

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

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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 CO₂-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 CO₂-eq/kWh battery, while studies using the “end-of-life” approach report emission reductions in the range of 16-32 kg CO₂-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

30

31

32 **1 Introduction**

33 Transport-related greenhouse gas (GHG) emissions have more than doubled since 1970, and have
34 increased at a faster rate than any other energy end-use sector. The transport sector consumed over half
35 of global primary oil and was responsible for nearly one-fourth of global energy-related CO₂ emissions
36 in 2010 (Sims et al., 2014). Light duty vehicles were responsible for around half of the total transport
37 energy use. From the current number of around one billion vehicles (Sousanis, 2011), the total light duty
38 vehicle ownership is expected to double in the next few decades (IEA, 2009). These patterns forecast a
39 dramatic increase in gasoline and diesel demands, and have implications for climate change, urban air
40 quality, and energy security. The projected increase in GHG emissions makes it particularly difficult for
41 the transport sector to reduce its emissions and oil dependency, and this has led to policies that mandate
42 more stringent fuel economy standards and encourage alternative drivetrain configurations and fuels
43 (Wallington et al., 2016). Electric vehicles have emerged as strong candidates among the available
44 transport alternatives (Hawkins et al., 2012). Compared to conventional vehicles, electric vehicles can
45 offer advantages in terms of powertrain efficiency, maintenance, and reduced tailpipe emissions.

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

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

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

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

91 **2 Life cycle inventory data and reported results**

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

96 **2.1 Production**

97 The different studies vary in how they report the breakdown of the GHG emissions due to production.
98 Where possible, we disaggregated emissions associated with cell materials (dark blue), other battery
99 components (pale blue), cell manufacture (dark green), battery pack assembly (pale green), and transport
100 (grey). For studies where fewer details are provided, we reported aggregated emissions associated with
101 production of cell materials and battery components combined (blue, striped), and aggregated emissions
102 associated with direct energy demand in cell manufacture and battery pack assembly combined (green,
103 striped). For two studies, we were unable to disaggregate into component- and energy-related emissions
104 and therefore, these results are reported aggregated battery pack emissions (turquoise). Figure 1 presents
105 the results in terms of kilograms of carbon dioxide equivalents per kWh of battery capacity (kg CO₂-
106 eq/kWh).

