Investigating Environmentally Sustainable Pathways for Passenger Vehicle Fleet in the UK

by

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Declaration

I, Mashael Kamran, confirm that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been indicated in the thesis.

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Abstract

To limit global warming to below 1.5 °C, global efforts are being made towards decarbonisation of energy sources, among which UK has placed itself at the forefront to achieve net zero greenhouse gas emissions by 2050. Transport is responsible for 26% of total GHG emissions, of which light duty vehicles represent >50%. Battery electric vehicles (BEVs) are seen as a pathway to achieve decarbonisation and improved local air quality of future transport at the point of use. However, there are environmental concerns about the production of BEVs because of toxic substances released during mining activities and the reliance on critical elements. The degree to which the use of BEVs can contribute to decarbonisation also depends upon the percentage of renewable energy sources in the electricity grid mix. The increased electricity demand and a growing vehicle fleet will entail greater energy investment and resource requirements to meet future targets. Strategies such as the uptake of shared mobility, to minimize the number of vehicles on the road, and circularity of resources, could reduce the wastage of raw materials and associated environmental and energy impacts. Given that lithium-ion batteries (LIBs) play a vital role in both the energy transition of transportation and energy storage for the electricity sector, respectively by enabling the electrification of transportation and supporting the increase of renewables in the electricity mix, there will be an increase in demand for critical battery elements. Reusing batteries to support the electricity grid mix and recovering battery materials for reuse in BEVs could be opportunities to ease some of the environmental concerns.

This thesis developed the case study of the entire passenger light-duty vehicle (LDV) fleet in the UK, set within the context of the co-evolution of transport and energy systems, up to the year 2050. The study investigates the environmental trade-offs of different BEV pathways by incorporating resource strategies, namely: the uptake of shared mobility, battery second life and closed-loop recycling. Initially, a systematic review of the sustainability supply challenges of battery cathode elements and other critical elements for supporting the low-carbon transition was conducted. To evaluate the consequences of resource strategies on the environment, a dynamic approach was taken to carry out a material flow analysis (MFA) and life cycle assessment (LCA) up to the year 2050. A dynamic MFA was used to track the changes in the mass flows of key chemical elements for each year, which laid the foundation for the LCA to assess the environmental consequence of the evolution of the whole UK LDV fleet over time. The interplay of several prospective changes were taken into consideration: (1) the transition of internal

combustion engine vehicles (ICEVs) to BEVs, (2) the transition to a low-carbon electricity grid mix, (3) the improvement in LIB technology, (4) the uptake of transport as a service (TaaS).

It was found that two strategies, namely recovering battery cathode materials and the successful uptake of TaaS, can play the most significant roles in reducing the overall demand for primary crtical materials for LIBs for both BEVs and grid-scale storage in the long term. Furthermore, second-life batteries were found to play a lesser role over time, due to the sheer requirement of battery elements by the BEV sector compared to grid storage battery requirements. In terms of environmental impact, the carbon emissions associated with the manufacturing of new BEVs are significantly higher than those for ICEVs. However, this is more than compensated by the positive effect of low-carbon electricity. Furthermore, the combined closed-loop recycling of cathode materials and potential emissions reduction due to TAAS also indicate a significant reduction potential in abiotic resource depletion and human toxicity potentials, whereas closed-loop recycling alone did not indicate the same outcome. Hence, this case study has highlighted the importance of applying both strategies synergistically to minimize the overall environmental impact of the transition to BEV for the passenger vehicle fleet.

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List of Abbreviations

Abbreviation and Acronym

A-CAES	Adiabatic Compressed Air Energy Storage
ACLA	Attributional Life Cycle Assessment
ADP	Abiotic Depletion Potential
AGR	Advanced Gas-cooled Reactor
AP	Acidification Potential
BECCS	Biomass Energy with Carbon Capture and Storage
BEV	Battery Electric Vehicles
bio-C	Biogenic Carbon
BMS	Battery Management System
c-Si	Crystalline Silicon
CAES	Compressed Air Energy Storage
CCGT	Conventional Combined Cycle Gas Turbine
CCC	Climate Change Committee
CCS	Carbon Capture and Storage
CdTe	Cadmium Telluride
CED	Cumulative Energy Demand
CF	Capacity Factor
CIGS	Copper Indium Gallium diSelenide

CLCA	Consequential Life Cycle Assessment
CNG	Compressed Natural Gas
CoMoUK	Collaborative Mobility UK
EoL	End of Life
EU	European Union
EV	Electric Vehicles
FCEV	Fuel Cell Electric Vehicles
FCV	Fuel Cell Vehicle
FES	Future Energy Scenario
FU	Functional Unit
GB	Great Britain
GWP	Global Warming Potential
HTP	Human Toxicity Potential
ICEV	Internal Combustion Engines
LAES	Liquid Air Energy Storage
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LFP	Lithium Iron Phosphate
LIB	Lithium-ion Battery
LiOH	Lithium Hydroxide
LMO	Lithium Manganese Oxide

