Identifying key assumptions and differences in life cycle assessment studies of lithium-ion traction batteries with focus on greenhouse gas emissions

Linda Ager-Wick Ellingsen[1], Christine Roxanne Hung, Anders Hammer Strømman

Norwegian University of Science and Technology, Program for Industrial Ecology and Department of Energy and Process Engineering, Sem Sælandsvei 7, NO-7491 Trondheim, Norway.

[1]Corresponding author, e -mail: linda.a.ellingsen@ntnu.no

Abstract

The various studies that consider the life cycle environmental impacts of lithium-ion traction batteries report widely different results. This article evaluates the inventory data and results to identify the key assumptions and differences in the studies. To aid the identification, we compile the reported life cycle greenhouse gas emissions of batteries. The studies find production-related emissions in the range of 38-356 kg CO_2-eq/kWh. One of the main sources of the large variations stems from differing assumptions regarding direct energy demand associated with cell manufacture and pack assembly. Further differences are due to assumptions regarding the amount of cell materials and other battery components. The indirect emissions associated with the use phase depend on the conversion losses in the battery, the energy required to transport the weight of the battery, and the carbon intensity of the electricity. Of the reviewed studies assessing the use phase, all estimate energy use associated with conversion losses while only one considers the mass-induced energy requirement. Although there are several industrial end-of-life treatment alternatives for lithium-ion batteries, very few studies consider this life cycle stage. Studies using the "recycled content" approach report emissions in the range of 3.6-27 kg CO_2-eq/kWh battery, while studies using the "end-of-life" approach report emission reductions in the range of 16-32 kg CO_2-eq/kWh battery. The uncertainty associated with the end-of-life results is high as the data availability on industrial process is limited. Based on our findings, we discuss how the life emissions of lithium-ion traction batteries may be reduced.

Keywords: life cycle assessment, life cycle inventory analysis, lithium-ion traction battery, electric vehicle, greenhouse gas emissions

Abbreviations

BEV	battery electric vehicle
EOL	end-of-life
GHG	greenhouse gas
LCA	life cycle assessment
LCO	lithium cobalt oxide
LFP	lithium iron phosphate
LMO	lithium manganese oxide
LTO	lithium titanium oxide
NCA	lithium nickel-cobalt-aluminum oxide
NCM	lithium nickel-cobalt-manganese oxide

PHEV	plug-in hybrid electric vehicle
SiNW	silicon nanowire

1 Introduction

Transport-related greenhouse gas (GHG) emissions have more than doubled since 1970, and have increased at a faster rate than any other energy end-use sector. The transport sector consumed over half of global primary oil and was responsible for nearly one-fourth of global energy-related CO_2 emissions in 2010 (Sims et al., 2014). Light duty vehicles were responsible for around half of the total transport energy use. From the current number of around one billion vehicles (Sousanis, 2011), the total light duty vehicle ownership is expected to double in the next few decades (IEA, 2009). These patterns forecast a dramatic increase in gasoline and diesel demands, and have implications for climate change, urban air quality, and energy security. The projected increase in GHG emissions makes it particularly difficult for the transport sector to reduce its emissions and oil dependency, and this has led to policies that mandate more stringent fuel economy standards and encourage alternative drivetrain configurations and fuels (Wallington et al., 2016). Electric vehicles have emerged as strong candidates among the available transport alternatives (Hawkins et al., 2012). Compared to conventional vehicles, electric vehicles can offer advantages in terms of powertrain efficiency, maintenance, and reduced tailpipe emissions.

Understanding the system-wide trade-offs of replacing conventional vehicles by electric vehicles requires a life cycle perspective. Environmental trade-offs that arise from the change in powertrain configuration are best analyzed using life cycle assessment (LCA) (Nealer and Hendrickson, 2015). As lithium-ion battery cells offer an unmatchable combination of high energy and power density, it makes them the battery of choice for electric vehicles (Nitta et al., 2015). Several studies have assessed the production impact of lithium-ion traction batteries (LIBs) as part of a battery electric vehicle (BEV), a plug-in hybrid electric vehicle (PHEV), or as its own product. Studies have mainly assessed LIBs with a graphitic anode in combination with a cathode of either lithium nickel-cobalt-manganese oxide (NCM), lithium iron phosphate (LFP), lithium nickel-cobalt-aluminum oxide (NCA), lithium manganese oxide (LMO), or a blended LMO-NCM cathode material. In addition, studies have also assessed a lithium titanium oxide (LTO) anode in combination with an LFP cathode and a silicon nanowire (SiNW) anode in combination with an NCM cathode. In contrast to production, the use phase and end-of-life (EOL) treatment of the battery are only evaluated in a few studies. Although several LCA studies have assessed LIBs, these assessments find significantly different results. Thus, there is much uncertainty associated with the data and results, making it difficult to provide direction for reducing environmental impacts of LIBs. Moving forward, it is important to understand why the studies obtain such widely different results. The main objective of this article is to identify the key assumptions and differences between the various LCA studies on LIBs. This will also allow us to identify potential issues that should be considered in future studies on LIBs and point out where further work is needed.

