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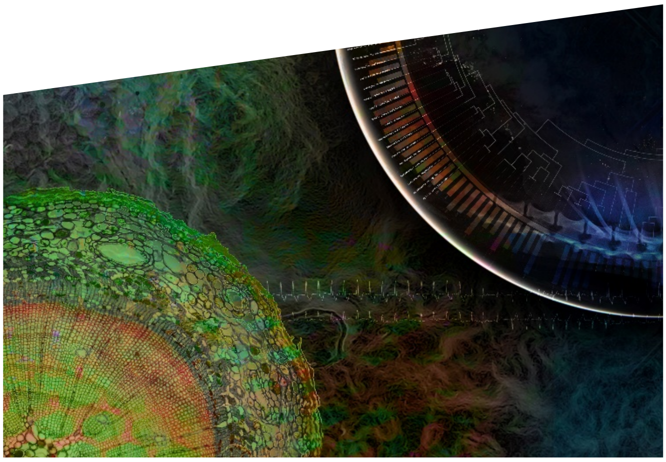

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State of the art within life cycle assessments of lithium-ion batteries

– An investigative study of key differences in previous studies and future needs of research

Nulägesrapport inom livscykelanalyser av litiumjonbatterier

- En undersökning av viktiga skillnader i tidigare studier och framtida forskningsbehov

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Abstract
The use of electric vehicles has seen a rapid growth in the past decade as it is expected to have an important role in mitigating greenhouse gas emissions from the transport sector. The increased demand for electric vehicles has in turn amplified the demand for traction batteries, especially lithium-ion batteries. However, to avoid problem-shifting it is important to consider the life-cycle-impact from the transition to an electrified transport fleet, since lithium-ion batteries are associated with its own environmental problems. Previous life-cycle assessments have been conducted, however, many studies lack in traceability and transparency regarding both battery composition, raw material supply and the processes included in the system boundaries. This makes it hard to compare the results and prevents the field from reaching a consensus in the best approach in conducting life-cycle assessments.

To enable future life-cycle assessments of lithium-ion batteries it is important to identify the key assumptions in the previous studies and understand why they differ to such large extent. This report will therefore assess how previous studies differ in two regards: 1. the methodological choices and 2. the environmental impact categories evaluated. By doing so, this report helps future studies identify how these choices affect the results and therefore aid in choosing the best options.

The results of the study show that the processes with the largest contribution to the environmental impact is the mining of materials, cell assembly and the use-phase and should therefore be assessed further in detail in future studies in order to reduce uncertainties. The impact categories that are concerned the most is the global warming potential, human toxicity potential, acidification potential, eutrophication potential and abiotic depletion potential. A large contributor to several of these impact categories is the electricity used in both the production- and use-phase. Consequently, using an electricity mix with higher shares of renewables have been stated as an efficient measure to reduce the life-cycle impact from lithium-ion batteries. Furthermore, there is a need for future studies that conduct full cradle-to-grave life-cycle assessments with primary data from manufacturers, preferably representable of large-scale production and data that better represents the current industry practice end-of-life treatment.

The best practice model to conduct future lifecycle assessment is dependent on the goal and scope of the study. Both the functional unit and the system boundaries should reflect the goal of the assessment. Cradle-to-grave assessment is to be preferred if the goal is to compare the environmental impact between using batteries in electric vehicles to internal combustion vehicles. In that case, the functional unit should also include the performance of the battery, such as charge efficiency and lifetime, and therefore the unit kilometres driven is preferred. Also, in order to include the most common environmental impact categories that are concerned, five categories should be included: the global warming potential, eutrophication potential, acidification potential, abiotic depletion potential and human toxicity potential.

Sammanfattning

Elektrifieringen av transportflottan sker runt om i världen för att minska utsläppen av växthusgaser från transportsektorn. Följaktligen har därmed även efterfrågan på litium-jonbatterier ökat markant runt om i världen. För att undvika att byta ut ett problem mot ett annat är det viktigt att förstå hur detta skifte mot en elektrifierad transportsektor påverkar miljön ur ett livscykelperspektiv.

Livscykelanalyser på litium-jonbatterier har gjorts tidigare, men resultaten från rapporterna varierar mycket. Många av studierna har även låg data-kvalité och transparens i metodval. Detta har resulterat i stor osäkerhet kring batteriernas egentliga miljöpåverkan och forskarna inom ämnet har efterlyst fler livscykelanalyser som har högre datakvalité och som kan minska osäkerheten inom området. Denna studie syftar därför till att kartlägga de metodologiska val som gjorts i tidigare studier för att undersöka viktiga skillnader som kan ha bidragit till den stora variationen på resultaten. Sju tidigare livscykelanalyser har analyserats och deras metodval såsom systemgränser, funktionell enhet, datakällor och metod för miljö-påverkansbedömning presenteras tillsammans med vilka miljöpåverkanskategorier som har undersökts. Studien avslutas med en diskussion kring hur dessa val har påverkat resultatet och hur framtida studier bör utföras med avseende på dessa val.

Studien fann att de processer som bidrar mest till miljöpåverkan från litium-jonbatterierna är råvaruutvinning, montering av cellerna och användningsfasen av batterierna. Genom att undersöka dessa tre processer mer noggrant i framtida studier kan därmed osäkerheten och variationen i resultatet minskas. De miljöpåverkanskategoriser som påverkades i störst utsträckning fanns vara den globala uppvärmningspotentialen, humantoxicitetspotentialen, försurningspotentialen, övergödningspotentialen och den abiotiska utarmningspotentialen. En betydande faktor till flera av dessa miljöpåverkanskategorier fanns vara den använda elmixen i både produktion- och användningsfasen. Desto större andel av förnyelsebara energikällor i elmixen, desto mindre var miljöpåverkan. Resultatet från studien visar också att det finns ett behov av framtida studier som utför fullständiga studier från vagga till grav. Dessutom finns det behov av studier som använder primärdata från industrin som bättre representerar verklig storskalig produktion.

Vilka val som framtida livscykelanalyser bör göra gällande metodvalen beror på studiens mål och omfattning. Både de valda systemgränserna och funktionella enhet bör återspegla studiens frågeställning och mål. För en studie som syftar till att jämföra batteriers livscykelmiljöpåverkan bör batteriets verkningsgrad och livslängd inkluderas och därmed passar en studie från vagga till grav bättre. Likaså bör då funktionella enheten även ta dessa faktorer i beaktning vilket enheten en kilometer körd sträcka gör. Denna studie har därmed bidragit med insikter kring vad framtida livscykelanalyser bör ta extra hänsyn till och inkludera för att ge ett mer tillförlitligt resultat.

Preface

This study has been conducted at the Swedish University of Agricultural Sciences as a 10-credit course part of the civil engineering degree program in Energy Systems. The study aims to provide a guidance in which methodological choices are important to consider for future life cycle assessment studies in lithium-ion batteries.

I would like to thank my supervisor Oscar Lagnelöv for all of his support and guidance which have been very valuable in order to execute and finish this report.

Furthermore, I would also like to thank my subject reader Gunnar Larsson for his much-appreciated insights and knowledge within the area of study.

Lollo Liu
Uppsala, 2020-03-20

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1. Introduction

The use of electric vehicles (EV) have seen a rapid growth in the past decade as it is expected to have an important role in mitigating greenhouse gas (GHG) emissions from the transport sector. In 2018 the global EV car fleet exceeded 5.1 million cars, which corresponded to a growth of over 60 % compared to the previous year (International Energy Agency, 2019). The Swedish government have set targets to reduce CO2 emissions from the transport sector by 70 % in 2030 compared to 2010 and to achieve overall net zero GHG emissions by 2045 (Government of Sweden, 2018). To reach these targets the Swedish government has been active in facilitating the expansion of EVs by investing in and enabling large scale production of lithium-ion batteries (LIBs). The market share of sold EVs and hybrid electric vehicles in Sweden was 8 % in 2018, which corresponds to the third largest market share in the world, after Iceland at 17 % and Norway at 46 % (International Energy Agency 2019).