MaaS	Mobility as a Service
MFA	Material Flow Analysis
NCA	Nickel Cobalt Aluminum oxide
NdFeB	Nodymium Iron Boron
NMC	Nickel Manganese Cobalt oxide
nr-CED	Non-renewable Cumulative Energy Demand
PGM	Platinum Group Metals
PHEVs	Plug-in Hybrid Electric Vehicles
PHS	Pumped Hydro Storage
PM	Permanent Magnets
POFP	Photochemical Oxidation Formation Potential
PV	Photovoltaic
PWR	Pressurized Water Reactor
REE	Rare Earth Elements
ReLIB	Recycling and Reuse of EV Lithium-ion Batteries
RER	Regional European
SMR	Small Modular Reactor
TaaS	Transport as a Service
TES	Thermal Energy Storage
UK	United Kingdom

USES-LCA	Uniform System for the Evaluation of Substances
USGS	US Geological Survey
V2G	Vehicle to Grid
VMT	Vehicle Miles Travelled
VRE	Variable Renewable Energy

Chemical Symbol and Unit

1,4-DB	1,4-dichlorobenzene
Co	Cobalt
CO_2	Carbon Dioxide
CO_2 eq	Carbon Dioxide Equivalent
CoSO4	Cobalt Sulphate
CTUh	Comparative Toxic Unit for human
Ethene eq.	Ethene Equivalent
kg	Kilogram
km	Kilometer
kWh	Kilowatt-hour
Li	Lithium
MJ	Mega-joules
Mn	Manganese
MnSO4	Manganese Sulfate

MWe	Megawatt Electrical
NH_x	Ammonia
Ni	Nickel
NiSO4	Nickel Sulfate
NOx	Nitrogen Oxides
Sb	Antimony
Sb eq.	Antimony Equivalent
SO_2-eq	Sulfur Dioxide Equivalent
SOx	Sulphuric Oxides
yr	Years

1 Introduction

1.1 Sustainability and Transport

“We can't solve problems by using the same kind of thinking we used when we created them.” - Albert Einstein

Besides being a crucial driver of economic and social development, transport is also one of the major contributors to several environmental impacts, which include climate change and depleting natural resources. The transport sector represents 64% of global oil consumption and 23% of the world’s energy-related carbon dioxide emissions (Mead 2021). The traditional approach is linear, whereby raw materials are collected, turned into products, and disposed of either by incineration or landfilled (Sillanpaa 2019); therefore, the technological progress of a linear economy has no accountability or concern for ecological footprint and consequences. This prioritizes mass production and consumption over sustainability (Sillanpaa 2019). This can be seen in the case of transport, where transport intensity and impact were loosely in line with economic growth, which indicates a lack of transport efficiency and environmental sustainability (Kopp 2013). While a linear approach to growth was successful in contributing to the human welfare till the 20th century, it has now shown detrimental effects on the natural environment and human health, which has accelerated since the industrialization in the 18th century (Sariatli 2017). Damaging ecological impacts of such practices can be noted in the use of non-renewable energy resources, intensive agricultural processes, current transport infrastructure, landfilling, overexploitation of natural resources, which are known to cause resource scarcity, the loss of ecosystem function and irreversible climate impacts (Harris 2012, Sillanpää 2019, Sariatli, 2017, Steffen et al. 2015).

In response to this, many researchers raised the concern of ecosystem contamination and resource scarcity dating back to the 1960s and the need for sustainability (Sillanpää 2019, Velenturf and Purnell 2021). The concept of sustainability arguably dates back to 1713, when a handbook of forestry brought in the concept of ‘Nachhaltigkeit’ translated as ‘sustained yield’, which is described as (Grober 2017) *“To fulfill our obligations to our descendants and to stabilize our communities, each generation should sustain its resources at a high level and hand them along* *undiminished. The sustained yield of timber is an aspect of man's most fundamental need: to sustain life itself."* (Grober 2017). This concept later was used for all things in our ecosystem (Purvis et al. 2019), starting from Boulding (1966), in which Earth was described as a spaceship with limited resources and finite endurance for pollution. This identified the need for technological progress in such a way that stocks or resources are maintained, and inputs are minimized, rather than maximized and then disposed of, this is being part of the ecological cycle where resources are capable of continuous reproduction (Boulding 1996), which brought about the concept of circularity (Sillanpää 2019). "The Limits to Growth" report was published in 1972 by the Club of Rome (a group of current and former politicians, United Nations administrators, diplomats, scientists, economists, and business leaders from around the globe), which was created to address the crises facing humanity and the planet (Club of Rome 2024). The report highlighted the physical limits on different finite natural sources and the limits on the ability of the Earth to absorb the pollution generated by those resources, which emphasizes the need to understand the ecological limits and design around them (Meadows et al. 1972). Both these studies raised the issue of unsustainable economic growth.

At the same time, the 1972 UN Conference on the Human Environment in Stockholm was the first global Earth summit to consider human impacts on the environment (UN 1972), which coined the term "eco-development" as an alternative in an attempt to reshape the pattern of the economy (Mellos 1988). The action plan included the need for alternatives to meet rapidly increasing urban transportation demands, including mass transport systems and services with reference to environmental development (UN 1972). In the 1980s, the World Conservation Strategy (WCS) was formed in collaboration with the International Union for Conservation of Nature, the United Nations Environmental Program and the World Wide Fund for Nature (WWF) with the objective of exploring strategies that aim to integrate economic and environmental management. This was when the first occurrence of "sustainable development" appeared, defined as "the need for economic development, with social and economic objectives, to take conservation into account by considering resource limitations and ecosystem carrying capacity" (Purvis 2019; IUCN 1980). It was not until the Brundtland Report (UN 1987) that sustainability was established as a critical part of economic development policy. This raised awareness of the need for integration between the economy and its reliance on Earth's ecosystems (Purvis 2019). However, sustainable development as defined by the Brundtland Commission has been considered to be open-ended, which led to multiple interpretations and in some cases misapplication (Daly 1996; Redclift 2005; Shi 2019).

