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Climate impacts of stationary NMC lithium-ion batteries for varying electricity mix

A comparative study of climate impact in the present and future from stationary NMC lithium-ion battery storages for varying scenarios of electricity mix, and provision of fast frequency reserve services

Master's thesis in Industrial Ecology

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Department of Electrical Engineering
CHALMERS UNIVERSITY OF TECHNOLOGY
Gothenburg, Sweden 2021

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Abstract

Stationary battery storages may come to increase in installations due to their potential to manage electricity system variability, which in turn is becoming more frequent because of the growth of variable renewable energy production. Therefore, an investigation of the climate impact of stationary Nickel-Manganese-Cobalt lithium-ion battery storages was conducted. The investigation was based on a literature study and calculations on the use phase of a stationary battery. The study concluded that implementing a battery can be advantageous to not doing so, under certain circumstances. A battery manufactured in China, USA or Europe is beneficial if the electricity which is being imported to Sweden can be used during the battery use phase, as it has large variations in its carbon intensity. It is also possible to reduce the climate impact if marginal electricity is applied during the use phase of a battery manufactured in Europe, 35 % extra emissions savings after the break-even point has been reached until the battery is at 80 % capacity could be seen. This is the situation both when the battery is solely an energy storage and when it provides the service of a fast frequency reserve during part of the year. Recycling and changing manufacturing location of the battery does not change the conclusion. However, if a repurposed electric vehicle battery is used, all electricity scenarios except the CHP and Wind scenarios, would benefit from implementing a battery. Assuming a 50/50 allocation between the electric vehicle and the stationary storage.

Keywords: Lithium-ion battery, NMC, LCA, FFR, Repurposed EV battery.

Acknowledgements

I would like to give many thanks to my supervisor Andreas Karlsson for having been a great support and giving me valuable guidance throughout the thesis project. I am very grateful for the help you have provided me to stay motivated and on schedule.
I would also like to thank Ola Carlson and David Steen for assisting me with the thesis project.
Furthermore, a thanks goes out to all who aided me with suggestions, articles and information.
Lastly, I would like to thank Erik Alksnis for all the encouragement you have given and always gives, it is unparalleled.

Amanda Ros, Gothenburg, June 2021

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Introduction

1.1 Background

With the target of zero CO_2 emissions coming decidedly closer, projects of transformation to reduce emissions become increasingly frequent. Within the Swedish industries there are projects such as HYBRIT (HYBRIT, 2021) and Råsjöbolagen (Råsjöbolagen, 2020), and within the transport sector there is the shift towards battery electric vehicles (Volvo Car Corporation, 2021). Though, further electrification in all of society is required for the target to be within reach.

Such an expansion of the electrification would entail increased installations of variable renewable energy production, due to their low cost (Gode et al., 2009). However, with a higher share of variable renewable energy in the electricity mix, there will be, as the name suggests, increased variability. The instability would encompass timescales from seconds to years and would be a major issue for the electricity system if not managed correctly (Göransson and Johnsson, 2018, Svenska kraftnät, 2021a).

The management of such instabilities would need to be adjusted depending on the circumstances, as for a grid with a high share of solar power different solutions may be optimal than for a grid with a high share of wind power (Ajanovic et al., 2020, Göransson and Johnsson, 2018). One potential solution, though, is in the form of stationary battery storages, where the potential is extra interesting due to batteries being able to handle variability within multiple timescales (Ajanovic et al., 2020).

Concerns surrounding the climate impact, as well as other environmental and social aspects of batteries, are currently being raised by for example the European Commission (European Commission, 2020). The importance of all these aspects cannot be denied. It is therefore vital with an understanding concerning what affects these aspects and hence, how to improve. With this motivation, this study was conducted. However, the main focus was set on climate impact to ensure enough depth to the study.

1.2 Aim of the study

The aim of the study is to investigate the climate impact of stationary battery storages in the electricity system. Specifically, the climate impact during usage of a lithium nickel manganese cobalt oxide cathode, NMC, battery is to be investigated and compared based on both present and future scenarios for electricity mix. Also, the climate impact from using lithium-ion NMC battery storages for ancillary services will be studied.

The first specific question which the study intends to answer is what the difference in climate impact would be if a battery storage could reduce high emission power production from the electricity system of 2020 and 2050.

The second specific question is what ancillary services a lithium-ion NMC battery storage could deliver, and the study aims to answer how this could motivate the usage of a battery storage both in present and future electricity systems.

The third specific question which the study aims to answer is what the climate impact of a lithium-ion NMC battery storage is today. Furthermore, the question of what the potential for future battery storages are in terms of their climate impact will be explored.

Last but not least the results regarding the previous questions will be converged, leading to overall conclusions regarding whether it is more sustainable to use a stationary battery than not and during what circumstances.

1.2.1 Limitations

The study will be based on already available life cycle assessments for lithium-ion NMC battery storages. This is to ensure a depth within the use phase of the battery and especially within the possibilities for both the future batteries and the future electricity mix.

Also, the main battery type which will be analysed in the study are lithium-ion batteries, specifically with a lithium nickel manganese cobalt oxide, NMC, cathode. This limitation choice was made due to that NMCs currently, as well as for the closest future, are the most common lithium-ion battery cathode in the world considering electric vehicles (Kelly et al., 2019, Romare and Dahllöf, 2017). There will however, be a smaller inclusion of vanadium redox flow batteries, VRFB, due to their future potential as stationary batteries.

A large limitation of the study is that it is assumed that the battery does not degrade during the period of the year where the battery acts as a fast frequency reserve, in short called FFR. A reason for this choice was that no articles could be found on how supplying FFR degrades a battery. Another reason was that there was a lack of data available on how much a battery unit is used for FFR, since FFR is so newly implemented in Sweden. The battery is however, assumed to degrade due to cycling when used for load balancing.

The carbon intensity of biomass generation is under debate and hence the choice of using only one value for the carbon intensity is a limitation. Depending on what time perspective is applied in a life cycle assessment on biomass generation, the resulting carbon intensity may be 18g/kWh, 50 g/kWh or even as high as 230 g/kWh (Axelsson et al., 2018). In this study a low value was assumed, and this is due to that Electricity Map, which is the source of the data, used a low value.

The study will not include a cost optimization. This is due to it being a common feature in studies covering battery storages in the electricity system. The focus of this study will instead be the climate impact. Although, it should be mentioned that there can be somewhat of a correlation between high cost of electricity and high emissions of CO_2, as production types with high operating costs typically are non-renewable and hence have high emissions.

Furthermore, the study will not include the provision of any electricity variation management strategies which could possibly be done by a battery storage. The potential for electric vehicle batteries to have a dual use, namely vehicle-to-grid, will be outside the limitations of this study. Neither will other storage solutions such as hydrogen or pumped hydro be included.

2
Theory

With the aim of ensuring a basis of mutual understanding between the author and the reader, the following sections contain a theoretical foundation for the study.

2.1 The Swedish electricity system

Currently, the electricity trading in the Nordic countries is mainly done through the liberalised spot market called Nord Pool, where the electricity per hour planned for production the next day is traded (Svenska kraftnät, 2021g). The average value of all electricity produced is called the average electricity, which will be one of the cases used in the results. The order of who gets to produce goes from the producer with the lowest bid, to the producer with the second lowest bid, and so on, up until the demand has been fulfilled (Gode et al., 2009). The price of electricity is then decided based on the bid of the producer with the highest bid that still is required to produce, this production is also what is named marginal electricity which will be one of the cases used in the upcoming results (Gode et al., 2009). This is the case for all four Swedish bidding areas respectively, hence, there can be varying electricity prices between the different bidding areas (Svenska kraftnät, 2021g). The reason for there being different bidding areas is that there are limitations to the transmission capacity between the areas and therefore there are limitations to how much electricity can be produced and utilised (Svenska kraftnät, 2021g).

If there are changes to the planned electricity production or consumption, there is the Nord Pool intra-day trading market (Svenska kraftnät, 2021g). Here, increased or decreased production of electricity can be traded on a short-term basis. There is also the possibility of importing or exporting electricity to handle the changes (de Maré, 2021). Currently, there are both collaborations and electricity exchanges with Denmark, Finland, Norway, Germany, Poland, Estonia, Latvia and Lithuania (Svenska kraftnät, 2021f). It is what is on the margin in terms of carbon intensity of the imported electricity which is the basis for the imported electricity case which will be used further on in the results of the report.

However, there are circumstances where the electricity demand overshoots the planned production and importation is not enough (Svenska kraftnät, 2021g). Such circumstances could be when the weather is cold during winter days, and especially if there at the same time is a lack of wind (de Maré, 2021). If this is the case, Svenska kraftnät has the responsibility to activate power reserves in the form of backup power plants, increased production or reduced consumption which have been purchased on a long-term basis instead of the ordinary day-ahead market (Svenska kraftnät, 2021h).

The balancing of the Swedish electricity system is largely done by hydro power, as can be seen in figure 2.1 which is based on data provided from Electricity Map. However, as the variable renewable power production grows, so will the challenges of balancing consumption and production (Svenska kraftnät, 2019). This is due to the weather dependency of the variable renewable energy sources resulting in somewhat unpredictable power production (Svenska kraftnät, 2021d).

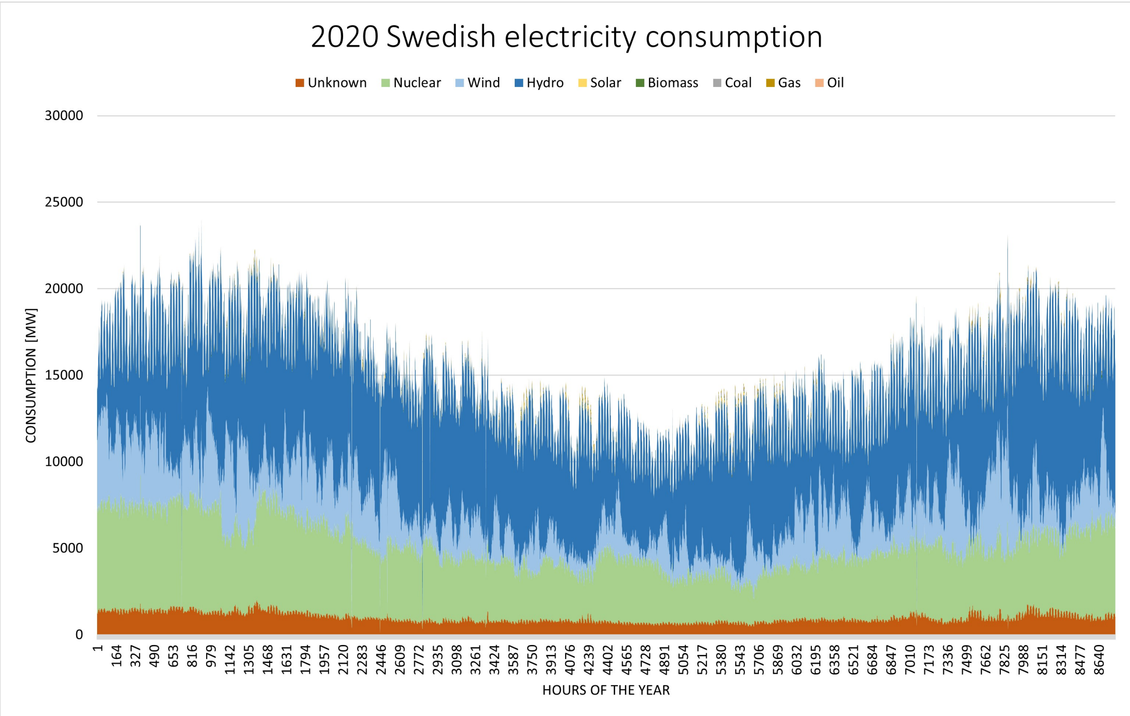

Figure 2.1: 2020 hourly consumption of electricity in Sweden per generation type.

The further balancing required in Sweden in the future may be handled in a slightly different way compared to today. With the EU directive 2019/944 on common rules for the internal market for electricity being on its way to become implemented into Swedish law, Energimarknadsinspektionen (2020) discusses the formation of flexibility markets. These are to be based on free competition around different flexibility services such as demand flexibility, production flexibility, and energy storage (Energimarknadsinspektionen, 2020). Specifically, the EU directive 2019/944 states that if avoidable, energy storages are not to be owned, developed, managed or operated by the electricity system authorities, instead, there are to be no obstacles for any supplier of flexibility services to trade on the market (Energimarknadsinspektionen, 2020). This is also supported by Svenska kraftnät (2021b), as there is an ambition to increase the number of flexibility suppliers as well as simplify for new technologies to set foot in the flexibility markets.

2.2 Shifting strategies in variation management

In order to handle the grid variability from increased wind and solar implementations, management strategies are required. Within electricity variation management there are different management strategies which based on the work by Göransson and Johnsson (2018) can be categorised into absorbing-, complementing- and shifting strategies. Out of these, it is the shifting strategy which is the most relevant when discussing batteries, as a battery is limited in the sense of how long the energy can be stored and is expensive in relation to absorbing- and complementing strategies for larger scales (Göransson and Johnsson, 2018).

Though, it should be mentioned that even though shifting is what batteries are believed to be optimal for, a future market for second-life electric vehicle batteries could change that. If the cost for the second-life battery is low enough, there may very well be a different situation for how to optimise the variation management strategies.

2.3 Frequency reserves in ancillary services

Ancillary services could be described as the uttermost short term variation management as it handles electricity system instabilities in the timescale of seconds to minutes. Different countries handle their ancillary services in a variety of ways, but for Sweden there are as of current five separate services. These five are fast frequency reserve, frequency containment reserve - normal, frequency containment reserve - disturbance, automatic frequency restoration reserve, and manual frequency restoration reserve (Svenska kraftnät, 2021e).

There are several of these ancillary services which a stationary battery could provide (Hollinger et al., 2018, Jankowiak et al., 2019). However, the fast frequency reserve, FFR, is the one which this study will focus on. One reason behind this decision is that FFR is believed to be required to grow in the future due to a decrease in rotational energy (Roupe et al., 2020). This is in turn due to increased installations of variable renewable energy production in combination with the discontinuation of nuclear power production as that results in fewer synchronous generators (Svenska kraftnät, 2019). Another reason behind the decision is that batteries are superior to conventional power production in that they are faster and more accurate in their reaction when supplying response power (Hollinger et al., 2018). Implementing a battery which provides FFR could hence be a route to the electricity system becoming 100 % renewable which in turn means a reduction of the climate impact.