The increased demand for electric vehicles has in turn amplified the demand for traction batteries, especially LIBs due to their high energy density and energy storage capacity compared to other batteries (International Energy Agency 2019). However, to avoid problem-shifting and increase production efficiency it is important to consider the life-cycle-impact from the transition to an electrified transport fleet as LIBs are associated with its own environmental problems. The cradle-to-gate processes, including raw material mining, component production, cell assembly and packaging is energy intensive and produces toxic pollutions (Ellingsen *et al.* 2017). Many of the necessary materials in LIBs, such as cobalt, aluminium, copper and nickel are especially coupled with energy consumption and critical geopolitical risks. The extraction often has a large negative regional effect depending on its country of origins (Kelly *et al.,* 2019).

Even though previous life-cycle assessments concur in the fact that the upstream processes are highly energy consuming (Majeau-Bettez *et al.* 2011; Kim *et al.* 2016; Ellingsen *et al.* 2017) the results from the studies vary considerably. Results between 38 and 487 kg CO_2-eq/kWh have been reported and the studies often lack in traceability and transparency regarding both battery composition, raw material supply and the processes included in the system boundaries (Kelly, Dai & Wang

2019). This makes it hard to compare the results and prevents the field from reaching a consensus in the best approach in conducting lifecycle assessments (LCA).

To enable future LCA on LIB it is important to identify the key assumptions made and understand why previous studies differ to such large extent. This report will therefore assess how previous studies differ in two regards: 1. the methodological choices and 2. the environmental impact categories evaluated. By doing so, this report helps future studies identify how these choices affect the results and therefore aid in choosing the best options.

2. Purpose

The purpose of this study is to evaluate previous LCA studies on LIBs and how they differ in two choices; 1. methodological choices and 2. environmental impact categories used, in order to determine how these choices have affected the result and which processes and choices are most important to consider in future LCA studies.

The methodological choices will be compared in how they affect the LCA outcome and will result in a brief discussion regarding how Swedish conditions, such as the energy mix and production capacity is expected to affect the methodological choices. By doing so, this report helps future studies identify how these choices affect the results and therefore give guidance in how to best design future LCA on LIBs.

A discussion regarding the global warming potential (GWP) from previous studies will be presented for the processes in order to map out which ones have significant impact on the LCA results. Furthermore, to give guidance in which environmental impact categories are important to consider in future studies in addition to the global warming potential, the environmental impact categories considered in the previous studies will be presented. Finally, a best practice within the methodological choices and environmental impact categories will be developed from the results of this report.

3. Method

This study aimed to identify the key assumptions in the previous reports that may have caused the large variance between the LCA results. In order to do this, eight reports were examined, of which, one was a comparative study of previous LCAs whilst the other seven reports were LCAs of LIBs. In order to assess how the studies differed in the choices regarding methodology and environmental impact categories this report compared the choices and a discussion regarding their effect on the results. In order to compare the studies, the results were given on a cradle-to-gate perspective since five of eight reports only included the processes up to and during the production. However, when possible the cradle-to-grave result was also presented.

Firstly, the methodological choices such as system boundaries, battery composition, functional unit and data source used was presented. The choices were compared and a discussion regarding their effect on the end results was given from Swedish perspective in order to add insight in how these choices will differ for future battery manufacturing within Sweden.

The system boundaries and the processes included in the studies was mapped out and presented. The contribution of each process to the environmental impact was evaluated and compared. Although, it is a complicated to compare studies with different assumptions regarding battery composition, system boundaries and assessment models, it was necessary to choose a simple metric for comparison. Therefore, the comparison in this study was based on the share of contribution to the total GWP. GWP is the most commonly used metric and the one used for comparison in all the reviewed studies. However, a discussion regarding the importance of other impact metrics used in the studies and how well they capture the intended impact affect were also given. The functional unit was chosen as one kWh of nominal capacity in order to compare between batteries with different efficiency and because it was the most commonly used metric in the reviewed studies. Although it may not be correct to compare batteries with different nominal capacity on a kWh basis, it can be assumed that the comparison is acceptable since the linear relationship is an adequate assumption when comparing battery packs and not cells.

Secondly, the environmental impact categories studied in the previous studies were presented and reviewed based on the 11 most commonly used impact categories, which are: 1. global warming potential, 2. ozone depletion potential, 3. acidification, 4. eutrophication potential, 5.photochemical oxidation potential, 6. ecological toxicity potential, 7. aquatic toxicity, 8. human toxicity potential, 9. abiotic resource depletion potential, 10. land use and 11. water use, which are all based on the methods mentioned in Matthews, Hendrickson & Matthews (2014). A discussion regarding the impact of each category was the given and how the Swedish conditions for future battery production can affect the environmental impact categories.

Lastly, a best practice within the choice of method and environmental impact was developed. The results should be viewed as guidance in what choices are suitable and useful in future LCA studies from a Swedish perspective.

4. Results

A comparison of previous studies in LCA of LIBs have been conducted by Ellingsen, Hung & Stromman (2017) but widely differing results are reported amongst the compared studies. The wide range of results can be derived from the various assumptions made in the studies regarding battery composition, choice of LCA method and system boundaries. Furthermore, due to lack of primary data from manufacturers, many of the studies show poor life cycle inventory resolution and have low data quality because they use second-hand data from previous studies and literature (Cusenza *et al.*, 2019). This has also resulted in many studies using the same data from studies such as Majeau-Bettez, Hawkins & Stromman (2011) and EPA (2013), which therefore gives higher uncertainties in how well the results actually reflects the reality (Ellingsen, Hung & Stromman, 2017).

The study conducted by Ellingsen, Hung & Stromman (2017) compares 9 different LCA studies based on the GWP-metric since it is the most commonly used metric and the only one that is consequently used in all studies. However, this makes the comparison somewhat subjective and only concerns the climate change impact category. Therefore, this study intends to review previous studies based on the 11 most commonly used impact categories which are: 1. global warming potential, 2. ozone depletion potential, 3. acidification, 4. eutrophication potential, 5.photochemical oxidation potential, 6. ecological toxicity potential, 7. aquatic toxicity, 8. human toxicity potential, 9. abiotic resource depletion potential, 10. land use and 11. water use. The impact categories are based on the methods mentioned in Matthews, Hendrickson & Matthews (2014).

Three different types of LIBs are assessed in the reviewed studies: lithium manganese oxide (LMO), lithium nickel manganese cobalt (NMC) and a composite cathode material of both LMO-NMC, as presented in table 1. Some studies, such as Majeau-Bettez, Hawkins & Stromman (2011) and Kelly, Dai & Wang (2019) also looked into Lithium Iron Phosphate (LFP) batteries, but the LFP batteries were excluded in this study because they have been debated to have lower potential in EVs due to their low energy density compared to NMC and LMO batteries (Ellingsen, Hung & Stromman, 2017). The processes included in the system boundaries for each study is presented in figure 1.

One of the most recent LCA studies by Cusenza *et al.* (2019) was a cradle-to-grave study on a LIB with a composite cathode material with both LMO and LMC as active material. The study aimed to present the first bill of materials with primary data for a composite cathode material LIB and to identify the critical processes in the life cycle since previous reports have mainly focused on either LMO or LMC batteries but not composite. The use-phase impact and the end of life (EOL) treatment was also taken into account in the report which adds to its further contribution to the LCA literature within LIBs. The authors used an attributional approach for the LCA.