In this article, we considered LCAs of LIBs from various literature sources. Studies assessing only the LIB as well as those examining BEVs and PHEVs studies were evaluated. Unfortunately, few of the BEV and PHEV studies provide a transparent inventory or a detailed contribution analysis of the LIB. Furthermore, many of these studies base their battery inventory on previously published studies and therefore do not contribute new data. Although there are fewer studies that assess only the LIB, these

studies more often include inventory data and a detailed contribution analysis. We mainly considered studies published in peer-reviewed journals, but we also included three grey literature cradle-to-gate studies. The first of these is the Volkswagen assessment of the battery used in the electric Golf (Volkswagen AG, 2012). Volkswagen has a long tradition of performing LCAs of their various vehicle models, and their reports are certified according to the ISO 14040 and 14044 standards. The other two studies were performed by the Paul Scherrer Institut (Bauer, 2010) and the United States Environmental Protection Agency (USEPA; 2013), two institutes that have extensive experience with LCA. To more easily pinpoint differences between the reviewed studies, we collected the reported cradle-to-gate results. Although most of the studies considered several different types of emissions, we limited our presentation to GHG emissions as global warming potential is the most consistently reported environmental impact category in the reviewed literature. Because the studies report GHG emissions based on different functional units, we recalculated the emissions for a common functional unit of 1 kWh of battery capacity. Even though there are much fewer studies that assess the use phase and EOL, which simplifies the search, we recalculate the reported GHG emissions for these life cycle stages where possible.

This article is divided into four sections, including this introductory section. In section 2, we examine the underlying assumptions and key parameters to uncover the causes of discrepancies in reported results. Section 3 discusses our findings, distil the information from the LCA literature, and use this to suggest measures that can succeed in reducing life cycle GHG emissions of LIBs. Finally, section 4 summarizes the most important findings, discusses knowledge gaps, and provides directions from the literature.

2 Life cycle inventory data and reported results

In the text below, we present the results and examine the life cycle inventories from the various studies. We start by reporting the compiled GHG emissions associated with production. Using the emissions as a starting point, we seek to identify and discuss key assumptions and differences among the various studies. Then, we examine the use phase and EOL treatment.

2.1 Production

The different studies vary in how they report the breakdown of the GHG emissions due to production. Where possible, we disaggregated emissions associated with cell materials (dark blue), other battery components (pale blue), cell manufacture (dark green), battery pack assembly (pale green), and transport (grey). For studies where fewer details are provided, we reported aggregated emissions associated with production of cell materials and battery components combined (blue, striped), and aggregated emissions associated with direct energy demand in cell manufacture and battery pack assembly combined (green, striped). For two studies, we were unable to disaggregate into component- and energy-related emissions and therefore, these results are reported aggregated battery pack emissions (turquoise). Figure 1 presents the results in terms of kilograms of carbon dioxide equivalents per kWh of battery capacity (kg CO_2-eq/kWh).