The common understanding of sustainable development is presented through a Venn diagram first introduced by H. Daly (1997), where the "economy" sits nested within "society", which in turn is entirely within the "environment", such that the economy is a subset of society, which depend on and is constrained by the environment. Sustainable development was emphasised in the 1992 UN Conference on Environment and Development (Earth Summit) in Rio de Janeiro (UN 1992), Agenda 21 intended as a guide to future sustainable development (Rogers 2007; Purvis 2019). This was also the first conference to highlight the need for sustainable development within the transport sector, where one of the main objectives was, “to limit, reduce or control, as appropriate, harmful emissions into the atmosphere and other adverse environmental effects of the transport sector (paragraph 9.14) (UN 1992)”. Following this, led to the world's first legally binding treaty of the 1997 Kyoto Protocol to reduce greenhouse emissions (UNFCC 1997), the 2015 Paris Agreement held in December 2015 was the first global agreement to combat climate change signed by 195 countries (UNFCCC 2015). In 2016, the first Global Sustainable Transport Conference took place, which highlighted the need to shift away from the traditional approach to meeting the transport demand to towards one of the principles of sustainability, which includes avoiding inefficient or unnecessary travel, shifting travel and transport towards efficiency and environmentally friendly mode and improving the environmental performance of transport (UN 2016). Regardless of the progress made in terms of environmental concerns and sustainable development, the main issues affecting the ecosystem remained a global concern (Wang et al. 2022). The recent 2022 Stockholm Conference (UN 2022) highlighted the need for environmental sustainability for a healthy planet in face of the still growing environmental challenges, “triple planetary crisis of climate change, biodiversity loss and pollution, which had been identified as one of the major obstacles to sustainable development, and which contributed significantly to poverty, food insecurity, climate crisis and spread of diseases, overall consequences had brought the Earth dangerously close to tipping points beyond which there would be little chance of recovery.” The conference highlighted the need to adopt and implement policies to promote circularity, resource efficiency, regenerative production approaches and nature-based solutions (UN 2022).

Starting from the 19th century with the great horse manure crisis, towards 20th century smog-infused cities and today’s 21st century efforts to tackle climate change, there is a significant need to understand the environmental implications of the transport sector with the on-going societal growth (Johnson n.d; Agbugba 2019). This PhD explores the resource use and environmental implications for the case of the UK in light of the on-going transport transition to mitigate the effect of climate change.

1.2 Transport Transition in the UK

Since the arrival of light duty vehicles in the 20th century, the number of motor vehicles has risen from almost nothing to nearly 40 million today in the UK, dominated by cars (Department of Transport 2019; Department of Transport 2023). These technologies require energy, which usually comes from the burning of fossil fuels; however, these fuels are declining and cannot be sustained forever, in addition, the use of fossil fuels is the main source of GHG emissions to the Earth atmosphere (Cahill M 2010). Light duty vehicles are also responsible for a significant increase in congestion and other emissions such as nitrogen oxides and particulate matter, leading to deteriorating air quality, and concentrated exposure of pollutants in urban setting responsible for causing various health impacts (Brand and Hunt 2018; Bakshi 2019). In 2019 nearly about over one fifth of the UK total GHG emission were due to road transport, mainly from petrol and diesel passenger cars (Department of Transport 2019). The government targets to ban all diesel and petrol cars sales by 2030, and potentially have all zero emission vehicles by 2050 (HM Government 2020).

As the UK aims to move towards a cleaner energy system, including the decarbonisation of the electricity grid, the battery electric vehicle (BEV) is seen as a growing market within the transport sector (Faraday Institute, 2022). Although BEV addresses some of the environmental concerns (Ghosh 2020) for improving local air quality and reducing GHG emission at the point of use, switching technology alone may not represent the most environmentally sustainable pathway for the future transport system (Zhang and Fujimori 2020). The level of emissions generated by electricity grid used to power BEV depends on the grid mix technologies and plays a significant role in the extent of decarbonisation of the transport sector. Furthermore, the production of BEVs and their batteries remains to be of serious environmental concern due to the reliance on critical elements and the impact associated with their mining (Notter 2010; Xia 2022).

Hence, the shift from fossil fuel-based vehicles to a growing electric vehicle fleet based on traditional linear approach will entail significant amounts of raw materials (Hawkins et al 2012), energy consumption and environmental impacts for the production of BEVs and grid mix technologies to support its uptake (Xia and Li 2022). Strategies such as the uptake of shared mobility to minimize number of vehicles on the road and circularity for resources could prevent the wastage of raw materials and associated environmental and energy impacts (Church and

Wuennenberg 2019; HM Government 2021; HM Government 2022). This is further discussed in chapter 1, section 2).