Another study from 2019 was conducted by Kelly, Dai & Wang (2019) which explores the regional effects on the production of NMC batteries and the associated environmental impacts from the regional supply chains with a cradle-to-gate perspective. The study compared how the European, American and the Chinese supply chain affected the life cycle pollutants, energy demand and water consumption. The authors did not specify if attributional or consequential approach was used in the LCA.

A third study was conducted by Kim *et al.* (2016) which presented the first cradle-to-gate LCA result on a mass produced LMO-NMC battery with primary data from the Ford Motor Company. The study aimed to compare the LIB in the Ford Focus battery electric vehicle (BEV) to the Ford Focus internal combustion engine vehicle (ICEV) and found that there is indeed much potential in LIBs to reduce the GWP from the transport sector. The authors used an attributional approach for the LCA.

Another cradle-to-gate LCA of an NMC battery was conducted by Ellingsen *et al.* (2014). The report aimed to provide an LCA based on primary data from the company Miljobil Grenland since preceding studies was mainly based on secondary data according to the authors. However, when data was not available, secondary data was used from Majeau-Bettez, Hawkins & Stromman (2010). The authors used an attributional approach for the LCA.

Amongst the older studies, there are three studies that provide the most complete life cycle inventories conducted by EPA (2013), Majeau-Bettez, Hawkins & Stromman (2011) and Notter *et al.* (2010). These three studies are the most regularly cited and sourced for secondary data in more recent studies (Cusenza *et al.*, 2019; Ellingsen, Hung & Stromman, 2017; Kelly, Dai & Wang 2019). The authors did not specify if attributional or consequential approach was used in the LCAs.

EPA (2013) conducted a full cradle-to-grave analysis on one LMO, one LFP and one NMC battery in collaboration with industry and academia. The aim was to identify potential opportunities to decrease environmental impact and to serve as a benchmark for further studies. The study was conducted with a mainly American data, such as the electricity mix used and the supply chain. Primary data was collected from manufacturers, suppliers and recyclers when possible and supplemented with second-hand data.

Majeau-Bettez, Hawkins & Stromman (2011) conducted a comparative LCA for three battery types: lithium nickel manganese cobalt (NMC), nickel metal hydride (NCM) and lithium Iron Phosphate (LFP) including upstream processes, production and use-phase. The data used was both industry figures as well as from literature. In this study, only the results regarding the NMC will be taken into consideration since the NCM and LFP is outside of the scope of this study.

Notter *et al*. (2010) conducted a full cradle-to-grave LCA on an LMO battery in order to map out the contribution of environmental impacts of LIBs to EVs. The report is taken from a European perspective and compares how the BEV performs against an ICEV.

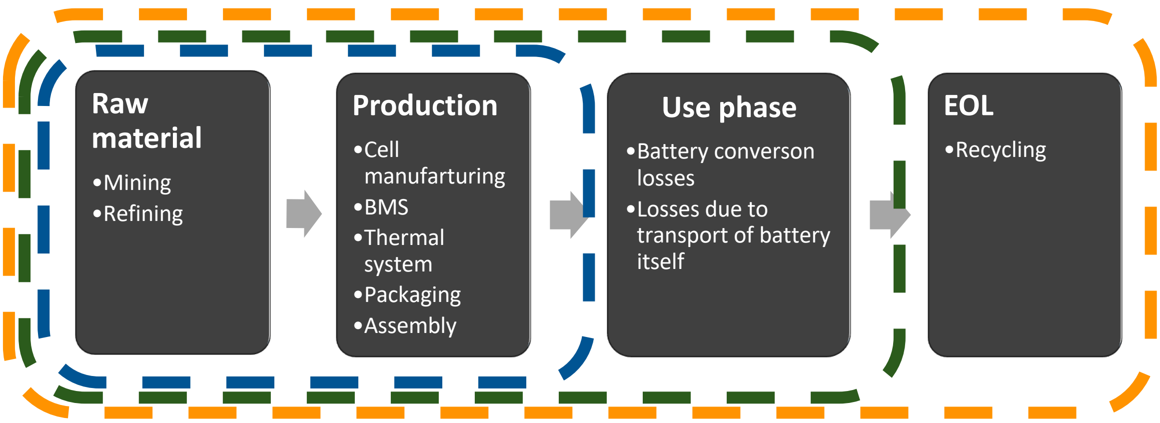

Figure 1. Simplified flow chart of the included processes in the reviewed studies. The system boundaries used in the reviewed studies are presented as the dotted lines, where the orange line represents the cradle-to-grave boundaries used in EPA (2013) and Notter et al (2010). The blue line represents the cradle-to-gate system used in Kelly, Dai & Wang (2019), Kim et al. (2016) and Ellingsen et al. (2014), whilst the green line represents the system boundaries used in Majeau-Bettez, Hawkins & Stromman (2011). BMS is the Battery Management System.

4.1. Methodological choices

The methodological choices in the reviewed studies differ widely in parameters such as battery characteristics, location of production, location of usage, electricity mix and chosen approach for the LCA. A comparison of the reviewed reports follows.

4.1.1. Functional unit

The analysed batteries in the reports all have different mass, number of cells, binders and material compositions. Also, the assumed lifetime and capacity in the batteries differ between 1,4 and 34,2 kWh, as presented in table 1. However, it can be assumed that the comparison can be done in a linear fashion between an 11,4 kWh and a 27 kWh battery since the relationship is linear when comparing battery packs.

The most common functional unit, chosen in four of the seven LCA studies, was one kWh of nominal capacity (Kelly, Dai & Wang, 2009; Kim *et al*., 2019; Ellingsen *et al*. 2014; Majeau-Bettez, Hawkins & Stromman , 2011) whilst two studies used one kilometre driven (EPA, 2013; Notter *et al*., 2010) and one used one battery pack (Cusenza *et al*., 2019). One battery pack as functional unit makes the comparison with other studies and other batteries more complicated, which is why Cusenza *et al*. (2019) also provided results in kWh in addition to the chosen functional unit. In some studies, multiple functional units where used, such as in Majeau-Bettez, Hawkins & Stromman, (2011). The authors used one main functional unit which was energy stored and delivered to powertrain, but they also presented the data in two other functional units; one kg battery and one kWh, in order give comparable results to other studies. Majeau-Bettez, Hawkins & Stromman, (2011) pointed out that the NMC battery in their research preformed similarly to the LFP battery on a per mass or nominal capacity basis but since LFPs have longer life expectancy, the LFP may result in a better performance on a per energy delivered unit.

The choice of functional unit is dependent on the goal and scope of the study. If the goal is to compare BEVs to other transportations techniques, there is a clear advantage in using kilometres driven as functional unit since the end use is the transported distance. When using distance as functional unit, the usage of the battery is taken into consideration and therefore energy efficiency and the lifetime of the battery is included in the evaluation. But if the goal of the study is to compare between different batteries it may be sufficient and easier to use nominal capacity since it reduces the need of assumptions regarding how to convert kWh to kilometres driven. The downside to using kWh is that the energy efficiency of the battery and the lifetime is not taken into consideration. Therefore, it is more suitable when comparing batteries with different application and as a supplement to other functional units for comparison between studies.