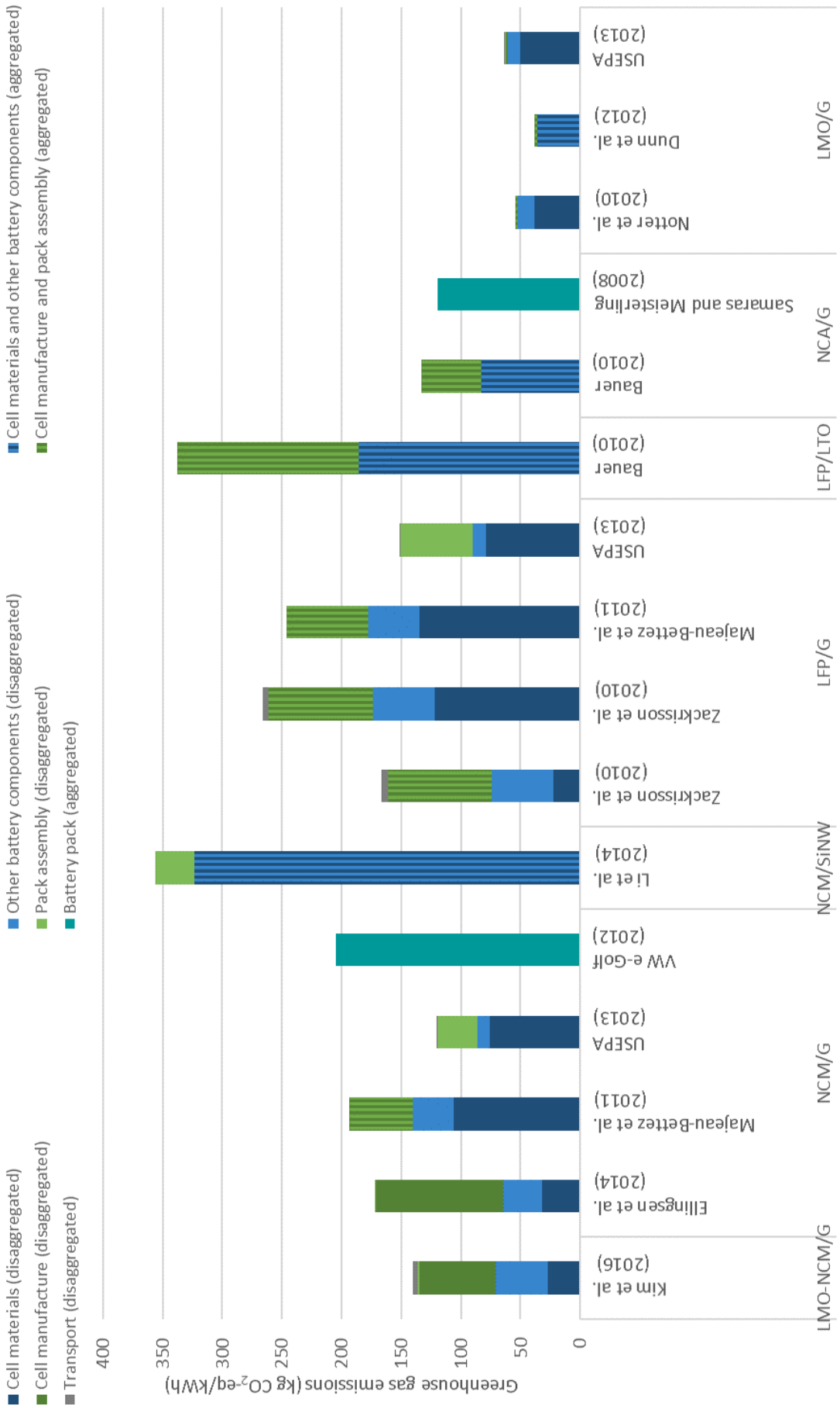

Figure 1 GHG emissions of battery production.

Figure 1 reports great variation in the overall production emissions with results between 38-356 kg CO_2-eq/kWh, corresponding to 0.9-8.6 tonnes of CO_2-eq for a 24 kWh battery. As can be seen in Figure 1, the studies also report different contributions from battery components and energy demand. To uncover the reasons for the different results, we examine the underlying data and assumptions in the different studies. Below, we go through some of the most important differences and contributing factors.

A major source of the difference in production-related GHG emissions stems from the energy demand in cell manufacture and battery pack assembly. Kim et al. (2016) report that use of utilities (electricity, natural gas, and water) in cell manufacture account for 45% of the total production GHG emissions. Ellingsen et al. (2014) found that electricity demand in cell manufacture causes 62% of the cradle-to-gate emissions. In contrast to the abovementioned studies, Dunn et al., (2012), Notter et al. (2010), and USEPA (2013) report insignificant GHG emissions associated with cell manufacture. Of the studies that report emissions associated with pack assembly, Kim et al. (2016), Ellingsen et al. (2014), Dunn et al. (2012), and Notter et al. (2010) are much lower (these are barely visible in Figure 1) than in Li et al. (2014). Interestingly, USEPA (2013) report that pack assembly of the NCM and LFP batteries contribute as much as 28% and 40% of total production emissions, while for the LMO battery, pack assembly has no contribution at all. Majeau-Bettez et al. (2011), Zackrisson et al. (2010), and Bauer (2010) respectively find found that cell manufacture and pack assembly combined contribute to 28%, 33-53%, and 38-45% of total production-related GHG emissions. To find out why the studies reach such different results, we examined the various approaches taken by the different studies in estimating energy demand in cell manufacture and pack assembly. Table 1 reports energy data and sources for cell manufacture and pack assembly from the studies that provided this information.