Once BEV batteries are retired, they still have 70–80% of their original capacity available, which opens up opportunity to reuse battery in grid storage applications to support the transition to low carbon electricity system, whereas recycling batteries could prevent the reduction in critical battery metals. Furthermore, vehicles stay idle for most of the time, a transition toward shared mobility (SKIFT REPORT, 2013, Sopjani et al 2020) could provide an opportunity to increase vehicle productivity and reduce congestion. Fewer vehicles overall and reusing and recycling batteries at end of life could reduce pollution during production and wastage of resources and energy.

The next few sub-sections provide background context to several synergies that will play a potential key role in the transport pathways with a focus on the light duty vehicle fleet:

- Uptake of Electric Vehicles
- Evolution of the Electricity Grid Mix
- Battery Evolution
- Closing the battery loop
- Transition from Vehicle Ownership to Transportation as Services

1.2.1 Uptake of Electric Vehicles

In 2020, light duty vehicles (4-wheel vehicles that can carry up to 8 passengers) represented 82% (Department for Transport 2022) of the vehicles on the road, where around 98% of these vehicles either run of diesel or petrol fuels. Figure 1.1 represents fuel type and propulsion of new vehicles registered each year in the UK. Over the recent years the numbers of new electric vehicles, i.e., BEV and PHEV have started to rise so noticeably in 2020 despite the COVID 19 impact from March 2020 onwards. Whereas worldwide, 2 million BEVs and PHEVs were sold in 2019, out of which 74% of these vehicles were BEVs (Deloitte 2020).

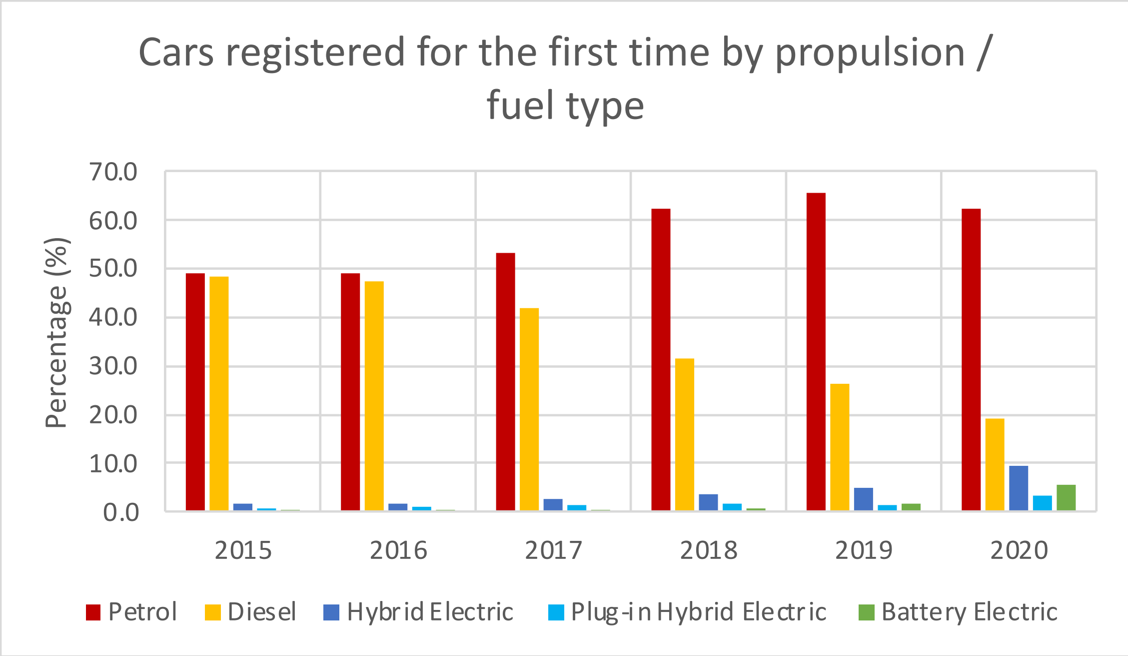

Figure 1.1: Car registered in the UK from 2015 to 2020, adopted from Department for Transport (2022).

The UK transition to BEV is still at its early stage. Currently, BEV represents a small share of the road transport market, however in the last five years, BEV sales have increased by 143% and they are expected to rapidly rise in the coming years (Department for Transport 2022). With policies on ban of fossil-fuelled vehicles, decreasing cost of batteries and increasing the number of charging points being installed around the UK, the growth of electric vehicles is inevitable (Blomberg NEF 2022). There are currently multiple initiatives taken to accelerate the deployment of electric vehicles worldwide: "The Electric Vehicles Initiative (EVI)", "The EV30@30 Campaign" and "Global EV Pilot City Programme" (IEA 2020). There is a wide possibility for the adoption of EVs based on different factors which such as, the grid mix, improvement in battery range, future battery prices, charging availability, growth of shared mobility services and efficient public transport (IEA 2022a; Deloitte 2020; Blomberg NEF 2022; FES 2022).