FFR is a service which handles the initially fast and transient changes in frequency which can occur during moments of low rotational energy in the electricity system (Svenska kraftnät, 2021i). In 2020, FFR was introduced in Sweden and the months in between which the FFR was active was June and September (Svenska kraftnät, 2021i). As mentioned, it is believed that the FFR will be required to expand in the future, however, the 2021 FFR activity is believed to become relatively similar to the

2020 FFR activity (Roupe et al., 2020). Though, it should be noted that ancillary services are not to be included in the previously described flexibility markets as they are managed by Svenska kraftnät (Energimarknadsinspektionen, 2020). This means that any supplier of ancillary services would need to do so through Svenska kraftnät.

2.4 Life cycle assessments

A life cycle assessment, LCA, is an extensive method for accounting and assessing the environmental impacts of products and services. In an LCA, all stages of the product life cycle, from raw material extraction, through production, transportation and use, to waste management, are to be quantified and evaluated for natural resource use and pollutant emissions. (Baumann and Tillman, 2004, Notter et al., 2010, U.S.EPA, 2013)

An international standard for LCAs was outlined in the International Standards Organization, ISO, 14040 series in 1997 (Baumann and Tillman, 2004, U.S.EPA, 2013). This standard has states that an LCA has four different components that are to be included in the analysis (U.S.EPA, 2013). These are the goal and scope definition, the life cycle inventory analysis, the life cycle impact assessment, and the interpretation of results (Baumann and Tillman, 2004). See figure 2.2 for an overview of the LCA components.

The goal and scope definition includes the purpose and aim of the study, a context and intended audience (U.S.EPA, 2013). It also describes the product system and decides on a functional unit related to the function of the product system described (Baumann and Tillman, 2004, Notter et al., 2010). The functional unit is particularly important for the purpose of comparing (Notter et al., 2010). It is also important to decide the system boundaries, such as spatial- and temporal boundaries (U.S.EPA, 2013). Also, what impact categories are to be included in the life cycle impact assessment is decided upon, for example environmental impact categories such as global warming potential, GWP, or acidification potential, AP (Baumann and Tillman, 2004). In this study, the focus will be on global warming potential, on request from Bengt Dahlgren AB as that allows for enough depth to be reached within the time constraints of the study. Last but not least, the data requirements are decided based on the desired level of detail (Baumann and Tillman, 2004). For example, this includes deciding whether the data collected for different purposes is to be from primary or secondary sources (U.S.EPA, 2013). If secondary sources are used there also need to be decisions regarding the usage of for example life cycle inventory databases such as Ecoinvent (Ellingsen et al., 2014, Kallitsis et al., 2020, Majeau-Bettez et al., 2011, Peters and Weil, 2018) or models such as GREET (Kelly et al., 2019). Also, whether the obtained data is to be processed in a software, for example GaBi, needs to be decided (Ciez and Whitacre, 2019, Dai, Kelly, et al., 2019, Gaines et al., 2011, Kallitsis et al., 2020, U.S.EPA, 2013).

In the life cycle inventory analysis, the environmental loads of all the product activities throughout its life cycle are quantified (U.S.EPA, 2013). To begin with, a

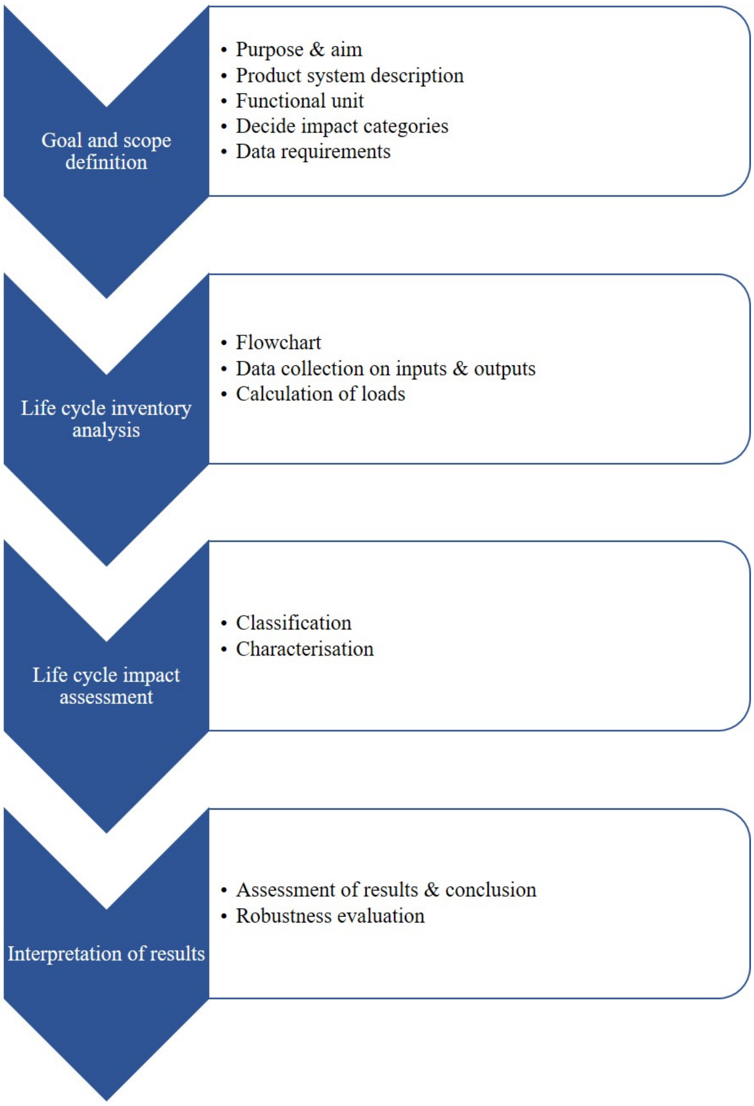

Figure 2.2: An overview of the main components in a life cycle assessment.

flowchart is constructed where the relevant flows are displayed between the activities. Then, data is collected on the inputs and outputs of all the activities. Lastly, the natural resource use and pollutant emissions are calculated per functional unit. (Baumann and Tillman, 2004)

The life cycle impact assessment is an evaluation of the results from the life cycle inventory analysis in the form of potential impacts on the environment (Baumann and Tillman, 2004). It is based on aggregation in several steps, where, according to the ISO 14040 standard, classification and characterisation are mandatory and weighting is optional (Baumann and Tillman, 2004, U.S.EPA, 2013). The classification takes the life cycle inventory analysis results and assigns them all to their respective type of environmental impact category (Baumann and Tillman, 2004). In the characterisation, the magnitude of the potential environmental impact in each category is calculated (U.S.EPA, 2013). This gives the characterisation results which can then be further aggregated, if desired, through weighting of the different impacts into for example a one-dimensional index (Baumann and Tillman, 2004, U.S.EPA, 2013).

The last LCA component according to the ISO 14040 standard is the interpretation of results, which is where the results are assessed and conclusions drawn in connection to the defined goal and scope (Baumann and Tillman, 2004). There should also be an inclusion of recommendations for improvements and further research (U.S.EPA, 2013). The assessment of the results should include an identification of issues in the study which have significantly affected the results, such as specific methodological decisions. It should also include a robustness evaluation, for example through completeness- or consistency checks, uncertainty-, sensitivity- or variation analysis, to verify the results (Baumann and Tillman, 2004).

3
Methods

In order to obtain relevant information on the subject, a literature study was performed. This information was then used both on its own and as a basis for calculations, to acquire results and hence answers to the stated questions of interest.

3.1 Literature study

The literature for this study was obtained through a number of sources. Firstly, the main source of literature were articles published in journals obtainable through the Chalmers Library. Secondly, sources were retrieved from agencies such as Svenska Kraftnät and the Swedish Energy Agency. The third source were article recommendations from scientists at Chalmers University of Technology, who were contacted in order to receive their experienced opinion on a specific question or in general.

For the emissions of CO_2-eq. per hour it was decided early on that the source of that data, for the year 2020, would be Electricity Map. This decision was taken based on that Electricity Map was able to deliver data on request for the emissions directly, as well as the electricity consumption per hour for each production type. This was not the case with similar data from for example Svenska kraftnät (2021c), where only the production per hour and production type were retrievable.

In order for the electricity mix of the future scenarios to cover many different possibilities, several different sources were used. This was done to broaden the results and hence make them more applicable to reality.

The retrospective global warming potentials of a NMC lithium-ion battery were retrieved by studying several different life cycle assessments. The choice of studying several sources was done to decrease potential bias to a specific LCA method such as bottom-up or top-down.

3.2 Calculations

With a basis in the literature study described above, the following calculations could be formulated. The order in which the calculations are described is from the calculations which were performed first to the calculations performed last.

3.2.1 CO_2-eq. emissions savings per kWh for a lithium-ion battery using varying electricity mix

Below, the calculations for the CO_2-eq. emissions savings from the entire use phase of the battery per kWh. The results from these calculations are to be compared to the emissions released during the cradle-to-gate of the battery life cycle.

3.2.1.1 Carbon intensity calculations

To begin with, the carbon intensity in g of CO_2-eq. per kWh had to be decided. For the year 2020, no calculations had to be performed since the data retrieved on request from Electricity Map were directly applicable.

For the year 2050 however, there had to be some calculations done. All sources chosen solely presented yearly generation from the various electricity generation types, $c_{i,tot}$, not the hourly generation. In order to transform the data from yearly to hourly the following calculations were performed. Firstly, a constant, k, for each generation type, i, had to be developed based on the total yearly consumption in 2020, $c_{i,tot}^{2020}$, and 2050, $c_{i,tot}^{2050}$.

$$k_i = \frac{c_{i,tot}^{2050}}{c_{i,tot}^{2020}} \tag{3.1}$$

This constant was then multiplied with the 2020 hourly share of emissions for that specific production type. The hourly share was calculated by multiplying the 2020 hourly carbon intensity, E^{2020}(h), with the 2020 hourly share of consumption, $\sigma^{2020}(h)$ from the specific production type, i. Both E^{2020}(h) and $\sigma_i^{2020}(h)$ were retrieved on request from Electricity Map.

$$E_i^{2050}(h) = k_i * E^{2020}(h) * \sigma_i^{2020}(h) \tag{3.2}$$

The generation type specific carbon intensity, E_i^{2050}(h), were then summed over all generation types for each hour. Hence, giving the desired data for further calculations.

$$E^{2050}(h) = \sum_i E_i^{2050}(h) \tag{3.3}$$

3.2.1.2 CO_2-eq. emissions savings for one year

Once the carbon intensity in g of CO_2-eq. per kWh for a specific electricity mix had been established, represented as E(h), they comprised the input data for the calculations of the emissions savings together with the battery reference information. This information was acquired, through Bengt Dahlgren, from a reference battery in the so called AWL building at Chalmers University of Technology. This battery has a maximum storage capacity, S_{max}, of 200 kWh, a minimum storage level, S_{min} of 20 kWh, a maximum charging power, CP_{max}, of 70 kW, and a maximum discharging power, DP_{max}, of 115 kW. These numbers were assumed to include any charge or discharge losses. Also, any losses due to self discharge were assumed negligible after a trial had been made using a self discharge of 0,1 % per day based on the work by the Danish Energy Agency (2020).

Prior to being able to compute any calculations, a carbon intensity limit had to be implemented in order to know when to charge or discharge the battery. This limit was guessed based on what carbon intensity was the most frequent for the specific electricity scenario. The limit would then be iterated at the end when all calculations had been computed in order to find the limit which provided the largest emissions savings per year.

With a limit guessed, the first calculations could be computed, where using several IF functions it could be determined whether the battery was charging or discharging. The first function states that if the carbon intensity during the current hour, E(h), are strictly less than the carbon intensity limit, EL, then the battery can charge, 1, otherwise it can discharge, 0.

$$\textit{IF } E(h) < EL \textit{ then } X(h)=1\textit{, otherwise } X(h)=0 \tag{3.4}$$

For the following functions the boundary limits of the battery were used, as it is not only the carbon intensity limit which restrict the usage of the battery. The second function stated that if the battery is allowed to charge and the sum of the battery charge level at the end of the previous hour, C(h-1), and the maximum charging power, CP_{max}, are strictly less than the maximum storage capacity, S_{max}, then the battery can charge with maximum charging power, otherwise it is zero. This allows for determination of whether the battery can be charged with maximum charging power or not.

$$\begin{gathered}\textit{IF } X(h)=1 \textit{ AND } C(h\text{-}1)+CP_{max}<S_{max} \\ \textit{then } Y_1(h)=CP_{max}\textit{, otherwise } Y_1(h)=0\end{gathered} \tag{3.5}$$

The third function also belongs to when the battery is allowed to be charged according to the first function, however, it handles when the battery cannot be charged with maximum power but instead the battery is charged to maximum storage level.

$$\begin{gathered}\textit{IF } X(h)=1 \textit{ AND } Y_1(h)=0 \textit{ AND } C(h\text{-}1)<S_{max} \\ \textit{then } Y_2(h)=S_{max}\text{-}C(h\text{-}1)\textit{, otherwise } Y_2(h)=0\end{gathered} \tag{3.6}$$

The following two functions cover the discharge of the battery, working in a similar manner to the previous two functions.

$$\begin{gathered}\textit{IF } X(h)=0 \textit{ AND } C(h\text{-}1)\text{-}DP_{max}>S_{min} \\ \textit{then } Z_1(h)=DP_{max}\textit{, otherwise } Z_1(h)=0\end{gathered} \tag{3.7}$$

$$\begin{gathered}\textit{IF } X(h)=0 \textit{ AND } Z_1(h)=0 \textit{ AND } C(h\text{-}1)>S_{min} \\ \textit{then } Z_2(h)=C(h\text{-}1)\text{-}S_{min}\textit{, otherwise } Z_2(h)=0\end{gathered} \tag{3.8}$$

Assuming the initial condition for the battery at the beginning of the year is it being fully charged, the charge level for each hour, C(h), was calculated by adding the total charge and subtracting the total discharge from the previous charge level each specific hour.

$$C(h) = C(h-1) + Y_1(h) + Y_2(h) - Z_1(h) - Z_2(h) \tag{3.9}$$

In order to calculate the emissions for the battery usage each hour, the present charge level was subtracted from the previous charge level, this was then multiplied with the present carbon intensity from the specific electricity mix.

$$EB(h) = (C(h-1) - C(h)) * E(h) \tag{3.10}$$

The emissions for the battery per hour were then summed for the entire year, giving the total savings of CO_2-Eq. emissions.

$$E_{tot} = \sum EB(h) \tag{3.11}$$

3.2.1.3 Battery capacity loss and CO_2-eq. emissions savings over several years

To be able to calculate the emissions savings during the entire use phase of the battery, the calculations described in the previous section were repeated. These repetitions were executed until the year the capacity of the battery went below 80%, as this is a limit recognized by several articles for when the usage of a lithium-ion battery is normally discontinued (Zhao, 2017, Kamath et al., 2020, Pusceddu et al., 2021, Richa, Babbitt, and Gaustad, 2017, Romare and Dahllöf, 2017).

