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Lifecycle impacts of lithium-ion batteries: a review

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Summary

This review considers life cycle assessment studies on lithium-ion batteries, summarizes the most sensitive assumptions of the studies, and discusses how the environmental performance of LIBs can be improved. We identify the main source of discrepancy and uncertainty of battery production impact to stem from differing assumptions regarding energy requirements in cell assembly. There are few studies assessing the use and end-of-life phases and the associated impacts are found to be smaller than that of the production phase, although there is some uncertainty associated with the findings. Access to primary data will provide more useful results with higher certainty.

Keywords: battery, BEV (battery electric vehicle), environment, LCA (Life Cycle Assessment), recycling

1 Introduction

Climate change and energy security are important global concerns. 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 CO₂-related emissions in 2010 [1]. Light duty vehicles (LDVs) were responsible for around half of the total transport energy consumption. From the current number of around one billion vehicles [2], the total LDV ownership is expected to double in the next few decades [3]. 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 [4]. Battery electric vehicles (BEVs) have merged as a strong candidate among the available transport alternatives [5]. Compared to conventional vehicles, BEVs offer advantages in terms of powertrain efficiency, maintenance, and zero tailpipe emissions.

Understanding the system-wide tradeoffs of replacing conventional vehicles by BEVs requires a lifecycle perspective. Environmental trade-offs that arise from the change in drivetrain configuration are best analyzed using life cycle assessment (LCA) [6]. The major differences between electric and conventional vehicles arise from the traction battery and the vehicle use phase. Several academic studies [7]–[14] as well as some grey literature studies [15], [16] and industry studies [17]–[21] have assessed the production impact of LIBs, either as part of an BEV or on its own. LCA studies have investigated LIBs with LMO, NCM, NCA, and LFP cathodes in combination with a graphitic anode [7], [8], [10]–[13], [15], [16], [22]. In addition, LFP in combination with an LTO anode [15] and NCM in combination with a silicon nanowire anode (SiNW) [12]

have been assessed. Moving forward, it is important to determine what is known about the environmental impacts of BEVs and to identify the most important knowledge gaps in order to support policy and technology developments.

This review considers the most recent LCA studies on LIBs, summarizes the most sensitive assumptions of the studies, and discusses how the environmental performance of LIBs can be improved. Emphasis will be placed on GHG emissions, owing both to global warming potential (GWP) being the only common impact category in the reviewed literature, and the fact that BEVs, to a large extent, are being promoted as a means to reduce GHG emissions.

This paper is divided as follows. Section 2 provides a brief overview of cell and battery components and aspects that must be considered in LCA studies of LIBs. Then we go through reported impact of production (Section 3), use (Section 4), and end-of-life (EOL) scenarios (Section 5). Section 6 sums up the results, identifies knowledge gaps, and provides directions from the literature.

2 Traction batteries

The battery cells are the main components of a traction battery. The cells consist of five subcomponents: anode, cathode, separator, electrolyte, and cell container. The anode is composed of a copper current collector with a coat of negative electrode paste. The negative electrode paste consists mainly of graphite (or in some cases, LTO), but also contains small amounts of binders. The cathode is composed of an aluminum current collector with a coat of positive electrode paste. The positive electrode paste consists mainly of the positive active material and small amounts of carbon black and a binder. The electrolyte is a mixture of organic solvents, such as ethylene carbonate and dimethyl carbonate, and an electrolyte salt compound, mainly LiPF_6 [23]. The separator is a thin microporous membrane made of a polyolefin [23]. The cell container can be a canister or a pouch, depending on design. Battery cell manufacturers normally do not produce any of the materials themselves, but order these from external suppliers. The cell assembly requires several processes (see Figure 1) and the associated energy requirements must be taken into consideration when performing an LCA.

