et al. 2017			
				Jones et al. 2020			
				Chowhurdy et al. 2020			
				Spanos et al. 2014			
Unspecified/ general/other	Wang et al. 2019	Wang et al. 2019	Peters et al. 2016	Lankey and McMichael 2000	Ahmadi et al. 2015	Peters et al. 2017	
	Cusenza et al. 2019	Cusenza et al. 2019	Casals et al. 2015		Cicconi et al. 2012	Sullivan and Gaines 2012	
	Sathre et al. 2015	Sathre et al. 2015	Ioakimidis et al. 2019		Bobba et al. 2018		
	Kamath et al. 2020	Kamath et al. 2020					

*Li-ion not primary battery technology evaluated.

(SOC) also contribute to accelerate battery aging. At the extreme ends of the SOC, active material loss in the positive electrode is a main driver of increased battery aging.^[18,25,105,107,108] The shelf life metric is defined as the number of years before a battery degrades to a specified condition while remaining inactive (e.g., in very underutilized batteries). These high-level metrics capture the combined effects of multiple battery degradation mechanisms on capacity fade^[105,109,110] and can be useful in estimating the total energy discharged over a battery's lifetime as a useful functional unit for LCAs.

Focusing exclusively on throughput (kWh discharged over a battery's lifetime) as a functional unit in LCA fails to account for another crucial dimension of the use phase: battery efficiency. A small fraction of energy in batteries cannot be recovered due to irreversible side reactions. Coulombic efficiency represents idealized slow charging and discharging, and is ~99% for LIBs, but in practice, the ratio of total charge extracted from the battery to total energy put into the battery during charging depends on the charge and discharge C-rate and temperature, and is generally closer to 90%.^[111] Frequent use of ultra-fast charging, for example, will decrease battery efficiency, as will rapid discharging. This means that the manner in which a battery is cycled will impact total throughput and energy losses over its lifetime. Round-trip efficiency can refer to the efficiency of a pack or system, as opposed to individual cells, but this terminology is used inconsistently. When incorporating energy losses during charging/discharging, researchers must be careful to avoid omitting or double-counting energy lost in the battery itself in addition to losses at inverters or in parasitic loads, such as thermal management systems. To further complicate matters, battery efficiency decreases over time, although capacity fade is generally the determining factor in deciding to decommission a battery.^[105,112] Impedance is often used as an indicator of LIB health because it can capture the effects of many of the mechanisms that drive capacity fade, power fade, and reduced efficiency,^[105,112,113] but this metric cannot be easily converted to any of the practical measures of efficiency and lifetime throughput that are needed to account for use-phase performance in an LCA.

Given the complexity of modeling battery behavior, and the lack of performance data from real-world applications, it is not surprising that many battery LCAs do not incorporate the use phase and most ignore battery aging. Sathre et al.,^[114] which focused on second life applications for LIBs from EVs, performed a sensitivity analysis to identify the battery parameters and their influence cumulative energy balance and cumulative greenhouse gas emissions. They noted considerable uncertainty in the timing at which LIBs reach an inflection point, when capacity fade accelerates. Hiremath et al.^[31] portrayed the different life-cycle impacts associated with LIBs used in different stationary storage applications. Their analysis provided the power rating, discharge duration, energy rating, and cycle frequency for multiple stationary storage applications, finding that use phase greenhouse gas emissions per MWh delivered can vary by nearly two orders of magnitude between applications. Longo et al.^[115] compared two theoretical EV batteries with one having a cycle life of 3000 cycles and a cycling frequency of 2 cycles per day, and the other having a cycle life of 3500 cycles and a cycling frequency of 1.6 cycles per day. This

resulted in a 5% difference in global warming potential during the use phase of these two battery scenarios. Although a more thorough accounting of use-phase cycling and its impact on lifetime and efficiency would be ideal, reliable data for use in LCAs is rare. Future studies would benefit greatly from a set of standardized scenarios that capture variations in C-rate, operating temperatures, SOC, and the expected impacts on capacity fade, battery lifetime, and efficiency.

