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Comparative Life Cycle Assessment of Second Life NMC Batteries

Master's thesis in Industrial Ecology

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Abstract

As the EV batteries reach their end of life, a large number of batteries will be available post 2030. One solution for retired batteries is to consider them for second life applications in Battery Energy Storage Systems (BESS). The viability of second life Nickel Manganese Cobalt (NMC) batteries, however, is a balancing act between their reliability due to fading performance and safety issues, costs and environmental impacts.

This Master's thesis investigates the viability by breaking it down into two research questions. Firstly, from the viewpoint of the impact categories energy demand and climate change, how the second life NMC battery compares to a new Lithium Iron Phosphate (LFP) battery for BESS applications. And secondly, regarding the material scarcity impact category, how beneficial it is to delay the recycling of the NMC battery by adding a second life. The first question is answered in a break-even analysis with LFP as a reference point by considering three different scenarios. The scenarios represent different allocations of environmental impacts between the first and second life of the NMC. Each scenario is further comprised of different recycling percentages. For the second question, the crustal scarcity indicator is used as a proxy to address the material scarcity hot-spots at the battery pack, cell and cathode level. Moreover, the supply risks and geopolitics surrounding the critical metals used in the battery chemistries are discussed.

According to the break-even analysis, it is only when 0% of the energy requirement for production and recycling are allocated to the second life that a second-life NMC battery can outperform a new LFP battery in terms of energy use. This is mainly due to a lower efficiency of the aged battery. In terms of climate change, the second-life NMC battery can potentially be a viable choice for both 0% and 20% allocation, depending on the local electricity mix. The carbon intensity of the electricity mix in the BESS was indicated as the most significant parameter. Regarding materials scarcity it was observed that in some situations second life use of NMCs could aggravate supply risks of nickel and cobalt. However, this depends on the existence and efficiency of recycling infrastructure, and hence on the time horizon.

Keywords: second life battery, lithium-ion battery, life cycle assessment, battery energy storage system, end of life management.

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Nadja Bergqvist & Bahareh Talakoob, Gothenburg, August 2022

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List of Abbreviations

BESS Battery Energy Storage System

CAM Cathode Active Materials

CE Circular Economy

CSI Crustal Scarcity Indicator

CSP Crustal Scarcity Potential

DRC Democratic Republic of the Congo

EFC Equivalent Full Cycle

EoL End of Life

EV Electric Vehicle

GHG Greenhouse Gas

GREET Greenhouse gases, Regulated Emissions, and Energy use in Technologies

GWP Global Warming Potential

LCA Life Cycle Assessment

LCI Life Cycle Inventory

LCIA Life Cycle Impact Assessment

LFP Lithium Iron Phosphate

LIB Lithium-ion Battery

NMC Nickel Manganese Cobalt

SOC State Of Charge

SOH State Of Health

1

Introduction

1.1 Background

Global society and our planet face many challenges, including climate change, air pollution, finite resources to support growing population, well-being and equity. While many candidate technologies could address the above-named challenges, renewable energy technologies, are typically identified as being among the least-cost approaches to energy sector transition (Heath et al. 2022). To achieve the climate goals, in addition to energy sector, decarbonisation of mobility system plays an important part. Road transportation across the mobility system is by far the biggest Greenhouse Gas (GHG) emitter. Therefore, electrifying cars, trucks, buses and other vehicles can considerably reduce emissions from transportation activity (Hannon et al. 2022). Consequently, the market for Electric Vehicles (EVs), both light and heavy duty vehicles, is increasing with the promise to meet net zero.

Energy storage technologies and among which batteries, are pivotal for the transition in energy and transportation sectors. On the one hand, with an increased share of renewable energy, there is a need for variability control. A solution to the variability problem is to have energy storage connected to the grid. Thereby, there is a need for storage and batteries specifically suited for stationary applications. On the other hand, Lithium-Ion Batteries (LIBs) are the state-of-the-art electrochemical energy storage technology for mobile electronic devices and electric vehicles. As the result, energy storage capacity, mostly from batteries, are projected to increase exponentially in different scenarios (BloombergNEF 2022).

Given the latest electrochemical energy storage technology, LIBs degrade significantly with service life cycles. With the current increase in the adoption of EVs, a large volume of retired LIB packs, which can no longer provide satisfactory performance to power an EV, will soon appear on the market (Zhu et al. 2021). Predictions of future rises in EV numbers suggest that by 2030, between 100 and 200 GWh will be available through batteries nearing their retirement age as they become unable to fulfil the specified requirements for EV use (Rahil et al. 2022). Battery deployment to this scale raises social concerns and has implications since it poses problems such as material supply risk, End of Life (EoL) management issues and environmental impacts across the life cycle, as well as economic costs to both consumers and society at large (Heath et al. 2022). However, retired battery stock has also been viewed as a possible resource that can be utilised to provide value in the energy and transportation sectors (Rahil et al. 2022).

The NMC chemistry is the most used battery technology in Europe, the United States and Japan (Dai et al. 2019). As NMCs are foreseen to accumulate and reach their EoL in large numbers by 2030, Volvo Group has been involved in pilot programs with different storage solutions for their worn out bus and truck NMCs to investigate the potential for use of those batteries and give them a second life before reaching their EoL. Notwithstanding the uncertainties of the ageing mechanism of batteries and the performance of their remaining capacity after the first life, Battery Energy Storage System (BESS) applications such as peak shaving and short and midterm storage seem suitable for a second life (Rahil et al. 2022). However, different reference capacity tests should be performed to check the State Of Health (SOH), State Of Charge (SOC) and battery durability for different grid applications to determine whether spent batteries would satisfy such applications in technical terms. While various EoL options such as recycling and recovery methods for spent batteries are under development (Zhu et al. 2021), in this study and in collaboration with Volvo Energy, the viability of second-life NMC for BESS, from environmental and resource intensity perspectives will be evaluated.

1.2 Aim and Research Questions

The aim of this study is to analyse the viability of a second life use of NMCs for BESS application.

The following research questions will be answered in the study:

- In terms of environmental impact, how do the second life NMC battery compare to other batteries for BESS applications?
- Is it beneficial to delay the recycling of the NMC battery by adding a second life?

1.3 Scope and Limitations

In order to assess the viability of the second life NMC, a reference point was needed. LFP was chosen for this purpose since improvements in its gravimetric energy density along with considerable cycle life, and relatively lower cost and high safety are a benefit for BESS. Moreover, China's investments on this technology as well as its production disconnect to issues surrounding the supply of cobalt and nickel makes LFP the most likely alternative choice in the BESS applications (Thakore 2020).

To analyse the environmental impact, the following impact categories were chosen;

- Energy use measured as electricity requirements,

- Climate change measured as GWP, and
- Crustal scarcity measured as CSI.

Furthermore, it is of interest to assess to whom the environmental impact and/or benefit is attributed to. Thus, this will be analysed firstly from the perspective of the electric utility company or BESS owner and/or operator and secondly from a system perspective. The former perspective includes the comparison between the second life NMC and the first life LFP in the BESS application. The system perspective however analyses the entire life cycle of the NMC battery, excluding the EV use, and the effects of delaying the EoL. Therefore, in addition to energy use and climate change impacts, the resource risks of lithium-ion battery metals that are at supply risk will also be discussed.

The considered time scope for this study is 2030 to 2040 when the substantial amounts of retired NMCs are expected to be available and when the logistics and facilities of recycling of the batteries are in operation. Thus, the business case of recycling LIBs are assumed to already be in place. Connected to this, the European Union (EU) regulations for 2030 onward regarding recycling is taken into account. Volvo Energy is considering the second life market of NMCs in the EU region currently as this being the first market for their electromobiliy vehicle products. Thus, the second life is assumed to be situated in the EU. However, the geographical scope includes a more global perspective since the supply chain of batteries is distributed across the globe.

2

The Lives of Batteries

2.1 Studied Battery Technologies

This study focuses on the NMC and LFP battery chemistries. These are two of the most used battery technologies for EVs today and among the most studied LIB chemistries (Aichberger and Jungmeier 2020). Yet, reflecting the higher market share of NMC (Mackenzie 2021), the research and publication count for NMC-based LIBs is higher than for LFP, or any other LIB chemistry. In the west, NMC is the most commonly used battery technology for EVs with an average energy density of 142 Wh/kg. LFP is the most common battery chemistry in China and has a lower average energy density of 116 Wh/kg (Emilsson and Dahllöf 2019; Quan et al. 2022). Thus, they are less desired in applications where light-weight along with high energy density technologies are required, such as EV applications. For stationary applications, this is for obvious reasons not as crucial. Lastly, it is important to emphasise that due to considerable recent investments on LIB technologies, improvements in gravimetric and volumetric energy densities are expected to occur. Thus, current available energy densities are moving targets which will excel over time (Doerrer et al. 2021; Gröger et al. 2015; Wikner and Thiringer 2022).

2.2 Battery Production

LIB manufacturers have different cell designs including cylindrical (e.g., Panasonic designed for Tesla), pouch (e.g., LG Chem, A123 Systems, and SK innovation), and prismatic (e.g., Samsung SDI and CATL). The cell manufacturing processes are very similar. An overview of the supply chain and processing steps of NMC battery production is shown in figure 2.1.

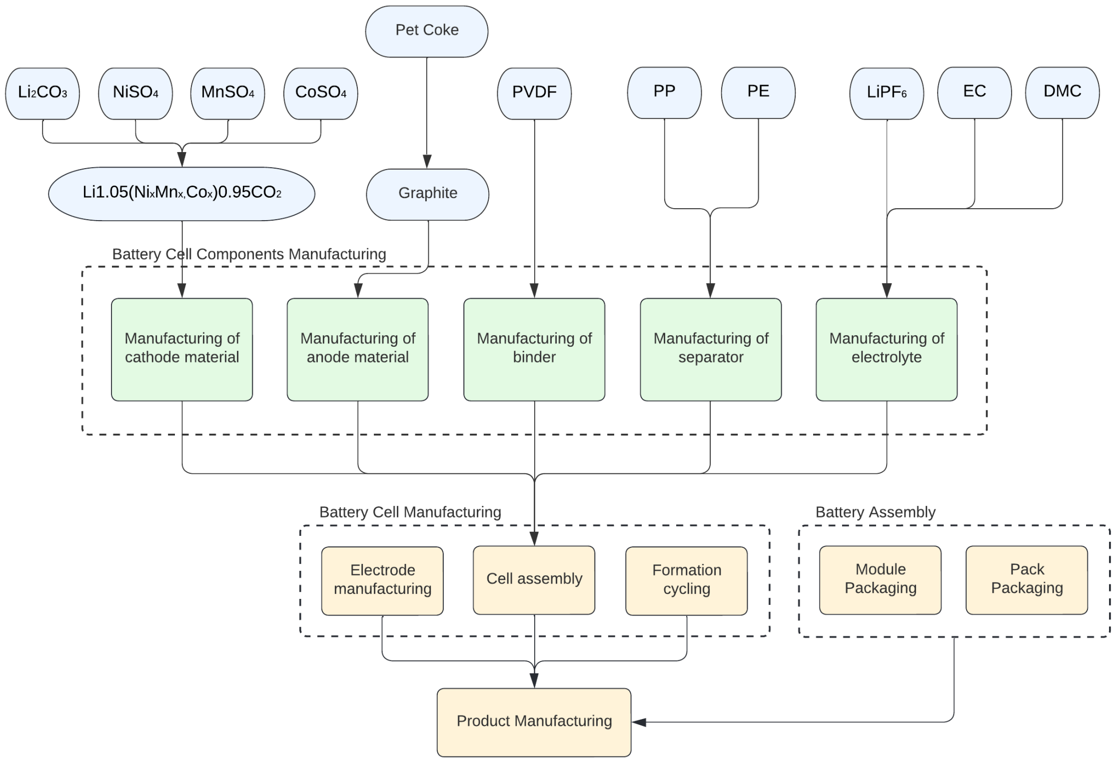


Figure 2.1: NMC Battery production, based on Lewrén (2019).

According to Emilsson and Dahllöf (2019), the impacts of LIB production are divided in the following categories:

- Mining and Refining
- Battery Material Production
- Cell Production and Battery Pack Assembly

Mining and refining often occur in separate locations and the material refining for one material can be executed in several smaller refining steps (Dai et al. 2018). Cell production occurs in a laboratory facility that needs strict controls on features like temperature, humidity and cleanliness. Battery pack assembly can be executed by the cell manufacturer or the battery pack components can be assembled by the automobile manufacturers (Ellingsen et al. 2017). Pack assembly does not have the same stringent requirements as cell production as the most sensitive parts have already been sealed in the cell production step (Ellingsen et al. 2014; Dai et al. 2019). Since different steps can occur in different facilities or locations, the choice of the local energy mixes for each processing step will affect the resulting GWP (Emilsson and Dahllöf 2019). Battery cell, battery module and battery pack are explained in the following sections.

2.2.1 Battery Cell

The four main components of a battery cell are: cathode, anode, electrolyte, and separator (Zubi et al. 2018). The properties of the battery can be configured by different combinations of cathode and anode materials (Romare and Dahllöf 2017). For cell manufacturing, the active material, conductive additive, and binder are mixed to form a uniform slurry with the solvent. The mix is then pumped into a slot die, coated on both sides of the current collectors which are aluminium foil for cathode and Copper foil for the anode, and then delivered to drying equipment to evaporate the solvent (Ahmed et al. 2016). The following calendering process adjusts the physical properties including conductivity, density and porosity of the electrodes.

The finished electrodes are stamped and slitted to fit the cell design regarding the required dimensions. The electrodes are then sent to the vacuum oven to remove the remaining water. The moisture level of the electrodes will be checked after drying as it is crucial to ensure that side reaction and corrosion in the cell are restrained. Once electrodes are prepared, they are sent to the dry room with dried separators for cell production. The electrodes and separator are stacked layer by layer to form the internal components of a cell. The aluminium and copper tabs are welded on the cathode and anode current collector, respectively. The cell stack is then sent for enclosure where the cell is filled with electrolyte before the final sealing and to complete the production. So far, the process for cell production does not have a consistent standard between different manufacturers and each has their preference based on a facility's operation or design and purpose of the cells.