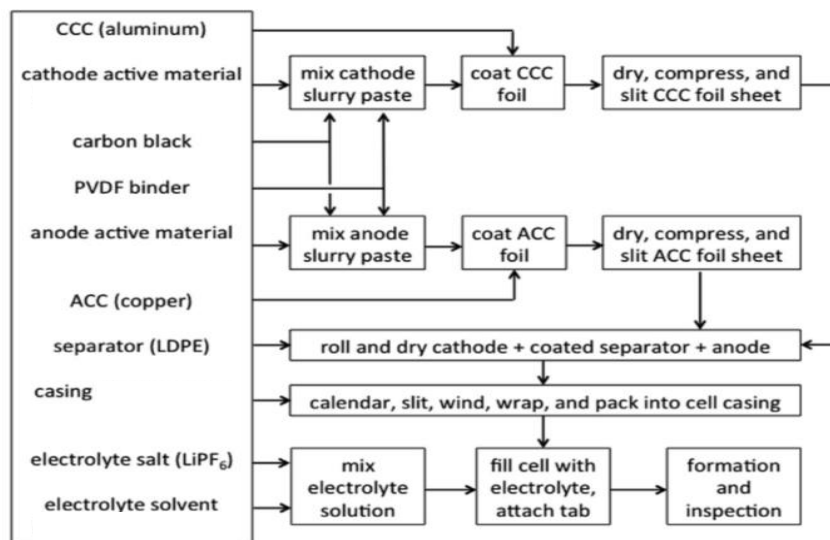


Figure1 Battery cell manufacture (figure adopted from [13])

In addition to the battery cells, a traction battery typically consist of packaging, a battery management system (BMS), and a cooling system. The latter may not be present in all BEV designs. In battery assembly, the component parts are combined to form the final vehicle battery pack.

During use, the battery has no direct emissions, but indirect emissions occur in connection with losses due to battery charging and weight. At the end-of-life (EOL), the battery is disposed of, which requires material and energy inputs and results in emissions.

In this review, we have grouped the lifecycle environmental impacts of batteries into three main lifecycle stages: production, use, and end-of-life (EOL) treatment.

3 Battery manufacturing

In this section, the environmental impact associated with battery production from 14 studies is presented. Battery capacity measured in kWh is used as a representative functional unit that allows for a consistent comparison between the studies. Figure 2 includes the result from four industry studies, eight academic studies, and two grey literature studies.