Loss in battery capacity is due to time- and cycling degradation (Wang et al., 2016). However, due to practicality considering the calculations, a function depending solely on the number of cycles was developed. This function was for the first one thousand cycles primarily based on the work by Elliott et al. (2020). For the following thousands of cycles the function was based on the work by Xu et al. (2018). The reason for using two different sources for the function was due to the work of Elliott et al. (2020) being very detailed in comparison to the work of Xu et al. (2018). However, the work by Elliott et al. (2020) only covered the first one thousand cycles, while the work by Xu et al. (2018) continued until the battery capacity was entirely nonexistent.

The battery degradation function which resulted from the compilation of the different sources was then implemented on the number of cycles the battery had done the previous year, in order to calculate the new degraded battery capacity. This capacity then replaced the previous capacity in the calculations for that year. Hence, for the first year calculations the battery capacity is at maximum, S_{max}, and for the following years the capacity is degraded with regards to the total number of cycles completed during the life of the battery. The function which the work resulted in can be seen below and illustratively in figure 3.1.

$$S = -7*10^{-11}*\left(\sum N\right)^3 + 2*10^{-6}*\left(\sum N\right)^2 - 2,28*10^{-2}*\left(\sum N\right) + 197,57 \tag{3.12}$$

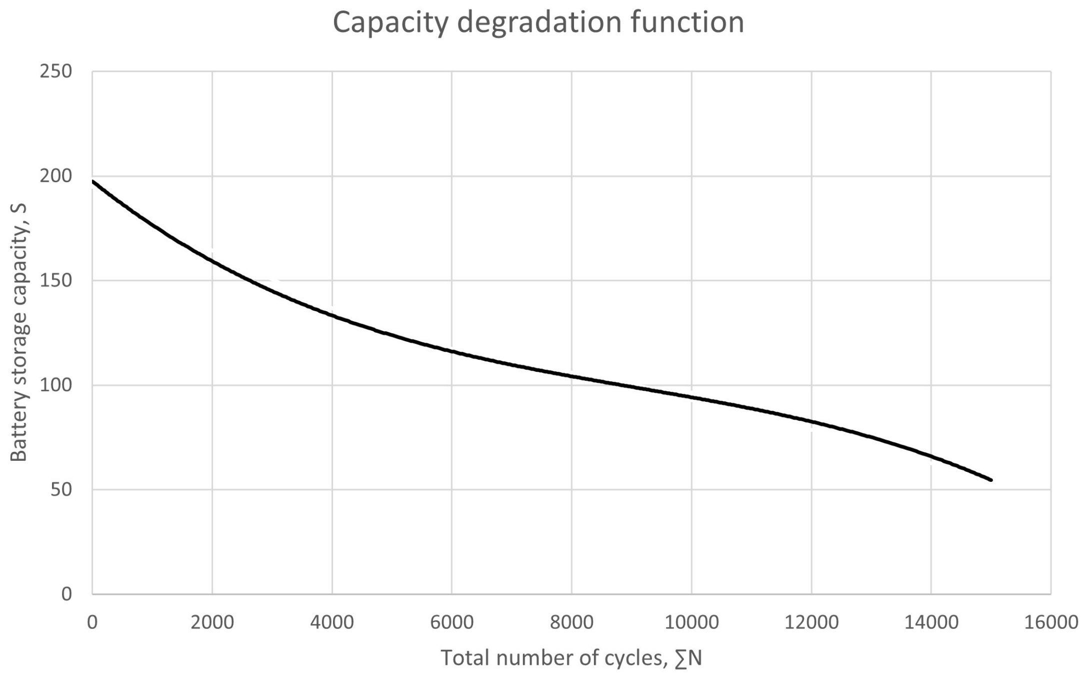

Figure 3.1: Lithium-ion battery storage capacity degradation function.

The number of cycles which the battery performed each year was calculated as the sum of all charging divided by the battery capacity for the beginning of that year (Taljegard et al., 2019).

$$N = \frac{\sum(Y_1(h) + Y_2(h))}{S} \tag{3.13}$$

The emissions savings when using the battery in the manner described can therefore be calculated for several years of usage and the sum of emissions saved for all years of battery lifetime can be divided by the maximum battery capacity in order to obtain the total emissions savings, TEE, which is the use phase emissions savings that could potentially nullify the cradle to gate emissions of the battery.

$$TEE = \frac{\sum E_{tot}}{S_{max}} \tag{3.14}$$

3.2.2 CO_2-eq. emissions savings per kWh for a lithium-ion battery using varying electricity mix and providing fast frequency reserve

The Swedish fast frequency reserve, FFR, regulations require the battery to be recovered and ready for a new cycle in 15 minutes (ENTSO-E, 2021). This means that the emissions saved during battery discharge will be equal to the emissions from charging the battery, assuming no losses. There is hence no reduction of climate impact to be made at this position in the system, even though there could be other benefits such as an increase in renewable power production or a reduction in deterioration of hydro power units (Nohrstedt, 2021).

As described in section 2.3, however, FFR was in 2020 only required during June-September (Svenska kraftnät, 2021i). Due to this, the battery can be assumed to be able to function as an energy storage, in the same way as described previously, during the other parts of the year. There is hence a potential for reduced climate impact during these parts of the year.

The calculations for the emissions savings were performed equivalently to those described previously. The only difference being that during the part of the year where FFR is active, the battery is untouched by the energy storage calculations mechanics.

4

Results and discussion

4.1 Electricity systems

In the following sections the basis for the calculations will firstly be described, in the form of what electricity scenarios were included in 2020 and 2050. Then the resulting CO_2-eq. emissions savings per kWh during the use phase of the battery will be presented both without and with provision of fast frequency reserve, FFR, services.

4.1.1 Present Swedish electricity system

As described in section 3.1, the hourly data for the 2020 carbon intensity as well as the consumption share per production type were retrieved on request from Electricity Map. For 2020 three sets of hourly data for the carbon intensity were chosen. Firstly, the carbon intensity from average Swedish electricity consumption, secondly, the carbon intensity from imported electricity to Sweden and thirdly, the carbon intensity of marginal Swedish electricity consumption.

The primary reasoning for choosing all three of these different data sets was that depending on how the issue is viewed, a certain choice of electricity data may be more suitable. If the battery is viewed as a shifting strategy, explained in section 2.2, using average electricity data may be more appropriate as there is no impact on what type of electricity production that is allowed to produce. On the other hand, if the battery is viewed from the perspective of an electricity system manager with possibilities to use the battery as a way of reducing the overall system CO_2-eq. emissions by substituting the types of production with the highest emissions for the battery, marginal electricity may be the most suitable choice. Furthermore, such a system manager may wish to reduce the share of imported electricity with high CO_2-eq. emissions by using a battery in its stead, this would mean imported electricity would be the appropriate choice.

Further reasoning for choosing the three different data sets was that they were very different from each other, see figure 4.1. The average carbon intensity data were somewhat lower than the imported carbon intensity data. However, the marginal carbon intensity data was substantially higher than both of them. On the other hand, the carbon intensity from the imported electricity varied much more than the carbon intensity from the average Swedish electricity and somewhat more than the carbon intensity from the marginal Swedish electricity. Due to these aspects there was hence, a desire to compare the differences in results between the three data sets.

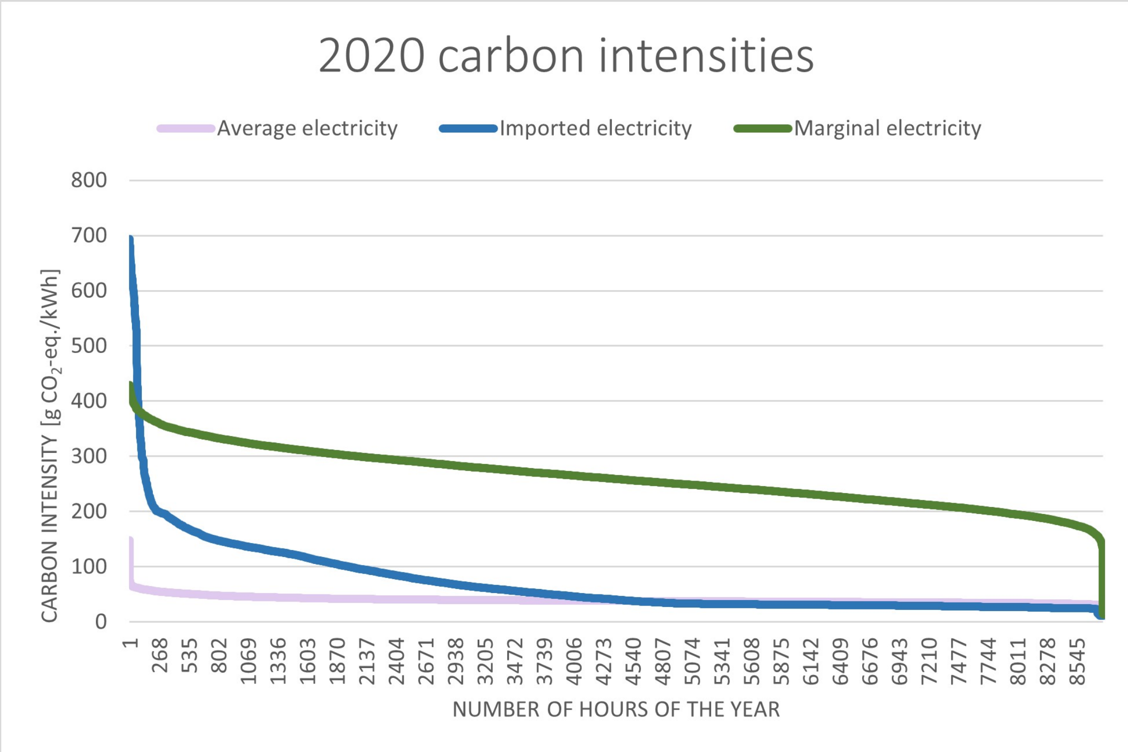

Figure 4.1: 2020 hourly carbon intensities from average, imported and marginal electricity.

4.1.2 Prospective electricity system scenarios

The choices of 2050 scenarios were made after some literature research had been executed. The main aim of the scenarios was that they present a good width of possible futures. The scenarios were also chosen based on an intention to retrieve them from a reasonable and trustworthy source. Hence, two different sources were chosen from the Swedish Energy Agency and from these two sources, five different scenarios were chosen.

From one of the sources from the Swedish Energy Agency (2021), the first two scenarios were chosen, namely Electrification and EU-Reference. These are both scenarios which predict a substantially higher production of electricity in 2050 compared to 2020, but also compared to the other 2050 scenarios. This is especially true for the Electrification scenario where the electricity production is estimated to increase to 282 TWh (Swedish Energy Agency, 2021).

In the Electrification scenario, it is assumed that investing in new nuclear power becomes profitable due to an increase in electricity price. This high price but also the increased demand are the reasons for the scenario having the largest electricity production. The high demand signify not only increased investments in industry located in Sweden but more importantly, a very large interchange of fossil fuels for electricity in all aspects of society. (Swedish Energy Agency, 2021)

On the other hand, the EU-Reference scenario is a more conservative scenario in terms of the assumed electrification of society. The basis for the EU-Reference scenario is business as usual, meaning no new policies are implemented. Though current policies, surrounding for example the EU emissions trading scheme, are continued as planned. This scenario is included in the study as it, to some extent, represents a worst-case scenario. (Swedish Energy Agency, 2021)

From the second source from the Swedish Energy Agency (2019), three scenarios were chosen, namely Wind, Solar and CHP. These are all scenarios with 100 % renewable power. This is a contrast to the first two scenarios were nuclear power and the so called unknown generation remains in usage, but it is especially contrasting to the EU-Reference scenario where also coal and natural gas power remains in production.

The total generation in the 100 % renewable scenarios is also smaller compared to the other two scenarios, this is due to that the Swedish Energy Agency (2019) assumes an increased efficiency in the usage of electricity. However, they also stress the uncertainty of how much the electricity usage will increase (Swedish Energy Agency, 2019).

The total generation of a specific production type for the Swedish electricity production in 2020 as well as the electricity production in all of the 2050 scenarios can be seen in table 4.1. The data for the 2020 generation was retrieved from the Swedish Energy Agency (2021).

Generation Type	Total Generation [TWh]					
	2020	Electrification	EU-Reference	Solar	CHP	Wind
Hydro	67	68	68	70	70	70
Wind	26	126	94	70	70	90
Nuclear	58	60	28	-	-	-
Biomass	13,1	13,8	13,7	15	35	15
Solar	0,8	11,0	9,7	25	5	5
Unknown	2,3	3,0	2,5	-	-	-
Coal	1,0	-	0,8	-	-	-
Natural Gas	0,9	-	0,2	-	-	-
Oil	0,3	-	-	-	-	-

Table 4.1: Total electricity generation per generation type in 2020 and in the five different 2050 scenarios.

As described in section 3.2.1, the total generation per generation type for each 2050 scenario was divided by the total 2020 generation per generation type, giving a generation type specific constant. This was then multiplied with the 2020 average hourly carbon intensity and share of consumption per generation type, which was then summed for all generation types. The resulting average hourly carbon intensities for the different 2050 scenarios can be seen in figure 4.2.