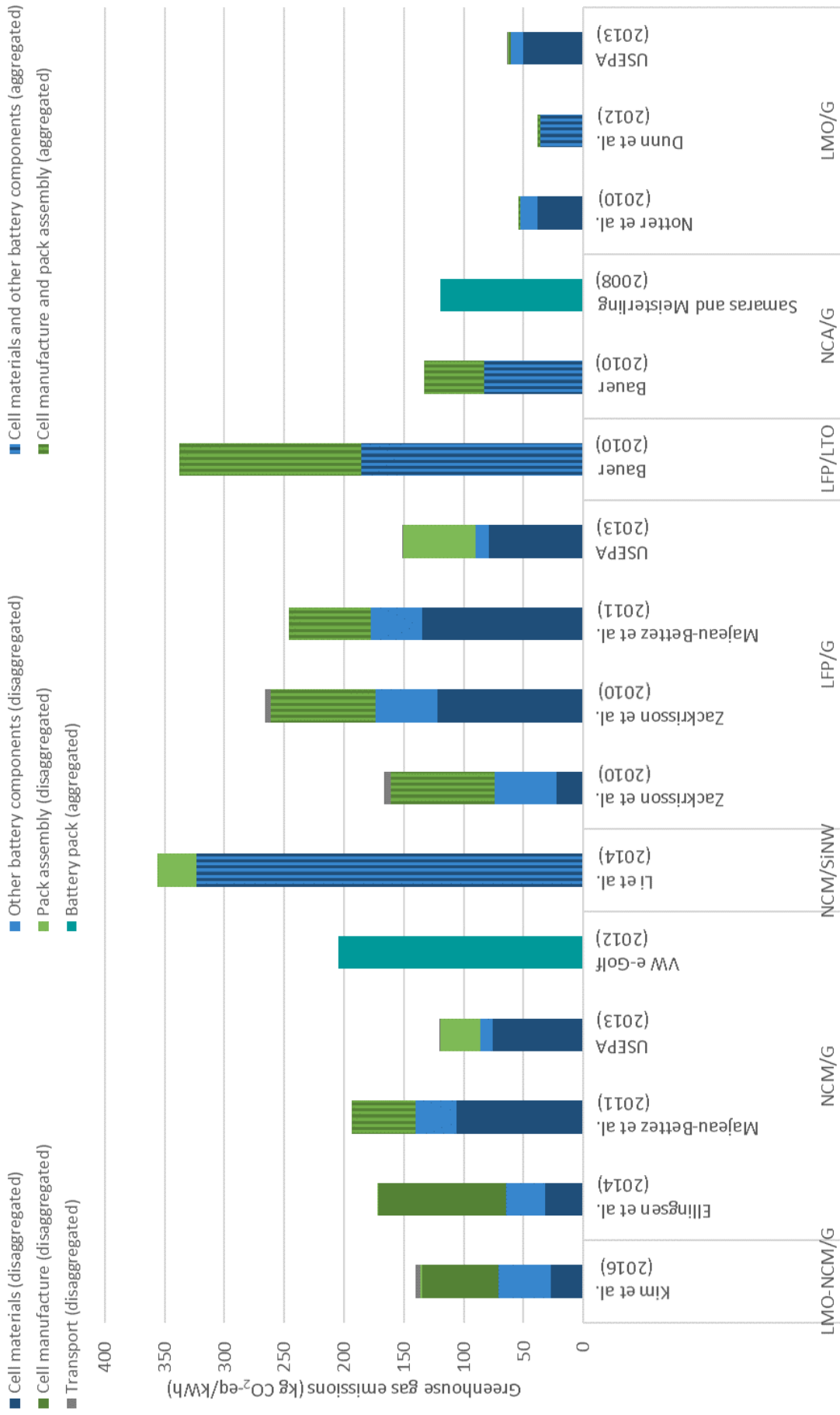


Figure 1 GHG emissions of battery production.

108 Figure 1 reports great variation in the overall production emissions with results between 38-356 kg CO₂-
 109 eq/kWh, corresponding to 0.9-8.6 tonnes of CO₂-eq for a 24 kWh battery. As can be seen in Figure 1, the
 110 studies also report different contributions from battery components and energy demand. To uncover the reasons
 111 for the different results, we examine the underlying data and assumptions in the different studies. Below, we
 112 go through some of the most important differences and contributing factors.

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

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

<i>Study</i>	<i>Cell manufacture (MJ/kWh)</i>	<i>Pack assembly (MJ/kWh)</i>	<i>Cell manufacture and pack assembly (MJ/kWh)</i>	<i>Energy sources</i>
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