4.1.2. Data sources

Due to lack of primary data, mainly in the production phase, many authors in the reviewed studies used second-hand data from literature or made assumptions regarding inputs and battery composition, as presented in table 1. This has resulted in low LCI resolution and higher uncertainties in the results (Ellingsen, Hung & Stromman, 2017; Kim *et al*, 2016). Furthermore, some of the data that have been used is for small-scale production such as laboratory scale and then scaled up, since data for industry production data is limited due to low transparency from industry (Ellingsen, Hung & Stromman, 2017). This has led researchers to use small-scale production data and assumptions regarding the scalability in order to mimic a future large scale-production (Kim *et al*., 2016). The data is therefore somewhat uncertain and many researchers (Ellingsen, Hung & Stromman, 2017; Kim *et al*., 2019) anticipate a more efficient production in the future when the industry are able to scale up the production. This will presumably lead to savings in both material and energy input. Therefore, there is a need for further cradle-to-grave LCA studies conducted with accurate industrial data, preferable for large scale-production on LIBs.

Table.1 Review of previous studies, with GWP for production, use-phase and EOL given in kg CO2-eq/kWh battery capacity.

	Data source	Battery type	Production	Use-phase	EOL	Total
Cusenza *et al*. 2019	Primary and Majeau-Bettez *et al*. (2010)	LMO-NMC (11,4kWh)	312	67	16	397
Kelly, Dai & Wang, 2019	Secondary from GREET	NMC (27kWh), LFP	66-100			
Ellingsen, Hung & Stromman, 2017	Investigative		38-356			
Kim *et al*, 2016	Primary	LMO-NMC (24kWh)	140			
Ellingsen *et al*. 2014	Primary and from Majeau-Bettez *et al*. (2010)	NMC (26,6kWh)	172-487			
EPA 2013	Primary, Majeau-Bettez *et al*. (2010) and Notter *et al*. (2010)	NMC (40 kWh)	121	580	-16	645
		LMO (40kWh)	63	581	-28	691
Majeau-Bettez, Hawkins & Stromman 2011	Primary and secondary from literature	NMC (N.A.)	200			
Notter *et al*. 2010	Primary, Estimates	LMO (34,2kWh)	53	-	-	-

GWP for production, use-phase and EOL is given in kg CO2-eq/kWh battery capacity.

The most used source of second-hand data is from Majeau-Bettez, Hawkins & Stromman (2011), which is used by Cusenza *et al.* (2019), Ellingsen *et al.* (2014) and EPA (2013). Therefore, the accuracy of these reports also highly depends on the accuracy of Majeau-Bettez, Hawkins & Stromman (2011). Majeau-Bettez, Hawkins & Stromman (2011) in turn used both primary data and literature data from preceding reports of Gaines & Cuenca (2000), Schexnayder *et al*. (2001) and Rydh & Sandén (2005).

Amongst the more recent studies, only Kim *et al*. (2016) used primary industry data. Cusenza *et al* (2019) used primary data for the bill of materials by dismantling a battery for evaluation of the composition. For the upstream-processes and production data, second-hand data was collected from both Majeau-Bettez, Hawkins & Stromman (2011) and Ellingsen *et al*. (2014). Kelly, Dai & Wang (2019) used only secondary data from GREET, which is a model specially created for LCA of vehicles and is commonly used when conducting wheel-to-wheel analysis (Argonne National Laboratory, 2018).

Furthermore, amongst the older reports only Notter *et al* (2010) used primary data from the industry whilst Ellingsen *et al*. (2014) used primary data but also estimates from Majeau-Bettez *et al* (2011). EPA (2013) used mostly primary data, but also Majeau-Bettez *et al* (2011) as source of second-hand data. As Ellingsen, Hung & Stromman (2017) stated there is a need for more accurate studies with real production data since most of the previous studies use second-hand data. This is still the case, since the more recent studies after 2017 also is highly dependent on these previous reports.

4.1.3. Environmental Impact Assessment Models

The most commonly used LCIA models was GREET and ReCiPe, but the chosen models vary significantly between the studies as can be seen in table 2. The GREET model is specially formed to take into consideration the important impacts for vehicles and is commonly used when conducting well-to-wheel analysis (Argonne National Laboratory, 2018). The GREET model is a US-Centric model and only takes into account the energy consumption, water consumption and air pollutants associated with the material and fuel usage for the product (Argonne National Laboratory, 2018). The ReCiPe model is however more complete and takes into account all the most commonly used categories: 1. global warming potential, 2. ozone depletion potential, 3. acidification potential, 4. eutrophication potential, 5.photochemical oxidation potential, 6. ecological toxicity potential, 7. aquatic toxicity potential, 8. human toxicity potential, 9. abiotic resource depletion potential, 10. land use and 11. water use, as presented in Matthews, Hendrickson & Matthews (2014).

Table 2. Environmental Impact assessment models used in previous LCA studies.

	IPCC	ReCiPe	USEtox	GREET	PEFCR
Cusenza *et al*. 2019	X		X		X
Kelly, Dai & Wang 2019				X	
Ellingsen 2017					
Kim 2016b				X	
Ellingsen 2014		X			
Majeau-Bettez 2011		X			
EPA 2013			X		
Notter 2010				X	

4.1.4. System boundaries

Three of the review reports, Cusenza *et al*. (2019), EPA (2013) and Notter *et al*. (2010) conducted full cradle-to-grave LCAs including mining of raw materials, production of battery cell and pack, use-phase and EOL treatment. The other five studies all conducted cradle-to-gate LCAs with the exception of Majeau-Bettez, Hawkins & Stromman (2011), which also included use-phase as was presented in Figure 1.

4.1.5. Cradle-to-gate

The reported cradle-to-gate GWP differs a lot for LMO-NMC batteries, Cusenza *et al*. (2019) reported 312 kg CO2-eq/kWh whilst Kim *et al*. (2016) with reported 140 kg CO2-eq/kWh as seen in table 1.

The average cradle-to-gate GWP for LMO batteries was 58 kg CO2-eq/kWh (Notter *et al.,*2010; EPA 2013) whilst the average GWP for NMC batteries was 147 CO2-eq/kWh (Kelly, Dai & Wang, 2019; Ellingsen *et al*., 2014) as presented in figure 2. This result points to the fact that NMC seems to have higher impact than LMO, but according to Kim *et al*. (2016) the reason for this is mainly due to the fact that the NMC batteries have been allocated higher energy demand during manufacturing compared to LMO batteries in the reports but there are no fundamental battery characteristics that causes the higher energy demand. Therefore, the differences are derived by methodological choices and assumptions in the studies according to Kim *et al*. (2019).

Moreover, in the study by EPA (2013) the NMC have higher GWP compared to LMO as well, but this is most likely due to the fact that EPA (2013) used data from Majeau-Bettez *et al*. (2011) for the NMC battery and data from Notter *et al*. (2010) for the LMO battery. According to Ellingsen, Hung & Stromman (2017) one reason why Notter *et al*. (2010) presented such low values for their lifecycle GWP, at only 52,6 kg CO_2-eq/ kWh, was because the authors neglected some processes in the cell manufacturing phase. Notter *et al*. (2010) calculated the energy consumption based on own estimates and did not consider the cell assembly that is associated with high energy consumption because of the dry room requirement, which can explain why the energy consumption is so low compared to other studies. Therefore, this misinterpretation also affects the EPA (2013) report.