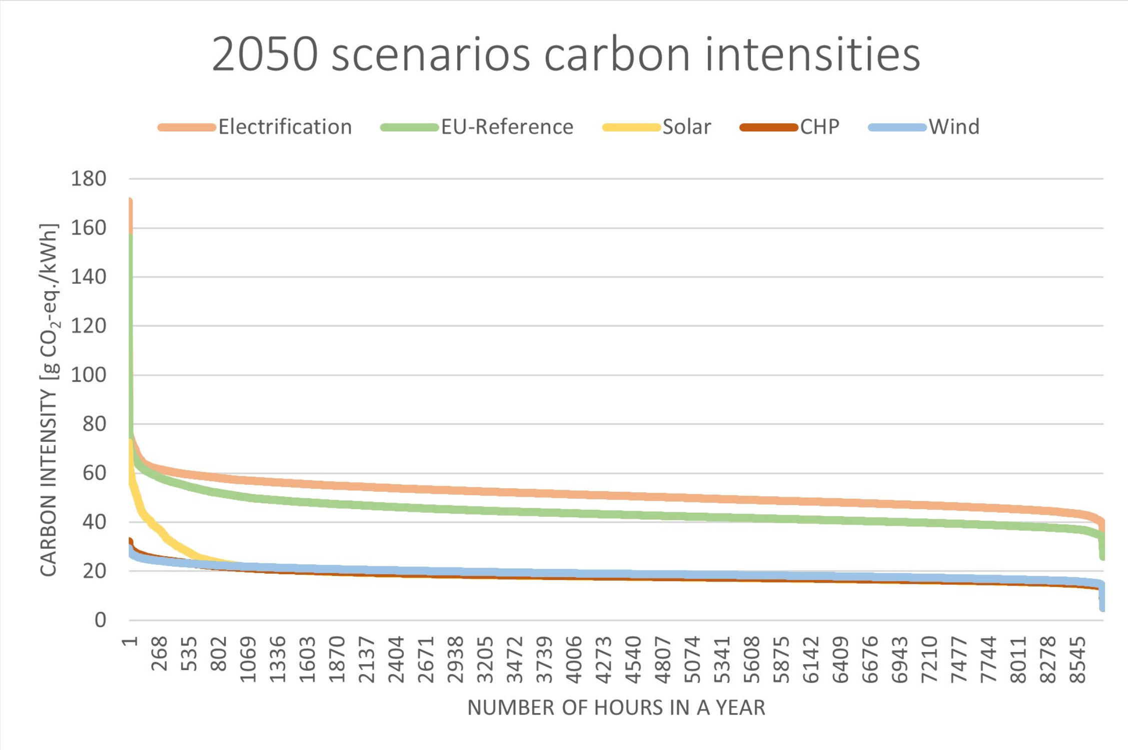

Figure 4.2: Average hourly carbon intensities from the 2050 electricity scenarios: Electrification, EU-Reference, Solar, CHP and Wind.

All three of the 100 % renewable scenarios have lower carbon intensities compared to the other two scenarios. This is positive from the perspective of the electricity system if viewed in solitude. However, if viewed in a more holistic way, another aspect may become relevant. Since, even though the largest carbon intensity can be seen from the Electrification scenario, the emissions from society overall may be smaller compared to the other scenarios due to the assumed interchange of fossil fuels for electricity. Hence, the 100 % renewable scenarios, although resulting in lower carbon intensity, may actually be coupled with less electrification in society overall. This aspect is further strengthened by the low total generation in the 100 % renewable scenarios compared to the Electrification scenario, as this would imply less of an electricity demand.

4.1.3 CO_2-eq. emissions savings per kWh for a lithium-ion battery using varying electricity system scenarios

On the basis of the results described in the previous sections 4.1.1 and 4.1.2, the emissions savings could be calculated as described in section 3.2.1, namely what size of battery cradle to gate GWP could be nullified during its use phase. The results of these calculations are presented in figure 4.3. It should be noted that these results do not take into consideration whether the generated electricity match the demand for each hour, as that would imply a different way of using the battery storage, which has not been done in this study.

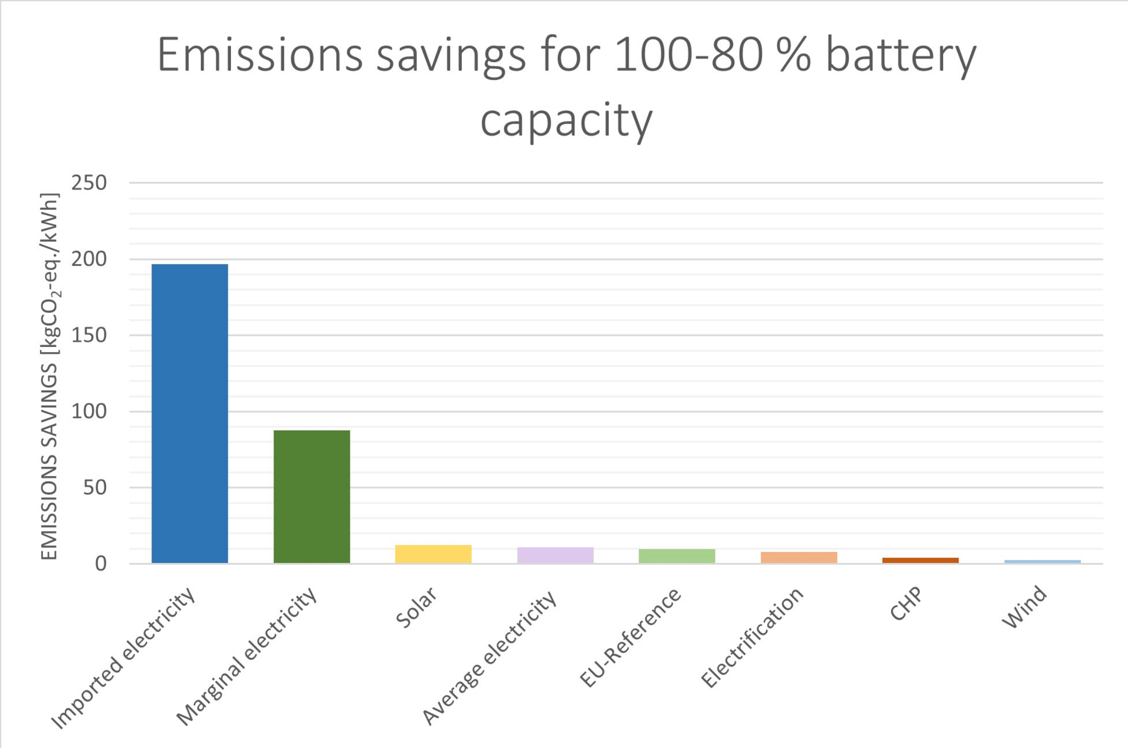

Figure 4.3: Emissions savings from a lithium-ion battery for the 2020 scenarios: Imported-, Marginal- and Average electricity and the 2050 electricity scenarios: Solar, EU-Reference, Electrification, CHP and Wind.

In figure 4.3, striking differences can be seen within the 2020 electricity data sets. The emissions savings from using imported electricity are more than double the emissions savings from using marginal electricity. In turn, the marginal electricity emissions savings are more than eight times the size of the average electricity emissions savings.

The reason for these differences become clear when comparing the carbon intensities from figure 4.1, as the imported electricity has a much larger spread in its carbon intensities compared to the other two data sets which have flatter curves. The imported electricity curve also has hours of extremely high carbon intensities most likely due to importation of coal and natural gas power, which have a carbon intensities of at least 740 g/kWh and 410 g/kWh respectively (Axelsson et al., 2018). Furthermore, roughly half the number of yearly hours have as low carbon intensities as the average electricity data set. There is hence, a possibility to save large amounts of CO_2-eq. emissions as there are both many hours with high carbon intensities and many hours with low carbon intensities.

For the 2050 scenarios, however, the differences in figure 4.3 are much smaller that what could be seen for the 2020 scenarios. The largest emissions savings are displayed by the Solar scenario, and just as for the 2020 electricity data sets this can be argued to be due to it having a rounder carbon intensity curve than the other 2050 scenarios, see figure 4.2. This would be due to the carbon intensity of solar power being roughly 41 g/kWh in comparison to wind power and biomass generation which have carbon intensities of roughly 12 g/kWh and 18 g/kWh respectively (Axelsson et al., 2018).

It should be noted though, that the carbon intensity of biomass generation is under debate. Depending on what time perspective is applied in a life cycle assessment on biomass generation, different results are achieved. The carbon intensity may be 50 g/kWh or even as high as 230 g/kWh (Axelsson et al., 2018). This could of course largely affect the results of this study and has hence been noted as a limitation.

The Electrification and EU-Reference scenarios have slightly lower emissions savings than the Solar scenario, even though they display a larger spectrum of carbon intensities in figure 4.2. The reason for why this is the case is because the highest carbon intensities are only occurring very few hours of the year. Thus, nearly all hours of the year, the carbon intensity curve is rather flat. The EU-Reference carbon intensity curve, in figure 4.2, does however, display a slightly rounder shape compared to the Electrification scenario. Hence, it is why its emissions savings are slightly larger.

The Solar scenario emissions savings are more than three times larger than the CHP emissions savings and more than five times the size of the Wind scenario emissions savings. As explained before, this is due to the broadness carbon intensities of the Solar scenario. The reason for why the CHP emissions savings are larger than the Wind emissions savings are due to that the CHP carbon intensities are slightly higher for the highest values compared to the Wind carbon intensities, see figure 4.2. This is due to the difference in carbon intensity between biomass and wind power production, as was discussed previously.

If, instead of looking at the data in figure 4.1 sorted from largest to smallest, looking at the carbon intensity per hour from the first hour of the year to the last, it becomes clear that the hours with high carbon intensity are spread out over the whole year, see figures 4.4 and 4.5 where the 2020 and 2050 respective carbon intensities per hour from the first hour of the year to the last can be seen. This holds true for all scenarios studied, although they all have lower top carbon intensities during the summer season.

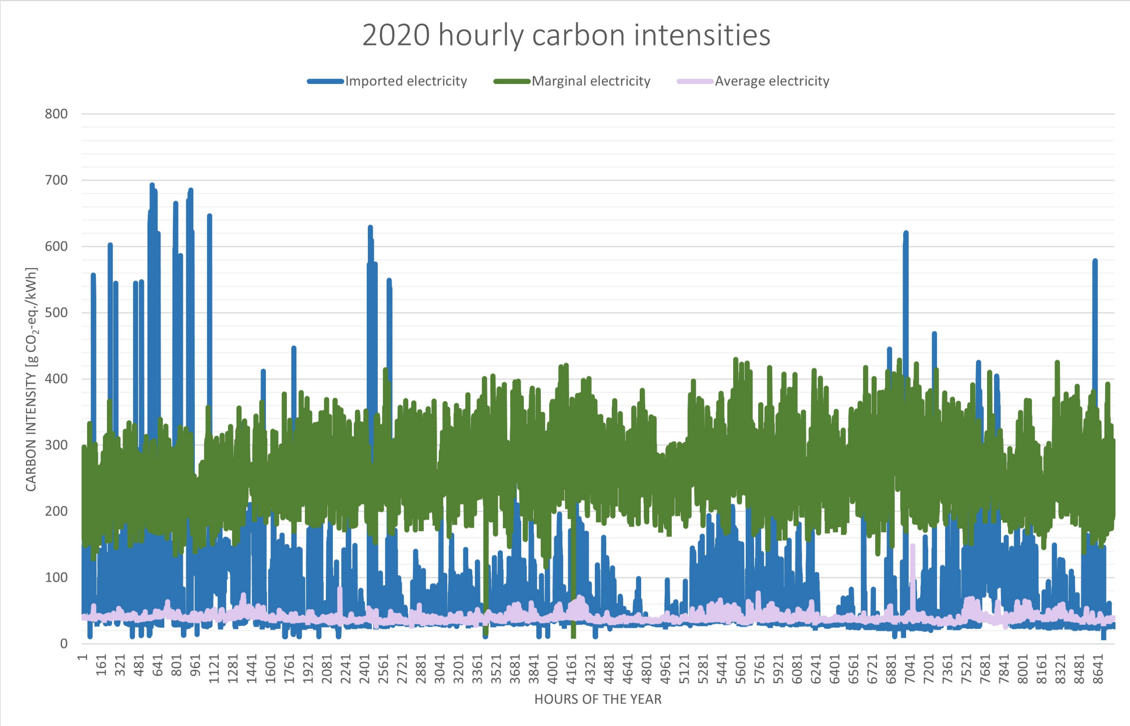

Figure 4.4: 2020 hourly carbon intensities for the scenarios: Imported- Marginal- and Average electricity.

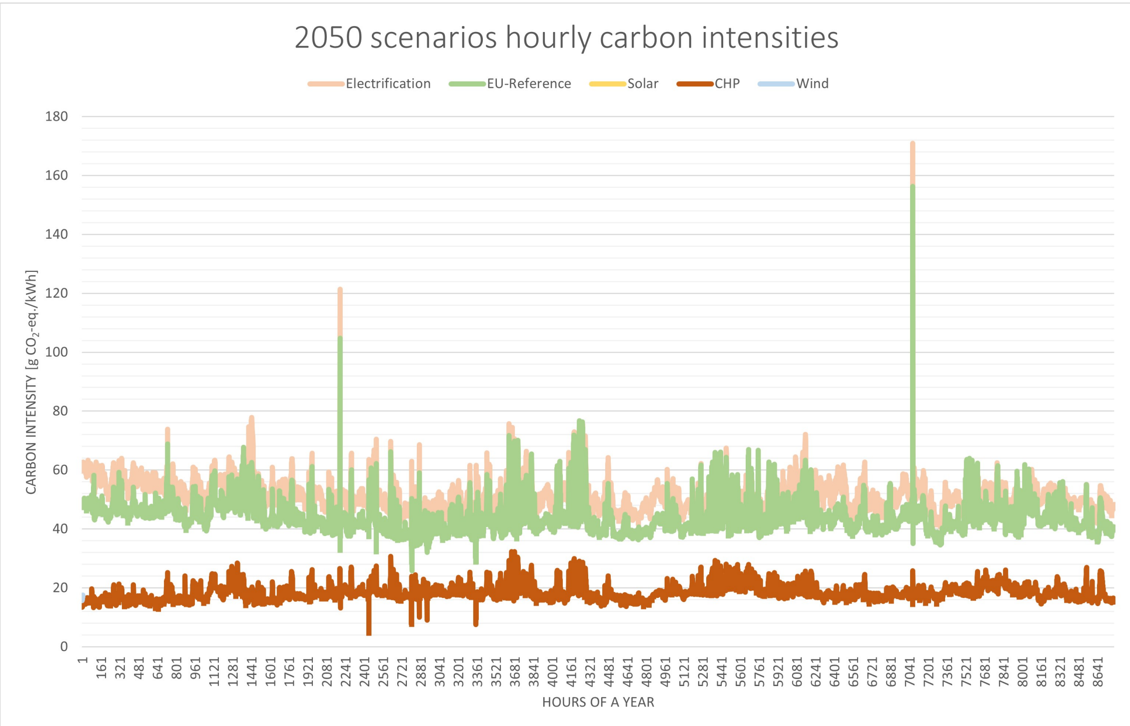

Figure 4.5: 2050 average hourly carbon intensities for the scenarios: Electrification, EU-Reference, Solar, CHP and Wind. Observe that the Solar and Wind lines lie behind the CHP line.

As the battery studied is rather small from a systems perspective, this means that there are few occasions where the battery goes unused for long periods of time, see figure 4.6 where an example of the hourly discharge and charge for a two week period can be seen, specifically when using marginal electricity. Both this and the spread of the carbon intensity tops are aspects which are positive, since the potential benefit of saving CO_2-eq. emissions which possibly could be gained from using a battery is increased. Also, the limitation of not regarding self-discharge is more acceptable than if there were less spread and fewer cycles.