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

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

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

155 Some of the studies made their own estimates for the processes involved in cell manufacture and pack
156 assembly. To determine what processes were considered in the various studies, we inspected the inventories.
157 In cell manufacture, Notter et al. (2010) considered heating of electrodes, calendaring, and a 70 % cell charge.
158 Thus, their estimate did not consider cell assembly, which is a process that places strict constraints on the
159 ambient environment and must take place in a dry-room (Schönemann, 2017; Wood et al., 2015). Furthermore,
160 it appears that Notter et al. (2010) underestimated the electricity required for formation cycling and aging. For
161 testing in battery pack assembly, Notter et al. (2010) assumed electricity for a single charge of the battery to
162 95 % of full capacity. Dunn et al. (2012) received an energy consumption estimate for a dry-room of 1860 m²
163 at 21 °C by a dry-room manufacturer, and assumed that the energy demand scales linearly with floor area. They
164 converted the values to a per-mass-of-battery basis for a room of 3000 m² with a cell production rate of six
165 million accepted cells per year. For the conditioning, Dunn et al. (2012) assumed a total of four cycles, one
166 formation cycle and three aging cycles. Their total energy demand was sufficient to fully charge their battery
167 five and a half times. Therefore, after the assumed four cycles, less than 30 % of their estimated total energy
168 demand is available for electrode production and cell assembly. This seemingly contradicts their assumption
169 that the operation of dry-rooms and conditioning account for 60 % of total energy demand. Thus, it seems that
170 both Notter et al. (2010) and Dunn et al. (2012) neglected processes in cell manufacture and therefore
171 underestimate the energy demand. Li et al. (2014) report results for a cell with an NCM cathode and a SiNW
172 anode. However, their cell inventory provides data for an LMO cathode and a graphite anode. For cell
173 manufacture, their low estimated energy demand correspond with the reported low GHG emissions, but the
174 low estimated energy demand in pack assembly do not align with the reported higher GHG emissions. Thus,
175 we suspect that Li et al. (2014) may have published the wrong sub-inventories for the cell and battery pack,
176 which, along with the lack of description for energy inputs, constrains further discussion of their energy data.

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

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

188 Another source of variation in production-related GHG emissions stems from assumptions regarding the cell
 189 materials and the amount of these. Bauer (2010), Dunn et al. (2012) and Kim et al. (2016) do not provide a
 190 contribution analysis of cell materials, which limits the discussion of cell-related emissions reported by these
 191 studies. The majority of the studies are in good agreement with respect to GHG emissions associated with
 192 graphite-based anodes, with emissions ranging between 7.5 and 9.9 kg CO₂-eq/kWh (Ellingsen et al., 2014;
 193 Notter et al., 2010; USEPA, 2013). Due to the binder used in the study by Majeau-Bettez et al. (2011), their
 194 study report higher anode emissions at 18.2 kg CO₂-eq/kWh. With a water solvent, Zackrisson et al. (2010)
 195 report only 1.6 kg CO₂-eq/kWh. Li et al. (2014) assess a LIB using silicon nanowire as anode material, which,
 196 due to the very high energy demand of its synthesis, emits 130 kg CO₂-eq/kWh. Except for the SiNW anode,
 197 the study by Li et al. (2014) appears to be exclusively based on the USEPA (2013) study; thus in the remainder
 198 of the text, we only refer to the results from the USEPA (2013) study. For the cathode, most studies find
 199 emissions in the range of 16-19 kg CO₂-eq/kWh (Ellingsen et al., 2014; Notter et al., 2010; Zackrisson et al.,
 200 2010), but Majeau-Bettez et al. (2011) report much higher emissions at 72 kg CO₂-eq/kWh, which again is due
 201 to the binder. The higher emission of 49 kg CO₂-eq/kWh reported by USEPA (2013) is likely attributable to
 202 the assumed larger amount of cathode materials in the cell. The studies find that the emissions related to the
 203 electrolyte are relatively low, ranging from 2.1 to 3.9 kg CO₂-eq/kWh (Ellingsen et al., 2014; Majeau-Bettez
 204 et al., 2011; Zackrisson et al., 2010), although USEPA (2013) reports higher impact at 14.6 kg CO₂-eq/kWh.
 205 The reported emissions associated with the separator are also low, ranging between 0.4 and 2.2 kg CO₂-
 206 eq/kWh.