Table 1 Energy data and sources for cell manufacture and pack assembly reported by LCA studies.

Study	*Cell manufacture (MJ/kWh)*	*Pack assembly (MJ/kWh)*	*Cell manufacture and pack assembly (MJ/kWh)*	*Energy sources*
Kim et al. (2016)			530*	Primary data
Ellingsen et al (2014)	586	0.01	586	Primary data
Majeau-Bettez et al. (2011)			463-590*	Secondary data
USEPA (2013)	0-10	0-400	10-403	Majeau-Bettez et al. (2011), Notter et al. (2010), and primary data
Li et al. (2014)	0.36	0.41	0.77	Own estimates
Zackrisson et al.			450*	Secondary data
Bauer	326-1060	109-278	434-1338	Secondary data
Notter et al. (2010)	3.1	0.11	3.21	Own estimates
Dunn et al. (2012)	2.7	2.9	5.6	Own estimates

*Study only provides the combined energy demand for cell manufacture and pack assembly

Due to the proprietary nature of the battery industry, the access to primary data provided by the industry is limited. Thus, only two of the studies used energy estimates that are based on primary industry data (Ellingsen et al., 2014; Kim et al., 2016b), while four of the studies used secondary energy data found in industry reports (Bauer, 2010; Majeau-Bettez et al., 2011; Samaras and Meisterling, 2008; Zackrisson et al., 2010). Other studies made their own estimates (Dunn et al., 2012; Li et al., 2014; Notter et al., 2010) or based their energy data partly on previous studies (USEPA, 2013). In the text below, we provide more information about the energy data used in the various studies.

To assess the manufacture of the Ford Focus battery cells, Kim et al. (2016) used primary energy data from LG Chem for the period between January to December 2014. The energy use by the Piston Group who assembled the LG Chem cells into Ford Focus battery packs was based on normal operation in their facility in Michigan. Kim et al. (2016) report that the combined primary energy demand from utility use (electricity, natural gas, and water) during cell manufacture and pack assembly amounted to 120 MJ per kg of battery. Using a primary to electric energy conversion factor of 0.35 (Kim et al., 2016b), we calculated the combined energy demand for cell manufacture and pack assembly. Based on the GHG emissions that Kim et al. (2016) found, one can deduce that most of the energy use stems from cell manufacture. Ellingsen et al. (2014) relied on primary energy data from a cell manufacturer covering an 18-month period in 2011 and 2012. For pack assembly, Ellingsen et al. (2014) received energy data that included the welding of cell tabs to busbars, but did not account for electricity use for testing of the modules or the initial charging process in battery pack assembly.

Next, we consider studies that based their energy estimates on industry reports. Bauer (2010) and Zackrisson et al. (2010) based their energy data on reports from Hitachi Maxell (2005, 2003) and SAFT (2008), respectively. Samaras and Meisterling (2008) and Majeau-Bettez et al. (2011) based their energy data on an energy analysis of a SAFT battery published by Rydh and Sandén (2005). While Bauer (2010) decided to divide the reported energy demand between cell manufacture (80%) and pack assembly (20%), Zackrisson et al. (2010) and Majeau-Bettez et al. (2011) assumed that the energy demand covered both cell manufacture and pack assembly.

Some of the studies made their own estimates for the processes involved in cell manufacture and pack assembly. To determine what processes were considered in the various studies, we inspected the inventories. In cell manufacture, Notter et al. (2010) considered heating of electrodes, calendaring, and a 70 % cell charge. Thus, their estimate did not consider cell assembly, which is a process that places strict constraints on the ambient environment and must take place in a dry-room (Schönemann, 2017; Wood et al., 2015). Furthermore, it appears that Notter et al. (2010) underestimated the electricity required for formation cycling and aging. For testing in battery pack assembly, Notter et al. (2010) assumed electricity for a single charge of the battery to 95 % of full capacity. Dunn et al. (2012) received an energy consumption estimate for a dry-room of 1860 m^2 at 21 ℃ by a dry-room manufacturer, and assumed that the energy demand scales linearly with floor area. They converted the values to a per-mass-of-battery basis for a room of 3000 m^2 with a cell production rate of six million accepted cells per year. For the conditioning, Dunn et al. (2012) assumed a total of four cycles, one formation cycle and three aging cycles. Their total energy demand was sufficient to fully charge their battery five and a half times. Therefore, after the assumed four cycles, less than 30 % of their estimated total energy demand is available for electrode production and cell assembly. This seemingly contradicts their assumption that the operation of dry-rooms and conditioning account for 60 % of total energy demand. Thus, it seems that both Notter et al. (2010) and Dunn et al. (2012) neglected processes in cell manufacture and therefore underestimate the energy demand. Li et al. (2014) report results for a cell with an NCM cathode and a SiNW anode. However, their cell inventory provides data for an LMO cathode and a graphite anode. For cell manufacture, their low estimated energy demand correspond with the reported low GHG emissions, but the low estimated energy demand in pack assembly do not align with the reported higher GHG emissions. Thus, we suspect that Li et al. (2014) may have published the wrong sub-inventories for the cell and battery pack, which, along with the lack of description for energy inputs, constrains further discussion of their energy data.