2.3.2. Battery End-of-Life and Recycling

Once a battery has reached its EOL, it must be safely disposed of or recycled. Incorporating reuse and recycling has long been a methodological challenge in LCA, raising questions of how credits for recovered materials, and the resulting avoided impacts of virgin material production, should be allocated.^[116] Recycling is categorized as closed-loop, meaning materials are recycled within the same production system (e.g., cathode materials recovered for use in new cathodes), or open-loop, where materials are recovered for use in other production systems. For batteries, most studies take a closed-loop approach to recycling and they explore one or more of the three main recycling approaches: pyrometallurgical, hydrometallurgical, and direct recycling.^[46,50,72,91,117,118]

Pyrometallurgical recycling is a smelting process capable of recovering transition metals, namely Co, Ni, and Cu, and is used to recycle LIBs as well as Ni Metal-Hydride (NiMH) batteries.^[119] Other materials are oxidized in the process to provide process heat and are not generally recovered; this includes Li and Al. Hydrometallurgical and direct recycling, which are based on leaching and physical separation processes, respectively, recover a greater fraction of battery materials by mass. Both processes are designed to recover the cathode (including Li, in addition to metals like Co or Mn), Al, and the anode, while only direct recycling can recover the electrolyte (through flushing of cells).^[119] With the exception of water use, hydrometallurgical recycling achieves greater savings across a wide range of life-cycle inventory metrics relative to pyrometallurgical recycling.^[117] Direct recycling is more challenging to compare because it is less commonly used and the process configuration and materials recovered vary. However, as Gaines points out, there is more of a continuum than a clear distinction between hydrometallurgical and direct recycling; as the Co content of LIBs declines, a hybrid direct/hydrometallurgical approach may become preferable to a pyrometallurgical process.^[119] Although not the focus of this study, EV batteries have the potential to be tested, refurbished as needed, and extend their service life in stationary storage applications.^[47,48,114,120–123] As noted in Sathre et al.,^[114] there are additional impacts associated with configuring vehicle batteries for use in stationary applications, including the installation and use of cell cooling systems, and capacity can decline rapidly once the battery reaches its inflection point.

The attractiveness of these recycling processes is ultimately contingent upon good use-phase performance of the recovered materials, and this is perhaps most uncertain with direct recycling processes. If recovered cathode materials cause a decrease in cycle life or round-trip efficiency, such impacts

could easily negate the benefits of offsetting raw material production. LCA studies focused on battery recycling to-date have relied in the assumption that recycled materials are functionally equivalent to new materials. This is understandable, as empirical data to support any other assumption is scarce or non-existent. An additional challenge is the establishment of a clear business-as-usual case for use as a baseline for comparison. The quantity of stationary and EV LIBs reaching their EOL remains small and recycling and disposal practices vary by country. Globally, it is estimated that 95% of LIBs are not recycled.^[124] As demand for energy storage in EV and stationary energy storage applications grows and batteries continue to reach their EOL, additional studies will be needed to track the date of these batteries and establish a clearer understanding of what processes are being used and what materials are ultimately recovered.

3. Recommendations for Battery LCA

It is surprising that, despite the publication of LIB LCAs dating back more than a decade, we are unable to point to a single study that sets the standard for best practices in battery LCAs. This is not meant to suggest that prior studies have not offered value and insight to the research community; the most widely-cited studies often perform well in one or more dimensions, but each has its own drawbacks. Most of the published LCAs have provided detailed data on the environmental impacts of raw materials extraction and processing.^[32,46,50,72,74] The shortcomings in our understanding of raw material extraction and processing are twofold. First, the studies we surveyed did not adequately account for geographic variation in mining practices and variations in the exposure risk for nearby populations. Some mining operations that comprise a minority share of production are likely to drive an outsized fraction of overall environmental impacts because of local conditions and practices. We liken these operations, which may be informal or loosely regulated, to the concept of “superemitters” in the natural gas industry.^[85] Second, there are inherent limitations in the underlying midpoint and impact methodologies; it is impossible for any LCA to conduct detailed fate and transport modeling for every emission to air, soil, and water, so studies rely on regional or global average factors that are likely to be one or more orders of magnitude different from the actual values. These uncertainties are compounded by the fact that documentation of where specific waste streams are discharged from mining and material processing operations is sparse. Moreover, we have yet to find any study that explores the differences between average, marginal, and incremental sources of key material inputs, and the implications for mining and processing-related energy use and emissions. This seems to be an obvious gap in the literature, and one that could be filled with data and market projections that are available today.