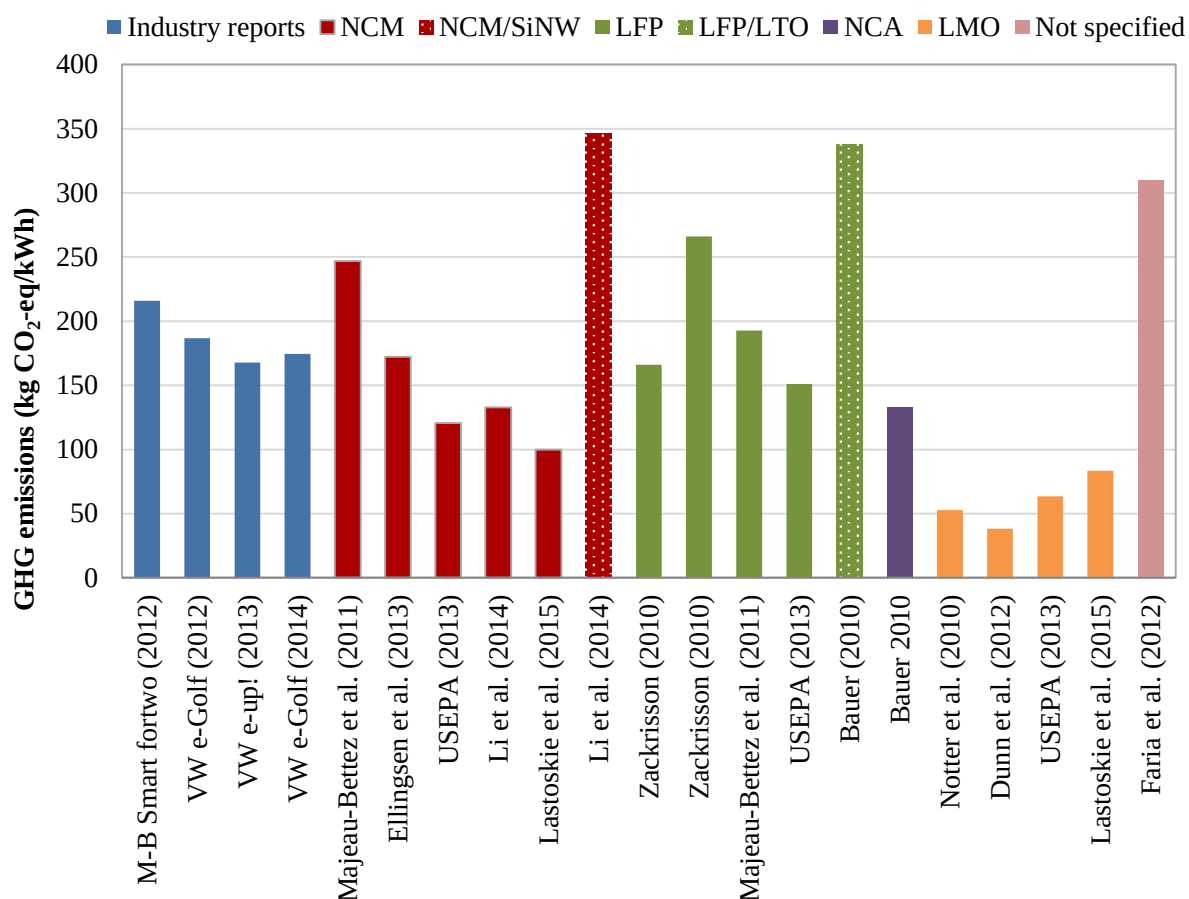


Figure2: GHG emissions of battery production

As can be seen in Figure 2, there is great variation in the reported production impact of LIBs, with results ranging from 38-354 kg CO₂-equivalents per kilowatt-hour (CO₂-eq/kWh), or 0.9-8.3 ton CO₂-eq for a 24 kWh battery. Although it may seem that the LMO chemistry is less environmentally intensive than the other chemistries, this variation can be attributed to a number of factors. Although the different chemistry might explain some of the variation, it is not the source of the discrepancy. Below we will explain some of main factors contributing to this difference.

When we examine the results from the various studies, we find that the main difference is due to energy consumption in battery cell assembly. Due to the proprietary nature of the battery industry, the access to primary data is limited. Therefore, most LCA studies use secondary data from the literature, databases, energy calculations, or proxies based on similar processes. The reported energy required for manufacture range from 2.4 to 1062 MJ/kWh battery cell (see Table 1). Bauer [15], Zackrisson et al. [7], and Majeau-Bettez et al. [9] based their energy data on industry reports from Hitachi Maxell [24], [25], Saft [26], and Rydh and Sandén [27], respectively. Note that the energy data reported in Zackrisson et al. [7] and Majeau-Bettez et al. [9] include battery assembly in addition to cell manufacture. Ellingsen et al. [11] based their study on primary production data and used a top-down approach to establish the energy usage. Notter et al. [8] made their own

process-based energy estimations. Dunn et al. [10] based their energy data on an energy quote for a dry room provided by dry room manufacturer SCS Systems and calculated energy required for formation cycling [28]. The USEPA [16] based their energy data partly on proprietary industry sources and partly on Majeau-Bettez et al. The process-level approach, used by Dunn et al. [10] and Notter et al [8], has the advantage of being process-specific and yields detailed results, but risks leaving out processes and may make uncertain estimates due to the lack of access to primary data. The top-down approach used by Bauer [15], Zackrisson et al. [7], Majeau-Bettez et al. [9], and Ellingsen et al. has the advantage of being complete with respect to inclusion of all relevant activities related to the producing industry, but data is often aggregated, which results in a lack of detail, and may include inhomogeneous products. Ellingsen et al., however, have well-defined boundaries for the battery cell manufacture in their study and homogenous products (i.e., the producer only manufactures Li-ion battery cells). The different approaches have resulted in two opposing understandings: Bauer [15], Zackrisson et al. [7], Majeau-Bettez et al. [9], and Ellingsen et al. find significantly higher energy requirements than Dunn et al. [10], Notter et al [8], and the USEPA [16]. The results from the different industry studies are in good agreement, and although the energy consumption in these studies is unknown, the reported results suggest high energy consumption.