1.2.2 Electricity Grid Mix

The UK Government have committed to fully decarbonising electricity generation by 2035 (CCC 2022) just few years after the sales of all fossil fuel vehicles will end (HM Government 2021). GHG emissions reductions in the electricity sector will provide a low carbon energy source which could then support the decarbonisation of various sectors via increased electrification. The degree to which the use of BEVs can contribute to decarbonisation depends upon the electricity consumption of BEV and the percentage of fossil fuel in the electricity grid mix (Held and Baumann 2011, Faria et al 2013), as well as the life cycle emission of the electricity grid (Lucas et al. 2012). Increasing electricity demand due to EVs will mean greater energy investment and resource requirement to meet the required grid capacity (Lucas et al 2012). Nevertheless, several initiatives on city planning, such as 15-min city that aims to reduce vehicle use altogether and well-integrated shared transport systems, furthermore V2G to help reduce the peak energy demand can be seen as solutions to some of these concerns.

There is already wide deployment of renewable and low carbon energy resources taking place in the UK electricity grid, of which 50% and 51% of the electricity is generated by renewable and low carbon technologies in 2020 and 2023 respectively (excluding import) (National Grid 2021; National Grid 2024). The electricity generation profile for 2021 is represented in Figure 1.2 (National Grid 2021). Renewable sources are intermittent and, in some cases, unpredictable and unlike the conventional types of generations such as coal-fired power plants and natural gas power plants, renewable power cannot be dispatched on demand. In the future, a reduced presence of conventional dispatchable power plants is expected to not only make it difficult to provide the flexibility needed in the electricity grid but also cause a reduction in the grid inertia, hence making the grid more susceptible to instability (Ulbig et al. 2014). The Climate Change Committee (CCC) 2022 highlights one of the challenges of 2030 will be the operation of the low-carbon electricity system which would include energy storages and ways to smooth demand to avoid excessive spikes (CCC 2022). There has been a lot of focus on grid balancing strategy, increasing efficiency of supply use, and reducing consumption to help reduce the overall emissions, ensure system stability, and deliver power from the grid in an affordable manner whilst meeting the carbon targets (HM Government 2018, National Grid 2019; CCC 2020). A major change in the evolution of the electricity sector is the transition of distribution network operators to distribution system operators (Western Power Distribution 2020); this will allow distributed energy resources such as electric vehicles and grid storage batteries to participate in enhancing the utilisation of renewables and thereby increase renewables in the grid mix (Element Energy 2019; National Grid ESO 2020; Ofgem 2023). Furthermore, there is an ongoing integration of various energy networks, such as gas and hydrogen, to improve flexibility and ensure a balanced and resilient in whole energy system (FES 2021). This integration aims to enhance the overall efficiency and reliability of energy supply systems. However, this thesis focuses exclusively on the electricity grid mix within the context of an evolving energy landscape.

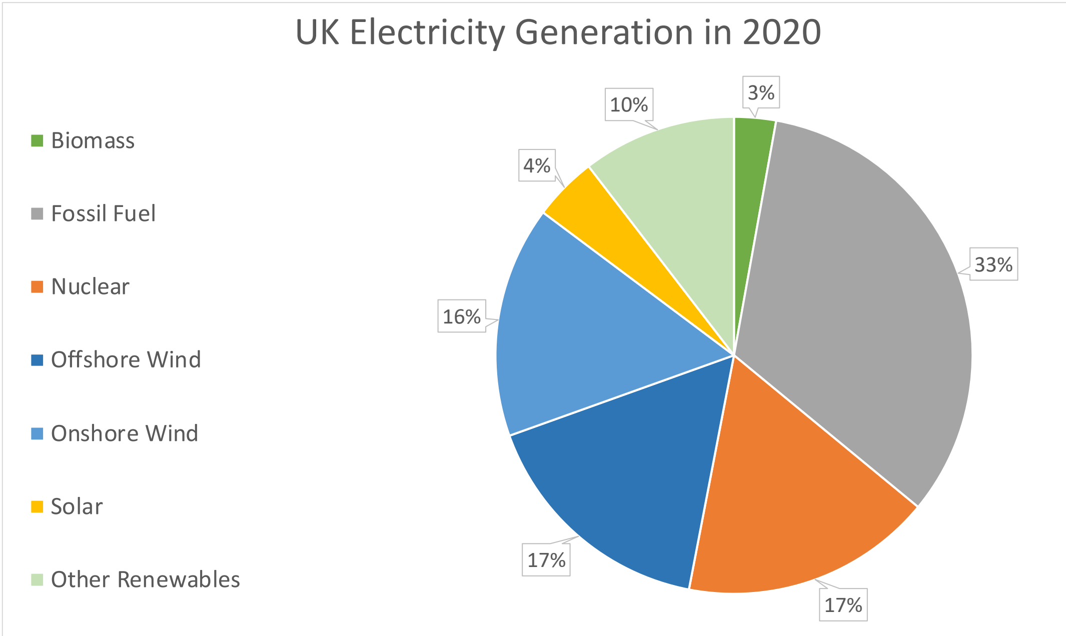

Figure 1.2: Electricity generation breakdown (%) in 2020, adopted from National Grid (2021).