Lastly, the formation and ageing process starts after cell production and before the use phase. It essentially consists of charging the cells to a relatively low voltage to protect the current collector from corrosion, followed by a rest session for electrolyte wetting. Consequently, the cells are stored on the ageing shelves to complete electrolyte wetting and Solid Electrolyte Interphase (SEI) stabilisation (Liu et al. 2021).

2.2.2 Battery Module and Pack

Battery modules allow for management of a smaller number of cells within a larger pack and protect them by a cassette which consists of an outer and inner frame (Ellingsen et al. 2014). While the purpose of having a battery pack is to be able to control certain units that are usually too large and complex to be handled only by cells or modules. Battery packs consist of the following four main components: battery cells contained in modules, battery management system (BMS), cooling system and packaging (Ellingsen et al. 2014). The BMS controls and monitors the battery whereas the cooling system maintains the temperature in certain limits.

Battery pack assembly is the assembly of the cells with other components. Dai et al. (2019) found that pack assembly was done manually in the factory they visited in China and they also noted that any energy used in the assembly step would be trivial compared to the energy used for the cell manufacturing and particularly in dry rooms (Dai et al. 2017).

2.3 Battery Energy Storage Systems

All over the world, the demand for energy capacity is increasing. At the same time, the need to reduce dependency on fossil fuels stresses the importance of increasing the share of renewables, like solar and wind in energy supply. However, due to the intermittent nature of renewable energy, stabilising factors are crucial to fulfil the new requirements of a grid system which faces new challenges like load imbalance problems with an increasing implementation of renewable energy sources (Kang et al. 2022). To address such imbalance challenges, the energy system needs to be accompanied by a BESS which can be used for charging during excess generation and discharged when needed (Abdi et al. 2017). Therefore, batteries play a crucial role in offering solutions for a more flexible and stable power system. Once LIBs are retired from their first use with around 80% remaining capacity, though insufficient for automotive application, they can be reused in stationary applications which have a much lower capacity limit and hence, are less demanding applications (Hein et al. 2012). Therefore, a large potential exists in utilising these battery systems for different applications in BESS.

2.3.1 BESS Applications

There are numerous studies on the several ageing mechanisms of LIBs. The ageing process that a battery undergoes and for how long it can sustain its performance under specific conditions are central in choosing the appropriate applications for which the battery can be used.

Factors such as calendar ageing, C-rate[1], DoD[2], SOC[3], temperature and voltage has been proven to impact the life of LIBs. Since these are factors influenced by the application, it strongly suggests that the application has a high impact on the life of a battery. Canals Casals et al. (2019) studied the estimated life remaining for second life NMC batteries in four stationary applications. These applications included self consumption, EV fast charge, self consumption with area regulation and transmission deferral. Supporting EV fast charge is a form of peak shaving since fast charging require a vast amount of energy over a short time. Thus, it requires high power that the transmission might have a problem to provide. To study these

[1]C-rate expresses the rate at which a battery is charged or discharged relative to its maximum capacity (MIT Electric Vehicle Team 2008).

[2]Depth of Discharge (DoD) expresses the percentage of capacity of which a battery has been used relative to its maximum capacity (MIT Electric Vehicle Team 2008).

[3]State Of Charge (SOC) expresses the current battery capacity as a percentage of maximum capacity (MIT Electric Vehicle Team 2008).

applications, the authors take advantage of an equivalent electric battery-ageing model that simulates the battery capacity fade through its use. They concluded that the batteries age significantly slower and consequently, the battery is more reliable in applications like fast EV charge support followed by BESS applications such as self consumption as opposed to grid-oriented services. Therefore, area regulation and transmission deferral are not recommended for second life usage since the predicted lifespan of battery is significantly shorter (Canals Casals et al. 2019).

Wikner and Thiringer (2018) further concluded that the SOC can have an impact on the calendar ageing. The authors analysed the impact of ageing when using various SOC levels. When accounting for the calendar ageing as well, this proved to be a large part of the total ageing (Wikner and Thiringer 2018).

Several studies confirm the technical and economic feasibility of using second life LIBs for stationary energy storage applications like intermittent renewable storage for self consumption or grid connected storage, area regulation, grid support and frequency regulation, peak shaving and power back-up (Heymans et al. 2014; Rahil et al. 2022; Canals Casals et al. 2019). Rahil et al. (2022) concluded that second life LIBs fully fulfil the requirements of peak shaving and also partly fulfil the requirements of frequency regulation services. However, BloombergNEF (2022) does not recommend the use of second life batteries for frequency regulation or energy shifting applications due to the non-linear capacity fading, an increase in internal resistance, low C-rate capability and short cycle life, making a second life use for these applications less reliable (Rahil et al. 2022).

According to Martinez-Laserna et al. (2018), since the ageing of retired EV batteries is still being evaluated, they are best suited for applications where characteristics such as reliability and safety are not of utmost importance (Martinez-Laserna et al. 2018). As many of the interviewees indicate, cell chemistry and design for power optimal vs. energy optimal performances fundamentally determine the expected life cycle of a battery and therefore affect the second life opportunities of used batteries. Residual cycle life, C-rate, recycling profitability, residual capacity, retention rate and safety were the factors the interviewees touched upon as heavy impact factors (Wikner and Thiringer 2022; Frith 2022; Campagnol 2022).

2.4 End of Life

In general, depending on battery ageing and application requirements, the life span of LIBs is 3–10 years (Yang et al. 2021). After EoL LIBs are collected, they can be evaluated for their potential for refurbishing, repurposing, and remanufacturing. In refurbishing, the LIB is collected, restored to its original functioning condition, and then used in its original application (Green Car Reports 2018).

Repurposing is when the energy storage capability of the LIB is restored through a series of steps at the end of life so that the LIB can be reused in another alternative application. For example, an EV LIB can be repurposed for stationary storage applications (Ahmadi et al. 2015; White et al. 2020; Heath et al. 2022). The series of steps involved in repurposing include collection at specific locations; presorting depending on battery chemistry, design, ageing and damage; disassembly and testing for degradation; performance assessment based on charge/discharge measurements; and classification for suitable second life applications leading to reassembly, and certification (Heath et al. 2022). If an evaluated LIB at this stage does not meet qualifications for refurbishing, repurposing, or remanufacturing, the next preferred Circular Economy (CE) strategy would be recycling. A proposed framework as a foundation for the development of a future circularity and recycling infrastructures by Neumann et al. (2022) is shown in 2.2.

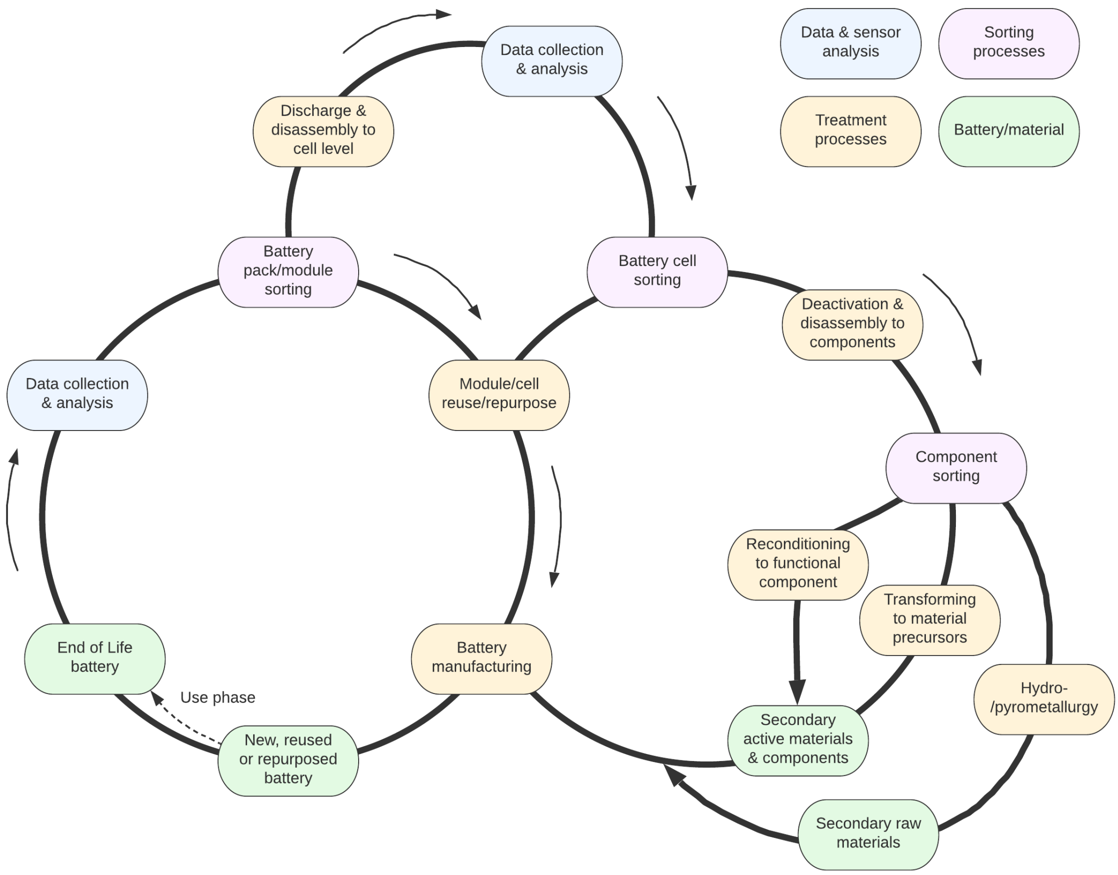


Figure 2.2: A proposed framework for the development of a future circularity and recycling infrastructures reproduced from Neumann et al. (2022).

There are several second life battery projects at pilot stage today with companies across the supply chain testing battery reuse. Nissan, for example, opened up 4R Energy Corp before the deployment of their LEAF model in 2010 with the goal to “refabricate, recycle, resell and reuse” Nissan EV batteries after their initial use. The health of the retired batteries are assessed and graded when entering the factory. Grade ’A’ is given to batteries which operate as new and can be used for new EVs. Slightly more worn batteries are given the grade ’B’ and can be used for stationary storage applications such as peak shaving or for industrial machinery. Batteries with the 'C' grade are e.g. used as backup systems during power outages in for example grocery stores (Nissan 2021).

2.5 Recycling

Due to many flammable organic and toxic substances in the spent LIBs, landfill of the worn batteries pose significant threat to human health and the environment. At the same time, spent LIBs contain many valuable metals such as cobalt, copper, aluminium, nickel and lithium. Recovering metal values from spent LIBs can not only reduce pollutants, but also supplement the metal sources, thus mitigating resource constraints and supply risks. It is generally acknowledged that recycling of spent LIBs is critical for the sustainable development of the LIB industry (Yang et al. 2021).

Recycling will thus become even more important in the future as the batteries produced today will eventually reach their EoL. When they do, it will become a higher priority to take responsibility from their resource flows. Recycling consists of three major steps in which there may be several smaller steps. The major steps are pretreatment, metal extraction and product preparation. The appropriate recycling infrastructure is not yet in place and it is currently under development. Thereby, it is difficult to find primary data and empirical research (Emilsson and Dahllöf 2019).

2.5.1 Recycling Technologies

Widely used LIB recycling methods (both in open and closed-loop applications) are hydrometallurgical, pyrometallurgical, and direct recycling (Chen et al. 2020). Although numerous technologies have been proposed to recycle spent LIBs, only limited number of processes have been in commercial operation. The products from hydrometallurgical processes are often metal sulfates (e.g. nickel, cobalt, manganese sulfates) or regenerated cathode materials or precursors. The products from pyrometallurgical process are usually metal alloys, such as Co-Cu alloy, Ni-Co-Cu alloy. Some companies further use hydrometallurgical process to refine those alloys (Yang et al. 2021).

With the exception of pyrometallurgical processing for certain recyclers (like Umicore), LIB recycling requires a common first step of mechanical preprocessing which entails sieving and crushing. In this first step, the LIB is crushed and further reduced in size, resulting in a mixture which consists of a coarse fraction (steel casing, plastics, metal foils) and a fine product called black mass. The latter consists of electrode materials (metal oxides) and carbon. Thereon, variations in properties such as ferromagnetism, density, and hydrophobicity are leveraged to separate the metals which black mass contains (Wang et al. 2016).

New recycling facilities being built today mainly use one of two technologies: hydrometallurgy or mixed pyrometallurgy-hydrometallurgy (BloombergNEF, 2021). Improving the recycling technologies of LIBs is a continuous effort and recycling is still far from maturity today. The complexity of lithium ion batteries with varying active and inactive material chemistry, designs and properties, interfere with the desire to establish one robust recycling procedure that is operational for all kinds of LIBs. Therefore, the current state of the art needs to be analysed, improved, and adapted for the variety of coming cell chemistries and components (Neumann et al. 2022). The recycling methods described in the literature use the processes which are described as follows.

2.5.2 Hydrometallurgical Recycling

Hydrometallurgical recycling uses low temperature chemical processes such as leaching, precipitation, ion-exchange, solvent extraction, and electrolysis to separate, recover and purify the metals from the black mass. Leaching is the critical step for metal value recovery in hydrometallurgical processes. The leaching methods include acid leaching, ammonia leaching, electrochemical leaching and bio-leaching. High metal recovery efficiencies are achievable via all these methods (Yang et al. 2021).