Table 1 Cell data, results, and energy sources

Study	Cell energy density (kWh/kg)	Direct energy use in cell production (MJ/kWh)	GWP due to direct energy use in cell production (kg CO ₂ -eq/kWh)	Total GWP of battery (kg CO ₂ -eq/kWh)	Energy reference
Notter et al. [8]	0.143	3.1	0.90	53	Authors' own estimates
Bauer [15]	0.069-0.176	326-1062	50-152	133-338	[24], [25]
Zackrisson et al. ^a [7]	0.103-0.104	451	87	166-266	[26]
Majeau-Bettez et al. ^a [22]	0.110-0.140	371-473	56-70	200-250	[29]
Dunn et al. ^b [10]	0.148	10.7	2.3	38	[28], [30]
USEPA ^c [16]	0.139-0.167	0-28.3	0-1.8	63-151	Proprietary, [22]
Ellingsen et al. [11]	0.174	586	107	172	Primary

^aThe studies have a combined cell and battery manufacturing process, and thus energy inputs and GWP shares includes both.

^bThe total weight of cells is assumed to be 90% of total the battery weight, which can be deducted from the original study.

^cThe total weight of cells is assumed to be 60% of total the battery weight, which can be deducted from the original study.

A smaller source of the variation in the reported results stems from assumptions regarding the cell materials and the amount of these. The majority of the impact results from graphite-based anodes are in good agreement, with impact ranging between 7.5 and 9.9 kg CO₂-eq/kWh [8], [11], [16]. Due to the binder used in the study by Majeau-Bettez et al., they report higher impact at 18.2 kg CO₂-eq/kWh [22]. Zackrisson used water as a solvent and report only 1.6 kg CO₂-eq/kWh. Li et al. assessed an NCM battery using SiNW as anode material, which due to the high energy consumption of the anode material, results in 130 kg CO₂-eq/kWh. Except for the SiNW anode, the study by Li et al. appears to be based on the USEPA study; thus in the remainder of the text, we only refer to the results from the USEPA study. As for the cathode, most studies find an impact in the range of 16-19 kg CO₂-eq/kWh [7], [8], [11]. Majeau-Bettez et al., report much higher impact at 72 kg CO₂-eq/kWh due to the same binder that was used in the anode [22]. The USEPA report 49

kg CO₂-eq/kWh, the higher impact is likely attributed to the higher share of cathode materials in the cell. The impact of the electrolyte is found to be fairly low at 2.1-3.9 kg CO₂-eq/kWh [7], [9], [11], although one study reports higher impact at 14.6 kg CO₂-eq/kWh [16]. Similarly, the impact of the separator is also found to be low, ranging between 0.4 and 2.2 kg CO₂-eq/kWh. For the packaging, BMS, and cooling system components, there is a large breadth in results. Combined, cell casing and battery packaging range between 3.0 and 23 kg CO₂-eq/kWh. Impact of the BMS range between 4.1 and 35 CO₂-eq/kWh. Ellingsen et al. assessed a battery with a cooling system and found it responsible for 3.6 CO₂-eq/kWh.

There are, of course, some differences in cathode production impacts as the cathodes consist of different materials and can be produced using differing synthesis methods. Cathode materials containing nickel and cobalt generally have higher environmental impacts per kg than cathode materials that do not contain these metals [13]. However, the energy content (kWh/kg) of cathode materials containing nickel and cobalt (e.g. NCM or NCA) is usually higher than those that do not (e.g. LFP or LMO), which can reduce the cathode amount necessary to achieve the same energetic output. Therefore, it is unlikely that the discrepancy in results seen in Figure 1 is due to battery chemistry and manufacturing processes, but rather reflects differences in data sources and methodological approaches among the studies.

4 Use phase of traction batteries

The use phase impact of traction batteries is influenced by a number of factors such as the energy efficiency, energy requirements needed to carry the battery weight, the carbon intensity of the electricity, and battery lifetime.