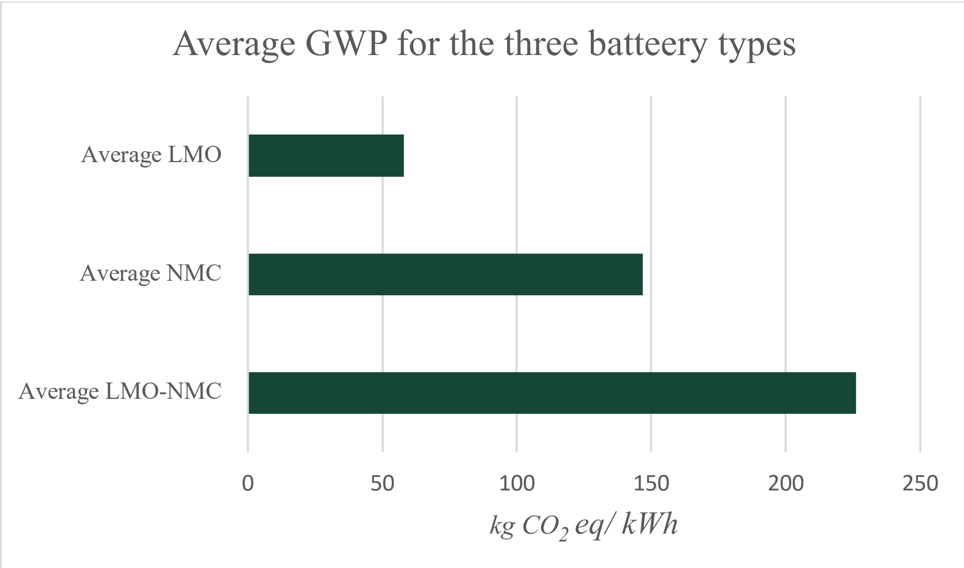

Figure 2. The GWP impact reported in the previous studies for the three battery types LMO-NMC, NMC and LMO. Lower values are presented if the report offered both an average value and a lower value.

In the study conducted by Ellingsen, Hung & Stromman (2017) the authors found that the key contributors to GWP in the cradle-to-gate analysis was the upstream processes, such as mining and raw material extraction. This is in accordance with Kelly, Dai & Wang (2019) that discovered that the upstream processes together with the cell assembly are the largest contributors to the overall environmental impact from LIBs in the cradle-to-gate perspective. Many of the reports reviewed in this study agree that cell assembly have high impact. According to Kim *et al*. (2016) cell assembly stood for 45 % of the total GWP emissions, 80 % according to Cusenza *et al*. (2019) and according to Kelly, Dai & Wang (2019) the cell assembly accounted for 18 % of total energy consumption. Despite the variation in impact, the reports all conclude that the cell assembly process have high environmental impact compared to the other processes. The reason why is because of the dry rooms that are required during the cell assembly. The dry rooms are highly energy consuming because of the large amount of air that needs to be temperature controlled and dried (Kelly, Dai & Wang, 2019; Ellingsen, Hung & Stromman, 2017).

According to the results of Ellingsen *et al* (2014), Notter *et al* (2010) and EPA (2013) the GHG emission from the graphite-based anodes lies within 7.5-9.9 kg CO_2-eq /kWh. The emissions corresponding to the cathode was reported to be between 16-19 kg CO_2 eq/kWh (Ellingson *et al*., 2014; Notter *et al*., 2010). However, Majeau-Bettez, Hawkins & Stromman (2011) reports significantly higher emissions, but according to both Ellingsen (2017) and Kim *et al*. (2016) this is probably due to the fact that Majeau-Bettez *et al* (2011) used a special binder material in their process which resulted in much higher emissions.

Kelly, Dai & Wang (2019) found that the active cathode material and energy use for cell assembly are the two main drivers of environmental impact for LIBs in the cradle-to-gate LCA. But this is also highly dependent on regional supply chain. The study found that a European supply chain generated 65 kg CO_2 eq /kWh, whilst a China-dominated supply chain generates 100 kg CO_2 e/kWh (Kelly, Dai & Wang, 2019). This indicated that the regional effects of supply chain and manufacturing have a large impact on how sustainable the LIBs are. Especially SO_x emissions related to nickel production are affected by regional differences. This is because the production of nickel can give high SO_x emissions, but these are easily reduced by using SO_x capturing efforts, like many of the production sited in Europe use. However, in Russia and China the SO_x are not captured, resulting in high emissions (Kelly, Dai & Wang, 2019). The study did not find one regional supply chain that outperforms all others in every aspect, but the ones powered by renewable electricity provides the largest emission reduction potential.

According to Ellingsen, Hung & Stromman (2017) the difference in GWP impact of the batteries in the reviewed studies were not mainly due to differences in the material compositions, instead the authors advocated that assumptions made regarding the different energy demand in battery production and battery components where the main drivers. But since use phase and EOL is often not considered in the previous studies, there is a need of more cradle-to-grave studies that can further evaluate those steps. Also, according to Ellingsen *et al*. (2017) many of the studies they reviewed were based on data from previously published studies and not on primary data. Therefore, the authors expressed a need for more accurate studies with real production data.

4.1.6. Cradle-to-grave

Amongst the reviewed studies that conducted full cradle-to-grave analysis, all three reports concur that the mining of raw materials, especially aluminium and copper, together with the cell-assembly are large drivers of environmental impact (Cusenza *et al*. 2019; Notter *et al*. 2010; EPA, 2013). However, there are some inconsistency regarding the share of contribution to the global warming potential from the use-phase and production.

According to EPA (2013) around 80-88 % of the GWP emissions is associated with the use-phase which implies that the use-phase is a very large contributor to the total life cycle impact from the batteries. In contrast, Cusenza *et al*. (2019) found that the use-phase actually had low impact in all categories, less than 20%, except for the impact of ionizing radiation on human health where it contributed to 55 % of the impact. Instead, the authors argued that the production-phase contributed to more than 60 % of all environmental impact in all assessed categories even in the cradle-to-grave analysis and it was therefore pointed out as the largest contributor in the life-cycle-impact. This might be an outcome due to that Cusenza *et al*. (2019) assumed a higher charge efficiency of 95% compared to 80% and 85% for Notter *et al*. (2010) and EPA (2013). Furthermore, the assumed lifetime and driven kilometres also differ between the reports. Cusenza *et al*. (2019) assumed about 140 000 km, whilst Notter *et al*. (2010) and EPA (2013) assumed 150 000 km and 193 000 km respectively.

Material extraction was the second largest contributor with between 9.7-11.1 % according to the EPA (2013) report. The production of aluminium for the thermal system and cathode was also presented as a key contributor, especially for energy consumption (EPA 2013) which was validated by Kelly, Dai & Wang (2019). However, the transportation of the batteries seems to have little impact (EPA, 2013). This is in line with findings from other studies as well (Ellingsen *et al*, 2014; Kim *et al* 2016).

Furthermore, Cusenza *et al*. (2019) also claimed that the energy loss due to the batteries own mass in the use-phase was very small compared to the efficiency losses in the battery. This means that the overall battery efficiency was more important than the losses due to the fact that the battery must carry its own weight in the transport as well (Cusenza *et al*., 2019). Majeau-Bettez *et al* (2010) also stated that the manufacturing-phase is a one of the main drivers of GWP for the batteries, but they also found that use-phase electricity consumption accounted for 40 % of the total GWP and fossil depletion impact. For eutrophication potential the use-phase electricity contributed to about 27-45 %, which indicated that the use-phase electricity mix does have a large impact on the total environmental impact of LIBs. This was calculated using the European electricity mix.

Furthermore, Notter *et al.* (2010) found that the production of LIBs was not a significant contributor to the overall lifecycle impact for BEVs, but rather the use-phase is the main source of impact. Especially if the use-phase used electricity produced by high share of fossil sources for charging.