Achieving consensus and clarity in battery manufacturing energy use and impacts is where prior studies largely fall short. Because there is little evidence to suggest appreciable non-combustion emissions to air, water, or soil during manufacturing, nearly all direct environmental impacts from this stage are expected to be tied directly to on-site combustion of fuels and

emissions from electricity generation. Dunn et al.^[46] provided the first clear and compelling discussion of dry room conditioning in an LCA context, and the reasons behind large differences in reported energy use, but did not provide a similarly detailed exploration of energy used for NMP recovery. Although its scope was more limited, focusing only on NMC-graphite LIBs, Dai et al.^[92] did provide a more thorough exploration of both dry room conditioning and NMP recovery. Battery recycling, by comparison to battery manufacturing, is relatively well studied and there is better agreement across the literature, although battery recycling LCAs must rely largely on estimated or simulated mass and energy balances because of the limited number of LIBs being recycled.^[46,72,117] It is possible that, when primary data becomes more widely available, it will reveal inconsistencies between simulations and primary data similar to those found in battery manufacturing. Although battery technologies will continue to evolve, and there will continue to be disagreements between primary and secondary data sources, we hope to provide recommendations for approaching these uncertainties in a manner that makes each study more interpretable, and simpler to replicate and update as battery technologies and the infrastructure supporting their production continues to develop.

3.1. Defining Appropriate System Boundaries

Defining the system boundaries requires researchers to weigh the value of comprehensiveness against the downsides of incorporating more assumptions that are not central to the battery technology itself. Expansive system boundaries that include the use- and end-of-life phases will result in the most complete assessment of the net environmental impacts. Inclusion of these phases can alter the conclusions of battery technology (or recycling technology) comparisons; if a less resource-intensive battery technology or directly-recycled cathode material results in reduced battery cycle life, a cradle-to-grave analysis captures these important differences. A cradle-to-gate analysis using only kWh of battery capacity as the functional unit, in this case, would be misleading. Similarly, a battery that relies on a larger quantity of Co may appear to be at a greater disadvantage in a cradle-to-gate analysis, but Co also has a higher likelihood of being recovered and this recovered material can offset the impacts of raw Co extraction and processing.

One can make a similarly compelling argument that cradle-to-grave LCAs carry, in some cases, considerable downsides. Tying results to a specific use case, can make results nearly impossible to compare across studies. This is especially true for stationary energy storage applications, where specific configurations and services provided vary. The layers of assumptions and uncertainty introduced while incorporating the use and EOL phases can dilute what might otherwise be a rigorous and clearly defined analysis of battery production impacts. For example, in analyses conducted based on novel battery technologies, the most viable use case may not be known and use phase performance is uncertain. For the purposes of cross-study comparisons and improving reproducibility of results, reporting cradle-to-gate results separately is valuable regardless of the

study's overall system boundaries. In cases where use-phase performance is not known, the development of alternative functional units and scenario analysis can ameliorate some of the drawbacks of the constrained system boundaries, as discussed further in Section 3.3 below.

3.2. Selecting Relevant Environmental Metrics

Above all, we recommend that future studies consider the goal and scope of the study, in addition to the availability of adequate quality data when selecting environmental metrics (inventory, midpoints, and endpoints). Presenting a long list of impacts without context or uncertainty analysis is likely to generate confusion and offer limited value to the broader research community. Omitting an impact category may lead readers to believe it is not important, but producing results that are inaccurate and/or convey false precision may lead to the same outcome. Making additional efforts to highlight the key drivers of each impact is also critical. There is an important distinction between environmental impacts tied to energy use (and combustion to generate that thermal or electrical energy) and impacts tied to other activities, such as non-combustion pollutant releases to water, air, or soil or depletion of finite resources through mining activities. GHG emissions and 100-year GWP for LIB production are dominated by combustion-related CO₂.^[11] The same is true for human health impacts from other air pollutant emissions, including particulate matter (PM), SO_x, and NO_x, and terrestrial acidification potential, which is driven by SO_x emissions.^[11] We strongly recommend that future LCAs make an attempt to separate impacts tied to energy use with those tied to other activities. If facilities shift their fuel use to lower-emission alternatives (e.g., from coal to natural gas, or natural gas to renewable fuels), making this distinction in published LCAs will make it easier to adjust the results accordingly. Furthermore, denoting which impacts are dependent on assumed grid mixes will make the use of LCA data for future studies considerably simpler; future researchers may wish to update underlying grid mixes or select mixes that are more representative of where production occurs. Emissions may also evolve depending on the tightening or loosening of emissions regulations in the location selected for analysis. Given the rapid decarbonization of electricity occurring in many countries, it is imperative that researchers be given the opportunity to update our understanding of battery production impacts in the context of current and future grid mixes.