Because it is difficult to find information regarding energy efficiency of LIBs, LCA studies often make an assumption regarding the battery energy efficiency and is often set to 90% [7], [13], [31], [32]. Ellingsen et al. based their study on industry data for a LIB with a measured energy efficiency of 95% [33]. When used for a lifetime of 200,000 km, Ellingsen et al. [33] found impacts associated with battery energy losses to be 0.6-0.9 ton CO₂-eq depending on car size, whereas Zackrisson et al. [7] found battery losses to be responsible for 2.3 ton CO₂-eq. The higher impact reported by Zackrisson et al. is partially due to the lower energy efficiency and some may be due to small differences in carbon intensity of the electricity used to charge the battery, although both studies were using the average European electricity mix. Zackrisson et al. [7] also assessed losses due to battery weight, which were found to be 0.4 ton CO₂-eq throughout the use phase of the battery.

5 End-of-life options

5.1 Battery recycling

At the EOL, the battery is disposed of and useful parts or materials can be reused or recycled. LIB recycling is typically a combination of two or more processes [34]. Very few LCA studies consider impacts associated with the recycling or reuse of LIBs. Hawkins et al. [35] modeled an EOL inventory consisting on dismantling and cryogenic chattering process. Ellingsen et al. [33] compiled a recycling inventory based on the pyrometallurgical treatment described by Dewulf et al. [36]. Li et al. [12] assumed a combined direct physical, hydro- and pyrometallurgical treatment process and considered energy requirements for this process. Faria et al. [14], Hawkins et al. [35], and Ellingsen et al. [33] report the impact of battery recycling activities to range between 0.1-0.2 ton CO₂-eq/battery, whereas Li et al. report 1.2 ton CO₂-eq/battery. There is very little data available for industrial recycling processes; therefore, the studies assessing recycling rely on secondary data. The higher impact reported by Li et al. is likely due to higher energy consumption, but could also be due to the difference in recycling processes.

In LCA, the credit for recycling can be given in two ways, either based on plausible collecting and recycling scenarios after the initial use of primary materials [37], or in the material production phase by using secondary materials [9], [11], [12], [38].

5.2 Battery re-manufacturing for second life

At the end-of-vehicle life, the battery can be reused in stationary applications. Reuse of traction LIBs extend the lifetime of the battery beyond the vehicle life and would enable the high production impact to be

distributed between two batteries uses, rather than just one. So far, only few studies have assessed the reuse of traction batteries for second life applications [39], [37]. Ahmadi et al. [39] found the impact associated with the re-functionalisation of the batteries to prepare for second life to be 0.5 ton CO₂-eq/battery.

6 Summary and conclusion

We have reviewed the available LCA literature on traction LIBs and compared the reported findings. Many studies have assessed the production impact, but only a few have assessed the battery impact of use and EOL options. Below we summarize the findings, point areas where more data is needed, and provide directions from the literature.

6.1 Production

There was much variation in the reported impact of production between the academic and grey literature studies, but the industry studies aligned well. This indicates that the lack of access to industry data is a source of the observed discrepancy. The lack of primary data is particularly clear when we look at the assumptions for energy requirements in the different academic and grey literature studies. Studies that assumed low energy requirements in cell assembly report low production impact and those that assume high energy requirements report high production impact. In the latter case, it is easier to point out impact reduction strategies; reduce energy requirements or use clean electricity sources in the battery assembly process [11]. The Tesla Gigafactory is currently under construction and is to employ both reduction strategies: the plant will take advantage of lower energy consumption in cell assembly due to high production volumes and will be self-sufficient with energy from solar, wind, and geothermal sources [40].

6.2 Use

Because of poor data availability, LCA studies often make assumptions regarding the use phase parameters. Thus far, LCA studies have not taken into account that different LIB cell types (e.g., pouch, cylinder, prismatic) and chemistries (LMO, NCM, LFP, NCA) can offer different efficiencies [41]. Often, studies have assumed an energy efficiency of 90%, which is 5% lower than the measured efficiency of the NCM pouch cell type analysed by Ellingsen et al. [33]. This seemingly small difference can, depending on the electricity used for charging and lifetime (i.e., total energy consumption), significantly influence the total lifecycle impacts of traction batteries. For a more accurate assessment of the use phase, increased access to measured energy efficiencies must be made available to LCA practitioners.