At the same time, in the future there will be large numbers of end-of-life (EoL) BEV batteries, with potential to provide energy storage for the grid (Engel et al 2019; Abdelbaky et al 2020; Greim et al 2020). Once EV batteries useable capacity is degraded by 20 to 25%, they are often considered unsuited for the use in EV, but they still have significant capacity for storage purposes (Haram et al 2021; Xu et al 2023). With EV numbers increasing rapidly, this could amount to terawatt hours of unused energy storage capacity which could provide up to ten years of services in a stationary battery energy storage system (Connected Energy 2022; IEA 2022b).

National Grid Electricity System Operator, the UK's largest utility company, produced "Future Energy Scenarios (FES)" which explores pathways to achieve fully decarbonised energy system by 2050 (FES 2022). Four different scenarios are outlined depending on policy support, customer engagement, technological development, economic growth, and energy efficiency: (1) "Leading the Way" (2) "Consumer Transformation" (3) "System Transformation" and (4) "Falling Short"**.** The most ambitious pathway is "leading the way" reaching decarbonisation of the electricity power sector by 2033 and full Net Zero of the whole energy system by 2047. Where Net Zero, is the commitment to bring all GHG emissions to zero from 1990 levels (UK Parliament 2023). The other two scenarios "Consumer Transformation" and "System Transformation" reaches Net Zero by 2050. Decarbonisation is slowest in the Falling Short Scenario, with emissions reduced by 80% from1990 levels. The "Leading the Way" Scenario sees the greatest increase in battery storage requirements from 1.6 GW in 2021 to 35 GW by 2050. Repurposing used EV batteries could generate significant value and benefit the grid-scale energy storage market (Engel et al 2019). Trials with second-life batteries have already begun. Currently the largest second-life initiatives are: "Daimler Mobility House" with 13 MWh of second-life batteries used for compensating power fluctuations (Daimler 2015); "Advanced Battery Storage" launching this year with 60 MWh of second-life batteries to facilitate the integration of renewables by 2020 (Groupe Renault 2018); and the "SmartHubs Connected Energy" pilot project launched in 2021 with 14.4 MWh of second-life batteries to provide grid balancing services (Connected Energy 2020).

1.2.3 Battery Evolution

Batteries play a vital role in enabling the electrification of vehicles, and the first rechargeable battery was invented in 1859 by the French physician Gaston Planté, known as the lead acid battery (Kurzweil 2010). Rechargeable lead acid batteries were used to power the first electric vehicles (Department of Energy 2014; Foresight 2018). The vehicles in the late 19th century and 20th century had a range of 50 miles, as the range at the time was not a problem until the expansion of motorways, which required vehicles to travel a higher distance (Department of Energy 2014; Warner 2015). Later on, as the price of oil increased in the late 20th century (1968 – 1973) and zero emission vehicle mandate was introduced in 1996, the trend to explore low emission alternatives begin and this led to the introduction of 'nickel metal hydride (NiMH) battery' BEV, with a higher range (70 to 160 miles) and half the battery weight as compared to the lead acid BEVs (Warner 2015). In the late 20st century, 1991 came the introduction of lithium-ion batteries (LIBs) invented by the John Goodenough group, this led to the higher travel range electric vehicle first mass manufactured by Tesla Motors in the early 21st century (Warner 2015; Manthiram 2020). These vehicles were able to travel a range up to 200 miles (Schneider 2007).

The use of LIB brought unique advantages, as lithium is both the lightest and is one of the most electropositive metals in the periodic table. The low density/specific gravity of lithium (Lide 2015) gives lithium-ion batteries higher gravimetric energy density and voltage, as compared to the other rechargeable batteries currently available commercially, such as silver–zinc and nickel–metal-hydride and NiCd (Kennedy et al. 2000; Tarascon & Armand 2001). LIBs are also known for withstanding a large number of charge cycles, low self-discharge rate and for little or no memory effect (Ding et al. 2019). The battery performance, safety, reliability, cost and climate targets are the main driving factors for the adoption of BEVs. The battery performance depends on the energy density which dedicates the driving range of the battery, power capabilities which entails how fast can BEVs accelerate and how long a battery can last. LiBs are currently the main battery technology used in the BEVs (Zeng et al. 2019; Breiter et al. 2022). The working of LIBs and comparison of their different cathode chemistry is provided in Appendix A.

There has been on-going research on developing new high-capacity batteries such as Ni-rich NMC cathode material, silicon-based anode material and new battery chemistries including all solid-state batteries (ASSBs) for future automotive industries to achieve higher performance of BEVs (Ding et al. 2019b). Nissan has already set its launch for the first production of solid-state batteries for its BEVs in 2028 (Tisshaw 2023). Similarly, there is on-going development for sodium-ion batteries due to cheaper, abundance and less toxic materials used in its making (Liu et al. 2022). To date, Nickel Cobalt Aluminum oxide (NCA) and Nickel Manganese Cobalt oxide (NMC) are the more dominate cathode materials for a BEV battery and will still dominate the BEV market in the foreseeable future given their maturity before new types of battery chemistries become mature enough for the automotive application (Zeng et al. 2019). Whereas Lithium Manganese Oxide (LMO) and Lithium Iron Phosphate (LFP) based LIBs are the dominant technologies for grid storage applications.