4.1.7. End-of-life

The EOL stage for LIBs is mostly focused on recovery of cobalt, nickel copper and steel (EPA, 2013; Cusenza *et al*. 2019). Both Cusenza *et al*. (2019) and EPA (2013) examined hydrometallurgical and pyrometallurgical treatment for recovering materials for the LIB, but EPA (2013) also considered a direct recycling process. In Cusenza *et al*. (2019) the EOL treatment was modelled by using the recyclability substitution approach which means that the recycled material was assumed to replace virgin materials as described in Allacker *et al*. (2014). However, in the study conducted by EPA (2013) the EOL was modelled with the end of life approach, which means that the recycled materials are not necessarily reused in new LIBs but gives benefits in any application anyways. The recycled material is assumed to displace virgin materials as well. The EOL data in EPA (2013) was gathered from industry recyclers and reflected the current processes at that time in 2013, which may not be representable since the industry is changing rapidly. The environmental credits from recycling in GWP was 8 % in the study by Cusenza *et al*. (2019) and 3- 4 % for the EPA (2013) study.

Within the EOL treatment Cusenza *et al*. (2019) found that the recycling of battery had a quite low impact on most impact categories with less than 11 % before credits was added, except from in the category freshwater ecotoxicity, where it contributed to a majority of the impact at 60 %, see Appendix 1. However, the recycling of materials such as cobalt, nickel and sulphates gave important environmental credits within marine eutrophication, human toxicity and resource depletion. EPA (2013) concurs in the fact that recycling considerably improves the batteries environmental impact.

In addition to recycling, according to Cusenza *et al*. (2019) 80 % of the capacity of the LIBs usually remain after retiring from BEVs. Therefore, they should first be reused in less critical and demanding applications, such as stationary storage, before recycling in order to further reduce the environmental impact of the batteries. However, this application has not been developed in Europe yet and the current course of action is to recycle the retired batteries according to EC Directive 2006/66/EC (Tytgat 2013).

4.2. Electricity mix

The results regarding the electricity consumption during production also differs due to different assumptions of electricity mix. This assumption is important, since the electricity mix gives high impact in many climate impact categories, especially GWP (Cusenza *et al.*, 2019; Kelly, Dai & Wang, 2019; Ellingsen, Hung & Stromman, 2017). Many of the reports highlight the electricity mix in both production and use-phase as crucial since the processes are very electricity intensive. Therefore, many researchers also state that using renewable electricity mix is the most efficient way to reduce impact from LIBs (Cusenza *et al.*, 2019; Kelly, Dai & Wang, 2019; Ellingsen *et al.*, 2017).
The electricity mix used in the studies are presented in table 3.

Table 3. Electricity mix used in reviewed reports.

	Production	Use-phase	EOL
Cusenza *et al.* 2019	Japan	EU	EU
Kelly, Dai & Wang, 2019	EU, US, China		
Kim 2016b	South Korea, US		
Ellingsen 2014	Ecoinvent mix *		
Majeau-Bettez *et al.* 2011	EU	EU	EU
EPA 2013	Canada & US	Canada & US	Canada & US
Notter *et al.*, 2010	EU	EU	EU

* Ecoinvent mix represented coal (46 %), coal (33 %), nuclear (15 %), gas (4.4 %), oil (1.4 %) and hydro (0.15 %) and renewables (0.28%) (Ellingsen *et al.*, 2014). China represents the People´s Republic of China.

Furthermore, in Kelly, Dai & Wang (2019) the authors found that when comparing the average US, EU and Chinese electricity mix, the European average mix resulted in a 9 % saving in GWP compared to the American, whilst the Chinese increased the GWP with 14 %, where the Chinese average represents the People´s Republic of Chinas (hereafter called China) average electricity mix. However, if the electricity mix was fully renewable a reduction of 29 % compared to the American scenario was obtained, however it was not specified how the renewable electricity was produced. Moreover, if the electricity mix was fully coal based, an increase of 31 % was reached. Ellingsen *et al*. (2014) reported even higher savings of 60 % by using all hydroelectric power compared to their base case. Notter *et al*. (2010) found that the use of hydroelectric power instead of the average European decreased the (EI99 H/A) impact with 40.2 %. This shows that the electricity mix assumed in the LCA reports impact the results significantly. Majeau-Bettez, Hawkins & Stromman (2011) found that changing the electricity mix used in production from the average European to Chinese increased the GWP with 10-16 %. Moreover, the Kelly, Dai & Wang (2019) and Ellingsen *et al*. (2014) report only takes into consideration the cradle-to-gate results and adding the impact of the electricity used in the use-phase and EOL treatment, the electricity mix should be even more important to consider.

Climate impact categories

Most of the reviewed studies used different LCIA models and therefore different climate impact categories was evaluated. However, the categories do correspond to similar effects on the environment. The 11 most commonly used impact categories, as described in Matthews, Hendrickson & Matthews (2014) and is presented with the impact from the reviewed studies in table 4, for exact values see appendix 1.

Table 4. Impact categories used in previous LCA studies.

Impact category	Cusenza (2019)	Kelly (2019)	Ellingson (2017)	Kim (2016)	Ellingsen (2014)	Majeau-Bettez (2011)	EPA (2013)	Notter (2010)
Global warming potential	X	X	X	X	X	X	X	X
Ozone depletion potential	X				X	X	X	
Acidification potential	X	X		X	X	X	X	X
Eutrophication potential	X			X	X	X	X	X
Photochemical oxidation potential	X				X	X	X	
Ecological toxicity potential	X	X			X	X	X	
Aquatic toxicity	X				X	X	X	
Human toxicity potential	X			X	X	X	X	X
Abiotic resource depletion	X	X	X	X	X	X	X	X
Land use								
Water use		X						

In all the reviewed studies, GWP was used as one of the main impact categories since one of the main intentions of using LIBs in EVs is to lower the GWP from transport. To measure GWP the unit kg CO2-eq is used in all studies and is also the most commonly used metric in most LCAs (Matthews, 2014). According to the reviewed literature, the energy consumption is a large contributor to the GWP caused by the LIBs. The production and use-phase are both high in energy consumption as stated in section 4.2. *System boundaries*, and the share of renewables used in the electricity mix is crucial for how large the impact is. Therefore, according to many of the reviewed studies the most effective solutions to reduce environmental impact from LIBs is to use renewable electricity mix, but also to use recycled materials and lowering the demand for energy in the production phase (Cusenza *et al*., 2019; Ellingsen, Hung & Stromman, 2017; Kim *et al*., 2016; EPA, 2013).

However, compared to ICEVs, the BEVs have significant reduction potential of the GWP, as well as in human toxicity potential (HTP) due to its high particulate matter emissions during the use-phase of ICEVs. This is because the ICEVs have high impact during the combustion of fuels (Kim *et al*., 2016; Notter *et al*., 2010). Therefore, the GWP impact is a very important category to consider, as well as the human toxicity potential in order to compare to ICEVs.

The acidification potential and eutrophication potential were also used in all reviewed studies. The reason why these two are important to consider is because of the high impact in acidification potential and eutrophication potential from upstream processes due to the high utility usage in production. The nickel production for NMC batteries is a large contributor to the environmental impact. The upstream processes of nickel give rise to high SO_x-emissions (Kelly, Dai & Wang, 2019) and human health impacts (EPA 2013) which is why it is important to include acidification potential in the LCIA. The production of aluminium used in both NMC and LMO batteries is highly energy demanding and electricity intensive (Kelly, Dai & Wang, 2019; EPA 2013). Therefore, the environmental impact depends heavily on the electricity mix used and its share of renewable sources.

The carbon intensity of the average electricity varies significantly between regions. The carbon intensity of electricity consumed at high voltage in Sweden is reported to be 45 g CO_2-eq/kWh, which is well under the average European intensity of 428 g CO_2-eq/kWh (Moro & Lonza, 2017). The average carbon intensity for the US-grid is close to 700 g CO_2-eq/kWh and the Chinese is 821 g CO_2-eq/kWh (Li *et al.,* 2017) as presented in Figure 3. This has a large impact on how much GWP emission the EVs are associated with. In order to include the benefits of using the Swedish electricity grid in production and use phase, further studies need to be conducted from a Swedish point of view but the expectations from the previous LCA studies show that the benefits should be extensive.