2.5.3 Pyrometallurgical Recycling

Pyrometallurgical recycling uses furnace- or smelter-based high-temperature processes such as incineration, calcination, pyrolysis, roasting, and smelting to separate and recover the metals in EoL LIBs. Pre-processing is optional for certain recyclers using pyrometallurgical methods for instance when the whole LIB is fed into a high-temperature furnace. In pyrometallurgical methods, the electrolyte and the organic materials including the separator and the plastics are combusted, which in turn provide energy for the process. Co, Ni, Cu and Fe are reduced and recovered in a residue called matte. Al and Li are typically oxidised, separated as slag, and then recovered through additional processing. Pyrometallurgical recycling requires subsequent hydrometallurgical processes to purify the metals which are available in the matte (Yang et al. 2021).

2.5.4 Direct Recycling

Direct recycling focuses on the recovery and enhancement of the Cathode Active Materials (CAM), which are subsequently used in the manufacturing of LIB cathodes. In contrast to hydrometallurgical methods, which dissolve the CAM into a solution, direct recycling maintains the morphology of crystals. The key processes in direct recycling are: obtaining the black mass, separating CAM from the rest of other materials via thermal and floatation processes, overcoming the PVDF binder to delaminate the CAM from the cathode, and finally, regeneration of the degraded CAM through relithiation (Ji et al. 2021). Unlike hydrometallurgical and pyrometallurgical processes, regenerated CAM from direct recycling can be used immediately in manufacturing new LIB. No further purification steps are needed which makes direct recycling an optimal choice regarding energy demand. Yet, lack of standardisation in battery cell and pack design and manufacturing is a significant obstacle for direct recycling implementation (Yang et al. 2021).

2.5.5 Recycling Policies

Battery recycling is encouraged by the legislation through different directives, mainly because of risks to human health or the environment deriving from hazardous battery constituents. In the EU, present regulations include the Battery Directive (Directive 2006/66/EC) and the Waste Electrical and Electronic Equipment Directive (Directive 2012/19/EU). These policies include a physical and financial Extended Producer Responsibility (EPR). Member countries are required to set up collection schemes for EoL portable batteries in the form of collection points located in the proximity of end-users. The costs for collection, treatment, recycling, and disposal must be financed by the battery producers (Neumann et al. 2022).

As part of the European Green Deal, in 2020 a legislative proposal was submitted by the European Commission to replace the 2006 Battery Directive. The proposed regulations considerably exceed previous legislation in many respects and are intended to support the development of the EU toward a modern, resource-efficient, and competitive economy. Accordingly, new collection targets for waste portable batteries are 45% by 2023, 65% by 2025, and 70% by 2030. Moreover, target material recovery rates of 95 % for cobalt, 95% for copper, 95% for lead, 95% for nickel, and 70% for lithium by 2030 have been defined (European Commission 2020). Further requirements include extended battery labelling, a battery passport for batteries with capacities above 2 kWh, minimum contents of recycled materials in new industrial and automotive batteries, minimum performance and durability requirements, and more (Neumann et al. 2022).

2.5.6 Recycling Economics

A major motivation for battery recycling is to reduce the adverse environmental impacts from the disposal of LIB wastes. For strategically important materials, recycling is also to secure their supply and alleviate price fluctuations and market control. To establish recycling at a commercial scale, however, the operation must ultimately be economically viable (Mayyas et al. 2019). The operating costs of recycling can be broken down into two broad categories, namely, processing costs and collection including transport costs (Yang et al. 2021). According to BloombergNEF (2022), the cost of transporting used battery packs can in some regions challenge the economic feasibility of recycling. Therefore, integrated supply chains can play a crucial role to reduce costs, where packs are collected and disassembled locally prior to sending the valuable material or the black mass for further processing.

Moreover, not all processes are economically viable for the recycling of any battery type. Due to the high investment costs, pyrometallurgical treatment, for example, is primarily suitable for the recycling of batteries with high cobalt and nickel content like NMC batteries. Lithium and aluminium end up in a slag and can only be recovered with considerable additional effort and does not provide an economic incentive. Hydrometallurgy, on the other hand, also enables the recovery of lithium while is applicable for different cathode types. However, due to the low intrinsic material value of LFP batteries, neither of the processes would be economically feasible to recover valuable parts from this battery type (Neumann et al. 2022).

Therefore, LFP batteries require a separate recycling process to those used for nickel-based chemistries as they don't contain the same high-value materials. The only valuable materials in LFP batteries are lithium and copper. Today, LFP batteries are very seldom recycled but due to increased regulations within the European Union and the prospect of a higher pressure on lithium market, it is likely that LFP batteries will be recycled in the future.

Lastly, due to uncertainty around recycling economics, many automakers have to pay recyclers to cover recycling costs and provide them with a profit (Neumann et al. 2022). Yet, as the economics of recycling improves, these fees are expected to become obsolete and result in revenue instead. Therefore, as the head of battery recycling at Volvo Energy indicated, the outlook of a cost-benefit trend shift is anticipated (Chabanne 2022).

2.6 Life Cycle Assessments of Batteries

Even though LIBs have been studied for a significant amount of time since their central role in the advent of EVs, there are discrepancies in the result of LCA studies made so far. According to Chordia et al. (2021), there are limited sources of primary data covering key aspects of LIB production and thus a high interdependency in the studies in terms of Life Cycle Inventory (LCI) data. Several studies published in the LCA literature on LIBs, present results based on the publicly available model called 'Greenhouse gases, Regulated Emissions, and Energy use in Technologies (GREET)'. This implies that several LCA studies using the GREET model (Dai et al. (2019), Yuan et al. (2017), Kelly et al. (2020), Raugei and Winfield (2018), and Wang et al. (2016)), so represent the same production facilities (and technical scopes) and rely on similar methodological assumptions.

Another commonly used data source is the Ecoinvent database, which provides datasets for modelling background processes for a wide variety of technologies and processes, including, but not specific to LIB production. However, current Ecoinvent datasets for LIB production are not representative of upcoming cathode chemistries and novel production developments (Chordia et al. 2021). Therefore, some parts of the knowledge is still in its infancy as new innovations are exponentially on the rise both in terms of battery design and chemistry, Giga-factories and production methods, EoL management, recycling and logistics that require more research.

Most of the recent studies made on LIBs has been with focus on the NMC chemistry, due to its prevalence in the western world Dai et al. (2019). Studies on LFP can be found but are fewer. Due to differences in the cycle life of LIBs of different chemistries, inclusion of the use phase in the LCA studies makes significant differences in the impact outcomes. This result underscores the importance of considering cycle life of LIBs in their environmental performances (Yang et al. 2021). In addition, the economic and environmental impacts of using LIBs to provide a wide range of grid services in second-life applications currently remain unquantified (Pellow et al. 2020). Regarding the EoL management, significant research efforts have been devoted to LIB recycling and recovery of metal values from the spent batteries. The focus, however, is predominantly on the development of technologies for materials recovery with limited attention to economic and environmental impact analyses (Yang et al. 2021).

2.7 Energy Demand

In the following section, some of the studies on the energy demand in the battery production and the recycling will be compiled and discussed.

2.7.1 Energy Demand in Battery Production

Process energy demand for LIB manufacturing has been identified as an environmental hotspot in many battery LCA studies. However, reported energy consumption for LIB manufacturing is based on engineering calculations or pilot-scale battery manufacturing facilities (Ellingsen et al. 2014), and therefore does not necessarily represent the actual energy consumption of the LIB industry. Moreover, there are discrepancies in assessment results regarding energy demand in battery production. This variation in assessment results can be explained by diverging technical scopes, the lack of representative data for key parameters such as battery lifetime, energy density and energy demand in cell production in the LCA studies (Chordia et al. 2021). Even so, in many articles, for proprietary reasons, LCI data for LIB manufacturing and assembly is not disclosed (Dai et al. 2019).

According to Chordia et al. (2021), the effect of upscaling LIB production can also impact the energy consumption and environmental impacts. Some equipment consumes about the same amount of energy regardless of the amount of materials going in or out, such as the over-dimensioned calcination kilns in the battery production plant reported by (Dai et al. 2019) and the dry-room (Dunn et al. 2014). Therefore, upscaling production reduced emissions by nearly 45% in their reference scenario due to a reduced energy demand in cell production (Chordia et al. 2021). To give an overview of energy demand in battery production, the three main studies often discussed and referred to by recent research and articles along with some of the latest studies and their results are compiled in table 2.1 and will be pointed out as follows.

Table 2.1: Previous studies on NMC and LFP batteries and their concluded energy use in the production phase. The studies indicated with grey are the main studies discussed and the ones indicated with blue are some of the latest LCAs in the field.

Reference	Direct energy use in LFP production [MJ/kWh]	Direct energy use in NMC production [MJ/kWh]	Comment
Dai et al. (2019)	-	30	Electricity in cell production
		140	Heat in cell production
Majeau-Bettez et al. (2011)	371 - 473	371 - 473	
Ellingsen et al. (2014)	-	960	Primary energy in cell production
Emilsson and Dahllöf (2019)	-	216	Cell and pack manufacture
		410	NMC powder production
		1127	Entire production
Carvalho et al. (2021)	1020	1951 - 2107	Entire production (of different NMC types)
Frith (2019)	105 - 378	460 - 500	Entire production

Dai et al. (2019) from Argonne National Laboratory is focused on the production phase of NMC batteries and uses industry data. For the cell and pack manufacturing as well as the NMC powder production, the LCI data has been gathered from one of the largest EV battery and NMC producers of the time.

Majeau-Bettez et al. (2011) uses data from the GREET model for the production phase of the batteries. Contrary to most studies on the topic, this study chose the functional unit 50 MJ energy stored but also provided two alternative functional units for the cradle-to-gate; nominal capacity in Wh and mass in kg. The data for the energy use in the manufacturing of the battery as well as the production of the virgin raw material is taken from Rydh and Sandén (2005). This data is for LIB in general and thus, it is assumed that the same amount of energy is used in the production of both LFP and NMC battery cells. However, cell density is accounted 0.110 and 0.140 kWh per kg battery, respectively (Majeau-Bettez et al. 2011).

Ellingsen et al. (2014) studied a NMC battery with primary data collected from Miljøbil Grenland, a smaller EV company. The cell density in this study is assumed to be 0.174 kWh per kg and the direct energy use in cell manufacture is significantly higher than any other reports.

Emilsson and Dahllöf (2019) focus on GHG emissions from the production of NMC for vehicles via an elaborate literature study on LCAs of lithium-ion batteries used in light-duty vehicles. They try to investigate the causes for the high energy usage and find a reliable estimate based on recent studies. The entire battery production is broken down into cell and pack manufacture, NMC powder production and upstream material acquisition (mining and refining). The reports indicates the central role of dry rooms and drying in high energy demand for battery cell production which is attributed to all LIB chemistries.

Carvalho et al. (2021) observed the need for an LCA on NMC and LFP batteries in Europe, especially Italy. Thus, the LCI data for the cell production is obtained from an Italian cell manufacturer. The study includes the EoL and subtracts the lesser need for virgin raw materials from the production. Disregarding the EoL, the NMC battery has a higher energy demand and environmental impact than the LFP battery. However, the NMC battery is at an advantage when considering the EoL due to the high impact of NMC powder (Carvalho et al. 2021).

Frith (2019) reports on LIB manufacturing energy use and emissions of different chemisty choices by using their proprietary model. The report indicates that synthesis of the CAM, requiring a high temperature, uses the most energy and this can vary across the manufacturing process and conditions used by 120%. The report then estimates relatively lower measures for LFP energy demand in manufacturing compare to NMC ones.

2.7.2 Energy Demand in Recycling

To the best of our knowledge, given that the recycling infrastructure is currently under development, valid and reliable data is not available in literature yet. Notwithstanding the difficulty of finding empirical research on recycling energy demand, it is anticipated that direct recycling would be the most energy efficient while the hydrometallurgical and pyrometallurgical methods are less energy efficient (Emilsson and Dahllöf 2019). Dai et al. (2019) reported new data on recycling for GREET based on Argonne's own research. This report indicates that pyrometallurgy has highest energy use, 1330 kWh/ton cells recycled, followed by 838 from direct recycling, 815 from hydrometallurgical with inorganic acid leaching and 2.20 from hydrometallurgical with organic acid leaching. There are more chemicals which are needed for leaching, however energy use for their production was not included in the indicated values. Also, European Commission (2019) - the European commission's document called as Product Environmental Footprint Category Rules (PEFCR battery study) - estimates that around twelve percent of the energy use and GHG emissions of a lithium-ion battery's lifetime occurs in the end of life stage in Europe.

2.8 Global Warming Potential

The climate change impact of LIBs measured as GWP varies significantly between different studies. Aichberger and Jungmeier (2020) compiled a literature review of 50

LCA studies and found the average greenhouse gas emissions for LIB production to be approximately 120 kg CO_2-eq/kWh. Ellingsen et al. (2017) have shown that there is a wide range of GWP results; from 38 to 356 kg CO_2-eq/kWh capacity. Due to differences in the cycle life of LIBs of different chemistries, inclusion of the use phase in the LCA studies makes significant differences in the impact outcomes. This result underscores the importance of considering cycle life of LIBs in their environmental performances (Yang et al. 2021). In addition, the environmental impacts of using LIBs to provide a wide range of grid services in second-life applications currently remain unquantified as understanding of battery ageing is yet insufficient (Pellow et al. 2020). Regarding the EoL, significant research efforts have been devoted to LIB recycling and recovery of metal values from the spent batteries. The focus, however, is predominantly on the development of technologies for materials recovery with limited attention on environmental impact analyses (Yang et al. 2021).

According to Nordelöf et al. (2014), variations of the GHG intensity of the electricity mixes explain 70% of the variability of LCA results. Regarding the climate change impact of LIB production, LCA literature lacks an actual understanding of how environmental burdens have changed over time. This, according to Chordia et al. (2021), is partly related to a transition to large-scale production which is not reflected properly in the LCA studies. Furthermore, the emissions would reduce by 55% if the energy is sourced from a low-carbon source (Swedish electricity mix) compared with their base scenario (South Korean electricity mix) in battery production, shifting almost all burden to upstream supply chains (Chordia et al. 2021). That being said, supply chains with high shares of renewable energy can also lower the GHG emissions from the upstream material production phase (Kelly et al. 2020).