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

210 *Table 2 Overview of GHG emissions (kg CO₂-eq/kWh) per component from various LCA studies.*

<i>Component</i>	<i>Kim et al. (2016)</i>	<i>Ellingsen et al. (2014)</i>	<i>Majeau-Bettez et al. (2011)</i>	<i>USEPA (2013)</i>	<i>Zackrisson et al. (2010)</i>	<i>Notter et al. (2010)</i>
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

211
 212 Although the overall emissions associated with other battery components (light blue in Figure 1) are similar,
 213 the contributions from the battery component groups vary significantly (Table 2). The studies based on primary
 214 data find similar emissions for cell materials and other battery components (Ellingsen et al., 2014; Kim et al.,
 215 2016b), while the studies based on literature or secondary data report significantly more variation in their
 216 results (Li et al., 2014; Majeau-Bettez et al., 2011; Notter et al., 2010; USEPA, 2013; Zackrisson et al., 2010).

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

226 **2.2 Use phase**

227 During use, the battery has indirect emissions that are influenced by the energy conversion losses, energy
228 required to carry the battery weight, and the carbon intensity of the electricity. The energy conversion losses
229 in the battery depends on the energy efficiency of the battery and the operational energy demand (Wh/km) of
230 the vehicles. Because it is difficult to find information regarding energy efficiency of LIBs, LCA studies often
231 make an assumption for the battery energy efficiency. Thus far, LCA studies have generally not taken into
232 account that different cell format types (e.g., pouch, cylinder, prismatic) and cathode materials (LMO, NCM,
233 LFP, NCA) offer different efficiencies (Mulder et al., 2013). USEPA (2013) assumed an energy efficiency of
234 85%, whereas Majeau-Bettez et al. (2011) and Zackrisson et al. (2010) assumed 90%. For the NCM pouch
235 cells in their study, Ellingsen et al. (2016b) used the measured energy efficiency of 95% (Ellingsen et al.,
236 2014). Depending on the carbon intensity of the electricity used for charging and the operational energy
237 demand of the vehicle, the differences can significantly influence the total life cycle emissions of LIBs. With
238 a total driving distance of 180 000 km, Ellingsen et al., (2016b) find the indirect energy demands due to
239 conversion losses to cause 638 kg CO₂-eq when charged with the average European electricity mix and 18 kg
240 CO₂-eq when charged with purely wind-based electricity. Zackrisson et al. (2010) also evaluated how much
241 of the operational energy demand can be ascribed to the weight of a battery. The study estimated the total
242 energy use due to the battery weight based on the ratio of battery to vehicle weight, the share of operational
243 energy use due to total vehicle weight, the operational energy demand of the vehicle, total driving distance,
244 and the share of the time the PHEV was in electric mode. When charged with the average European electricity
245 mix for 180 000 km, indirect emissions associated with the battery weight results in a total of 236 kg CO₂-eq.

246 **2.3 End-of-life**

247 At the EOL, useful components and materials can be reused or recycled. LIB recycling is typically a
248 combination of two or more processes (Hanisch et al., 2015). There are several competing industrial LIB
249 recycling processes, but very few LCA studies consider the emissions associated with the EOL treatment of
250 LIBs. Hawkins et al., (2012) model EOL treatment consisting of dismantling and a cryogenic chattering
251 process. Ellingsen et al. (2016b) compiled a recycling inventory based on secondary industry data for the
252 pyrometallurgical treatment described by Dewulf et al. (2010). Li et al. (2014) assume a combined direct
253 physical, hydro- and pyrometallurgical treatment process, but only consider the direct energy requirements for
254 this process. Dunn et al. (2012) and USEPA (2013) consider hydrometallurgical, intermediate physical, and
255 direct physical recycling. Dunn et al. (2012) report purchased energy consumption for this process. USEPA
256 (2013) obtained primary data from battery recyclers, but do not provide an EOL inventory. While Hawkins et
257 al., (2012), Ellingsen et al. (2016b), and Li et al., (2014) use the “recycled content” approach and report
258 emissions associated with recycling, USEPA (2013) and Dunn et al. (2012) use the “end-of-life” approach and
259 respectively report the reduction in emissions and total energy consumption that may be possible through
260 various recycling scenarios. Hawkins et al. (2012) and Ellingsen et al. (2016b) report 3.6 and 8.0 kg CO₂-
261 eq/kWh battery associated with recycling, respectively, while Li et al., (2014) report emissions of 27 kg CO₂-
262 eq/kWh battery. USEPA (2013) report emission reductions in the range of 16-32 kg CO₂-eq/kWh battery for