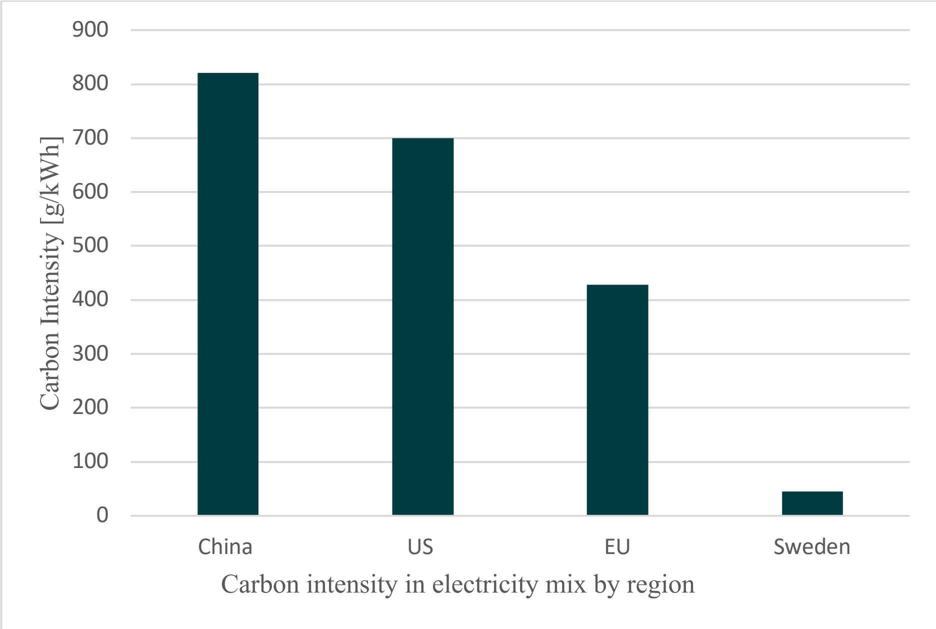

Figure 3. The carbon intensity of the electricity mix in different regions.

A problem that Kelly, Dai & Wang (2019) highlighted was that power grids with high share of hydroelectrical power provided high impact in water consumption even though it resulted in dramatically reduced pollutant emissions. However, it was not specified in the study if the water used was related to local water scarcity, which is an important factor since only water associated with local scarcity is an environmental issue (European Commission, 2013). Therefore, if the production of batteries is located in an area with local water scarcity, the impact category of water use should then be included, however otherwise not. Moreover, except from in Kelly, Dai & Wang (2019) which included water usage, neither water nor land usage was included in the other reviewed reports. The reason why land use is not included is because the uncertainty is high, and the results would most likely be inaccurate (Cusenza *et al*. 2019).

5. Discussion

The literature review showed that the most commonly used perspective in LCA on LIBs are the cradle-to-gate perspective, including upstream processes, cell manufacturing, BMS, thermal system, packaging and cell assembly. The reason why use-phase and EOL is often excluded is because there are little data on these stages, which gives higher uncertainties and calls for more assumptions. Even industry data for the production-phase is hard to come by since the information often is under industry confidentiality. For this reason, much of the reviewed literature is based on either secondary data from research or a mix of primary data from industry and secondary data. Only two studies conducted by Kim *et al*. (2016) and Notter *et al*. (2010) was based primarily on industrial data. Of the two studies, only Notter *et al*. (2010) was a full cradle-to-grave LCA, whilst Kim *et al*. (2016) conducted a cradle-to-gate research.

Furthermore, the data that have been used is mostly for small-scale production since data from large scale industry (>5GWh/ year) production is scarce. This has led researchers to use small scale production data and assumptions regarding the scalability in order to mimic a future large scale-production. The data is therefore somewhat uncertain. However, large scale production is expanding and growing all over the world and many regions have plans to develop production with >5GWh production capacity per year before 2025 (Lutsey, Grant, Wappelhorst & Zhou, 2018). In the light of this, many researchers (Ellingsen *et al*., 2017; Kim *et al*., 2019; Kelly, Dai & Wang, 2019) anticipate a more efficient production in the future when the industry is able to scale up the production and this will presumably lead to savings in both material and energy input. Since a large-scale production of at least 32 GWh is under construction by Northvolt AB in Sweden (Northvolt, 2020), the environmental impact from Swedish production should therefore most likely be smaller than calculated by previous reports. Therefore, to understand how this affects the results, there is a need for further cradle-to-grave LCA studies conducted with accurate industrial data for large scale-production of LIBs.

The production phase has generally been accepted as a key contributor to LIBs life cycle impact since it is the phase where most of the input material is added and production processes are often energy intense. Within the production, the most critical processes seem to be material mining and cell assembly. The mining of raw material causes high impact in abiotic depletion, especially the aluminium extraction and lithium refining is contributing to the abiotic depletion potential (EPA 2013). The mining of raw material is also energy demanding and cause pollutions such as SO_x, whilst the cell assembly is a very energy intensive process due to the need of a dry room. These processes are all high in electricity demand, which increases the importance of the electricity mix used.

The cradle-to-gate assessment is highly dependent on the assumptions regarding how the batteries are used and for what application. In these studies, the energy efficiency and the lifetime of the battery is taken into account. Therefore, the use-phase are naturally an important contributor to the lifecycle environmental impact. However, Cusenza *et al*. (2019) did not find that use-phase was a large contributor which might be due to the fact that the authors assumed higher charge efficiency. In the cradle-to-gate assessment there are two important assumptions, which is battery charge efficiency and the lifetime of the battery. The energy loss due to the fact that the battery also must carry its own weight in the transport is however negligible. Also, the use-phase is highly depending on what electricity mix is used.

Depending on regional electricity mix used in the production phase, use-phase and EOL-phase, the environmental impact varies significantly because the carbon intensity of the electricity mix varies between regions (Kelly, Dai & Wang, 2019). The European electricity mix has an average carbon intensity of 428 g CO_2-eq/kWh (Moro & Lonza, 2017), whilst the US grid has a carbon intensity of close to 700 g CO_2-eq/kWh (EPA 2013). The Swedish carbon intensity, however, is only 45 g CO_2-eq/kWh (Moro & Lonza, 2017). According to many of the studies reviewed, changing the electricity mix to one with a higher fraction of renewables is the most efficient way of reducing the GWP of LIBs. Therefore, it would be interesting to conduct cradle-to-grave LCAs of a LIB production in regions such as Sweden, where conditions are very well suited for a sustainable production.

Moreover, the EOL-phase has also been highly neglected in many previous reports due to the lack of data available. Little available industry data exist regarding EOL treatment. However, previous studies have been conducted (Bobba *et al*., 2018; Gohla-Neudecker *et al*., 2015; Heymans *et al*., 2014; Richa *et al*., 2017) indicating that both the available research and understanding is expected to grow in the near future (Cusenza *et al*., 2019). The production of LIBs is growing rapidly, and the EOL treatment is also developing. Amongst the reviewed reports, three reports: Cusenza *et al*. (2019), EPA (2013) and Notter *et al*. (2010) included the EOL process. The research findings were mainly that the recycling of materials is indeed giving credits to the LIBs and further attempts to improve recycling methods are desired. The authors expressed the need for further studies which include EOL, especially with industrial data in order to receive more certain results.

To compare between the most commonly used impact assessment model, GREET and ReCiPe, the goal of the study is important. If the goal is to compare LIBs in EVs to IVECs, the GREET model may be sufficient since the impact from transport is considered. However, if the goal is to minimise the risk of problem shifting, the use of ReCiPe offers a fuller cover of the categories that are most commonly used.