2.9 Material Constraints

2.9.1 Battery Supply Chain Risks

With increasing green technology deployment, the demand for and dependence on certain minerals is projected to increase. According to IRENA and Economic and Social Affairs (DESA) (2019), the energy transition will have “major social, economic and political implications which go beyond energy sector” and “redraw the geopolitical map of the 21 st century”. Kalantzakos (2020) highlights the economic importance and strategic significance of certain metals. Moreover, according to Månberger and Johansson (2019), the supply of such metals have geopolitical implications as a result of geographical concentration of reserves, the existence of strategic supply chain bottlenecks, the import dependence of several governments, and potential for internal tensions and vulnerabilities. Overland et al. (2019) highlights location and transportation routes which hold a key role in relying on a robust supply chain for batteries. Besides, the extractions of some of these metals have more steps than others, which adds up to even greater supply risks, albeit as temporary deficits (Emilsson and Dahllöf 2019).

Kushnir and Sandén (2012) argue how the rate of extraction which is needed to build up a large societal stock over a given time period results in material constraints that are then projected for LIBs. As historical examples indicate, institutional inefficiency can be a major mechanism driving transitory scarcity. Limited resources and the rise of raw material prices have therefore become a bottleneck problem which directly threatens the sustainable development of the LIB industry (Yang et al. 2021). Thus, geopolitical supply risk especially in case of critical resources as well as material scarcity, market structure of producers and time lags in supply system response (Kushnir and Sandén 2012) should be taken into account while devising and implementing strategies for battery supply chain.

Six countries (Australia, Chile, Democratic Republic of the Congo (DRC), China, Brazil and Russia) together hold a large share of cobalt (66%), copper (33%), lithium (84%), nickel (52%), rare earths (70%) reserves. These countries are situated in different continents and are heterogeneous regarding the level of economic development and political systems and priorities (Månberger and Johansson 2019). Kalantzakos (2020) highlights the Chinese stronghold on rare earths, with majority of the world's material production (97%) and processing of them, leading to Chinese leverage power.

The world supply distribution of certain metals can thus affect the whole battery supply chain, especially when they are difficult or impossible to substitute with other materials. Recycling batteries is one method to increase the supply of battery metals that does not involve sourcing virgin metals (Emilsson and Dahllöf 2019). However, recycling is not the only way to address material supply challenges even if it is the most obvious EoL solution. A broader conception of such solutions is posited as the CE. According to Heath et al. (2022) three CE strategies including closed-loop recycling, open-loop recycling, and re-manufacturing, bring materials from the EoL phase of LIBs back into the manufacturing phase. Yet, more information on the supply risks of different metals is needed, as well as traceability of the metals in the existing batteries, so that sustainable production can eventually be achieved and guaranteed. In the following sections, some metals of the highest significance will be discussed.

2.9.2 Cobalt

According to Kalantzakos 2020, Cobalt is a key resource used in electric vehicles and thus a crucial part of the energy transition. LIB demand for cobalt is expected to be approximately 50% greater than all current supply by 2025 (Campagnol et al. 2018). Driven mainly by strong demand from consumers in the rechargeable battery, average annual cobalt prices increased in 2018 due to limited availability of cobalt metal (Yang et al. 2021). So increasing prices compared to the existing levels can be expected in face of supply risks.

Like all critical materials, cobalt is geographically constrained to only a few global locations with the DRC having the largest reserves (3.6 million tons) which accounts for almost over 60% of global cobalt production. The DRC is among the poorest African countries and is economically and politically unstable. The production of cobalt thus has significant social, environmental, and economic costs and many reports criticise unethical mining practices in the DRC (Kalantzakos 2020).

Out of all critical materials, cobalt is the most likely to be 'so called conflict material'. Månberger and Johansson (2019) argue, because it does not need industrial separation, can therefore be easily looted by militant groups which in turn provides them with revenue and results in destabilising the DRC economically and politically. Moreover, due to the criticality of cobalt, importing countries will try to gain influence over the DRC's cobalt production and exports. Kalantzakos (2020) remarks on efforts on the part of China for instance to gain leverage e.g., by forming a union of mining companies. Apart from the internal and geopolitical issues that might threaten global cobalt supply, cobalt is mainly produced as a by-product of nickel and copper. This associates cobalt production to market dynamics of these materials and thus turns it into the most unstable and unsteady market which can lead to cobalt supply shortages especially in the short term (Pozybill 2022).

2.9.3 Nickel

Average nickel content is expected to increase and cobalt content to decrease in newer NMC batteries as the industry trend is to move towards higher energy density (towards NMC622 and NMC811 and away from NMC111). Therefore, apart from cobalt which is at supply risk, nickel may become at risk too (Emilsson and Dahllöf 2019). Projected LIB growth by 2025 is expected to drive the demand for nickel to levels greater than current supply, and above the projected supply increases (Campagnol et al. 2018). Moreover, according to Lazzaro (2022), nickel prices may significantly influence the relative price competitiveness between next generation NMC batteries and the other alternative LIB batteries like LFP given the projected nickel price sensitivity of the battery industry.

2.9.4 Lithium

In 2018, worldwide lithium production increased by an estimated 19% to 85,000 tons in response to increased lithium demand for battery productions (Yang et al. 2021). It is assumed that lithium may become a bottle neck metal for certain periods (Emilsson and Dahllöf 2019).

From the view point of supply, lithium is concentrated in the South American countries of Argentina, Bolivia, and Chile, 'the so-called lithium triangle' and can also be found in Australia and China (Kalantzakos 2020). The lithium triangle does not only possess the largest reserves of lithium (55%) but is also a region of social and political unrest due to inequality and environmental issues, thereof a threat to the lithium market. An uncertain variable in future lithium supply is Bolivian lithium production which is currently non-existent but could potentially have large influence on the lithium market due to Bolivia possessing the largest reserves. Interdependence between countries is then politically problematic and will cause competition and confrontation (Pozybill 2022; Kalantzakos 2020) which adds up to the uncertainty and risks of lithium supply.

2.9.5 Copper

Copper scarcity is expected to result in deteriorating ore quality, which will consequently lead to a higher gross energy requirement for copper production (Harmsen et al. 2013). DRC is the Africa's largest copper producer with exceptional 3% to 4% copper grades (Denina and Reid 2021). Cobalt together with copper makes up 80% of Congo's export revenue (Kalantzakos 2020). In 2015, big share of the copper-cobalt mining in the DRC was artisanal (Dai et al. 2018), meaning that the workers were not officially employed by a company or the state, thus child labour or unethical labour practices were involved (Emilsson and Dahllöf 2019). Copper is used as cell, module and pack component of different LIB batteries. In mechanical pre-treatment of recycling batteries, the main scope is to separate metallic particles (casing, copper and aluminium foils) and further concentrate the black mass (Neumann et al. 2022).

3

Theory & Method

As explained in the previous chapter, the battery can go through several stages in its life: production, first use phase, removal and repurposing, second use phase, collection and EoL. These assumptions are set in this study for the viability analysis of a second life use of NMCs for BESSs. To answer the research questions, the analysis is performed in reference to LFP battery by using a break-even analysis to find the number of cycles that a second life NMC should perform for the assigned BESS application in order to outcompete the LFP battery. Moreover, the CSI is used to assess the mineral resource scarcity and the impact of extending the use of NMC battery by adding a second use phase and consequently locking in the scarce metals by delaying the recycling. The use of LCAs studies, CSI and different assumptions incorporated into the model will be further explained in the following sections.

3.1 Life Cycle Assessment

LCA stands for Life Cycle Assessment and is a comprehensive method to analyse environmental impacts related to a product or a service. It consists of goal and scope definition, followed by inventory analysis, impact assessment (classification, characterisation and weighting) and interpretation (Baumann and Tillman 2004). The goal specification of an LCA study is a central part which shapes the scope. While in the scope definition, the object of the LCA is defined and described. In the inventory analysis, the data collection and modelling of the product system is conducted in line with the goal and scope definition. The LCI is typically the most time-consuming step requiring the highest efforts and resources. The data collection includes, for example, gathering information on energy and material flows between the processes along the product life cycle (Lewrén 2019). In a Life Cycle Impact Assessment (LCIA) the results on the environmental loads from the LCI are translated into potential environmental impacts (Baumann and Tillman 2004).

According to Arvidsson et al. (2018), LCA can be very useful for assessing emerging technologies and systems and guiding early development, but it has to be adapted to this purpose, giving rise to a particular type of LCA methodology which is called prospective LCA. An LCA is prospective when the emerging technology studied is in an early phase of its development like small-scale production, but the technology is modelled at a future and more-developed phase like a large-scale production.

Methodological choices in prospective LCA must be adapted to reflect the defined goal regarding the assessment of environmental impacts of emerging technologies. This deviates from the typical goals of conventional LCA studies.

Since an emerging technology needs to be modelled at some projected future point in time in order to portray the technology's environmental performance when it is produced and used on a relevant scale, a prospective LCA will always rely on generating scenarios. Predictive scenarios may be considered if there is a sound basis for predictions, and in some cases, status quo may serve as a valid proxy for assumptions representing the future (Arvidsson et al. 2018). Thus, as summarised in Table 3.1, predictive scenarios and scenario ranges are two general approaches to prospective inventory modelling of both foreground and background systems in order to avoid temporal mismatch.

Table 3.1: Summary of recommendations for conventional and prospective LCA reproduced from Arvidsson et al. (2018).

Aspect	Conventional LCA	Prospective LCA
Definition	System modelled at a current or near-by time	System modelled at a future time
Technology alternatives	Currently existing technologies are studied	Emerging technologies with relevance for the future are studied
Foreground system data	Current foreground system and production scale are modelled. Common data sources include: • Life cycle inventory databases • Previously conducted LCA studies	A future scenario of the foreground system and production scale is modelled. Valuable data sources include: • Scientific articles • Patents • Expert interviews • Unpublished results • Process simulations
Background system data	Current background system is modelled	Avoid temporal mismatch between foreground and background systems by using predictive scenarios and scenario ranges

In this study, the likely improvements in different battery life stages are considered in relation to the relevant time aspects by predictive scenarios. This has been supported by reviewing scientific articles and a variety of expert interviews from both academia and the industry. In the case of second life NMC battery to be available on 2030 onward, it is assumed that the production has taken place in early years of the current decade. Therefore, the results by the recent conventional LCA studies are used for the energy demand in NMC's production. Whereas in the case of LFP battery, since it is assumed to be produced in 2030, its production is considered to be further optimised in terms of its manufacturing process and technologies and thus improvements are expected in its energy use. Besides, LFP battery technology is assumed to be further enhanced so that its performance for BESS applications would last for a range containing relatively higher number of cycles.

Recycling is also considered to be more established in Europe with investments on hydrometallurgical method for NMC batteries. The role of the EU's directives and the use of scarce metals in NMC cathode motivate such investments. Thus, different recycling scenarios regarding different percentages of recycled material input for NMC battery production are incorporated into the model for prospective inventory modelling. However, given the current ecosystem of battery technologies and the industry outlook as well as the inclination of European market with respect to NMC battery as opposed to other LIB options, the recycling of LFP battery is unlikely to be available in due course in Europe. In addition, the significant role of China in LFP production and its global battery market share outlook add to the fact that relatively low economic incentives are speculated to exist in Europe to invest on LFP recycling infrastructure.

3.2 Break-even Analysis

In order to compare the LCA of second life NMC battery to LFP battery for BESS applications, break-even analysis is used regarding the impact categories energy demand and climate change. These impact categories are measured by electricity requirement and GWP, respectively. The incorporated scope of life cycles of NMCs and LFPs in the model is illustrated in figure 3.1.

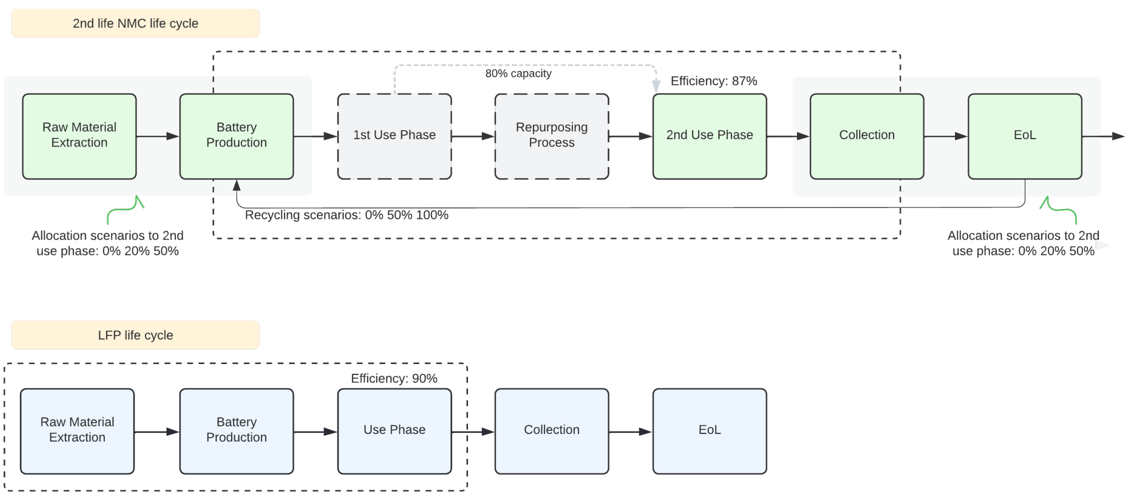


Figure 3.1: Scope of LCA of second life NMC and LFP batteries for break-even analysis in this study. The system boundaries are indicated with a dashed line and the grey boxes indicate that the processes are to some extent within the system boundary, representing the allocation of these processes.