Only one of the studies attempted to account for the battery's share of energy consumption. We deem the uncertainty of this finding to be relatively high due to the difficulties in determining the contribution of weight to total vehicle operation energy requirements. Regardless, reducing battery weight is a means of decreasing battery-related impacts and can be achieved through higher gravimetric energy density (kWh/kg).

There are both near and long term strategies to reduce the impacts associated with the use phase. In the short-term, impact reductions can be achieved indirectly by improving the energy consumptions (kWh/km) of the vehicle. Increasing the energy efficiency of batteries reduces the impact associated with energy losses. In the long-term, the use of cleaner energy sources in electricity production will also reduce impacts associated with energy losses.

6.3 End-of-life

There are several competing industrial LIB recycling processes, but the access to primary data on these recycling processes is lacking. LCA studies assessing battery recycling consider different recycling processes, but generally suffer from poor data quality. As there is very little data available, there is high uncertainty with respect to the considered options, particularly with respect to solvent use in the hydrometallurgical treatment and energy use in the pyrometallurgical processes.

As the majority of the life cycle battery impacts occur during the production phase, reuse of batteries is expected to lead to substantial environmental benefits, spreading the initial environmental burden over a greater functionality. How the impacts of production are ascribed between the first and the second life is a question of allocation. In this review, we found only one study that evaluates the reuse of batteries. Ahmadi

et al. report low impact associated with re-functionalization and is much lower than the production of a new battery. When assessing a second life in addition to the first life in the vehicle, it becomes even more pertinent to use measured energy efficiencies as the battery's operational lifetime is prolonged in order to minimize the uncertainty associated with impact due to losses.

6.4 Conclusion

Differences in reported battery lifecycle impacts can to a large extent be explained by the lack of access to primary data. Production of batteries is found to be the most important source of lifecycle impacts. Therefore, R&D efforts should focus on reducing impacts of production. The progress of the LIB technology is still moving rapidly and continued efforts in assessing the lifecycle impacts of these devices are necessary to provide updated information regarding improvement opportunities and the sustainability of BEVs. LCAs can provide valuable guidance to the battery industry and policy makers alike, but in order to perform useful analysis with high certainty, increased co-operation from the battery industry is needed.

References

- [1] R. Sims, R. Schaeffer, F. Creutzig, X. Cruz-Núñez, M. D'Agosto, D. Dimitriu, M. J. F. Meza, L. Fulton, S. Kobayashi, O. Lah, A. McKinnon, P. Newman, M. Ouyang, J. J. Schauer, D. Sperling, and G. Tiwari, "Transport," in *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, T. Z. and J. C. M. Edenhofer, O., R. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, A. Adler, I. Baum, S. Brunner, P. Eickemeier, B. Kriemann, J. Savolainen, S. Schlömer, C. von Stechow, Ed. Cambridge, United Kingdom and New York, NY, USA.: Cambridge University Press, 2014, pp. 1–115.
- [2] J. Sousanis, "World Vehicle Population Tops 1 Billion Units," 2011. [Online]. Available: http://wardsauto.com/ar/world_vehicle_population_110815.
- [3] IEA, *Transport, Energy and CO2: Moving Toward Sustainability*. 2009.
- [4] T. J. Wallington, J. E. Anderson, R. D. De Kleine, H. C. Kim, H. Maas, A. R. Brandt, and G. A. Keoleian, "When Comparing Alternative Fuel-Vehicle Systems, Life Cycle Assessment Studies Should Consider Trends in Oil Production," *J. Ind. Ecol.*, vol. 00, no. 0, p. n/a–n/a, 2016.
- [5] T. R. Hawkins, O. M. Gausen, and A. H. Strømman, "Environmental impacts of hybrid and electric vehicles—a review," *Int. J. Life Cycle Assess.*, vol. 17, no. 8, pp. 997–1014, May 2012.
- [6] R. Nealer and T. P. Hendrickson, "Review of Recent Lifecycle Assessments of Energy and Greenhouse Gas Emissions for Electric Vehicles," *Curr. Sustain. Energy Reports*, pp. 66–73, 2015.
- [7] M. Zackrisson, L. Avellan, and J. Orlenius, "Life cycle assessment of lithium-ion batteries for plug-in hybrid electric vehicles - Critical issues," *J. Clean. Prod.*, vol. 18, no. 15, pp. 1519–1529, 2010.
- [8] D. A. Notter, M. Gauch, R. Widmer, P. Wäger, A. Stamp, R. Zah, and H.-J. Althaus, "Contribution of Li-ion batteries to the environmental impact of electric vehicles," *Environ. Sci. Technol.*, vol. 44, no. 17, pp. 6550–6, Sep. 2010.
- [9] G. Majeau-Bettez, T. R. Hawkins, and A. H. Strømman, "Life cycle environmental assessment of lithium-ion and nickel metal hydride batteries for plug-in hybrid and battery electric vehicles.," *Environ. Sci. Technol.*, vol. 45, no. 10, pp. 4548–54, May 2011.
- [10] J. B. Dunn, L. Gaines, J. Sullivan, and M. Q. Wang, "Impact of recycling on cradle-to-gate energy consumption and greenhouse gas emissions of automotive lithium-ion batteries," *Environ. Sci. Technol.*, vol. 46, no. 22, pp. 12704–12710, Oct. 2012.
- [11] L. A.-W. Ellingsen, G. Majeau-Bettez, B. Singh, A. K. Srivastava, L. O. Valøen, and A. H. Strømman, "Life cycle assessment of a lithium-ion battery vehicle pack," *J. Ind. Ecol.*, vol. 18, no. 1, pp. 113–124, 2013.
- [12] B. Li, X. Gao, J. Li, and C. Yuan, "Life Cycle Environmental Impact of High-Capacity Lithium Ion Battery with Silicon Nanowires Anode for Electric Vehicles.," *Environ. Sci. Technol.*, vol. 48, pp. 3047–3055, Feb.