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

267 **3 Discussion of LCA findings**

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

272 **3.1 Evaluation and analysis of inventories and results**

273 Energy demand in battery production has been much discussed in the LCA literature (Dunn et al., 2015, 2012,
274 Ellingsen et al., 2015, 2014; Kim et al., 2016b; Majeau-Bettez et al., 2011) and the literature reports two
275 opposing views. On one hand, we have studies that assume low energy demand in cell manufacture and find
276 that energy use has insignificant contributions to production-related GHG emissions. On the other hand, we
277 have studies that report high energy demand and consequently find this to be a main source of GHG emissions.
278 Below, we consider what we know of the cell manufacturing process and review the associated energy
279 demands. Cell manufacture is a complex and protracted process that places constraints on ambient conditions
280 and therefore requires strictly controlled environments (Schönemann, 2017; Wood et al., 2015). Due to the
281 proprietary nature of the industry, the information available on the various production steps and requirements
282 is limited and data are scarce. As a result, estimating energy demands of the various production steps in cell
283 manufacture is a formidable task. Having examined the inventories, we find that studies attempting to estimate
284 the energy demand seemingly omit or underestimate energy inputs for some production steps. Furthermore,
285 two independent LCA studies based on primary industry data report significant energy use. Although some
286 have suggested that the high energy demand may be due to low production volumes (Dunn et al., 2015), this
287 has been disputed (Ellingsen et al., 2015) and proved not to be the case (Kim et al., 2016b). Thus, we conclude
288 that cell manufacture is an energy-intensive process and a main contributor to production-related GHG
289 emissions of LIBs. Unlike cell manufacture, battery pack assembly does not place constraints on ambient
290 conditions and the main energy requirements are associated with welding, testing, and charging (Schönemann,
291 2017). The low energy inputs for pack assembly reported by the majority of the LCA studies thus seem
292 reasonable.

293 Further disagreements among the studies were due to assumptions regarding cell materials and other battery
294 components. For cell materials, some of the differences are the result of assumptions regarding material types
295 and the amount of these. In general, the studies agree that the cathode and the anode have, in that order, the
296 highest cell GHG emissions and that the contributions from the electrolyte and the separator are about an order
297 of magnitude smaller. At this point, we should also mention that not all of the cathode materials assessed by
298 LCA studies are used in LIBs for electric vehicles. Although LMO is a low-cost material that has safer
299 characteristics than NCA and NCM (Kim et al., 2012), it is only used as a blended cathode material with either
300 NCM or NCA (Anderman, 2016a) as it does not provide adequate energy density or lifetime for electric vehicle
301 applications when used alone (Ellingsen et al., 2016a). LFP provides excellent cycle stability and lifetime, but
302 the material is no longer used in commercial BEVs due to its low energy density (Anderman, 2016a; Ellingsen
303 et al., 2016a). Initially, LFP was used in the Chevy Bolt battery, but in 2015 it was substituted with NCM for
304 its higher energy density (Anderman, 2016a; Voelcker, 2014). LFP is, however, still used in some PHEVs
305 (Anderman, 2016b). Of the current cathode materials, NCM and NCA have the highest energy density
306 (Ellingsen et al., 2016a) and are the only unblended cathode materials used in commercial BEVs (Anderman,
307 2016a). Regarding other battery components, studies based on literature or secondary data seemingly misjudge

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

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

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

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

337 **3.2 Potential for emission reductions**

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

353 life cycle GHG emissions as production impacts may increase (Ellingsen et al., 2016a; Kim and Wallington,
354 2013). Therefore, lightweighting efforts of electric vehicles should be carefully evaluated through a holistic
355 life cycle perspective. Evidently, the use of cleaner energy sources in electricity production will reduce indirect
356 GHG emissions associated with the battery use phase. Unfortunately, the LCA literature on EOL treatment
357 offers little guidance, as there is limited access to primary data on any of the numerous recycling schemes.