A model is created for the break-even analysis to find the number of cycles a second-life NMC should last to outcompete a LFP battery for BESS application regarding the energy use impact category. For the climate change impact category however, the break-even analysis points out the local electricity mix for BESS application and recycling operation that a second-life NMC should be supplied with to outperform the LFP battery. Therefore, the conclusion is made regarding the viability of replacing a LFP BESS with a BESS consisting of second life NMCs. The break-even analysis is conducted and visualised in the software MATLAB by MathWorks. The break-even point is then compared to reasonable cycle numbers for NMC batteries according to previous research to analyse viability. The Break-even analysis regarding the impact categories energy demand and GWP is based on the functional unit and Equivalent Full Cycles (EFCs) of the batteries.

3.2.1 Functional Unit

To the best of our knowledge, most of the research, especially newer research, consider the functional unit kWh of capacity for LCAs on the production of LIBs and kWh delivered when considering the use phase (Dai et al. 2019; Quan et al. 2022; Emilsson and Dahllöf 2019). Additionally, studies with a focus on EoL and the recycling phase mainly use the functional unit kg EoL battery (Carvalho et al. 2021; Mohr et al. 2020; Dai et al. 2018; Dai et al. 2019; Quan et al. 2022). Therefore, the functional unit has been chosen as kWh battery capacity for the break-even analysis and the energy density of NMC.

3.2.2 Production

In this study, the energy use and GWP of mining and refining, battery material and cell production and battery pack assembly are being considered. The LFP production is assumed to have 70% of the energy use in the production of the battery pack compared to NMC batteries. This assumption is based on literature review and conducted interviews for this study. In addition to the comparison of the state of the art of the LIB production, the outlook of improvements in future battery production supports this view (Carvalho et al. 2021; Campagnol 2022; Frith 2022). In case of LFP, the fraction between material upstream acquisition and battery material and cell production is assumed to be equal to the one of the NMC battery. Energy demand in production is converted to kWh electricity requirement in the model.

3.2.3 First Use Phase

The environmental impact from the first use phase is in this study fully allocated to the EV usage and thus not accounted for. Once the first use phase ends, it is assumed by Volvo Energy that the battery would be removed at 80% capacity.

3.2.4 Repurposing

According to Quan et al. (2022), the repurposing process is unavoidable to proceed to the second use phase. However, Nicolo‘ Campagnol, Solutions manager of Battery Insights at McKinsey, stated that the batteries do not go through a repurposing process in practice since it is too costly (Campagnol 2022). Due to this uncertainty, the repurposing process is excluded from the study.

3.2.5 Second Use Phase

While generally all applications call for high energy and power density, low cost, and safety, the relative importance of these characteristics varies significantly depending on the specific requirements of each application (Bresser et al. 2020). The potential second use phase of the NMC batteries in this study is assumed to replace a LFP BESS in the following applications;

- Self consumption,
- Peak shaving.

These two applications were chosen on the basis that they require different properties of the batteries while they are appropriate options for battery reuse given that they don’t require frequent charging and discharging and are suitable for small-size systems as opposed to large grid-scale systems. Self consumption requires a more energy optimised battery whereas peak shaving mainly requires a power optimised battery. Thus, the two represent two ends of the spectrum of specifications regarding potential applications. For different BESS applications, it was mainly based on interviews to assume that the LFP battery as the reference point would probably last for 5000 to 7000 EFCs before its EoL (Wikner and Thiringer 2022). In the break-even analysis for GWP, second-life NMC is assumed to last between 2000 to 4000 cycles.

3.2.6 End of Life and Recycling

The NMCs are collected and further transported to a recycling facility after second use. The collection rate is assumed to be 100% since this is what to be aimed for. Hydrometallurgy has been assumed as the sole recycling method since this technology is expected to be the most likely case for the medium term time horizon.

The recycling is represented by recycled material input to the production phase of NMCs while for LFPs, the assumption is to exclude the recycling phase from the model since it is unlikely that by 2030, recycling becomes cost-efficient for this battery chemistry. Thus, only energy use in the production of NMC regarding the upstream material acquisition is affected by the share of recycled material input. In this study, a share of recycled material input on a range from 0% to 100% is evaluated. This is incorporated into the break-even analysis for energy use as three different scenarios represented by three graphs of 0%, 50% and 100% recycled material input. Thereby, it is easy to see the effects of recycling on the energy use in the life cycle of the NMC battery. 10% is however seen as the reasonable upper limit to recycling of NMC batteries by 2030 from Volvo Energy's perspective since production of new NMCs is expected to increase quite rapidly (Heath et al. 2022). Thus, 10% recycled material input is used in the break-even analysis for GWP.

3.2.7 Model

The energy loss for a battery of 1 kWh capacity per 1 EFC is

$$E_{loss} = 1 \text{ kWh} \cdot (1 - \eta),$$

where η is the charge-discharge efficiency. Thus, the energy use per cycle for using a second-life NMC battery with 1 kWh capacity i amount of cycles is

$$E_i = \left((E_{prod} + E_{rec}) \frac{a}{0.8} + iE_{loss} \right) / i,$$

where E_{prod} and E_{rec} is the energy use in production and recycling respectively and a is the allocation of the energy use to the second life. The energy of production and recycling is converted from MJ to kWh electricity. It is then divided by 0.8 to obtain the functional unit of 1 kWh capacity remaining in the second life as opposed to the initial capacity. This is compared with the energy use of a first life LFP battery

$$E_i = (E_{prod} + E_{rec} + iE_{loss}) / i, \tag{3.1}$$

where E_{prod} and E_{rec} is the energy use in production and recycling respectively, E_{rec} is considered zero in the model due to exclusion of recycling phase of LFP for reasons described above. η is the charge-discharge efficiency and i is 5000-7000 cycles depending on the BESS application. For NMC, i is set as a range for the energy use to analyse how many cycles are needed for the second life NMC battery to outperform a new LFP battery. The LFP battery is assumed to have a charge-discharge efficiency of 90%, which is the efficiency mostly used in LCA literature on LIBs (Quan et al. 2022; Lewrén 2019). Since the NMC battery is in its second life, its charge-discharge efficiency is assumed to be slightly lower, at 87%.

For the impact category GWP, the part of the electricity use in production phase that is not originated from the material acquisition and is related to cell and pack production is multiplied with a factor of 1 kg CO_2-eq/kWh electricity since the batteries used within the temporal system boundaries of the study are most likely produced in China or South Korea (Emilsson and Dahllöf 2019). For the use phase of both NMC and LFP batteries and the recycling of NMC, the electricity use is multiplied with a factor of 0-1 kg CO_2-eq/kWh electricity as the minimum and maximum values are the furthest extremes of different sources (Emilsson and Dahllöf 2019). With China's electricity mix representing the higher end which equates to 1 kg CO_2-eq/kWh electricity while the average electricity mix of the EU countries equates to 229 g CO_2-eq/kWh electricity (European Environment Agency 2022; Emilsson and Dahllöf 2019). Therefore, the carbon intensity of the electricity supply in the use phase and recycling affects the break-even point for GWP. This impact is visualised in a separate analysis.

3.2.8 Allocation Scenarios

The environmental impacts from the production of the NMC battery along with its recycling are allocated between the first and the second life. Depending on how the ownership of the battery changes over its lifetime, how BESS market values second-life batteries and from which viewpoint the environmental impacts are assessed, the most intuitive and apparently logical allocation method may vary. That said, the impact of the allocation method on the results could be significant. To that end, different allocation percentages representing three different scenarios used in this study are further explained. Each scenario can be of interest to different involved parties and stakeholders.

1. 0% allocation (system perspective): no environmental impact is allocated to the second life, representing extended life cycle via system expansion. Therefore, answering the question that to what extent the addition of a secondary reuse application can minimise the net environmental impacts across the battery life cycle. The system expansion explains the effect on society as a whole if using the second life NMC batteries for BESS as a substitute for LFPs and thereby avoiding the emission of a whole life cycle of newly made LFPs. This viewpoint is particularly valid when a second-life battery adoption could be a possibility, and thus the potential of a market formation while it has not been intended beforehand.

2. 20% allocation (baseline scenario): as the second-life BESS market is expected to be in place by 2030 due to the availability of retired batteries, an economic value is expected for second-life NMCs. In this scenario, 20% of the production along with recycling is allocated to the second life since it shows a sensible point in the range (0% to 50% allocation). Besides, this lower-mid point in the range can be reasonably attributed to the battery performance of an already aged NMC as opposed to a newly made NMC battery, which is optimised in its designed and aimed for a particular use in its first life. This scenario is considered as the base scenario in the break-even analysis.

3. 50% allocation: the impact is split evenly between the first and second life. This represents the case where both uses are seen as equally valuable in the

NMC life cycle. This view point can be of interest to EV makers in particular which intend to share the environmental burdens of battery production with other parties.

3.3 Delay of End of Life

To analyse the environmental impacts on mineral resources due to prolonging the life cycle of NMC battery by applying a second use and delay the EoL, the CSI is used. This method provides assessment results which are relevant for decisions with >100-year implications (Arvidsson et al. 2020). Thereof, the scarcity index is used in this study as a proxy for shorter time supply risks and as a way for making a comparison between CSI per 1 kWh available capacity of the batteries. This indicator is then used to address the material scarcity hot-spots in the battery pack, cell and CAM level. This is indicated by calculating the fractions of CSI caused by different battery components or materials that a cell or cathode of NMC and LFP are comprised of. Thus, it is used to point out which materials are relatively causing the main impact regarding material scarcity and to what extent. However, the electronic parts and plastics are excluded from the calculation as they are either in common between both battery types or has negligible significance regarding material scarcity impact in case of plastics. In the following, the rationale behind CSI method is further described.

3.3.1 Crustal Scarcity Indicator

The assessment of mineral resources has been much discussed in the field of LCA. The discussions have included how to best assess the impacts of mineral resource extraction from different perspectives and even what a mineral resource impact is to begin with (Arvidsson et al. 2020). In order to assess the mineral resource impact, a method that captures the perspective of long-term global scarcity of elements is suggested by Arvidsson et al. (2020). This method is a midpoint-level mineral resource impact assessment and is called the CSI. The main rationale behind the method is to provide a long-term perspective on mineral resource scarcity and provide assessment results which are relevant also for decisions with >100-year implications.

The developed CSI is based on characterisation factors called Crustal Scarcity Potential (CSP) which for element i is

$$CSP_i = \frac{1/C_i}{1/C_{Si}},$$

where C denotes the crustal concentration (in ppm). Thus, the crustal concentration of the analysed element is referenced to the crustal concentration of Silicon (Si) to be compared with other elements. The CSP is further multiplied with the mass of element i, in kg, to obtain the CSI (Arvidsson et al. 2020).

3.4 Inventory Analysis

In the following sections, the data used in the model will be explained.

3.4.1 Energy Use in NMC Production

This study uses data on the energy use of NMC production from Emilsson and Dahllöf (2019) which is based on Dai et al. (2017), Dai et al. (2018), and Dai et al. (2019). According to Emilsson and Dahllöf (2019), 1127 MJ (313 kWh) of energy is used in the production of NMC batteries per kWh capacity. The upstream materials acquisition attribute to 910 MJ (253 kWh) of the total and the remaining 216 MJ (60 kWh) represents the battery production. The battery pack assembly is assumed by Dai et al. (2019) to be done manually and thus does not require a significant amount of energy.

To add up different types of energy to a sum, they have to be comparable. In Dai et al. (2019), the total energy amount of 60 kWh for the manufacture was obtained from adding energy use of both steam and electricity. Dai et al. (2019) considered the steam to be created by a natural gas boiler with an efficiency of 80%. However, to analyse how the production of the NMC battery is affected by the electricity mix Emilsson and Dahllöf (2019) considered the steam to be created by electricity. Since an electric boiler has an efficiency of 100%, the amount of steam energy is equivalent to the amount of electricity. Therefore, the energy use from both steam and electricity is directly comparable and thus can be added.

3.4.2 Energy Use in Recycling

There are discrepancies and huge data gaps in the literature regarding the energy requirement of recycling methods since the operations are not yet in place and have not been scaled to industrial level. Moreover, such evaluation depends on the logistics of recycling regarding collection, transportation, location of facilities and the design of recycling process per se (Emilsson and Dahllöf 2019; Neumann et al. 2022; Heath et al. 2022; Frith 2022; Petranikova 2022; Chabanne 2022). Yet, 140 Wh/kg EoL NMC is considered as electricity requirement in the model (Carvalho et al. 2021). The energy density is multiplied with 80% to obtain the energy density of 114 Wh/kg second life NMC. To convert this to kWh battery capacity, the energy density of 143 Wh/kg NMC is used (Quan et al. 2022; Dai et al. 2019).

3.4.3 Global Warming Potential

The climate change impact of LIBs life cycle, measured as GWP, varies significantly between different studies for reasons already stated. That said, in this project, the results from Emilsson and Dahllöf (2019) are used for LCI regarding GWP resulting from the production phase of NMC batteries. This is due to a more updated and transparent data that the study uses and its elaborate methodology for incorporating emission factors of different electricity mix and energy supply scenarios. As noted before, the production phase of LFP battery in this study is assumed to have 70% energy demand of that of NMC. Same rate is assumed with respect to GWP of the two batteries in production phase.

Emilsson and Dahllöf (2019) estimates 61-106 kg CO_2-eq/kWh battery capacity which is calculated for NMC battery production as the most common type of battery chemistry. This emission range is mainly based on the energy requirement published by the most recent report by Dai et al. (2019) which equates 1127 MJ/kWh. The energy for the materials and processes of cell and pack production from Dai et al. (2019) are used to calculate the GWP for different energy mixes. The authors extrapolated the energy from a factory at 75% capacity use and then calculated the resulting emissions. Only 30 MJ out of 170 MJ come from electricity in Dai et al. (2019), and the rest was estimated as heat produced with natural gas. Further, the GHG emissions are calculated considering the use of 100% virgin raw materials. The energy for materials and cell manufacture processes are illustrated in table 3.2.