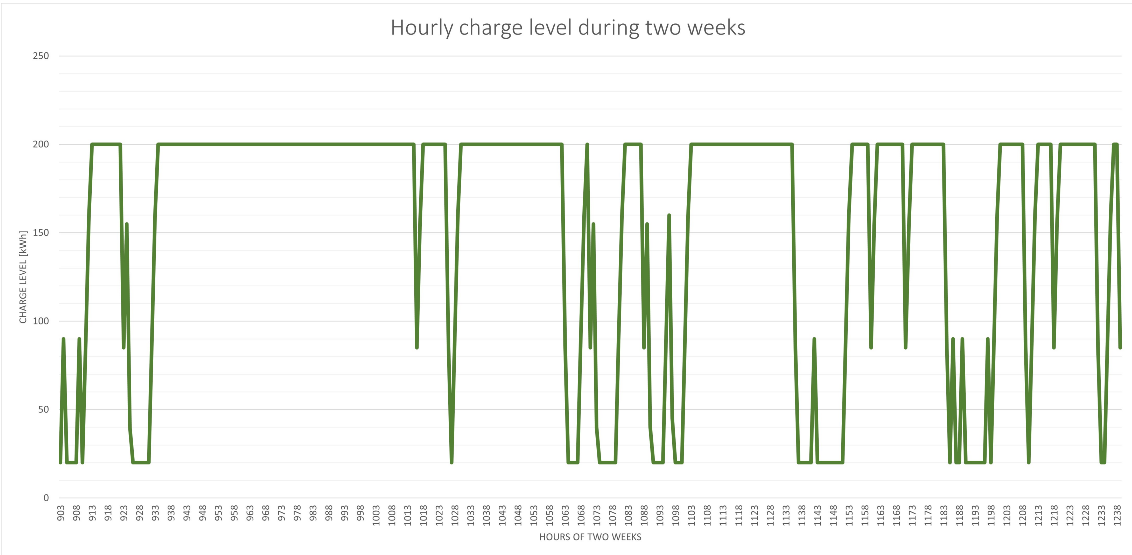

Figure 4.6: Hourly discharge and charge of a lithium-ion NMC battery for a two week period using 2020 Marginal electricity.

The time taken for the battery to go from 100 % to 80 % of the maximum storage capacity must be mentioned as there are large differences between the various scenarios. For marginal- and imported electricity the time taken is 4 and 8 years respectively. For the CHP and EU-Reference scenarios, it took 9 years. The Wind scenario and the average electricity had a time of 10 years. Lastly, the Electrification and Solar scenarios had a time taken of 11 and 16 years respectively.

When comparing the 2020 results with the 2050 results, a possible deduction could be that the utility of having a battery with the aim of reducing emissions by moving the high emission tops will be lowered in the future. The reason for this would be that there would be a lack of high emission power production to reduce. Hence, the reason for implementing a battery must be another, such as the supply of a fast frequency reserve. The times taken before 80 % capacity has been reached must, therefore, be taken with a grain of salt. The reason for this is that the time would increase if the utility is decreased.

4.1.4 CO_2-eq. emissions savings per kWh for a lithium-ion battery using varying electricity system scenarios and providing fast frequency reserve

Again, using the results from section 4.1.1, the emissions savings could be calculated, see figure 4.7. This time with the inclusion of providing fast frequency reserve, FFR, during June-September, as described in section 3.2.1.

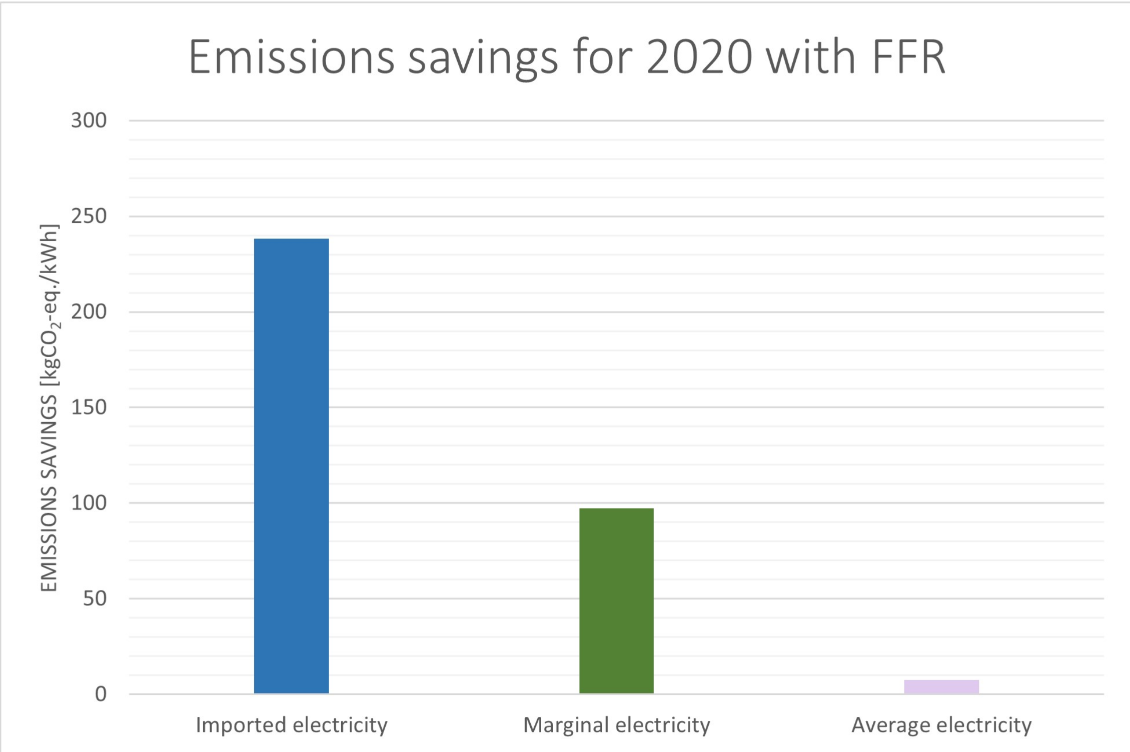

Figure 4.7: Emissions savings from a lithium-ion battery supplying fast frequency reserve during June-Sept for the 2020 electricity scenarios.

If comparing the results presented in figure 4.7 to the results for the 2020 electricity data sets from figure 4.3, it can be seen that the emissions savings become larger for imported- and marginal electricity use, but smaller for average electricity, when supplying FFR during part of the year than when FFR is not supplied.

The first reason for this is that there became an increase in years during which the battery is able to operate before the capacity got below 80 %. For the imported electricity there was an increase from 8 to 12 years, the marginal electricity had an increase was from 4 to 7 years and for the average electricity the increase was from 10 to 13 years. However, if viewing the results based on a chosen number of operating years, instead of when the battery has reached below the 80 % capacity, it is clear that the emissions savings become smaller when supplying FFR than without supplying FFR regardless of which electricity was used, as is to be expected.

The second reason for the increase in emissions savings is that since the FFR is supplied during the summer season, the least efficient season for using the battery as an energy storage is removed, see figure 4.4. This improves the overall efficiency of the emissions savings.

An important notation that must be made is that the results in figure 4.7 only represents the emissions savings for a fictive case where there is no degradation during the period of the year where FFR is supplied. This would not be the case in reality (Hollinger et al., 2018). Though, because of the lack of data available due to FFR being so newly implemented in Sweden, this limitation had to be accepted.

4.2 Retrospective climate impact of NMC lithium-ion batteries

As described in section 3.1, a number of different life cycle assessments, LCAs, were studied in order to receive a result with little bias to a certain LCA method. The global warming potentials, GWPs, of the acquired LCAs are presented in table 4.2, where they are ordered by size. For each GWP value, the source and the electricity mix is presented.

GWP [kg CO_2-eq./kWh capacity]	Electricity mix	Source
487	Coal	(Ellingsen et al., 2014)
240	Natural gas	(Ellingsen et al., 2014)
220	China	(Majeau-Bettez et al., 2011)
200	USA	(Romare and Dahllöf, 2017)
200	Europe	(Majeau-Bettez et al., 2011)
180	China	(Kallitsis et al., 2020)
172	USA	(Ellingsen et al., 2014)
150	USA	(Romare and Dahllöf, 2017)
140	China	(Kallitsis et al., 2020)
121	USA	(U.S.EPA, 2013)
100	China	(Kelly et al., 2019)
73	USA	(Dai, Kelly, et al., 2019)
65	Europe	(Kelly et al., 2019)
60	USA	(Dunn et al., 2015)
59	Europe	(Notter et al., 2010)
42	USA	(Ciez and Whitacre, 2019)
42	USA	(Hendrickson et al., 2015)

Table 4.2: Global warming potential for a NMC lithium-ion battery.

The resulting mean and median GWPs, based on all values in table 4.2 except the ones for 100 % coal or natural gas based electricity, can be seen in figure 4.8 for the different electricity mix areas.

What can be seen when observing table 4.2 and figure 4.8 is that for China, the GWP values are the most consistent of the different electricity mix areas as the mean and the median are equal. This may be due to that China has been producing

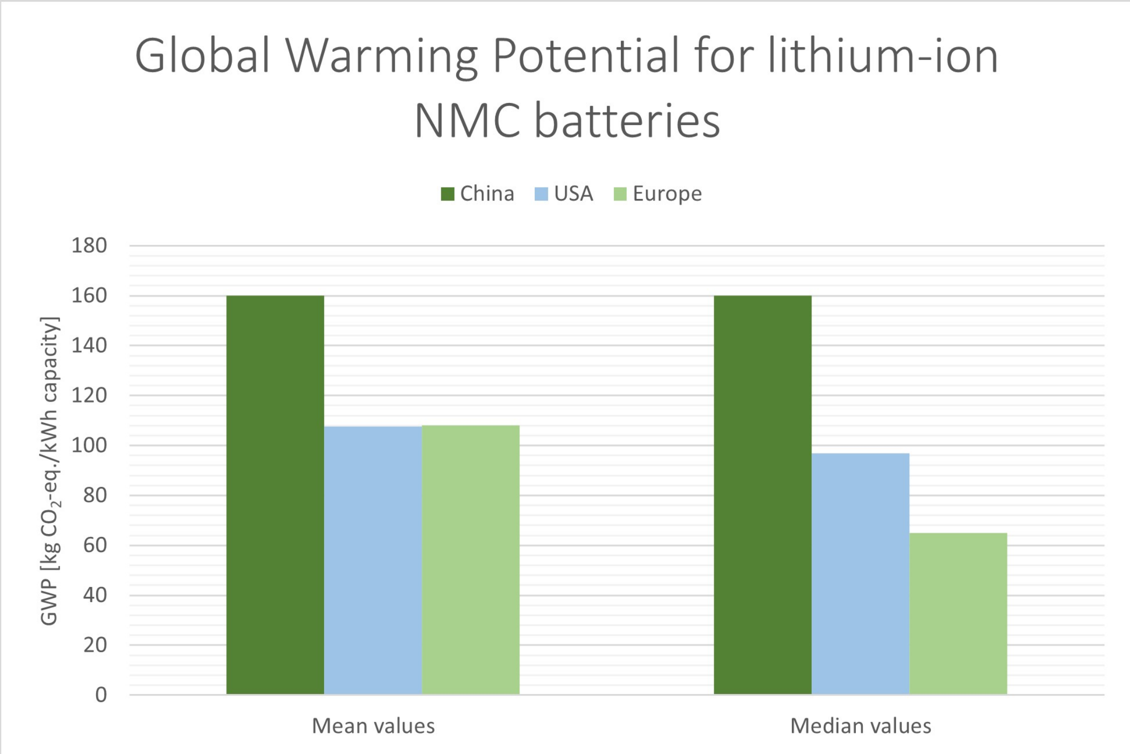

Figure 4.8: Mean and median global warming potentials for Chinese-, USA- and European electricity mix from different sources of life cycle assessments.

lithium-ion batteries for longer than the other two areas. Hence, there could be a possibility that a stream-lining effect has occurred where the manufacturers have become more similar in how a battery is produced or perhaps in how and where the resources are retrieved and processed.

For the GWP results for USA, there is a slight difference between the mean and median values. However, the most striking is how large the span of GWP values is in table 4.2. Romare and Dahllöf (2017) suggests that the choice of LCA methodology may play a role in such differences, where a top-down approach results in a higher GWP than a bottom-up approach.

The European GWPs in table 4.2 are, just like the USA values, spread out, but in this case it is only one value which is much larger than the other two values. This shows in figure 4.8 where the mean value is a lot larger than the median value. The reason for the high value is likely that Majeau-Bettez et al. (2011) assumes a slightly different battery chemistry compared to other sources. For both the positive and negative electrode pastes, Majeau-Bettez et al. (2011) assumes polytetrafluoroethylene and substitutes it with tetrafluoroethylene. In comparison, Ellingsen et al. (2014) assumes that the positive paste is made up of polyvinylidenefluoride and uses polyvinylfluoride as a proxy. The effect is a five- and four times smaller GWP for the positive- and negative electrode pastes respectively (Ellingsen et al., 2014).

It should also be noted however, that obtaining primary data from battery manufacturers is very difficult (Romare and Dahllöf, 2017). The differences between the GWP values within all three areas studied, could therefore depend on how the lack of primary data has been solved.

If comparing the emissions saving results from figures 4.3 and 4.7 with the GWPs for the Chinese electricity mix in figure 4.8, it can be seen that it is only when using imported electricity, with and without providing FFR, during the battery use phase that there becomes a negative climate impact overall. This can be seen in figures 4.9-4.12 where the percentage emissions savings away from the break-even point is displayed. Meaning, it is better to use the battery in the studied manner than not using a battery. The time taken until the break-even point has been reached is 8 and 7 years for with and without providing FFR respectively.

Similarly, comparing the USA GWPs with the emissions saving results, it can be seen that it is more beneficial to use the battery than not with the imported electricity, with and without providing FFR. Though, the time taken before break-even is reached for the mean value GWP is now 6 and 5 years, with and without providing FFR respectively. For the median GWP value it is only 5 and 4 years respectively for with and without providing FFR. For the median GWP value this is also true for the marginal electricity when supplying FFR, see figure 4.12. The time taken is then 7 years.