358 **4 Conclusion**

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

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

383

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387

388 **References**

389 Anderman, M., 2016a. Extract from The Tesla Battery Report.

390 Anderman, M., 2016b. Extract from the Battery Pack of Modern xEVs Report.

391 Bauer, C., 2010. Ökobilanz von Lithium-Ionen Batterien [Life cycle assessment of lithium-ion batteries], Analyse der
392 Herstellung von Energiespeichern für den Einsatz in Batteriefahrzeugen. PSI, Technology Assessment / GaBE,
393 Villigen, Switzerland.

- 394 Dewulf, J., Van der Vorst, G., Denturck, K., Van Langenhove, H., Ghyoot, W., Tytgat, J., Vandeputte, K., 2010.
395 Recycling rechargeable lithium ion batteries: Critical analysis of natural resource savings. *Resour. Conserv. Recycl.*
396 54, 229–234. doi:10.1016/j.resconrec.2009.08.004
- 397 Dunn, J.B., Gaines, L., Kelly, J.C., James, C., Gallagher, K.G., 2015. The significance of Li-ion batteries in electric
398 vehicle life-cycle energy and emissions and recycling's role in its reduction. *Energy Environ. Sci.* 8, 158–168.
- 399 Dunn, J.B., Gaines, L., Sullivan, J., Wang, M.Q., 2012. Impact of recycling on cradle-to-gate energy consumption and
400 greenhouse gas emissions of automotive lithium-ion batteries. *Environ. Sci. Technol.* 46, 12704–12710.
401 doi:10.1021/es302420z
- 402 Ellingsen, L.A.-W., Hung, C.R., Majeau-Bettez, G., Singh, B., Chen, Z., Whittingham, M.S., Strømman, A.H., 2016a.
403 Nanotechnology for environmentally sustainable electromobility. *Nat. Nanotechnol.* 11, 1039–1051.
404 doi:10.1038/nnano.2016.237
- 405 Ellingsen, L.A.-W., Majeau-Bettez, G., Singh, B., Srivastava, A.K., Valøen, L.O., Strømman, A.H., 2014. Life cycle
406 assessment of a lithium-ion battery vehicle pack. *J. Ind. Ecol.* 18, 113–124. doi:10.1111/jiec.12072
- 407 Ellingsen, L.A.-W., Majeau-Bettez, G., Strømman, A.H., 2015. Comment on “The significance of Li-ion batteries in
408 electric vehicle life-cycle energy and emissions and recycling's role in its reduction” in *Energy & Environmental*
409 *Science*. *J. Ind. Ecol.* 19, 518–519. doi:10.1111/jiec.12309
- 410 Ellingsen, L.A.-W., Singh, B., Strømman, A.H., 2016b. The size and range effect: lifecycle greenhouse gas emissions of
411 electric vehicles. *Environ. Res. Lett.* 11, 1–8. doi:10.1088/1748-9326/11/5/054010
- 412 Hanisch, C., Diekmann, J., Stieger, A., Haselrieder, W., Kwade, A., 2015. Recycling of Lithium-Ion Batteries, in:
413 *Handbook of Clean Energy Systems*. pp. 1–24. doi:10.1002/9781118991978.hces221