2014.

- [13] C. M. Lastoskie and Q. Dai, “Comparative life cycle assessment of laminated and vacuum vapor-deposited thin film solid-state batteries,” *J. Clean. Prod.*, vol. 91, pp. 158–169, 2015.
- [14] R. Faria, P. Moura, J. Delgado, and A. T. de Almeida, “A sustainability assessment of electric vehicles as a personal mobility system,” *Energy Convers. Manag.*, vol. 61, pp. 19–30, Sep. 2012.
- [15] C. Bauer, “Ökobilanz von Lithium-Ionen Batterien [Life cycle assessment of lithium-ion batteries],” Paul Scherrer Institut, Labor für Energiesystem-Analysen, 2010.
- [16] United States Environmental Protection Agency, “Application of life- cycle assessment to nanoscale technology : Lithium-ion batteries for electric vehicles,” EPA 744-R-12-001. Washington, D.C., 2013.
- [17] Volkswagen AG, “The e-mission. Electric mobility and the environment,” 2012.
- [18] Daimler AG, “Environmental Certificate Mercedes-Benz B-Class Electric Drive,” Stuttgart, Germany, 2014.
- [19] Daimler AG, “>> Einfach elektrisch. smart fortwo electric drive.,” Stuttgart, Germany, 2012.
- [20] Volkswagen AG, “The e-up ! Environmental Commendation – Background Report,” 2013.
- [21] Volkswagen AG, “The e-Golf Environmental Commendation - Background Report,” 2014.
- [22] G. Majeau-Bettez, T. R. Hawkins, and A. H. Strømman, “Life cycle environmental assessment of lithium-ion and nickel metal hydride batteries for plug-in hybrid and battery electric vehicles.,” *Environ. Sci. Technol.*, vol. 45, no. 10, pp. 4548–54, May 2011.
- [23] A. Yoshino, “Development of the Lithium-Ion Battery and Recent Technological Trends,” in *Lithium-Ion Batteries*, Elsevier, 2014, pp. 1–20.
- [24] Hitachi Maxell Ltd., “Annual report,” Tokyo, Japan, 2003.
- [25] Hitachi Maxell Ltd., “Environmental report,” Tokyo, Japan, 2005.
- [26] Saft, “Annual Report 2008,” Saft Group SA, Bagnolet, France, 2008.
- [27] C. J. Rydh and B. A. Sandén, “Energy analysis of batteries in photovoltaic systems. Part I: Performance and energy requirements,” *Energy Convers. Manag.*, vol. 46, no. 11–12, pp. 1957–1979, Jul. 2005.
- [28] J. B. Dunn, M. Barnes, L. Gaines, J. Sullivan, and M. Q. Wang, “Material and energy flows in the materials production, assembly, and end-of-life stages of the automotive lithium-ion battery life cycle,” 2012.
- [29] C. J. Rydh and B. A. Sandén, “Energy analysis of batteries in photovoltaic systems. Part I: Performance and energy requirements,” *Energy Convers. Manag.*, vol. 46, no. 11–12, pp. 1957–1979, Jul. 2005.