As discussed previously, most of the non-energy-related environmental impacts in the life cycle of LIBs are tied to mining and material processing operations. There are various midpoint and endpoint metrics aimed at characterizing depletion of non-renewable resources. We argue that these multipliers fail to capture the nuances of some of the key inputs to battery production, where availability itself may be a secondary or tertiary concern and the more likely outcome is a long-term shift toward more costly and energy-intensive extraction methods. Rather than attempting to quantify resource depletion in a single metric, we recommend that future LCAs develop a set of current average, marginal, and incremental scenarios for the recovery and processing of a few key material inputs (including

Li, for example) and use these scenarios to illustrate the long-term implications of continuing to extract these materials without recovering and recycling them at the battery end-of-life. The other impact most relevant to raw material extraction we have discussed here is eutrophication potential. We hesitate to recommend that this metric be included in future studies, in part because the required data on relevant waste stream discharges may not be of sufficient quality to draw meaningful conclusions from the results. Ozone depletion potential, which we briefly touch on, is not likely to be a useful metric to quantify given the ongoing phase-out of ozone-depleting substances such as CFC-11.^[88] Human toxicity does not feature prominently in non-energy-related impacts within the studies we surveyed, but as noted earlier, published values regularly rely on data provided by large mining companies^[86] and those datasets likely reflect best industry practices and fail to account for the impact of outliers, particularly in artisanal and small mining operations. Omitting such outliers is a known problem in emissions and environmental impact accounting^[85] and must be mitigated in future LCAs, particularly in the context of material extraction and processing.

Water consumption and withdrawals associated with battery production can be substantial, yet it is often overlooked in LCAs.^[68] Battery electric vehicles are associated with over 50% more water use relative to internal combustion engine vehicles over the course of their lifetime. This is mostly associated with the electricity use associated with vehicle charging, but a large contribution of water consumption is attributable to the LIB itself, consisting of 5–10% of the total water consumption depending on the battery chemistry.^[125] It is estimated that 752 liters of water are consumed per kWh of Li-ion battery pack produced, with roughly 50% of this attributable to aluminum used as housing and 30% attributable to the cathode active material for NMC-111 cathodes.^[92] Electrolysis during aluminum production is responsible for 65% of the water use attributable to aluminum production if produced through thermal power,^[126] and mineral extraction is responsible for a large majority of water use attributable to the cathode active material. In particular, Co production consumes the most water, representing 50% of the embedded water consumption in NMC-111 cathode active materials.^[92] Water use, while not the central focus of most battery LCAs, is worthy of further exploration for studies seeking to broaden their scope beyond energy use and GHG emissions, particularly if the values can be weighted based on a water stress index or similar metric aimed at capturing local water scarcity impacts.^[127]

3.3. Defining Functional Units for Analysis

Defining functional units for battery LCA presents a challenge; the closer the functional unit is to representing the actual service a battery provides (e.g., powering a vehicle to travel one km), the more underlying assumptions, none of which are standardized, must be made. This makes cross-comparisons in the literature labor-intensive or impossible. Conversely, it is common for studies to report results per kg of battery mass,^[32,46,74] which has no direct relationship to the service a battery provides but it does provide for straightforward

comparison across multiple studies. Functional units used in prior studies include battery or material mass (kg), individual battery pack, energy capacity (kWh of battery capacity), energy throughput (kWh passed through the system over the battery lifetime), and distance driven (km) for battery electric vehicles. Ellingsen et al.^[74] set a useful precedent in reporting their results across multiple functional units (per pack, per kWh, and per kg of battery mass).

We suggest that the time has come to phase out the use of kg of battery mass as a functional unit in LCAs. Normalizing results per kWh of battery capacity offers similar potential for cross-comparison while also serving as a more logical functional unit because storage capacity is tied to the core service provided by rechargeable batteries. Normalizing LCA results in terms of lifetime energy throughput is another compelling alternative because it incorporates differences in cycle life and round-trip efficiency that a per unit-capacity analysis does not. One can imagine fascinating assessments of the tradeoffs between, for example, a shift from NMP to aqueous solvents and the resulting impact on lifetime kWh throughput if phasing out NMP negatively impacts cycle life. As long as underlying assumptions about cycle life are clearly documented, we suggest that studies would be well served to report results in these two formats (per kWh of capacity and per kWh of lifetime throughput). This being said, studies aimed at capturing the contribution of battery-related impacts in the context of a larger system may justifiably choose to report their results differently, including per-km traveled or at the individual pack level.