- 414 Hawkins, T.R., Gausen, O.M., Strømman, A.H., 2012. Environmental impacts of hybrid and electric vehicles—a review.
415 *Int. J. Life Cycle Assess.* 17, 997–1014. doi:10.1007/s11367-012-0440-9
- 416 Hawkins, T.R., Singh, B., Majeau-Bettez, G., Strømman, A.H., 2012. Comparative Environmental Life Cycle Assessment
417 of Conventional and Electric Vehicles. *J. Ind. Ecol.* 17, 53–64. doi:10.1111/j.1530-9290.2012.00532.x
- 418 Hitachi Maxell Ltd., 2005. Environmental report. Tokyo, Japan.
- 419 Hitachi Maxell Ltd., 2003. Annual report. Tokyo, Japan.
- 420 IEA, 2009. *Transport, Energy and CO2: Moving Toward Sustainability*.
- 421 Kim, H.C., Wallington, T.J., 2013. Life-Cycle Energy and Greenhouse Gas Emission Benefits of Lightweighting in
422 Automobiles : Review and Harmonization. *Environ. Sci. Technol.* 47, 6089–6097.
- 423 Kim, H.C., Wallington, T.J., Sullivan, J.L., Keoleian, G.A., 2016a. Life Cycle Assessment of Vehicle Lightweighting:
424 Novel Mathematical Methods to Estimate Use-Phase Fuel Consumption. *Environ. Sci. Technol.* 50, 11226–11233.
425 doi:10.1021/acs.est.5b01655
- 426 Kim, H.C., Wallington, T.J., Arsenaault, R., Bae, C., Ahn, S., Lee, J., 2016b. Cradle-to-Gate Emissions from a Commercial
427 Electric Vehicle Li-Ion Battery: A Comparative Analysis. *Environ. Sci. Technol.* 50, 7715–7722.
428 doi:10.1021/acs.est.6b00830
- 429 Kim, T.-H., Park, J.-S., Chang, S.K., Choi, S., Ryu, J.H., Song, H.-K., 2012. The Current Move of Lithium Ion Batteries
430 Towards the Next Phase. *Adv. Energy Mater.* 2, 860–872. doi:10.1002/aenm.201200028
- 431 Li, B., Gao, X., Li, J., Yuan, C., 2014. Life Cycle Environmental Impact of High-Capacity Lithium Ion Battery with
432 Silicon Nanowires Anode for Electric Vehicles. *Environ. Sci. Technol.* 48, 3047–3055. doi:10.1021/es4037786
- 433 Majeau-Bettez, G., Hawkins, T.R., Strømman, A.H., 2011. Life cycle environmental assessment of lithium-ion and nickel
434 metal hydride batteries for plug-in hybrid and battery electric vehicles. *Environ. Sci. Technol.* 45, 4548–54.
435 doi:10.1021/es103607c
- 436 Mulder, G., Omar, N., Pauwels, S., Meeus, M., Leemans, F., Verbrugge, B., De Nijs, W., Van Den Bossche, P., Six, D.,
437 Van Mierlo, J., 2013. Comparison of commercial battery cells in relation to material properties. *Electrochim. Acta*
438 87, 473–488. doi:10.1016/j.electacta.2012.09.042
- 439 Nealer, R., Hendrickson, T.P., 2015. Review of Recent Lifecycle Assessments of Energy and Greenhouse Gas Emissions
440 for Electric Vehicles. *Curr. Sustain. Energy Reports* 66–73. doi:10.1007/s40518-015-0033-x