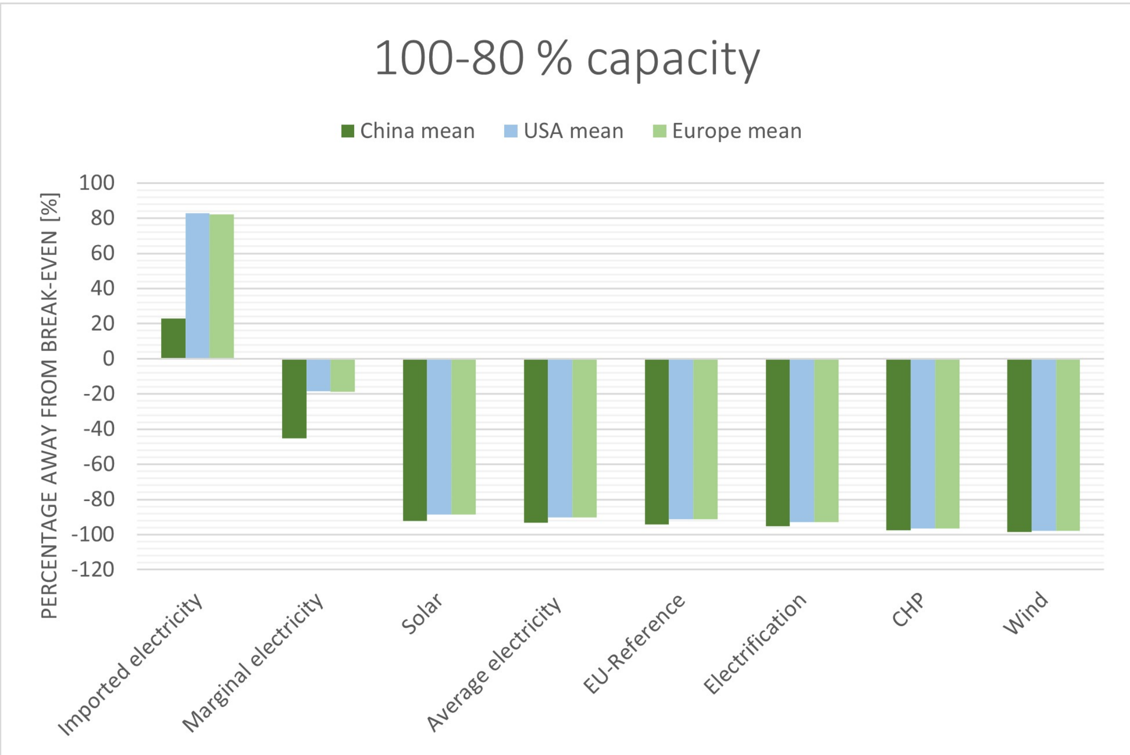

Figure 4.9: The percentage away from the break-even point of the mean GWPs for the different electricity scenarios.

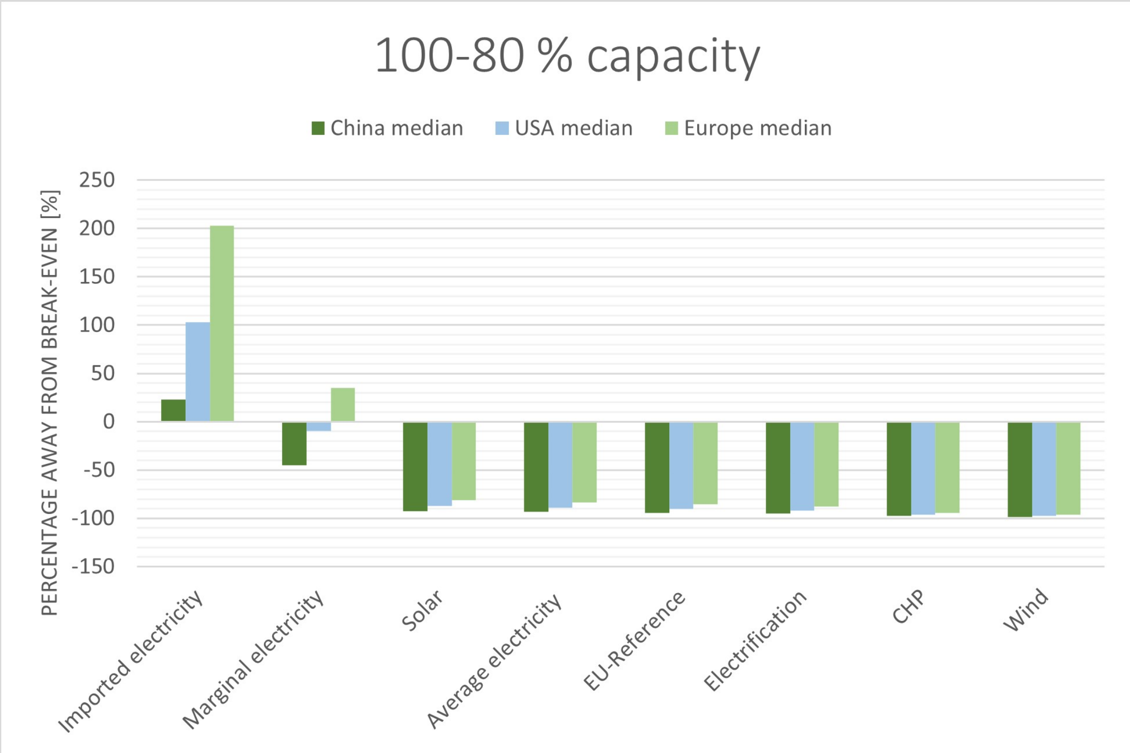

Figure 4.10: The percentage away from the break-even point of the median GWPs for the different electricity scenarios.

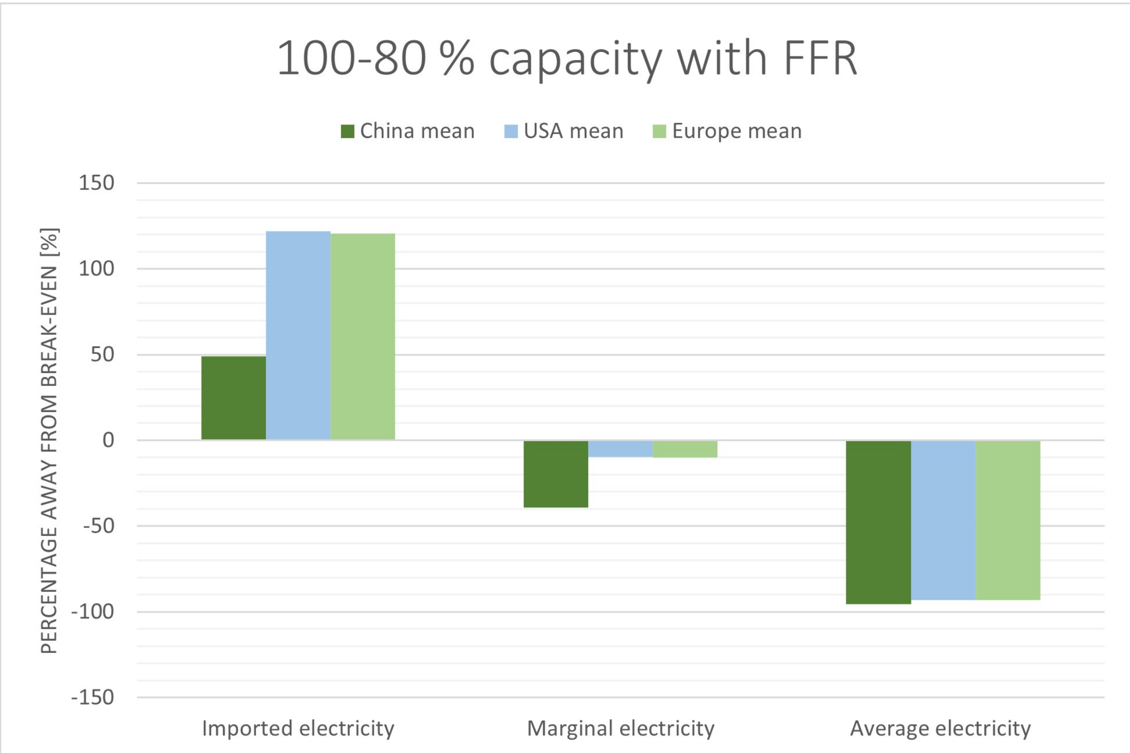

Figure 4.11: The percentage away from the break-even point of the mean GWPs for the different electricity scenarios when supplying FFR.

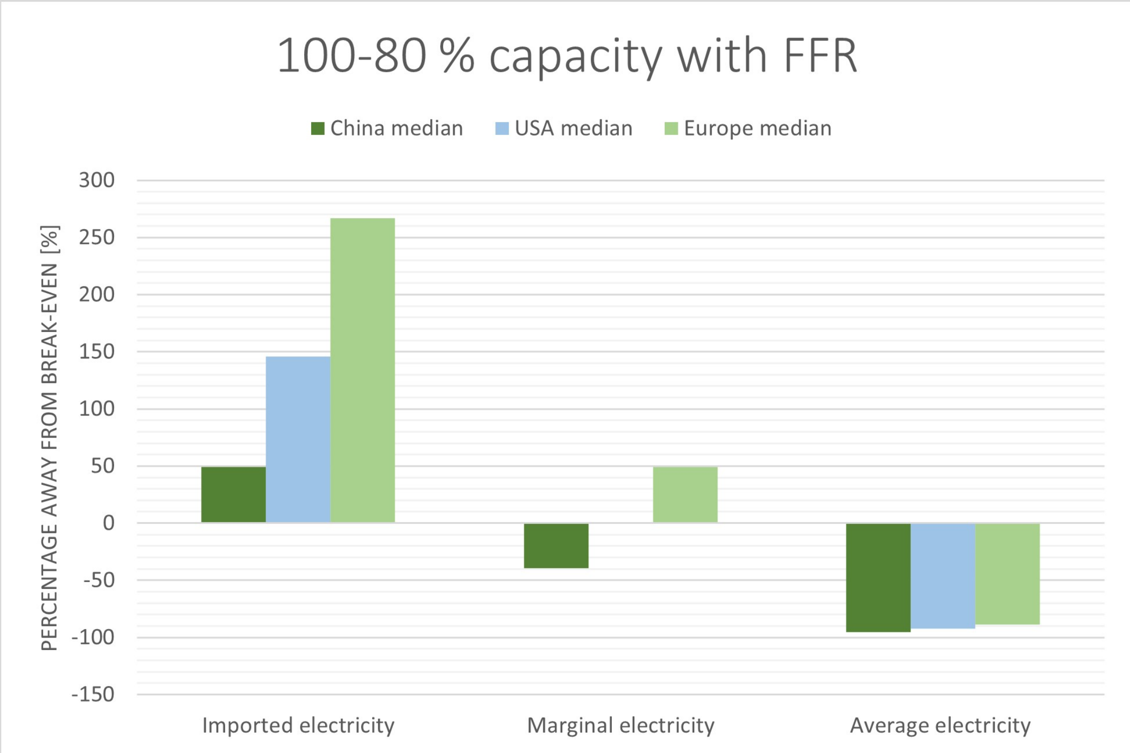

Figure 4.12: The percentage away from the break-even point of the median GWPs for the different electricity scenarios when supplying FFR.

Just as for the Chinese and USA GWPs, the European GWPs also lead to a battery being beneficial for the imported electricity, with and without providing FFR. The time taken before the break-even point is reached for the mean GWP value is 6 and 5 years respectively for with and without providing FFR, just as for the mean USA GWP value. For the median GWP value the time taken with and without providing FFR is 4 and 3 years respectively. However, for the European median GWP, both marginal electricity with and without providing FFR result in a battery being beneficial, in contrast to the median USA GWP value, see figures 4.10 and 4.12. The time taken for the median GWP value when providing FFR is 5 years and when not is 4 years.

These results are, however, directly dependent on there being a possibility to choose which electricity to use. Normally, a consumer would not be able to choose for example imported electricity over marginal or average electricity. This leads to the requirement of it being an electricity system authority, or similar, using the battery. This is therefore, contrary to how the EU Directive 2019/944 states energy storages should be operated, as was described in section 2.1 (Energimarknadsinspektionen, 2020).

What would most likely be assumed for a Swedish battery owner, who is not an electricity system authority, is that marginal electricity is being used to charge and discharge the battery. A battery produced in Europe would hence be the only viable option if a climate impact benefit is to be achieved. This would also work well within the EU Directive 2019/944 regulations for a flexibility market as there would be no rule broken considering the ownership of the battery storage (Energimarknadsinspektionen, 2020).

In figures 4.9 and 4.10, it can be seen that none of the 2050 scenarios resulted in a positive, or even zero, percentage away from the break-even point. The usage of a battery storage to remove high emission electricity production could therefore be deemed limited in the future. Instead the battery storages are more likely to be necessities within variation management and ancillary services (Göransson and Johnsson, 2018, Roupe et al., 2020). This likely necessity stems from the 2050 scenarios requiring short- and long-term management in order to be practically functional.

A likely necessity is not, however, equal to a must. As seen in figure 4.8, the battery manufacturing brings with it quite a climate impact. It must therefore be put against its usefulness as well as alternative options. When it comes to variation management, battery storages often compete against demand side management or vehicle-to-grid services. Two options which are often much cheaper and have less of an environmental burden (Göransson and Johnsson, 2018). Regarding ancillary services, these are often provided by hydro power units, at least in Sweden (Nohrstedt, 2021), which also have low costs and environmental impacts due to them already being installed. For battery storages to be able to compete as a viable option, their costs and environmental burdens need to be reduced.

4.3 Prospective climate impact of NMC lithium-ion batteries

There are many possibilities for reducing the environmental impacts of a lithium-ion battery life cycle, such as changing production location, using second-life electric vehicle batteries or increase recycling.

4.3.1 Changing production location

In section 4.2, it could be argued that there is only a slight consensus between different life cycle assessments on what effect changing the production location has for the climate impact. However, many of the articles studied do discuss or imply the effect changing electricity mix in the production could have for the overall GWP.

Several articles, such as Ellingsen et al. (2014), Kallitsis et al. (2020), Kelly et al. (2019), and Romare and Dahllöf (2017), argue that changing the electricity mix can result in 60 % less GWP. Saving between 60-70 % is furthermore, what the newly established Swedish battery manufacturing company Northvolt aims to achieve with their manufacturing using Swedish electricity mix instead of Chinese (Peplow, 2019).

If calculating a 60 % reduction to the Chinese results in figure 4.8 this would lead to a GWP of 64 kg CO_2-eq./kWh. This is roughly the median GWP value when using

European electricity mix. Hence, the result from comparing the emissions saving and GWP results from section 4.2 remains. Only imported- and marginal electricity, with and without providing FFR, leads to negative climate impacts.

Calculating a 60 % reduction of the lowest GWP value with Chinese electricity mix in table 4.2 from Kelly et al. (2019), results in a GWP of 40 kg CO_2-eq./kWh. Even though this is a very low value, it still isn't enough to change the already established imported- and marginal electricity being the two relevant.