To calculate emissions of upstream materials and cell production and battery pack assembly, the entire energy demand from the cell production and battery pack assembly are added to the reported 59 kg CO_2-eq/kWh capacity from upstream materials (Dai et al. 2019). The emissions range for cell production and battery pack assembly are calculated with a renewable electricity mix, estimated at 0.05 kg CO_2-eq/kWh consumed electricity, and a fossil-fuel rich mix, estimated at 1 kg CO_2-eq/kWh consumed electricity. The range of emission values is wide, and the authors believe that the upper range is an overestimate because electricity is unlikely used for heating in processes that could be heated with more energy efficient alternatives, such as natural gas or other fuels. However, some exceptions could be if renewable sources are purposely being used to lower emissions, or if the electricity happens to be cheaper than fuels such as natural gas in those locations where the operation occurs.

It is suggested by Emilsson and Dahllöf (2019) that adjusting only the electricity mix for cell production and battery pack assembly reflects how cell production facilities may influence the emissions. Given that only a small portion (30 MJ out of 170 MJ) of energy use comes from electricity in Dai et al. (2019), varying only the electricity mix will not have a significant effect on the resulting GWP. Since heating can also be sourced by electricity, it is also interesting to see how the GWP is affected if the heating sources were varied. Results by Emilsson and Dahllöf (2019) are presented in Table 3.3 where the heating comes from natural gas or electricity and the electricity mix is varied from a renewable energy mix to a fossil-rich mix. The fossil-rich mix can be interpreted quite similar to the electricity mix in China (Emilsson and Dahllöf 2019).

Table 3.2: Energy for materials and cell manufacture processes reproduced from Emilsson and Dahllöf (2019). The energy for battery pack assembly is assumed to be insignificant compared to the total energy demand.

Materials and Processes		Energy [MJ/kWh capacity battery]
Cell components	NMC111 powder	409.9
	Graphite	88.6
	Carbon Black	10.7
	Binder (PVDF)	5.5
	Copper	35.7
	Aluminum	50.7
	Electrolyte: LiPF6	20.3
	Electrolyte: EC	3.2
	Electrolyte: DMC	11.8
	Plastic: PP	6.1
	Plastic: PE	1.4
	Plastic: PET	1.1
Module components	Copper	0.8
	Aluminum	37.4
	Plastic: PE	0.6
	Insulation	0.1
	Electronic parts	19.0
Pack components	Copper	0.2
	Aluminum	116
	Steel	1.3
	Insulation	0.8
	Coolant	6.0
	Electric parts	83.4
Solvent	NMP (recycled)	0
Cell Production and Battery Pack Assembly		216.2
Total		1127

Finally, the sum of the results of the two scenarios by Emilsson and Dahllöf (2019) for cell production and pack assembly from Table 3.3 and the upstream material GWP from Dai et al. (2019) which is 59 kg CO_2-eq/kWh consumed is shown in Table 3.4. As noted already, given the energy required for the production per kWh battery capacity, total GWP ranges from 61-106 kg CO_2-eq/kWh battery capacity when varying the electricity mix in the first scenario. However, with varying the electricity only when natural gas is used for heating, the emissions range from 70-77 kg CO_2-eq/kWh battery capacity as shown in the second scenario.

Table 3.3: Total GWP from cell and pack manufacture in scenarios varying only the heat source. The electricity used is 30 MJ/kWh capacity and the heat is 140 MJ/kWh capacity. Reproduced from Emilsson and Dahllöf (2019).

Scenarios: different energy source for heat	Energy sources of cell and pack manufacture	kg CO_2-eq /kWh consumed	GWP, 30 MJ electricity consumed /kWh capacity	GWP 140 MJ heat consumed /kWh capacity	Sum GWP from cell and pack manufacture [kg CO_2-eq /kWh
Scenario 1	Electricity: Renewable mix – fossil-fuel rich mix	0.05–1	0.4–8.3		2-47
	Heat: Electricity, Renewable mix – fossil-fuel rich mix	0.05–1		2.0–38.8	
Scenario 2	Electricity: Renewable mix – fossil-fuel rich mix	0.05–1	0.4–8.3		11-18
	Heat: Natural gas with boiler efficiency 80%. Calculated from (EIA, 2016).	0.26		10.1	

Table 3.4: GWP of different energy scenarios in the production phase (Emilsson and Dahllöf 2019).

Scenarios: different energy source for heat	Sum GWP from cell and pack manufacture [kg CO_2-eq/kWh]	Total GWP [kg CO2-eq /kWh]
Scenario 1	2-47	61-106
Scenario 2	11-18	70-77

3.5 Sensitivity Analysis

To assess the impact of uncertainty in parameters and assumptions used in the model and the consequential changes in results, a sensitivity analysis is conducted. Energy use in the production phase of LFPs and charge-discharge efficiency of second life NMCs are tested in a reference scenario where average number of cycles are set for both batteries; 6000 cycles for a LFP battery and 3000 cycles for a second-life NMC. Additionally, allocation is set to be 20% to second life as in the base scenario and recycled input for the NMC production is considered to be 10%.

4

Result & Analysis

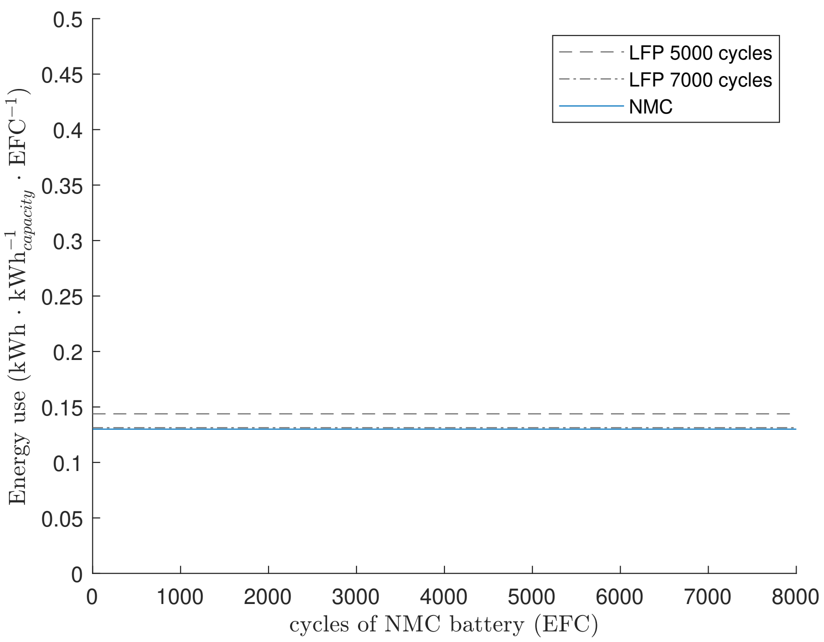
The following sections will present the results of the break-even analysis, as well as the CSI.

4.1 Break-even Analysis

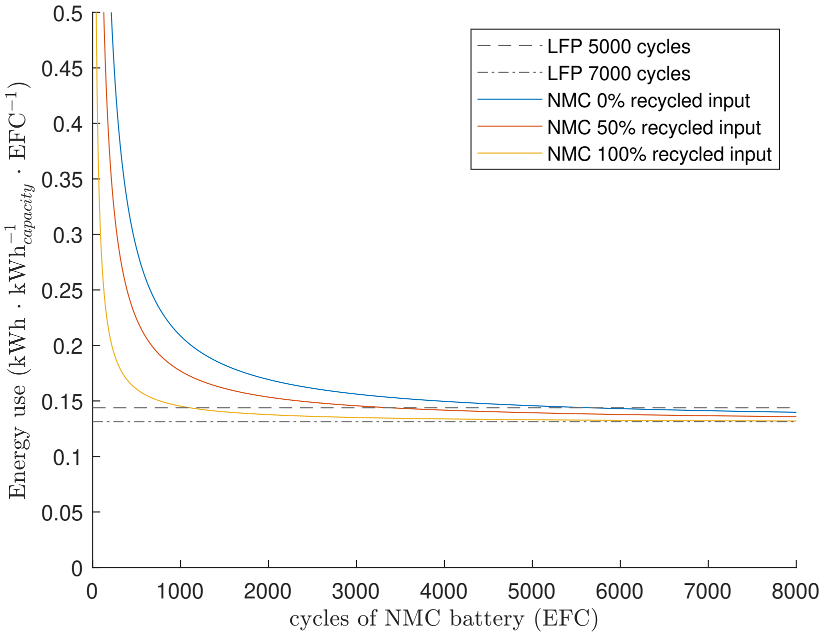
The break-even analysis has been conducted for two impact categories, energy use and GWP. In the first category, the energy use is analysed based on the amount of EFCs the NMC battery can last and for the GWP, the carbon footprint is based on the electricity mix.

4.1.1 Energy Use

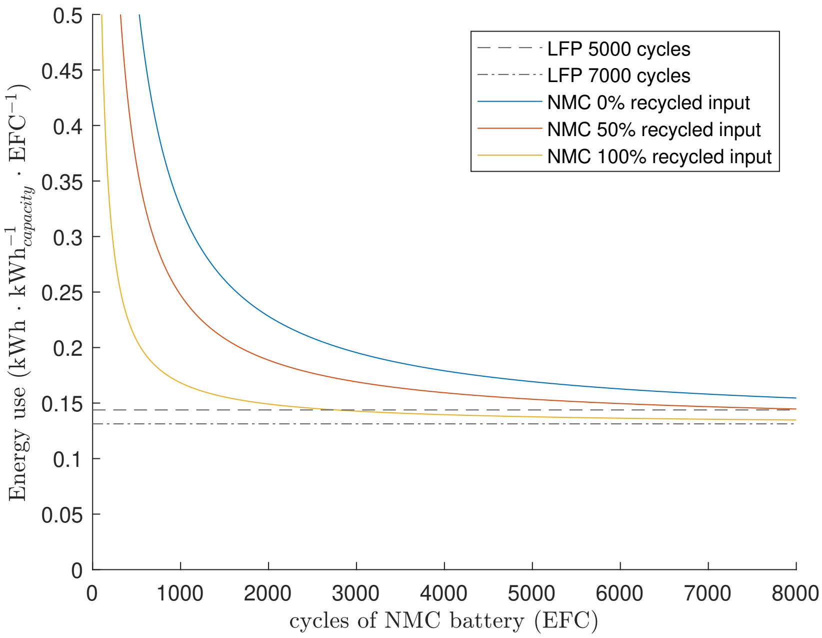
In Figure 4.1, the break-even analysis of the energy use of the NMC battery with varied cycle life compared to the LFP battery with fixed 5000 or 7000 cycle life can be seen. The analysis is presented for three different scenarios, each representing a different allocation to the first and second life of NMC.



(a) 0% allocated to second life, representing system expansion.



(b) 20% allocated to second life.



(c) 50% allocated to second life.

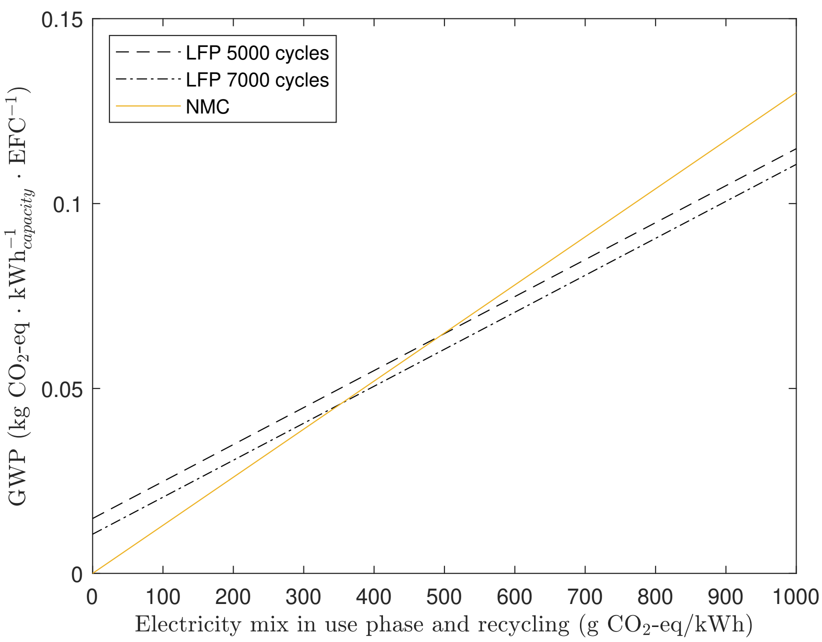
Figure 4.1: Break-even analysis of the energy use of the NMC battery compared to the LFP battery with varied allocation percentages.

In the system expansion scenario where 0% allocation is attributed to second-life, Figure 4.1a, the NMC battery requires less energy per EFC and kWh capacity than the LFP battery for either 5000 or 7000 cycles. Thus, it is in terms of energy demand beneficial to society to use the NMC battery for a second life instead of producing a new LFP battery for a BESS.

When considering that the second life of the NMC battery should account for some of the production and recycling energy use, the outcome changes. With the chosen allocation percentages 20% and 50% in Figures 4.1b and 4.1c respectively, the LFP battery outperforms the second life NMC battery. At a realistic limit of 10% recycling, the second-life NMC would need to last for about 5000 cycles in the 20% allocation case to have a lower energy consumption than the 5000 cycles LFP battery. This would according to the current research in battery ageing not be a realistic cycle number for any application. However, with 100% recycled material in the production, the cycles needed to outcompete the new LFP battery might be within the reasonable range. Thus, an individual battery pack with 100% recycled input could be the environmental choice but in the larger scope, this recycling percentage is not feasible. For 50% allocation however, even the NMC battery produced with 100% recycled input is struggling to be justified as the choice.