USEPA (2013) based their energy data partly on Majeau-Bettez et al. (2011), Notter et al. (2010), and primary data. The energy data used in the study are not found in the report. Therefore, the primary to electric energy conversion factor of 0.35 was used to estimate the energy demand based on the reported primary energy use. We find that USEPA (2013) report very different energy use associated with cell manufacture and pack assembly for the NCM, LFP, and LMO batteries. Unfortunately, USEPA (2013) offers no explanation for why

the energy demands for these processes are so different themselves, but Kim et al. (2016) suggested that they may have misinterpreted the industry information. For the NCM and LFP batteries, it appears USEPA (2013) ascribed the aggregated energy demand for cell manufacture and battery pack assembly reported by Majeau-Bettez et al. (2011) entirely to battery pack assembly and nothing to cell manufacture. For the LMO battery, USEPA (2013) modelled low energy use for both processes, which corresponds with the energy data Notter et al. (2010) used for their LMO battery.

Another source of variation in production-related GHG emissions stems from assumptions regarding the cell materials and the amount of these. Bauer (2010), Dunn et al. (2012) and Kim et al. (2016) do not provide a contribution analysis of cell materials, which limits the discussion of cell-related emissions reported by these studies. The majority of the studies are in good agreement with respect to GHG emissions associated with graphite-based anodes, with emissions ranging between 7.5 and 9.9 kg CO_2-eq/kWh (Ellingsen et al., 2014; Notter et al., 2010; USEPA, 2013). Due to the binder used in the study by Majeau-Bettez et al. (2011), their study report higher anode emissions at 18.2 kg CO_2-eq/kWh. With a water solvent, Zackrisson et al. (2010) report only 1.6 kg CO_2-eq/kWh. Li et al. (2014) assess a LIB using silicon nanowire as anode material, which, due to the very high energy demand of its synthesis, emits 130 kg CO_2-eq/kWh. Except for the SiNW anode, the study by Li et al. (2014) appears to be exclusively based on the USEPA (2013) study; thus in the remainder of the text, we only refer to the results from the USEPA (2013) study. For the cathode, most studies find emissions in the range of 16-19 kg CO_2-eq/kWh (Ellingsen et al., 2014; Notter et al., 2010; Zackrisson et al., 2010), but Majeau-Bettez et al. (2011) report much higher emissions at 72 kg CO_2-eq/kWh, which again is due to the binder. The higher emission of 49 kg CO_2-eq/kWh reported by USEPA (2013) is likely attributable to the assumed larger amount of cathode materials in the cell. The studies find that the emissions related to the electrolyte are relatively low, ranging from 2.1 to 3.9 kg CO_2-eq/kWh (Ellingsen et al., 2014; Majeau-Bettez et al., 2011; Zackrisson et al., 2010), although USEPA (2013) reports higher impact at 14.6 kg CO_2-eq/kWh. The reported emissions associated with the separator are also low, ranging between 0.4 and 2.2 kg CO_2-eq/kWh.

Another source of differences in production-related GHG emissions is due to assumptions regarding other battery components, such as battery electronics, thermal system, and packaging. The studies report a large range of results for battery component groups (Table 2).

Table 2 Overview of GHG emissions (kg CO_2-eq/kWh) per component from various LCA studies.

Component	*Kim et al. (2016)*	*Ellingsen et al. (2014)*	*Majeau-Bettez et al. (2011)*	*USEPA (2013)*	*Zackrisson et al. (2010)*	*Notter et al. (2010)*
Cell materials	27	32	110	76	22	39
Electronics	13.5	8.2	27	4.1*	50	10.1
Thermal system	5.9	3.6				
Packaging	25	20	6.0	6.3	2.2	4.0

*Emissions for electronics and thermal system combined

Although the overall emissions associated with other battery components (light blue in Figure 1) are similar, the contributions from the battery component groups vary significantly (Table 2). The studies based on primary data find similar emissions for cell materials and other battery components (Ellingsen et al., 2014; Kim et al., 2016b), while the studies based on literature or secondary data report significantly more variation in their results (Li et al., 2014; Majeau-Bettez et al., 2011; Notter et al., 2010; USEPA, 2013; Zackrisson et al., 2010).