6. Conclusions

The electrification of the global car fleet has caused a rapid growth of utilisation of LIBs. In order to avoid problem shifting, it is important to understand the life cycle impact on the environment from the batteries. This review has shown that within the system boundaries, previous studies find that both production-phase and use-phase are large contributors to the GWP and other impact from the batteries. The upstream processes and the cell assembly are two main drivers for environmental impact during the production-phase and are thus important to investigate thoroughly in future studies. In the use-phase, the most important factor to consider is what electricity mix is used.

The electricity mix used in the LCA will have a large effect on the results both for use-phase and for production which is why it will be one of the main assumptions to evaluate in future studies. Since the Swedish electricity grid is mainly fossil free, the gain from using the Swedish electricity mix should be extensive. However, depending on region, the water use can also be an important factor to include in the LCIA. Although, the Swedish hydroelectric power is not based on scarce water and should therefore not result in negative environmental impact. But this may not be the case in other regions where water scarcity can be a bigger issue.

Furthermore, most previous studies only include the cradle-to-gate processes and many of the previous studies are based on second-hand data and figures from literature. This causes high uncertainties. The EOL stage is also frequently excluded due to lack of data. It is therefore important to include the entire life cycle process from cradle-to-grave, preferably with real production data from primary sources in order to reduce the uncertainties and find a more accurate LCA result. In addition to using first-hand data, there is also a need for using data reflecting large-scale production to better mirror the economies of scale and more efficient use of input material and energy in such production. The same reasoning applies to the EOL-phase, where more accurate data regarding how the recycling or re-usage for large scale EOL treatment is needed. Doing this will most likely lead to lower environmental impact from LIBs in future studies.

Since it is important to avoid problem shifting, the use of ReCiPe as impact assessment model covers more impact categories and is therefore the recommended model. According to the assessed studies, the impact categories that the LIBs have a large impact within is GWP, human toxicity potential, acidification potential, eutrophication potential and abiotic depletion potential. These categories are therefore most important to discuss in depth in the LCA in order to capture the impact caused by the batteries. However, if the goal is to compare the results with ICEVs, the particulate matter formation is also very important since it will show a clear reduction in this category compared to ICEVs.

With the findings from this literature review, the best practice of future lifecycle assessment is found to be dependent on the goal and scope of the study. If the goal is to compare the batteries in electric vehicles application, the functional unit used should include the charge efficiency and lifetime of the batteries, such as the unit kilometres driven. However, the unit one kWh nominal capacity is recommended to be used as a complement in order to compare the results to other batteries and studies.

Regarding the system boundaries chosen, it is also dependent on the goal and scope of the study. The choice of using cradle-to-gate is good for comparing batteries that are to be used for different applications or the main objective is only to find production hot spots. However, it does not include the performance of the battery in later stages.

The processes that are important to include is the main drivers of impact, which was found to be 1. the mining of raw materials, especially aluminium and copper, 2. the manufacturing of batteries, especially the cell assembly and 3. the use-phase. These processes are all large contributors to the environmental impact and should therefore be assessed in detail in order to give a lifecycle result with more accuracy. In addition, most of the studies only consider the cradle-to-gate emissions and does not include use-phase or the EOL. The use-phase has been called out by many of the reports (Ellingsen, Hung & Stromman, 2017; EPA, 2013; Notter *et al.*, 2010) that it is likely to be the phase where many of the environmental impact savings occur compared to ICEVs. The BEV will likely outperform the ICEVs in many environmental impact categories, such as eutrophication potential, acidification potential and human toxicity during the use-phase. Furthermore, the EOL stage have shown to provide credits to the LIBs by reducing need of raw material. Since mining is one of the main drivers of impact, the EOL stage is very important to further evaluate and research. To include the entire life cycle is important in order to understand system wide trade-offs and credits for LIBs.

Furthermore, there is a need for lifecycle assessment that are more representative of industry production, such as for large scale production. Also, overall there is a need for studies that use industry data and not second-hand data from literature.

The environmental impact method chosen is also depending on the goal and scope for the study. If comparing transportation usage, the GREET model is sufficient, however, the ReCiPe model is more exhaustive and give a more system wide overview.

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Appendix 1

Table of environmental impact results from the reviewed studies

Impact category	Unit	Cuzenza 2019	Kelly 2019	Kim 2016b	Ellingsen 2014	Majeau-Bettez 2011	US EPA 2013 LMO	US EPA 2013 NMC	Notter 2010
Climate change	kg Co2 eq	3,12E+02	4E+01-1,4E+02	1,41E+02	1,72E+02-4,87E+02	2,00E+02	6,34E+01	1,21E+02	5,31E+01
Ozone depletion potential	kg CFC-11 eq	2,33E-05			1,4E-5 - 2,4E .5	2,00E-03	2,40E-06	2,13E-06	
Acidification potential	Kg mol H^+ eq	2,72E+00	4,5E02-9.E-02	9,15E+01			1,82E+01	9,51E+01	
Terrestrial acidification potential			1E02-1,3E02 g NO_x	4,04E+02 g NO_x	1,9E00-3,2E00 kg SO_2 eq	1,6E00 kg SO_2 eq			
Eutrophi-cation potential	kg N eq						-6,29E-03	8,56E-03	
Freshwater eutrophi-cation potential	kg P eq	1,63E-01		9,15E+01	3E-01-4,2E-01	3,40E-01			
Marine eutrophi-cation potential	Kg N eq	5,50E-01			2,4E-01-2,9E-01	2,90E-01			
Terrestrial eutrophi-cation potential	mol N eq	3,20E+00				4,3E-02 kg 1,4 DCB eq			
Photo-chemical oxidation potential	Kg NMVOC eq	9,91E-01			6,8E-01-1,4E00	5,10E-01	3,52E00 kg O_3 eq	7,83E00 kg O_3 eq	
Ecological toxicity potential							8,05E+00	1,01E+00	
Freshwater toxicity potential	kg 1,4 DCB eq	5,37E+03 CTUe			9,6E00-1,16E01	7,1E00			
Marine toxcity potential	kg 1,4 DCB eq				5E-02-5,9E-02	7,8E00			
Human toxicity potential	kg 1,4 DCB eq			8,7E+01 g VOC	5,96E+02-6,81E+02	6,90E+02	1,06E-09 (unit increase in morbidity	2,30E-09 (unit increase in morbidity	

							in human population)	in human population)	
Ionising radiation	kBqU235 eq	2,35E+01							
Non cancer effects	CTUh	1,94E-04					6,12E+02 unitless	1,54E+02 unitless	
Cancer effects	CTUh	3,38E-05							

Continued table of environmental impact results from the reviewed studies

Impact category	Unit	Cuzenza 2019	Kelly 2019	Kim 2016b	Ellingsen 2014	Majeau-Bettez 2011	US EPA 2013 LMO	US EPA 2013 NMC	Notter 2010
Particulate Matter	kg PM2.5 eq	2,19E-01		1,81E+01 kg (PM not specified)	5,8E-01-9,7E-01 kg PM10	4,6E-01 kg PM10 eq	6,09E+02 unitless		
Abiotic resource depletion potential	kg Sb eq	6,64E-03					3,67E-01	8,86E-01	2,28E-03
Energy consumption	MJ	4,83E+03					8,69E+02	1,96E+03	6,42E+00
Metal depletion potential	kg Fe eq				1,54E+02-1,57E+02	1,50E+02			
Fossil depletion potential	kg oil eq				4,95E+01-1,366E+02	4,20E+01			

Units are given in the column “unit” unless otherwise specified.

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