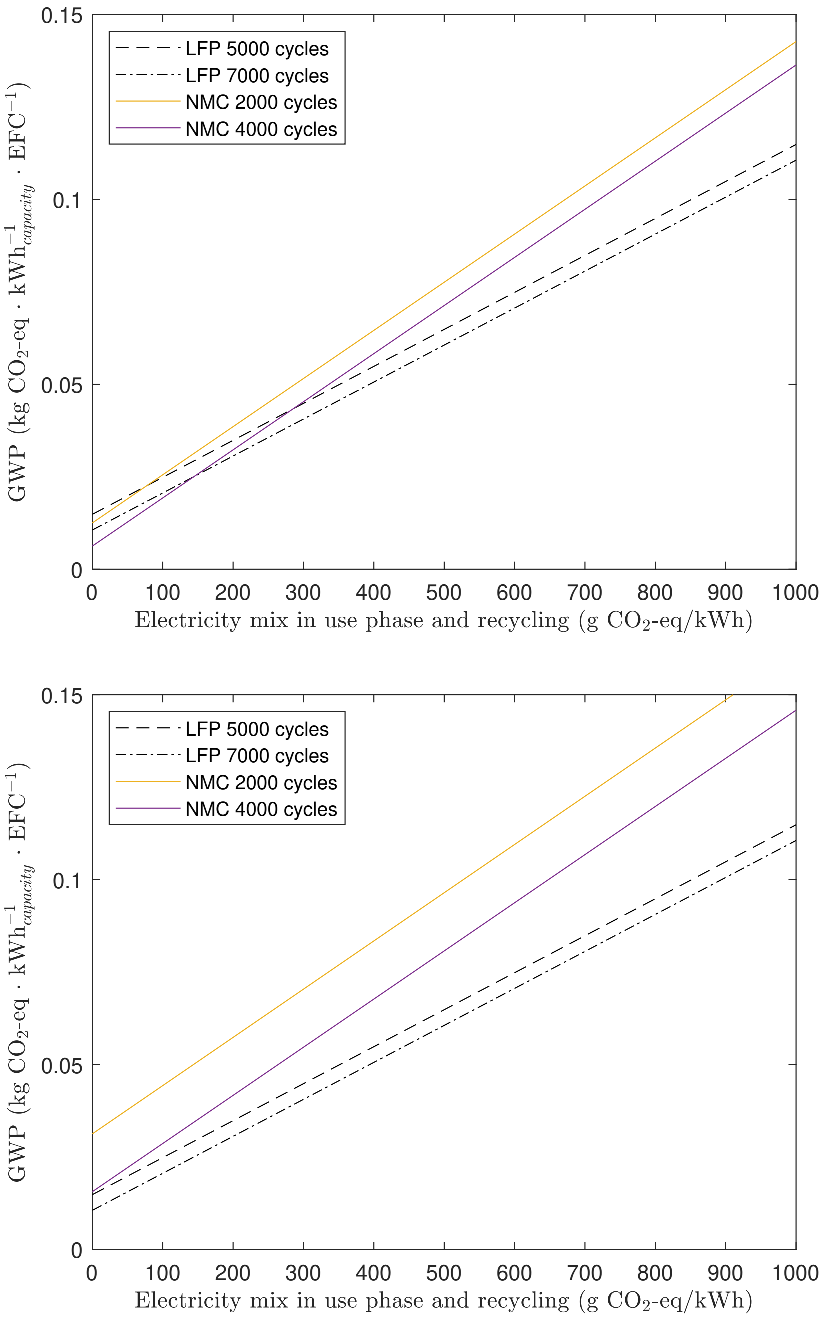
Both of these choices of allocation percentages clearly show that recycling can play a key role in the future of NMC batteries, especially when considering extended life cycle. However, prolonged use also represents a delay of EoL and recycling. Thus, as NMC batteries are increasingly used in a second use phase, less recycled material will be available to replace raw material extraction.

4.1.2 Global Warming Potential



(a) 0% allocated to second life, representing system expansion.

Considering the impact category GWP, the analysis switches from the focus on EFCs to the electricity mix being used in the use phase and recycling. Therefore, the significance of a low carbon intensity can be analysed. In Figure 4.2, the results of break-even analysis of the GWP of the NMC battery with 2000 or 4000 cycle lives and 10% recycled input compared to the LFP battery with 5000 or 7000 cycles life are shown.



(c) 50% allocated to second life.

Figure 4.2: Break-even analysis of the GWP of the NMC battery with 10% recycled input compared to the LFP battery with varied lifetime.

In the system expansion scenario shown in Figure 4.2a, it can be observed that the NMC battery has a lower carbon footprint per functional unit and cycle than the LFP batteries for a carbon intensity of 350 g CO_2-eq/kWh in the electricity mix. This carbon intensity is higher than the European mean of 229 g CO_2-eq/kWh in 2020 (European Environment Agency 2022). Thus, within the scope of this study, the NMC battery is the more environmentally beneficial choice in most parts of Europe.

The scenarios where allocation of the production and recycling emissions are included are visualised in Figures 4.2b and 4.2c. For the 20% allocation, the NMC battery most likely has a lower carbon emission than the LFP battery. If considering the average life cycles of the batteries, the NMC battery would be the better choice in countries with a very low carbon intensity in the electricity mix. One of those countries is Sweden, with a carbon intensity of 8 g CO_2-eq/kWh in 2020 (European Environment Agency 2022). Considering that BESSs is more in demand in countries with a high share of renewables for a time horizon of 2030 onward and thus lower carbon intensities, this makes a strong case for the second life NMC. When 50% of the production and recycling emissions are allocated to the second life, the LFP battery with either 5000 or 7000 cycles are the better choice than the NMC battery with 2000-4000 cycles.

In all the figures in 4.2, it can be seen that the NMC battery reacts more to the carbon intensity of the electricity mix than the LFP battery. This is due to the fact that the energy loss in the use phase is higher with a relatively lower charge-discharge efficiency of aged batteries and thus a higher fraction of the total energy use would be dependent on the electricity mix. The LFP battery however has a higher initial carbon footprint due to the fact that the entirety of the production is included given that the production is assumed to have a carbon intensity of 1 kg CO_2-eq/kWh.

4.1.3 Sensitivity Analysis

To study the changes in results when changing two critical and especially uncertain parameters, a sensitivity analysis was performed. The baseline scenario represents the case when the LFP lasts for 6000 cycles, LFP production uses 70% as much energy as the NMC production, 20% of the production and recycling is allocated to the second life NMC battery, the charge-discharge efficiency of LFP is 90% and 87% for the NMC, and the recycling percentage of NMC is 10%. Further, the cycle number of the NMC battery in the GWP analysis is 3000 cycles. The two scenarios represent when the efficiency of the second life NMC battery is changed from 87% to 90% (the same as for a new LFP battery) and when the LFP production require as much energy as the NMC production instead of only 70%. The visualisations of these scenarios can be seen in Appendix A.

Table 4.1: The sensitivity analysis of the break-even analysis for a change in two parameters. For energy demand, the number indicates the minimum amount of cycles that the NMC battery has to last for to outperform the LFP. In the case of GWP, the NMC battery is the more beneficial choice for regions with a lower or equal carbon intensity to the one indicated.

Scenario	Break-even point Energy demand [EFCs]	Break-even point GWP [g CO_2-eq/kWh]
Baseline	11085	134
Efficiency 90%	1979	∞[a]
Production 100%	3259	310

[a] The GWP is lower for any carbon intensity.

The results of the sensitivity analysis presented in Table 4.1 show that the change in efficiency has the greatest impact on the results. With a change in charge-discharge efficiency from 87% to 90%, the NMC battery outcompete the LFP battery when surviving for 1977 or more cycles, which is well within a reasonable cycle number for NMC. Additionally, comparing the 3000 cycle NMC battery to the 6000 cycle LFP battery, the GWP will always be lower for the NMC since the energy loss in the use phase is equivalent for the batteries. When the production of the batteries require as much energy, the NMC battery has a significantly better outlook compared to the baseline scenario, having to last for about 3300 cycles or more to be the more environmentally beneficial choice in terms of energy use. In regard to the GWP with this parameter change, the NMC battery would have a lower GWP with a carbon intensity lower than 310 g CO_2-eq/kWh in the electricity mix. This is higher than the average European electricity emission of 229 g CO_2-eq/kWh in 2020 (European Environment Agency 2022). Thus, the NMC battery would not only be the beneficial choice in Sweden, but in most of Europe where the projected electricity mix would have even lower emissions by 2030 onward.

4.2 Crustal Scarcity Indicator

Current LIBs suffer from the use of several scarce metals with associated environmental challenges. Moreover, for some metals the criticality is coupled with potential supply disruption and geopolitical risks.

Table 4.2: The CSI of 1 kWh of the batteries.

Battery Chemistry	CSI (Si-eq)
LFP	15100
2nd life NMC	21300

As shown in Table 4.2, second life NMC has a relatively higher crustal scarcity impact per 1 kWh available capacity compared to LFP. While for both battery chemistries, materials used in cells are the main contributors to that cause.

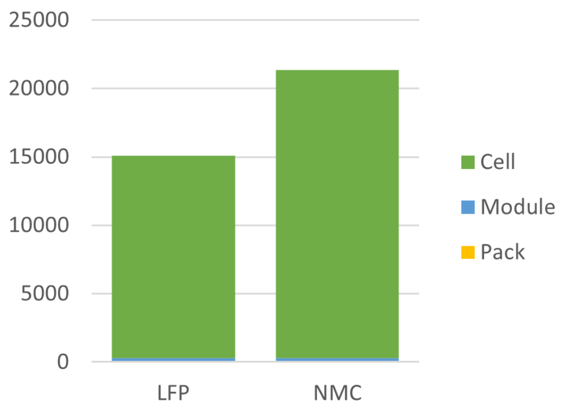


Figure 4.3: Fraction of CSI that is caused by different parts of the batteries, indicating that the CSI of pack and module is relatively insignificant.

As shown in Figure 4.3, for both NMC and LFP batteries, cells contribute to more than 98% of resource scarcity impact. Given the significance of cells in material scarcity impact of LIBs, it is important to indicate the metals which are relatively crucial. Thus, Figure 4.4 shows which materials in the cell contribute the most to the CSI.

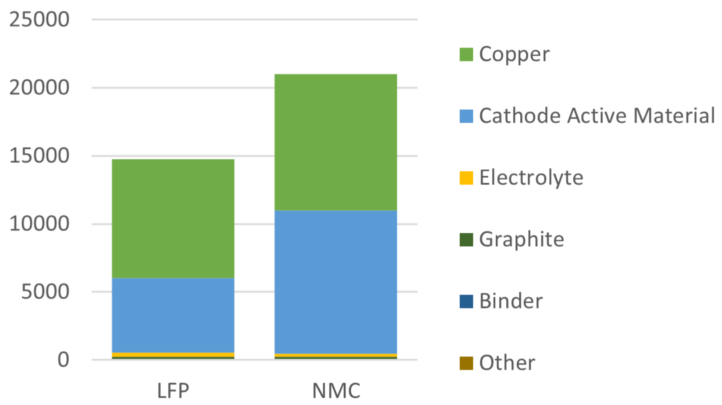


Figure 4.4: Fraction of the CSI of the cells that is caused by different materials.

As depicted in Figure 4.4, for LFP cells, copper accounts for almost 60% of the impact while the CAM account for less than 40%, whereas for NMC cells, the CAM account for more than half of the impact. Since the cathode is such a large contributor to the CSI, the CSI of the metals making out the CAM are presented in Figure 4.5.

For LFP, lithium has the highest impact. However for NMC, cobalt, lithium and nickel are the main contributors to the CSI. Given the risks attributed to supply of cobalt, nickel and lithium, the criticality of these metals beyond the crustal scarcity impact, becomes more pronounced in NMCs. Therefore, extending the life cycle

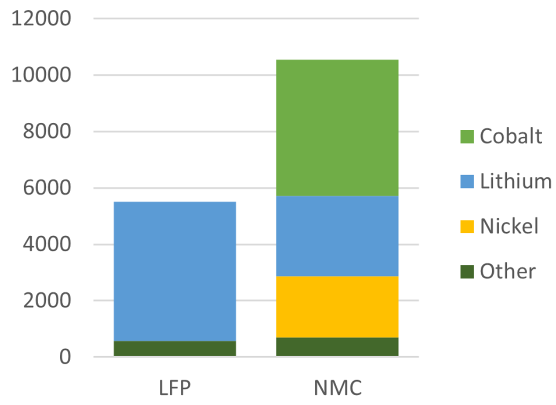


Figure 4.5: Fraction of the CSI of the active cathode materials caused by different critical metals.

of NMC batteries by applying them to a second use and consequently delaying the EoL can have implications for material supply for the new production. That said, recyclability of batteries becomes central in this analysis and as recycling infrastructure is under development, devising strategies should be in accordance with different time horizons based on how feasible and economically viable the outlook of recycling will be.

5

Discussion

The production data in this study has been collected from current data associated to energy requirements of production today. It should also be noted that there is a lack of primary data in existing literature and substantial inter-dependency between different sources. As this study has a focus on 2030-2040, the time mismatch between the production of the two different battery chemistries is only reflected by taking energy for LFP production equal to 70% of that of NMC (Frith 2022). To obtain a more correct conclusion, the data for the reference battery, the LFP, would have to be taken from a prospective LCA study. These are not readily available but considerations of development in production of LIBs has been reflected on in some previous research.

Chordia et al. (2021) contemplated over this and observed some prospects of improvements in production methods. Since the battery factory studied had a certain amount of fixed emissions, from e.g. dry rooms, independent of the amount of batteries produced, an increase in production in the factory would significantly decrease the emission per battery produced. In the same manner, giga-factories can reduce the emissions drastically (Aichberger and Jungmeier 2020; Chordia et al. 2021). Moreover, Ellingsen et al. (2014) noted the importance of production location in terms of transportation and electricity mix. This study considered both of the batteries being produced in China with a carbon intensity of 1 kg CO_2-eq/kWh battery capacity. However, the emissions from production would decrease drastically if another electricity mix was chosen.