On the other hand, the high climate impact for a NMC battery manufactured using average European electricity mix found by Majeau-Bettez et al. (2011), see figure 4.8, was described to only increase with 10-16 % if the manufacturing electricity was changed to the average Chinese mix. There is hence, an uncertainty of to what extent moving the manufacturing can influence the climate impact of the battery. Though, there being an effect is certain, even though there are articles which claim lower climate impact when using Chinese electricity mix than what Majeau-Bettez et al. (2011) claims the climate impact is with European electricity mix.

However, it should be noted that changing the production location often comes with changes in where the resources are extracted. Such changes may result in increases of other impacts, apart from climate impact, hence resulting in sub optimal decisions from a more holistic point of view. (Gode et al., 2009)

An example of this is when changing the production location to Europe, the nickel extraction may be moved to Russia where the emissions of SO_2 are much higher than in other countries where the nickel was extracted previous to the location change (Dai, Kelly, et al., 2019). This leads to a significantly increased acidification potential, which arguably would counteract the positive change on climate impact brought about by the shift in location.

What size of change to other impact categories is required before the climate impact change could be considered inadequate to motivate a production location change is difficult to answer. Such an answer would only be possible to retrieve using subjective methods, for example weighting which is sometimes applied within life cycle assessments (Baumann and Tillman, 2004, U.S.EPA, 2013). However, it should be noted that while applying such a subjective method would result in the impact categories being comparable, which could be useful, it nonetheless undermines their objectivity.

4.3.2 Repurpose of electric vehicle batteries

Instead of operating a fresh battery as a stationary storage from 100-80 % of its capacity, it could be possible to implement a second-life electric vehicle battery. According to Zhao (2017), this would mean operating the battery from 80-40 % of its capacity. The resulting emissions savings can be seen in figure 4.13. These results where calculated as described in section 3.2.1 but for a battery capacity starting at 80 % and ending at 40 % of its maximum capacity.

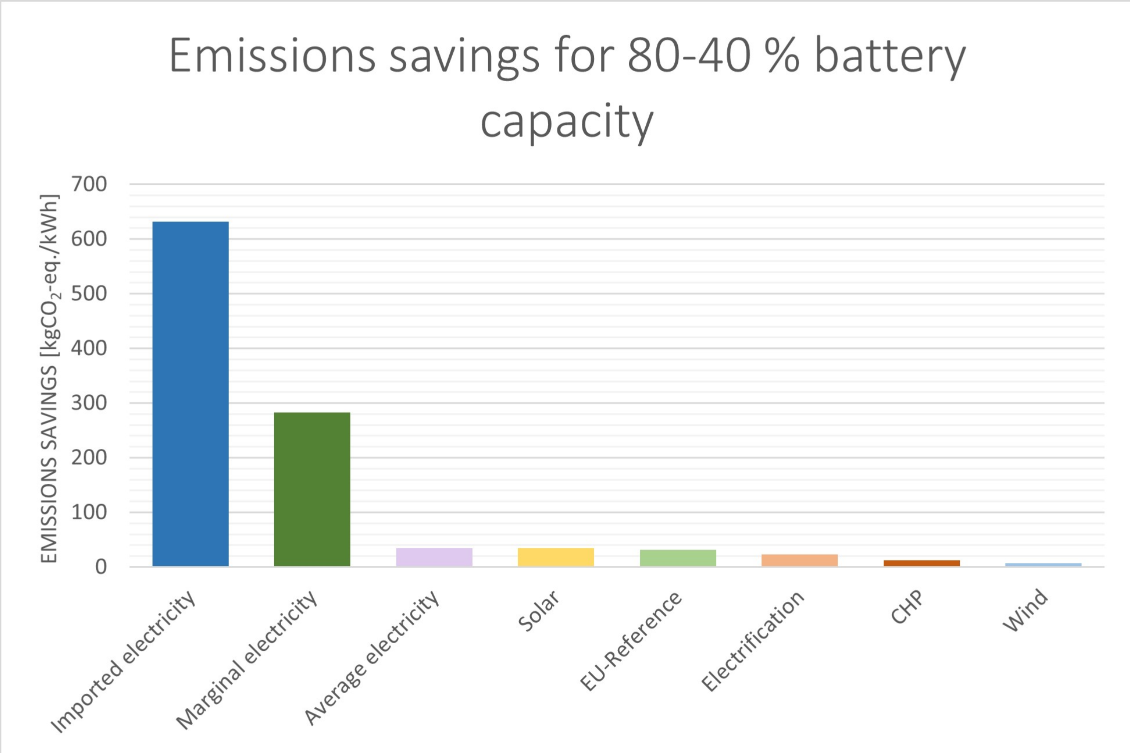

Figure 4.13: Emissions savings from 80-40 % of a lithium-ion battery capacity for the 2020 and 2050 electricity scenarios.

What can be seen in figure 4.13 is that all electricity scenarios have roughly three times larger emissions savings than in figure 4.3. The gain from doubling the range within which the capacity is allowed is therefore larger than just a doubling. The reasoning behind why this is the case can be found in figure 3.1, which shows the battery capacity degradation. The degradation is namely the fastest for the first few thousands of cycles, then it slows down somewhat for the next few thousands of cycles and eventually it picks up speed again for the last thousands of cycles. The number of cycles which the battery is allowed before the capacity is at the lower limit is hence, more than double for the 80-40 % case compared to the 100-80 % case.

With all emissions savings for the 80-40 % capacity case being roughly three times larger than the 100-80 % capacity case, the possibility of finding a battery with a GWP which equals to, or is smaller than, the emissions savings is increased.

Furthermore, there are other potential environmental benefits from reusing an electric vehicle battery. Firstly, from a circular economy perspective, there would be less new batteries manufactured as the already existing batteries are repurposed. This would lead to less resource extraction and hence less of an environmental impact.

Secondly, from a life cycle perspective there may be a decreased environmental impact from the second-life battery compared to a single-use battery, or the environmental impact of the second-life battery may even be nonexistent. This depends on how the emissions are allocated between the first and second use of the battery.

Romare and Dahllöf (2017) argues that due to the present situation with no second-life battery market, the climate impact is completely allocated to the electric vehicle. According to Richa, Babbitt, Nenadic, et al. (2017) this would be a type of cutoff allocation where the intention of producing the battery was solely for using it in an electric vehicle means that the whole life cycle climate impact is allocated to the electric vehicle. This would hence lead to a 0 kg CO_2-eq./kWh GWP and evidently would result in all electricity scenarios from figure 4.13 having less of a climate impact when using a second-life battery than not, regardless of how long the second-life battery was allowed to operate. The use of the cutoff allocation approach could be supported by it traditionally being the most frequently used approach and by it being simple to implement (Nordelöf et al., 2019).

If a market for second-life batteries is established, however, the question of allocation is not as simple. As this is a situation which is most likely in 2050, it hence must be taken into consideration. One allocation method which could be relevant is the so called 50/50 approach, where according to Richa, Babbitt, Nenadic, et al. (2017), it is assumed that the percentage of new batteries to repurposed electric vehicle batteries is 50 %. The climate impact of the life cycle would, therefore, be allocated equally between the electric vehicle and the stationary storage (Richa, Babbitt, Nenadic, et al., 2017). This would result in a minimum GWP of 21 kg CO_2-eq./kWh, which is half the lowest value from table 4.2. If comparing with the emissions savings from figure 4.13, it can be seen in figure 4.14 that all electricity scenarios except the CHP and Wind scenarios would benefit from implementing a battery. The time taken before the break-even point is reached would be at least 11 years, as seen when using imported electricity, and at most 51 years, which is the case when using the Electrification scenario electricity. When using imported electricity this would then mean that there are 31 years left to use the battery before the capacity drops below 40 %. It should be noted, though, that the degradation from time, regardless of the number of cycles, may impact these results.

The positive aspects of implementing second-life batteries instead of new batteries are as can be seen plentiful. Not only are there environmental benefits to be gained, there are also savings in terms of the cost of stationary batteries which can be made. The positive aspects have also been acknowledged by the Swedish energy company Fortum which is in this year installing second-life electric vehicle batteries next to a hydro power unit to observe and evaluate their function compared to using new batteries (Nohrstedt, 2021).

It should be noted however, that there are currently impediments to a broad utilisation of second-life batteries. Firstly, there may be doubts around the security of second-life batteries. According to Romare and Dahllöf (2017), it may be difficult to ensure the safety and stability of second-life batteries leading to less of a demand. Also Kamath et al. (2020) supports the importance of how the customers view the second-life batteries and suggests that it may impact the market diffusion.

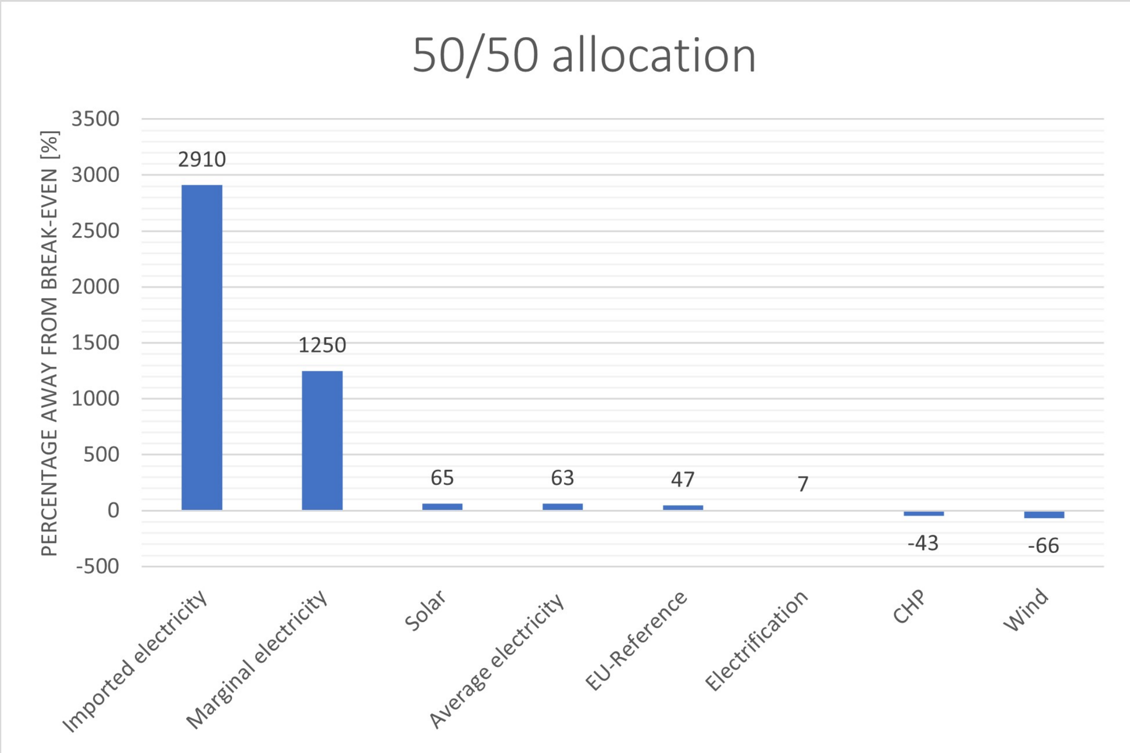

Figure 4.14: The percentage away from the break-even point of the lowest GWP for the different electricity scenarios at 50/50 allocation.

Secondly, in order for there to be a market for second-life batteries, businesses need to be established which focus on diagnosis of the batteries to obtain information regarding for example what remains of the initial capacity or what the condition of the battery is (Romare and Dahllöf, 2017). This impediment may however, be further hindered by the lack of end-of-life electric vehicle batteries (Kamath et al., 2020). Though, this would be a diminishing issue as time moves on and more electric vehicles are built.

The third and last note is that there needs to be regulations put in place on how the previous two notes are to be evaluated and documented (Jankowiak et al., 2019, Romare and Dahllöf, 2017). Currently, repurposing of batteries is not defined by the EU Battery Directive nor by the EU end-of-life vehicle directive (Richa, Babbitt, and Gaustad, 2017). However, there are changes on the way (European Commission, 2020). There are also, according to Kamath et al. (2020), strategies required on battery design, both regarding standardisation and dismantling.

4.3.3 Recycling of lithium-ion batteries

Presently, it is the case that if a lithium-ion battery is recycled this is mainly done by pyrometallurgy (Romare and Dahllöf, 2017). This is a recycling method in which the battery is incinerated in a smelter, normally with the main objective of obtaining cobalt and nickel but also copper and iron (Dai, Spangenberger, et al., 2019, Dunn et al., 2015, Gaines et al., 2011, Hendrickson et al., 2015). These metals in the so called matte are then processed further by acid leaching, solvent extraction and precipitation (Dai, Spangenberger, et al., 2019). The resulting slag contains for example lithium and oxidised aluminium, however, the slag is seldom processed further to recover these metals due to lack of an economic incentive (Dunn et al., 2015, Gaines et al., 2011).

By using pyrometallurgy Romare and Dahllöf (2017) and Ciez and Whitacre (2019) claim there is a net increase in the overall climate impact of 15 and 3 kg CO_2-eq./kWh respectively. Hendrickson et al. (2015) on the other hand, argues that there is a 10 kg CO_2-eq./kWh net decrease in overall climate impact.

An alternative recycling method which is currently being commercialised is hydrometallurgy (Hendrickson et al., 2015). This method consists firstly of discharge and disassembling of the batteries (Dai, Spangenberger, et al., 2019). Then, a hammer mill crushes and shreds the batteries to pieces(Romare and Dahllöf, 2017). Thirdly a calcination process incinerates the binder and electrolyte at low temperature (Dai, Spangenberger, et al., 2019). Next step is further physical separation (Dai, Spangenberger, et al., 2019). Then a leaching process occurs, potentially using lithium brine, to chemically leach the metals aimed to recover (Hendrickson et al., 2015, Romare and Dahllöf, 2017). Lastly, the solvent is extracted, sometimes followed by a precipitation (Dai, Spangenberger, et al., 2019). With hydrometallurgy it is possible to obtain various valuable metals, though common is copper, aluminium, cobalt, nickel, manganese and Li_2CO_3 (Dai, Spangenberger, et al., 2019, Romare and Dahllöf, 2017).

By using hydrometallurgy Romare and Dahllöf (2017) claims that there is a 12 kg CO_2-eq./kWh net decrease in climate impact. Ciez and Whitacre (2019) also claims there is a net decrease, however, only of 5 kg CO_2-eq./kWh. The reason for this difference may be that there are, as of yet, mainly laboratory scale processes installed (Dunn et al., 2012). This leads to it being difficult to estimate the energy required and hence the CO_2 emissions. A more likely reason though, is that Ciez and Whitacre (2019) assumes that all materials not recovered are incinerated.