The latter studies seem to underestimate the amount of packaging that LIBs require and consequently report lower packaging GHG emissions. The studies that find lower emissions from electronics and packaging had very limited sub-inventories for these components. Furthermore, many of the studies did not include a thermal system. The underestimation of the amount and weight of other battery components in LIBs results in the studies modeling batteries with considerably higher gravimetric energy density than actual LIBs currently used in electric vehicles. Because commercialized batteries have lower gravimetric energy density than that assumed by many of the academic studies, they are heavier and therefore demand more energy during BEV operation. Thus, the assumed high energy density by can potentially lead to underestimation of the GHG emissions associated with both the production and the use phase.

2.2 Use phase

During use, the battery has indirect emissions that are influenced by the energy conversion losses, energy required to carry the battery weight, and the carbon intensity of the electricity. The energy conversion losses in the battery depends on the energy efficiency of the battery and the operational energy demand (Wh/km) of the vehicles. Because it is difficult to find information regarding energy efficiency of LIBs, LCA studies often make an assumption for the battery energy efficiency. Thus far, LCA studies have generally not taken into account that different cell format types (e.g., pouch, cylinder, prismatic) and cathode materials (LMO, NCM, LFP, NCA) offer different efficiencies (Mulder et al., 2013). USEPA (2013) assumed an energy efficiency of 85%, whereas Majeau-Bettez et al. (2011) and Zackrisson et al. (2010) assumed 90%. For the NCM pouch cells in their study, Ellingsen et al. (2016b) used the measured energy efficiency of 95% (Ellingsen et al., 2014). Depending on the carbon intensity of the electricity used for charging and the operational energy demand of the vehicle, the differences can significantly influence the total life cycle emissions of LIBs. With a total driving distance of 180 000 km, Ellingsen et al., (2016b) find the indirect energy demands due to conversion losses to cause 638 kg CO_2-eq when charged with the average European electricity mix and 18 kg CO_2-eq when charged with purely wind-based electricity. Zackrisson et al. (2010) also evaluated how much of the operational energy demand can be ascribed to the weight of a battery. The study estimated the total energy use due to the battery weight based on the ratio of battery to vehicle weight, the share of operational energy use due to total vehicle weight, the operational energy demand of the vehicle, total driving distance, and the share of the time the PHEV was in electric mode. When charged with the average European electricity mix for 180 000 km, indirect emissions associated with the battery weight results in a total of 236 kg CO_2-eq.

2.3 End-of-life

At the EOL, useful components and materials can be reused or recycled. LIB recycling is typically a combination of two or more processes (Hanisch et al., 2015). There are several competing industrial LIB recycling processes, but very few LCA studies consider the emissions associated with the EOL treatment of LIBs. Hawkins et al., (2012) model EOL treatment consisting of dismantling and a cryogenic chattering process. Ellingsen et al. (2016b) compiled a recycling inventory based on secondary industry data for the pyrometallurgical treatment described by Dewulf et al. (2010). Li et al. (2014) assume a combined direct physical, hydro- and pyrometallurgical treatment process, but only consider the direct energy requirements for this process. Dunn et al. (2012) and USEPA (2013) consider hydrometallurgical, intermediate physical, and direct physical recycling. Dunn et al. (2012) report purchased energy consumption for this process. USEPA (2013) obtained primary data from battery recyclers, but do not provide an EOL inventory. While Hawkins et al., (2012), Ellingsen et al. (2016b), and Li et al., (2014) use the “recycled content” approach and report emissions associated with recycling, USEPA (2013) and Dunn et al. (2012) use the “end-of-life" approach and respectively report the reduction in emissions and total energy consumption that may be possible through various recycling scenarios. Hawkins et al. (2012) and Ellingsen et al. (2016b) report 3.6 and 8.0 kg CO_2-eq/kWh battery associated with recycling, respectively, while Li et al., (2014) report emissions of 27 kg CO_2-eq/kWh battery. USEPA (2013) report emission reductions in the range of 16-32 kg CO_2-eq/kWh battery for

the average EOL treatment options. Dunn et al. (2012) find that the use of recycled battery materials offers reduced energy consumption compared to the use of virgin materials. Due to the lack of access to industry data, there is large uncertainty associated with the use of materials (e.g., solvents) and energy. As a result, there is also significant uncertainty associated with the reported results across all of the reviewed studies.