Batteries with a second life and their implementations are a relatively new subject. Thus, more research on calendar and cyclic ageing and the analysis on the SOH of batteries is needed to identify second life applications. Moreover, it is important to ensure the safety. As a result, more studies should be conducted on the handling and transportation of retired batteries to eliminate the risk of accidents (Shahjalal et al. 2022). This is furthermore an important characteristic for the use of second life batteries, as indicated by the expert interviews held (Campagnol 2022). Regarding economic viability of repurposing, an overview of cost analysis and profitability of second life battery applications should be taken into account. Moreover, as the use of second life NMCs accelerates, more standardisations and regulations are expected to be developed (Zhao et al. 2021). Lastly, the location of implementation of second life batteries matters in terms of their environmental impact. This is due to transportation needed to handle batteries in addition to the electricity mix in use, as is analysed in this study.

The data available on LIB recycling is limited. This is due to the fact that recycling is in development and that the logistics for it is not in place yet. Thus, the impacts of collection, sorting, pre-treatment and dismantling are uncertain and transportation involved in the steps of recycling has in most cases not even been studied. Zhou et al. (2018) showed that the transportation in some cases can have a substantial impact, providing an example of retired batteries being sent between Europe and Australia for processing. This transportation alone would generate approximately 300 kg CO_2-eq per tonne of batteries (Zhou et al. 2018). Thus, the geographical distribution of recycling facilities could play a substantial role in the impact ofLIBs. However, for a recycling facility to run profitably, it needs a sufficient supply of LIBs. Since hydrometallurgy is a chemistry customised process and LFPs are not commonly used in Europe, this potentially poses a problem for the future of LFP recycling in the region. The recycling of NMCs have a promising outlook in Europe however, with the increase of EV market share.

As NMC batteries are increasingly used in a second use phase, less recycled material will be available to replace raw material extraction. Nevertheless, the use of recycled materials instead of virgin materials to manufacture LIBs does not always generate an environmental benefit. This could be due to a variation in factors such as collection rate, recovery efficiencies and purity of the recycled material, recycling energy requirements and carbon intensity of the electricity and distance between recycling facilities and battery manufacturing locations (Ciez and Whitacre 2019). Besides, the cost of recycling and transporting used battery packs can have a large impact on the economics of recycling. Yet, to enhance circularity of batteries, design for circularity should be aimed for by standardising designs to facilitate easier disassembly and more efficient recycling. This can clear the way for direct recycling, which in turn will be more energy efficient. Moreover, traceability and battery labelling will enable more efficient allocation of the EoL batteries to the most suitable recycling process (Heath et al. 2022).

The choice of allocation can, as shown in this study, greatly impact the results. There are many factors that can affect the choice of allocation between the first and other potential use phases of the battery including performance, available capacity and economic value (Ryberg 2022; Janssen 2022). However, there is no correct allocation method and prolonged lifetime creates ambivalence (Nordelöf 2022). Since not only different stakeholders aim for motivating different allocations between the first and second life of batteries, how the market evolves for second life will also have indications on which rationale to go for. Thus, allocation comes down to who is the beneficiary.

The 0% allocation was observed as the system expansion case in this study. However, which system is being discussed and which reference point or time frame are set as premises are highly important to note. The system expansion in this study was under the reasoning that the NMC battery is already in existence and a second application has not been aimed for in production. Thereby, would it be most beneficial to the environment to keep using the battery or to produce a new LFP battery to provide the same service? This reasoning does however contest the motivation behind crustal scarcity since there is an additional negative impact to society in prolonging the life of the batteries by avoiding the availability of recycled materials to replace raw material extraction.

Second life batteries can only have a large scale adoption if they are economically profitable and certain business models are likely to be more favourable for battery reuse. The cost efficiency of retired batteries further highly depends on the charge-discharge efficiency, especially in countries with a higher electricity cost. With more research in the field of battery ageing and performance, this will however be more understood and a more relibale cost benefit analysis can be performed. Given a lower lifespan of used compared to new LIBs, to ensure profitability, the logistics of second life has to be quite efficient and cost-effective regarding collection from the first use, transportation, SOH analysis, repurposing and distribution to second life to reach the user and then collection by the end of life.

Nevertheless, by the time NMCs reach the EoL, as recycling infrastructure is under development today and is presumed to be so until 2030s, that can be a stronger case for prolonging the use of retired batteries. This delays the EoL until when recycling is commercially viable. On the other note, the growth in battery demand has driven up raw material prices and given rise to concerns about potential cobalt, nickel, and lithium scarcities. Mining, battery manufacturing, and the automotive industry will become increasingly interwoven. Thus, such inter-dependency between intra-sectoral players will add to the uncertainty of market and economic outlook of second-life batteries (Berbner et al. 2022; Campagnol et al. 2018).

In this study, transportation has been excluded in general. This is due to the lack of data in this regard as well as the lack of infrastructure in place for second life and EoL logistics. The production data taken from Emilsson and Dahllöf (2019) and Dai et al. (2019) do not include transportation and neither does the reviewed recycling literature (Carvalho et al. 2021; Heath et al. 2022; Quan et al. 2022). Given that the supply chain of LIB is still relatively new and emerging, transportation could affect the results to a significant extent.

It is important to emphasise the importance of time horizon when devising strategies. Recycled material may be insufficient to meet the growing demand in the short to medium term if the use of batteries continue to increase exponentially. It is projected that by mid-2040's, recycled material can meet a growing share of demands causing the demand for virgin materials to reach a plateau and then decline. Moreover, technology development and reduction of metal intensity would enable the number of supply sources to increase over time (Månberger and Johansson 2019). However, rapid technological advancements in the realm of energy storage and also in the case of LIBs can alter the points of reference. Implementation of policies, adoption of new regulations and rising uncertainties regarding scarce metals and market dynamics can also add to such complexity. Therefore, it is strategically important to emphasise the development of a robust supply chain of NMCs which can be adaptable to both extending the life cycle by creating a second market as well as efficient management of the EoL.

6

Conclusion

The results show that the choice of allocation and recycling input for the NMC battery greatly impacts the outcome. As a result, assuming 20% of the production and recycling energy use being allocated to the second life and a foreseeable 10% recycling input, the LFP battery requires less energy per EFC than the second life NMC battery within the reasonable cycle life. However, the impact of cyclic and calendar ageing on battery performance should be better anticipated. Thus, solely based on energy demand, second life NMCs does not seem to outcompete LFP.

However when varying the carbon intensity of the grid mix in the use and recycling phases, the outlook is nuanced. Since the second lifeNMC battery has a lower charge-discharge efficiency, a greater share of its impact originates in the use phase compare to a new battery. Therefore, it is more impacted by the carbon intensity of its used electricity. That said, with the same assumptions, the second life NMC battery has a lower GWP impact per 1 kWh capacity and EFC than the LFP battery in countries with a carbon intensity of about 134 g CO_2-eq/kWh or lower. Several countries within Europe has an electricity mix with a carbon intensity below this number, including Sweden.

Prolonging the life of NMCs has its advantages and disadvantages. Advantages include displacing other technologies to provide BESS services and the prospect of more developed recycling infrastructure at the end of second life. Depending on the displaced reference technology, the environmental benefit can however vary. A disadvantage to prolonged life is that more virgin raw material will have to be extracted to produce new NMCs since less recycled material will be available due to a delayed EoL. However, uncertainties in recycling development makes it speculative if fully closed-loop recycling will be possible by 2030.

Since this analysis only covers the impact categories of energy demand, climate change and material scarcity, further studies are needed to fully grasp the scope of environmental impacts associated with the second life NMCs. Additionally, more research is needed on many fronts in this new field. The ageing of the batteries has to be studied to conclude which applications the second life batteries are most suited for. There is also a need to analyse the dynamics of economy surrounding this product to determine its profitability. Last but not least, the recycling has to develop to a point of operation. This is critical in terms of circular economy of batteries and from the view point of material scarcity.

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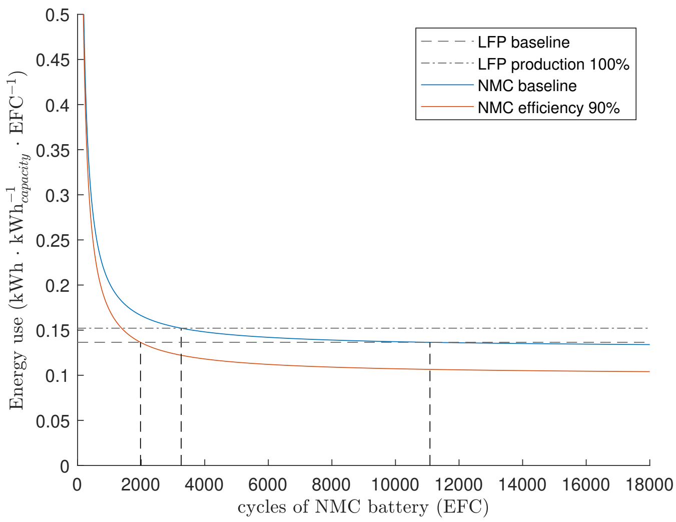
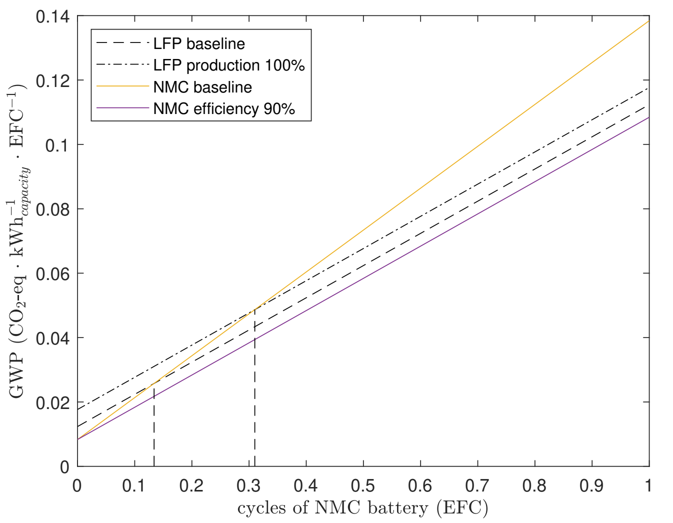
A

Sensitivity Analysis



(a) Impact category Energy Demand.



(b) Impact category Global Warming Potential (GWP)

Figure A.1: Sensitivity analysis comparing the baseline scenarios to two changes in the parameters.

B

Material Scarcity

Table B.1: The CSI of 1 kg of LFP powder.

Substance	Mass [kg]	CSI [kg Si-eq]
Lithium hydroxide (LOH)	0.46	2399.861
Phosphoric Acid (H_3PO_4)	0.65	133.544
Iron sulphate ($FeSO_4$)	1	149.744
Total	-	2683.149

Table B.2: The CSI of 1 kg of NMC powder.

Substance	Mass [kg]	CSI [kg Si-eq]
Lithium Carbonate (Li_2CO_3)	0.383	1303.918
NMC precursors		
Nickel Sulphate ($NiSO_4$)	0.535	1052.020
Manganese Sulphate ($MnSO_4$)	0.535	151.613
Cobalt Sulphate ($CoSO_4$)	0.522	2258.649
Sodium Hydroxide (NaOH)	0.845	5.826
Ammonium Hydroxide (NH_4OH)	0.118	0.737
Total	-	4772.764

Table B.3: The CSI of 23.5 kWh initial capacity LFP and NMC batteries (Quan et al. 2022; Dai et al. 2018; Dai et al. 2019; Arvidsson et al. 2020).

Battery Types	LFP			NMC		
Capacity (kWh)	23.5			23.5		
Energy Density (Wh/kg)	115.8			142.4		
Cell components						
Material	Mass [kg]	CSP [kg Si-eq/kg]	CSI [kg Si-eq]	Mass [kg]	CSP [kg Si-eq/kg]	CSI [kg Si-eq]
Cathode active material (CAM)	48.21	2683.15	129354.61	41.52	4772.76	198165.14
Graphite	24.85	140	3479	23.18	140	3245.2
Carbon Black	3.25	140	455	2.8	140	392
Binder (PVDF)	4.02	355.14	1427.68	3.55	355.14	1260.76
Copper	20.51	10000	205100	18.84	10000	188400
Aluminum	11.13	3.4	37.84	9.8	3.4	33.32
Electrolyte: $LiPF_6$	4.98	1228.17	6116.27	2.66	1228.17	3266.92
Electrolyte: Ethylene Carbonate	13.89	57.29	795.69	7.43	57.29	425.63
Electrolyte: Dimethyl Carbonate	13.89	56.00	777.89	7.43	56.00	416.11
Subtotal: cells	147.59	-	347543.98	119.77	-	395605.08
Module components sans cell						
Material	Mass [kg]	CSP [Si-eq/kg]	CSI [Si-eq]	Mass [kg]	CSP [Si-eq/kg]	CSI [Si-eq]
Copper	0.51	10000	5100	0.43	10000	4300
Aluminum	9.39	3.4	31.93	7.22	3.4	24.55
Subtotal: module sans cell	11.33	-	5131.93	9.06	-	4324.55
Pack components sans module						
Material	Mass [kg]	CSP [Si-eq/kg]	CSI [Si-eq]	Mass [kg]	CSP [Si-eq/kg]	CSI [Si-eq]
Copper	0.11	10000	1100	0.09	10000	900
Aluminum	26.34	3.4	89.56	22.33	3.4	75.92
Steel	1.44	428.73	617.37	1.02	428.73	437.30
Subtotal: Pack sans module	44.09	-	1806.92	36.14	-	1413.22
Total: Battery Packs	203.1	-	354482.84	164.99	-	401342.85

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