3.4. Recommendations for Future Work

Although we are not the first to highlight the difficulties in achieving consensus on methods for conducting battery LCAs,^[7–11] we hope this review provides the most comprehensive exploration of the underlying reasons behind inconsistent battery LCA results. Quantifying the environmental impacts of battery production can seem enormously complicated and we recommend that future studies simplify and prioritize their efforts based on the processes and materials that are the largest contributors. Comprehensiveness has value, but it must be acknowledged that it also comes at a cost; selecting a large number of inventory, midpoint, or endpoint categories increases the likelihood that researchers will rely on inventory data and characterization factors that are not technologically, temporally, and/or geographically correlated with the details of the study. Selecting life-cycle inventory, midpoint, and/or endpoint metrics that are likely to yield the greatest insights (and have sufficiently high quality data to support those conclusions) will make future battery LCAs more interpretable and impactful. We also urge researchers go beyond representation of industry best practices and develop datasets that capture outliers or “superemitters,” particularly in mining and material processing. Disaggregating environmental impacts by location and type of operation can provide better transparency and accuracy, and also establish a framework by which companies that carefully manage their supply chains to avoid such suppliers are able to be recognized in their estimated environmental footprints.

Improving the interpretability and impact of future battery LCAs will also require that every study conduct a sensitivity analysis across a range of manufacturing facility scales. It is clear from our review of the literature that this point, and the resulting deviations in estimated manufacturing energy use, causes more confusion than any other parameter. We recommend that future LCAs define two or three facilities scales, on the order of 0.1, 1, and 10 GWh per year of battery capacity output and generate results across these different scales. Specific facility scale scenarios could be chosen based on economic “tipping points” for a change in design of, for example, the NMP recovery system or dry room conditioning equipment. Clearly indicating the likely breakdowns of thermal energy use versus electrical energy will also provide enormous value, as many studies do not differentiate between the two. An LCA that makes use of market reports to estimate global-average energy use for battery manufacturing, and ideally projects potential trends, is also sorely needed to illustrate the gap between the current literature and current/future practices in industry.

A final conclusion from this review is that a rigorous, complete cradle-to-grave LCA of multiple battery technologies can be made more tractable by the production of consensus-based scenarios to address some of the major sources of uncertainty for these analyses. Specifically, scenarios that capture critical raw material availability, the geographic distribution of near- and long-term sources, and any expected shifts in extraction/processing methods would reduce reliance on sub-standard data sources and enable easier cross-comparisons between different battery studies. The same is true for the battery use-phase; most LCA researchers and practitioners do not have the resources and subject matter expertise to develop detailed scenarios for battery cycling, operating temperatures, and SOC, nor can such a scenario easily be translated to expected shifts in capacity fade, efficiency, and lifetime. However, if a collection of experts were able to devise a set of scenarios that reflect the most likely use cases for batteries in transportation and stationary applications, these would be widely used and further improve the ability to compare studies and externally validate results. Ambitious harmonization projects are not unheard of^[128] and, through a partnership between systems analysis experts and technology experts, the community can ensure that future analyses of battery technologies further our understanding of their impacts on the environment.

Acknowledgements

Funding for this work was provided by the Energy & Biosciences Institute (EBI) through the EBI-Shell program. This work was also supported by the U.S. Department of Energy (DOE) Office of Science, Advanced Manufacturing Office, Office of Electricity, and the Office of Energy Efficiency and Renewable Energy’s Strategic Analysis Team under Contract No. DE-AC02-05CH11231. The United States Government retains and the publisher, by accepting the article for publication, acknowledges that the United States Government retains a non-exclusive, paid-up, irrevocable, world-wide license to publish or reproduce the published form of this manuscript, or allow others to do so, for United States Government purposes.

Conflict of Interest

The authors declare no conflict of interest.

Keywords

carbon footprint, environmental impacts, life-cycle assessments, lithium-ion batteries

Received: March 6, 2021

Revised: June 23, 2021

Published online:

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