- 441 Nitta, N., Wu, F., Lee, J.T., Yushin, G., 2015. Li-ion battery materials: present and future. *Mater. Today* 18, 252–264.
442 doi:10.1016/j.mattod.2014.10.040
- 443 Notter, D.A., Gauch, M., Widmer, R., Wäger, P., Stamp, A., Zah, R., Althaus, H.-J., 2010. Contribution of Li-ion batteries
444 to the environmental impact of electric vehicles. *Environ. Sci. Technol.* 44, 6550–6. doi:10.1021/es903729a
- 445 Rydh, C.J., Sandén, B.A., Sande, A., Sandén, B.A., 2005. Energy analysis of batteries in photovoltaic systems. Part I:
446 Performance and energy requirements. *Energy Convers. Manag.* 46, 1957–1979.
447 doi:10.1016/j.enconman.2004.10.003
- 448 Saft, 2008. Annual Report 2008, Management. Saft Group SA, Bagnolet, France.
- 449 Samaras, C., Meisterling, K., 2008. Life cycle assessment of greenhouse gas emissions from plug-in hybrid vehicles:
450 implications for policy. *Environ. Sci. Technol.* 42, 3170–6. doi:10.1021/es702178s
- 451 Schönemann, M., 2017. Battery Production and Simulation, in: Herrmann, C., Kara, S. (Eds.), *Multiscale Simulation*
452 *Approach for Battery Production Systems*. Springer International Publishing, pp. 11–37. doi:10.1007/978-3-319-
453 49367-1
- 454 Sims, R., Schaeffer, R., Creutzig, F., Cruz-Núñez, X., D’Agosto, M., Dimitriu, D., Meza, M.J.F., Fulton, L., Kobayashi,
455 S., Lah, O., McKinnon, A., Newman, P., Ouyang, M., Schauer, J.J., Sperling, D., Tiwari, G., Bruckner, T.,
456 Bashmakov, I.A., Mulugetta, Y., Chum, H., Navarro, A. de la V., Edmonds, J., Faaij, A., Fungtammasan, B., Garg,
457 A., Hertwich, E., Honnery, D., Infield, D., Kainuma, M., Khennas, S., Kim, S., Nimir, H.B., Riahi, K., Strachan,
458 N., Wisser, R., Zhang, X., 2014. Transport., in: Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Farahani, E., Kadner,
459 S., Seyboth, K., Adler, A., Baum, I., Brunner, S., Eickemeier, P., Kriemann, B., Savolainen, J., Schlömer, S.,
460 Stechow, C. von, Zwickel, T., Minx, J.C. (Eds.), *Climate Change 2014: Mitigation of Climate Change. Contribution*
461 *of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*.
462 Cambridge University Press, Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
- 463 Sousanis, J., 2011. World Vehicle Population Tops 1 Billion Units [WWW Document]. URL
464 http://wardsauto.com/ar/world_vehicle_population_110815 (accessed 9.16.15).
- 465 United States Environmental Protection Agency, 2013. Application of Life-Cycle Assessment to Nanoscale Technology:
466 Lithium-ion Batteries for Electric Vehicles.
- 467 Voelcker, J., 2014. 2015 Chevrolet Spark EV Switches Battery Cells; 82-Mile Range Remains [WWW Document]. *Green*
468 *Car Reports*. URL [http://www.greencarreports.com/news/1092094_2015-chevrolet-spark-ev-switches-battery-](http://www.greencarreports.com/news/1092094_2015-chevrolet-spark-ev-switches-battery-cells-82-mile-range-remains)
469 [cells-82-mile-range-remains](http://www.greencarreports.com/news/1092094_2015-chevrolet-spark-ev-switches-battery-cells-82-mile-range-remains) (accessed 7.22.16).
- 470 Volkswagen AG, 2012. The e-mission. Electric mobility and the environment. Wolfsburg, Germany.
- 471 Wallington, T.J., Anderson, J.E., De Kleine, R.D., Kim, H.C., Maas, H., Brandt, A.R., Keoleian, G.A., 2016. When
472 Comparing Alternative Fuel-Vehicle Systems, Life Cycle Assessment Studies Should Consider Trends in Oil
473 Production. *J. Ind. Ecol.* doi:10.1111/jiec.12418
- 474 Wood, D.L., Li, J., Daniel, C., 2015. Prospects for reducing the processing cost of lithium ion batteries. *J. Power Sources*
475 275, 234–242. doi:10.1016/j.jpowsour.2014.11.019
- 476 Zackrisson, M., Avellan, L., Orlenius, J., 2010. Life cycle assessment of lithium-ion batteries for plug-in hybrid electric
477 vehicles - Critical issues. *J. Clean. Prod.* 18, 1519–1529. doi:DOI 10.1016/j.jclepro.2010.06.004
- 478