The usage of hydrometallurgy on a large scale is currently being hindered by its process complexity and costs (Hendrickson et al., 2015). Hence, the potential it has for lowering the environmental impacts of lithium-ion batteries may yet take some time before being realised. Though, there are companies which have begun combining their pyrometallurgical process with hydrometallurgical process steps which may increase the speed with which the hydrometallurgical method evolves (Romare and Dahllöf, 2017).

A recycling method with great potential for lowering the environmental impact of lithium-ion batteries is direct physical recycling (Ciez and Whitacre, 2019). This is a recycling method in which the battery firstly is discharged, disassembled and perforated. Then, the electrolyte is obtained through super-critical CO_2 extraction and the remains of the battery are shredded. Plastics, anode, cathode, and metals are then retrieved through several physical separation processes such as density separation and froth flotation. Lastly, the obtained cathode material is relithiated, resulting in a cathode powder which could be used as an input in battery manufacturing. (Dai, Spangenberger, et al., 2019)

For direct recycling Dunn et al. (2015) argues that a net reduction of 17 kg CO_2-eq./kWh can be achieved specifically for the cathode material if compared to a virgin material. A much smaller value is obtained by Ciez and Whitacre (2019), a net reduction of 3 kg CO_2-eq./kWh. A likely reason for the large difference between the values is that Dunn et al. (2015) assumes not only the cathode material, but also metals and electrolyte are recovered while Ciez and Whitacre (2019) assumes that all but the cathode material is incinerated.

It should be noted that specifically for the reductions described by using the direct physical recycling, only the cathode material is considered. There is a possibility that even further climate impact savings could be made for other recyclables. This would, hence, lead to even lower GWPs.

However, direct physical recycling is currently being developed and is not yet commercialised (Dunn et al., 2012). Just as for the hydrometallurgical method, the direct physical method may require some time before being possible to realise. There is also a question yet to be answered regarding whether the obtained cathode material has the same performance in a battery as a virgin cathode material (Dunn et al., 2012). If this is not the case, the development and diffusion of the method may be hindered due to lack of desirability of the recovered cathode material from the battery manufacturers.

The largest GWP reduction claimed is 17 kg CO_2-eq./kWh (Dunn et al., 2015). If subtracting this value from the lowest results in table 4.2, the remaining climate impact is 25 kg CO_2eq./kWh. Even though this value is much lower than the retrospective GWPs, the reduction is not enough for there to be a difference to what electricity mix should be used for a battery, between the storage capacity of 100-80 %, to be preferred over just using the electricity directly. The time taken for the break-even to be reached is between 1 year, when using imported electricity, and 2 years, when using marginal electricity.

It is important to note in what manner the LCAs have handled when the recycling is assumed to have occurred. This is vital for there not being any double counting, but also because the time aspect could be of importance. In the results of this study, the trends did not change when recycling was implemented and the time taken before the break-even point is reached is fairly short. Hence, it is not of much importance if the recycling is assumed to have occurred at the end of the life cycle or if the battery has used recycled material in its production.

4.4 Climate impact of vanadium redox flow batteries

As could be seen in section 4.2, the global warming potentials, GWPs, of lithium-ion NMC batteries are rather high. An interesting battery chemistry with high future potential for stationary storage is vanadium redox flow batteries, VRFBs. The reason for this potential is the low climate impact of 0,0382 kg CO_2-eq./kWh according to Weber et al. (2018) or 0,0402 kg CO_2-eq./kWh according to Denholm and Kulcinski (2004) which can be claimed by the VRFBs. This can be compared to the lowest GWP value from this study of 42 kg CO_2-eq./kWh.

Though the two studied sources for climate impact are in agreement with one another, it should be noted that it is difficult to obtain trustworthy data on VRFBs, as there, according to Weber et al. (2018), is no large scale manufacturing in place at the present. This also leads to there being a lack of detail in the life cycle assessments as well as oversimplifications (Weber et al., 2018). The reason for including the GWPs of VRFBs despite the issues described, was that unlike previous LCAs on the topic, the two sources were more comprehensive and transparent of their inventory data.

The GWP from Weber et al. (2018) is based on the assumption that the mining and extraction of vanadium is located in South Africa which has a largely coal based electricity and hence constitutes 46 % of the climate impact. If assuming the mining and extraction of vanadium is located in Sweden and the transformation of the Swedish mining industry has taken place, those 46 % could be nearly entirely removed, resulting in an even lower GWP.

Another aspect in favour of the VRFBs, apart from the low GWP, is the potential recyclability. Due to VRFBs being possible to dismantle in their entirety by mechanical processes, high recycling efficiency can be achieved with little climate impact (Weber et al., 2018). It is still however, uncertain how much the electrolyte degrades over the battery life time and hence how much processing is required before it can be reused, though Weber et al. (2018) nevertheless claims the electrolyte will be possible to reuse.

If comparing the average GWP of a VRFB with the emissions saving results for a lithium-ion NMC battery from figure 4.3 it can be seen that even with the lowest emissions savings of 2,3 kg CO_2-eq./kWh, which were obtained from the Wind scenario, there would be benefits from having a VRFB. Hence, in all electricity mix scenarios studied there would be less of a climate impact if a VRFB is used than if it is not used, see figure 4.15. Also, the time taken before the break-even point is achieved, for a battery starting at 100 % storage capacity, is less than a year for all scenarios of electricity studied.

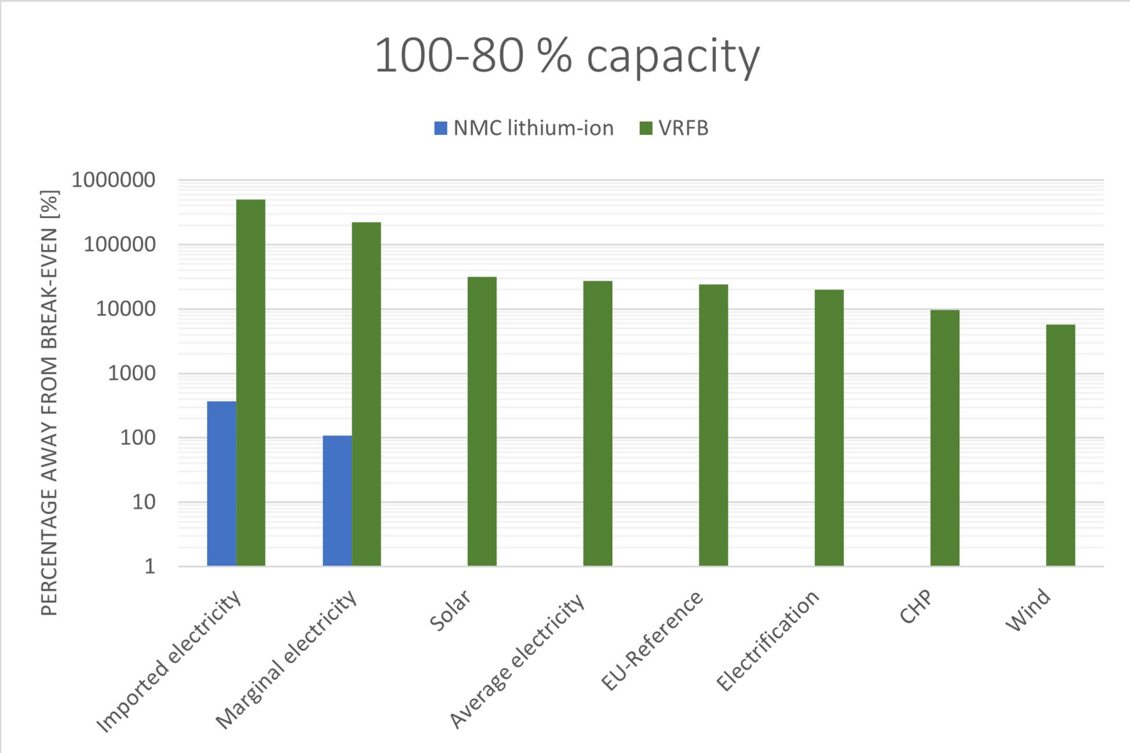

Figure 4.15: The percentage away from the break-even point of an average VRFB GWP and the lowest GWP of a NMC lithium-ion battery for the different electricity scenarios, observe the logarithmic scale.

5

Conclusion

What can be concluded from this study is that depending on the circumstances, deciding to implement a stationary NMC lithium-ion battery can be advantageous from a climate impact perspective compared to not doing so. The circumstances are however, rather limiting.

To begin with, one of the circumstances with the largest effect comes from the electricity used to charge and discharge the battery during the use phase of the life cycle. Large differences in carbon intensity throughout the year, results in a larger GWP being possible from the other phases of the battery life cycle. In this study, the largest carbon intensity differences could be seen from Imported electricity, and secondly from Marginal electricity.

It can be concluded that regardless of if the battery being implemented was manufactured in China, USA or Europe, it is beneficial if it is Imported electricity which is used during the use phase. This is the situation both for the case when the battery is providing FFR during the year and for the case where no FFR is provided. For the USA median with FFR and the European median, both when supplying FFR and when not doing so, GWP results, it is also beneficial with Marginal electricity.

The European median GWP without providing FFR would have a time taken before the break-even point has been reached of 4 years. It would also provide 35 % extra emissions savings after the break-even point has been reached until the capacity is at 80 %. This is the lowest of the positive results for a battery produced in Europe, even so, it could be large enough for it to be realistic to implement such a battery in order to reduce the overall climate impact. The short time taken before the break-even point is reached is also relevant for the emissions goal to be within reach before it is too late. It should be noted, though, that this is under the premise that the battery can be used optimally from the perspective of the whole electricity system, and not for example a single building. Also, based on the results for FFR it can be concluded that Sweden has unfavourable conditions during the summer and further optimisation should be possible.

There could be seen some effect of recycling or moving production location on the GWP of the battery. However, in solitude the effects were not large enough for there to be changes to what electricity could be applied during the use phase with the result of a battery being beneficial to implement.

On the other hand, a large effect on the results could be seen from changing from fresh batteries, being used from 100-80 % capacity, into repurposed electric vehicle batteries, used from 80-40 % capacity. Depending on what allocation is chosen, the extent of the effect varies. All electricity scenarios presented in this study would benefit from implementing a battery if the climate impact is solely allocated to the electric vehicle. With the more conservative allocation of 50/50, though, a battery would be advantageous for all electricity scenarios except the CHP and Wind scenarios.

The limitation of assuming that no degradation of the battery occurred during its provision of FFR must be acknowledged. Depending on to what extent the FFR service is being utilised, this limitation could largely influence the results. This could lead to the European median GWP value being the only value which is low enough for Marginal electricity to be applicable during the use phase and hence beneficial for the overall life cycle. The Chinese and USA GWP values, as well as the European mean GWP value, would hence all require Imported electricity for a battery to be beneficial, limiting the conclusions even further.

If the development and diffusion of vanadium redox flow batteries are progressed, it could be concluded that they would be very beneficial regardless of electricity scenario. An example for the future could be the Solar scenario, where implementing a VRFB could result in over 200 000 % extra emissions savings after the break-even point has been reached until the battery is at 80 % capacity. Also, the break-even point would be reached within less than a year, concluding a very large potential for the emissions target to be within reach.

5.1 Recommendations for future work

It would be recommended for future studies to look at other impact categories than global warming potential, such as acidification-, ecotoxicity-, human toxicity- or abiotic resource depletion potential. By doing so, a more holistic view of the eventual usage of a stationary battery would be achieved, and hence more optimised decisions can be taken in order to minimise the overall environmental impact.

One example of an aspect which could be affected by this is the recycling of the batteries, which in this study had inconsistent results depending on recycling method. The reason for it being affected is because a regard for abiotic resource depletion potential would lead to recycling being viewed in a consistently more positive light compared to if only global warming potential is regarded. This is due to that recycling, regardless of method chosen, results in less abiotic resource depletion than if no recycling is executed (Richa, Babbitt, and Gaustad, 2017).

Another example is what was described in section 4.3.1, where a change in manufacturing location may result in an increased acidification potential due to additional location changes of the materials extraction. If acidification potential is taken into consideration, it may lead to other decisions being made by the battery manufacturers, such as pressure being put on the potential materials extraction company to improve before a shift in location can be accepted.

It would also be of value to study social impacts in addition to the environmental impacts. Time and time again it can be seen how social factors are neglected in favour of low costs, the fast fashion industry and the metals extraction industry being just two examples. If the social impacts of a battery life cycle are studied in a so called social life cycle assessment (United Nations Environment Programme, 2009), there may be a major potential for improvements. However, this depends on a diffusion of social life cycle assessments performed by third parties throughout industries, as well as the spread of their results to consumers for there to be a shift in demand.

Since there were limitations to how true to life the emissions saving results when providing FFR were, it would be interesting with a follow-up study in a couple years time. This study would have the opportunity to assess the actual degradation of the battery capacity and hence its life time. Furthermore, the change in demand over the years up until the new study for FFR services during a year could be utilised in order to construct possible 2050 scenarios of the demand. These scenarios would most likely be more probable than scenarios constructed without this information.

It would also be of interest to investigate how the plethora of possible flexibility services which a battery could provide could be optimised for certain circumstances. This would include the supply of a single flexibility service as well as the potential combination of flexibility services. Provision of several ancillary services on the same supply unit has occurred, for example with fast frequency reserve and frequency containment reserve - disturbance (Svenska kraftnät, 2020). However, it would be interesting to study the impact this would have on a battery.

Another aspect, which would be interesting to explore in the future, is the effect which the implementation of a flexibility market will have. As was described in section 2.1, Energimarknadsinspektionen (2020) is currently formatting the flexibility markets retrieved from the EU directive 2019/944. Once such markets have been enforced, there may be a whole new situation for smaller actors wanting to participate in the supply of flexibility services.

On the same note, it would be of value to investigate how the requirements for flexibility- or ancillary services change over time, as the electricity system evolves. This may be of interest from an international perspective where lessons learned could be shared and knowledge exchanged. Specifically the questions of whether battery storages are required, and if so, to what extent and during what circumstances could be valuable to answer. Previous studies regarding variation management strategies have been optimised based on cost (Göransson and Johnsson, 2018), it would in the future therefore be interesting to look into optimisation based on environmental burdens.

Last but certainly not least, is a recommendation to study the formation of a second-life electric vehicle battery market. This could be done in the sense of an accounting study, observing the current trends and transitions. On the other hand, it would also be valuable to study how such a formation of a market can take place and hence, what would be required. Something which was lightly touched upon in section 4.3.2.

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