1.2.4 Closing the Battery Loop

Over the years the cost of LIBs has decreased quite significantly. On average, the battery accounts for 30 to 40% of the cost of an EV (Faraday Institution 2022, Wentker et al 2019) and the raw material cost 50-70% of the cell manufacturing in the battery pack (Houache et al. 2022). The raw material cost of the cell depends on the availability of the material. A more abundant material would have a slightly lower raw material cost. Many elements are used in lithium-ion batteries, this includes lithium, cobalt, nickel, manganese, graphite, aluminium and copper some of which are considered to be critical (Olivetti et al. 2017). There are already concerns over increased demand for batteries due to the supply risk associated with battery materials as some of the materials are not mined in large amounts, limited by the number of reserves currently available to mine, or are mined in the countries with high geopolitical risks (European Commission 2017, Olivetti et al. 2017, Mayyas et al. 2019). Cobalt and lithium are one of the critical materials used in batteries, while on-going efforts are taking place to reduce cobalt content in batteries and increase the use of high nickel cathodes (NCA and NMC 622 and NMC 811). This on the other hand could spike the nickel demand that may have consequences on the supply of nickel (Houache et al. 2022).

The circular economy is one of the main drivers for decarbonisation (Barrett and Scott, 2012), which aims ‘to keep resources in productive use for as long as possible, extract the maximum value from them whilst in use, then recover and regenerate products and/or raw materials at the end of their service life’ (Velenturf and Purnell 2021). Closing the battery loop can provide an opportunity to reduce the cost of extracting new raw materials, alleviate some of the supply risk of critical raw materials and the environmental and energy impacts associated with the production of new batteries, thus accelerating the transition to BEVs. There are three potential pathways for EoL BEV batteries: (1) repairing the battery packs for use in EVs, extending their lifetime. (2) Reusing the batteries in second-life stationary applications such as grid or home storage. (3) Recycling battery raw materials for the manufacturing of new EV batteries that can ease pressure on specific raw materials (Nordelöf et al. 2014; McKinsey 2022b). The extent to which these pathways or a combination thereof could help reduce demand on specific raw materials is uncertain as grid storage batteries do not necessarily require the high energy density provided by NCA and NMC technologies. There is also delay between the time when batteries are manufactured and available to be recycled after BEV EoL. When considering second-life use of batteries, this could prolong the delay when raw materials are available from recycling, thus it is uncertain what implication this may have on the raw material demand.

1.2.5 Shared Mobility

Travel and car ownerships have increased significantly over the years in most of the developed countries and has become one of the major needs of an urban lifestyle (Banister, 2005). Car dependence in the UK has grown massively since the 1950s as motorisation expansion has grown which has led to various issues, such as congestion, traffic collusion and series of environmental impacts (Banister 2005, Department of Transport 2019). Department of Transport (DfT) estimates the car fleet will grow from the current 27 million vehicles to somewhere between 37 and 40.5 million by 2050 (Department for Transport 2018). With UK goal to achieve net zero, this will require massive production of BEV and much greater generation capacity in the electricity grid to support the transition of growing BEV fleet (Marsden et al. 2019). Furthermore, private cars are only in-use for 3–4% of the time. During peak times in the morning, the largest proportion of the car fleet in use at any one time is just 15%, and 62% of car trips are done with a lone driver

(Marsden et al. 2019). There is substantial potential to utilize energy and resource in a more efficient way.

“Shared Mobility” is mobility in a shared economy, where the latter is defined as a *“phenomenon of turning unused or under-used assets owned by individuals into productive resources”* (Santos 2019). Shared Mobility can be seen as a way towards a more environmentally friendly transport system if they are adopted successfully, allowing a more intensive use of fewer vehicles (Santos 2019), thereby contributing to less congestion and resource use, whilst also being seen as a more cost-effective alternative in some cases (Marsden et al 2019). Nevertheless, there are several barriers to the successful adoption of shared mobility, such as concerns about its rebound effects, which might impact the use of more sustainable transport modes such as buses and bicycles. Additionally, there are concerns about the frequent need for replacement due to the intense usage of shared mobility options. These are further discussed and explored in chapter 2 section 2. Advances in shared mobility concept opens an opportunity for self-driving prospects, increased safety, and efficient travel patterns for a better development of vehicle usage (Banister 2005; Goodall et al 2017).

Another concept related to shared mobility is Mobility as a Service (MaaS) or Transport as a Service (TaaS) (Butler et al 2021) which is defined as “*a digital interface to source and manage the provision of a transport related service which meets the mobility requirements of a customer’* (Foresight 2018b). TaaS focuses on making existing transportation network more efficient and user-friendly by integrating on demand and shared services in a single platform which can be tailored to customers choices and convenience (Goodall 2017; Butler et al 2021). The purpose of TaaS is to provide a shared and flexible mobility which attempts to minimize the private use of light duty vehicles, hence enabling a shift away from private vehicle ownership and potentially reducing the number of vehicles on the road, to solves challenges regarding congestions in cities and making transport more accessible (Butler et al 2021; Jittrapirom et al 2017).