- [30] K. Tagawa and R. J. Brodd, “Production processes for fabrication of lithium-ion batteries.,” in *Lithium-Ion Batteries Science and Technologies*, M. Yoshio, R. J. Brodd, and A. Kozawa, Eds. New York: Springer, 2009, pp. 181–194.
- [31] R. Arvidsson, D. Kushnir, S. Molander, and B. a. Sandén, “Energy and resource use assessment of graphene as a substitute for indium tin oxide in transparent electrodes,” *J. Clean. Prod.*, 2015.
- [32] G. Majeau-Bettez, T. R. Hawkins, and A. H. Strømman, “Life Cycle Environmental Assessment of Lithium-Ion and Nickel Metal Hydride Batteries for Plug-In Hybrid and Battery Electric Vehicles,” *Environ. Sci. Technol.*, vol. 45, no. 10, pp. 4548–4554, Apr. 2011.
- [33] B. S. Linda Ager-Wick Ellingsen and A. H. Strømman, “The size and range effect: lifecycle greenhouse gas emissions of electric vehicles.”
- [34] C. Hanisch, J. Diekmann, A. Stieger, W. Haselrieder, and A. Kwade, “Recycling of Lithium-Ion Batteries,” *Handb. Clean Energy Syst.*, no. 2011, pp. 1–24, 2015.
- [35] T. R. Hawkins, B. Singh, G. Majeau-Bettez, and A. H. Strømman, “Comparative environmental life cycle assessment of conventional and electric vehicles,” *J. Ind. Ecol.*, 2012.
- [36] J. Dewulf, G. Van der Vorst, K. Denturck, H. Van Langenhove, W. Ghyoot, J. Tytgat, and K. Vandeputte, “Recycling rechargeable lithium ion batteries: Critical analysis of natural resource savings,” *Resour.*

Conserv. Recycl., vol. 54, no. 4, pp. 229–234, Feb. 2010.

- [37] K. Richa, C. W. Babbitt, N. G. Nenadic, and G. Gaustad, “Environmental trade-offs across cascading lithium-ion battery life cycles,” *Int. J. Life Cycle Assess.*, 2015.
- [38] D. A. Notter, M. Gauch, R. Widmer, P. Wäger, A. Stamp, R. Zah, and H.-J. Althaus, “Contribution of Li-ion batteries to the environmental impact of electric vehicles,” *Environ. Sci. Technol.*, vol. 44, no. 17, pp. 6550–6, Sep. 2010.
- [39] L. Ahmadi, A. Yip, M. Fowler, S. B. Young, and R. a. Fraser, “Environmental feasibility of re-use of electric vehicle batteries,” *Sustain. Energy Technol. Assessments*, vol. 6, pp. 64–74, Jun. 2014.
- [40] L. A.-W. Ellingsen, G. Majeau-Bettez, and A. H. Strømman, “Comment on ‘The significance of Li-ion batteries in electric vehicle life-cycle energy and emissions and recycling’s role in its reduction’ in *Energy & Environmental Science*,” *J. Ind. Ecol.*, p. n/a–n/a, 2015.
- [41] G. Mulder, N. Omar, S. Pauwels, M. Meeus, F. Leemans, B. Verbrugge, W. De Nijs, P. Van Den Bossche, D. Six, and J. Van Mierlo, “Comparison of commercial battery cells in relation to material properties,” *Electrochim. Acta*, vol. 87, pp. 473–488, 2013.

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