3 Discussion of LCA findings

We have reviewed the relevant LCA literature on LIBs and examined the key assumptions and differences between the studies. We found that all of the reviewed studies have assessed the production impact, while only a few have assessed the use and EOL stages. Below, we discuss the findings and evaluate the reliability of the results and inventory data before we point out measures that can reduce life cycle GHG emissions of LIBs.

3.1 Evaluation and analysis of inventories and results

Energy demand in battery production has been much discussed in the LCA literature (Dunn et al., 2015, 2012, Ellingsen et al., 2015, 2014; Kim et al., 2016b; Majeau-Bettez et al., 2011) and the literature reports two opposing views. On one hand, we have studies that assume low energy demand in cell manufacture and find that energy use has insignificant contributions to production-related GHG emissions. On the other hand, we have studies that report high energy demand and consequently find this to be a main source of GHG emissions. Below, we consider what we know of the cell manufacturing process and review the associated energy demands. Cell manufacture is a complex and protracted process that places constraints on ambient conditions and therefore requires strictly controlled environments (Schönemann, 2017; Wood et al., 2015). Due to the proprietary nature of the industry, the information available on the various production steps and requirements is limited and data are scarce. As a result, estimating energy demands of the various production steps in cell manufacture is a formidable task. Having examined the inventories, we find that studies attempting to estimate the energy demand seemingly omit or underestimate energy inputs for some production steps. Furthermore, two independent LCA studies based on primary industry data report significant energy use. Although some have suggested that the high energy demand may be due to low production volumes (Dunn et al., 2015), this has been disputed (Ellingsen et al., 2015) and proved not to be the case (Kim et al., 2016b). Thus, we conclude that cell manufacture is an energy-intensive process and a main contributor to production-related GHG emissions of LIBs. Unlike cell manufacture, battery pack assembly does not place constraints on ambient conditions and the main energy requirements are associated with welding, testing, and charging (Schönemann, 2017). The low energy inputs for pack assembly reported by the majority of the LCA studies thus seem reasonable.

Further disagreements among the studies were due to assumptions regarding cell materials and other battery components. For cell materials, some of the differences are the result of assumptions regarding material types and the amount of these. In general, the studies agree that the cathode and the anode have, in that order, the highest cell GHG emissions and that the contributions from the electrolyte and the separator are about an order of magnitude smaller. At this point, we should also mention that not all of the cathode materials assessed by LCA studies are used in LIBs for electric vehicles. Although LMO is a low-cost material that has safer characteristics than NCA and NCM (Kim et al., 2012), it is only used as a blended cathode material with either NCM or NCA (Anderman, 2016a) as it does not provide adequate energy density or lifetime for electric vehicle applications when used alone (Ellingsen et al., 2016a). LFP provides excellent cycle stability and lifetime, but the material is no longer used in commercial BEVs due to its low energy density (Anderman, 2016a; Ellingsen et al., 2016a). Initially, LFP was used in the Chevy Bolt battery, but in 2015 it was substituted with NCM for its higher energy density (Anderman, 2016a; Voelcker, 2014). LFP is, however, still used in some PHEVs (Anderman, 2016b). Of the current cathode materials, NCM and NCA have the highest energy density (Ellingsen et al., 2016a) and are the only unblended cathode materials used in commercial BEVs (Anderman, 2016a). Regarding other battery components, studies based on literature or secondary data seemingly misjudge

the amount of other battery components required for LIBs. Compared to the two studies based on primary industry data, the other studies particularly underestimate the amount of packaging. In addition, many of the studies have not included a thermal system. Thus, we deem the studies based on primary data sources to provide higher certainty in the contribution of these components than those based on literature or secondary data sources.

LIBs can be made with various anode and cathode materials and, as seen in Figure 1, this is reflected in the LCA literature. If one only considers the overall production impact without examining the underlying assumptions and differences, one could mistakenly draw the conclusion that the difference between the studies is due to electrode materials. However, as we have uncovered in the text above and seen in Figure 1, the difference in electrode materials is not the main driver to differences between the various studies, instead it is primarily due to assumptions regarding energy demand and battery components.

For the use phase, studies mainly evaluate emissions as a product of energy conversion losses and the carbon intensity of the electricity used for charging, while omitting emissions associated with the energy required to transport the weight of the battery. The energy conversion losses depend on the energy efficiency of the battery and the operational energy use of the vehicle. Although energy efficiency is cell-specific, a generic approach assuming a certain energy efficiency has been used by most of the reviewed studies. To make evaluations of the energy losses that are more realistic, studies should attempt to obtain and use energy efficiency data that are representative for the analyzed electrode materials and cell format types. Similarly, rather than using energy consumption data for vehicles based on driving cycle tests (e.g., the New European Driving Cycle) that often underestimate operational energy use, using measured energy consumption data could provide higher confidence in estimated energy conversion losses. Only one of the reviewed studies estimate the energy required to transport the weight of the battery. A recent publication presents a new physics-based model that captures the mass-induced energy demand for vehicles with different powertrains (Kim et al., 2016a), which can be used to estimate the energy required to transport the battery in future studies.

LCA studies assessing battery recycling consider different recycling processes and generally suffer from poor data quality. This results in high uncertainty with respect to the considered recycling alternatives, particularly with respect to the use of materials and energy. LCA studies suggest that production of secondary metals from battery recycling is less energy demanding than extraction of primary metals and that recycling in this way is beneficial with respect to GHG emissions.

3.2 Potential for emission reductions

Based on the findings reported by the studies, we can point out various measures that can reduce the life cycle GHG emissions of LIBs. Studies that report high energy demand in cell manufacture propose reducing energy demand or using renewable energy sources as the most efficient measures to reduce GHG emissions associated with battery production. Studies that assume low energy demand recommend recycling as an important measure as it can reduce impacts associated with extraction of virgin metals. While lowering energy demands and the use of recycled metals can both reduce the production-related impacts, the most efficient measure to reduce GHG emissions is perhaps to manufacture cells in facilities that are supplied entirely by renewable energy sources. Future cell manufacturing practices following this strategy are likely to produce battery cells with lower GHG emissions than current practices that use electricity from the grid and heat from natural gas. Battery recycling is an important source of secondary metals and can reduce the need of extracting primary metals. As a result, battery recycling and the use of recycled metals in batteries may offer reductions in GHG emissions and other environmental impacts. In the use phase, emissions can be reduced by lowering the energy conversion losses. Thus, increasing the energy efficiency of the cells and reducing vehicular operational energy demand can lower use phase emissions. Making lighter vehicles through material substitution and higher energy density batteries may reduce operational energy demand, but it does not necessarily reduce the total

life cycle GHG emissions as production impacts may increase (Ellingsen et al., 2016a; Kim and Wallington, 2013). Therefore, lightweighting efforts of electric vehicles should be carefully evaluated through a holistic life cycle perspective. Evidently, the use of cleaner energy sources in electricity production will reduce indirect GHG emissions associated with the battery use phase. Unfortunately, the LCA literature on EOL treatment offers little guidance, as there is limited access to primary data on any of the numerous recycling schemes.

4 Conclusion

In this article, we have examined the inventory data and results from LCA studies assessing the life cycle GHG emissions of LIBs. Based on this, we conclude that the production phase is the main contributor to life cycle GHG emissions of LIBs, while the use phase end EOL treatment hold much smaller contributions. Because the various LCA studies report widely different results for the production of LIBs, there has been some disagreement with respect to the amount and main sources of production-related emissions. As a result, the LCA community has not been able to provide a unified answer about the production-related emissions of LIBs. However, as the battery industry has provided primary data for recent studies, the data quality is much improved and this increases confidence in the results. Furthermore, the studies based on primary data obtain similar results and conclude that cell manufacture is energy-intensive. Consequently, we now have improved information about the amount and main sources of production-related GHG emissions. As the production is the main contributor to life cycle GHG emissions of LIBs, reducing production-related emissions is particularly important and the use of renewable energy sources in cell manufacture may be a particularly effective measure.

Although the data quality and certainty of LCA studies assessing LIBs have increased, knowledge gaps still exist. Primary energy data provided by the industry has proven that cell manufacture is energy demanding, but more details about use of heat and electricity for the various production steps in cell manufacture is still desirable. Such information could provide further insights as to how the battery industry can reduce energy demand and GHG emissions. Furthermore, very few studies assess the use phase and EOL treatment, and the data quality of these studies is low. Further research efforts are therefore required for these life cycle stages to obtain higher confidence in the reported results, but this demands increased transparency and data from the battery industry and recyclers. On their part, LCA practitioners should strive to be updated on the LIB technology so that they assess relevant battery technologies, rather than chemistries that have already been ruled out for use in LIBs for electric vehicles. The LIB technology is still progressing and continued LCA efforts are necessary to provide updated information regarding improvement opportunities and the sustainability of LIBs and electric vehicles.

Acknowledgement

The work was supported by the Research Council of Norway through the Centre for Sustainable Energy Studies.

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