The recent report released to the UK parliament on progress to reduce emissions (CCC 2022b) highlighted the need to include shared form of transport in the new upcoming planning reforms. There are already several pilots shared electric mobility schemes being proposed or running in the UK such as proposed E-mobility hub project to start in Nottingham and Derby to facilitate different EV sharing schemes (Cenex 2022), co-car sharing club in Exeter (CoMoUK 2023) and car club and bike share mobility hub for express bus services to Edinburgh (CoMoUK 2023). Furthermore, cities are changing their approach to urban development to reduce carbon emissions.

The 15-minute city is a potential framework that could help achieve support the uptake of TaaS. Glasgow is looking to create 28 Liveable Neighbourhoods, providing access to key services within a 20-minute walk or public transport journey (SKEDGO 2022). The combination of shared mobility service with electrified transportation system can be seen as an opportunity to use less amount of primary energy and resources whilst reducing carbon emission.

1.3 Research Questions

This thesis is part of the Faraday Institution - ReLIB (Recycling and Reuse of EV Lithium-ion Batteries) project. The project investigates how to reuse the batteries and their materials, to make better use of global resources, and ultimately increase the impact of batteries in improving air quality and decarbonization. This includes understanding the sustainability supply challenges of battery cathode elements and other critical elements for supporting the low-carbon transition in addition to the potential role of shared mobility towards mitigating environmental impacts.

An increasing number of countries around the world are planning full phase out the sale of new ICEVs, UK is one of the 9 countries in Europe aiming to ban ICEVs in near terms (ICCT 2021). With the UK target to achieve Net Zero, i.e., 100% reduction of GHG emissions by 1990 level, there will be a wide deployment of low carbon energy sources to meet the carbon target at the 'point of use'. Therefore**,** net zero do not count for indirect GHG emissions, other environmental impacts or the availability of critical resources. To address the future environmental transport challenges, it is vital to understand and quantify environmental trade-off of the future transition to BEVs from a whole transport system perspective, this is taking account of the entire light duty vehicle fleet in the combined transport and electricity system. The UK aims to achieve battery circularity and improve vehicle utilisation (Department of Transport 2021); therefore, it has been chosen as a case-study to understand the implications of fleet transition and explore strategies for resource efficiency.

The overall aim of this research is to investigate the energy and environmental implications of a transition to BEVs in the UK over the time period to 2050, and the associated demand for key battery materials when considering simultaneous strategies for resource efficiency, closed-loop battery recycling, second-life battery re-use, and the uptake of shared mobility.

The thesis will start of by undertaking a systematic review of the existing literature to understand the sustainability supply challenges related to the supply of raw materials for the wider low-carbon transition. Then, it will focus on conducting an environmental analysis on the transition to BEVs in the UK, with focus on resource efficiency strategies for BEV batteries and their materials. A material flow analysis will be carried out to quantify the raw material demand of battery in the combined electricity grid and transport sectors. Based on these findings, a full prospective life cycle assessment will then be used to assess the environmental performance for the BEV transition. The combination of both will allow to quantify the environmental concerns on resource constrains and ecological consequences associated with the transition of the light duty vehicle fleet and the result of closing the battery loop and the uptake of shared mobility to reduce the number of vehicles on the roads. The research presented in this thesis provides scientific evidence to inform strategies for decision-making on environmental policies for energy and transport pathways by providing an understanding on the environmental impact of these strategies to the transition of light duty fleet.

The research questions are defined as following:

- What are the overall supply challenges of critical raw materials required in the transition to EVs and low carbon electricity grid mix?

- Does reusing retired EV batteries in second-life grid storage delay when battery raw materials become available from recycling? What implication may this have on the raw material requirement for EV batteries?

- When considering the effect of minimizing the use of private vehicles through uptake of TaaS, what implication may this have on the raw material requirement for BEV batteries and battery recycling opportunities?

- What is the overall energy and environmental trade-off of the transition to BEV?

1.4 Thesis Structure

Chapter 2: Represents a systematic review on the current challenges with the supply raw materials for supporting the low carbon transition to address the first research question. The second part represents the literature groundwork on the evolving landscape of EV batteries and end-of-life treatments, current shared mobility strategies and life cycle assessment methods to form the basis for developing the research framework. This chapter is largely based on the following journal article: Kamran, M., Raugei, M., & Hutchinson, A. (2023). Critical Elements for a Successful Energy Transition: A Systematic Review. Renewable and Sustainable Energy Transition, 100068. https://doi.org/10.1016/j.rset.2023.100068

Chapter 3: Describes the methodology to understand the consequences of resource use and environmental trade-off over time to answer the following three research questions. The section outlines the research framework on carrying out a material flow analysis (MFA) and life cycle assessment (LCA) of the transition to electric mobility within the context of UK passenger fleet.

Chapter 4: Carries out the dynamic MFA to track the changes in various mass flows of passenger vehicles, battery requirement and all the key lithium-ion battery (LIB) metals. This later forms the inventory groundwork for the overall life cycle assessment for passenger vehicle fleet in section 6). The chapter is based on the following journal article: Kamran, M., Raugei, M., & Hutchinson, A. (2021). A dynamic material flow analysis of lithium-ion battery metals for electric vehicles and grid storage in the UK: Assessing the impact of shared mobility and end-of-life strategies. Resources, Conservation and Recycling, 167, 105412. https://doi.org/10.1016/j.resconrec.2021.105412

Chapter 5: Carries out the LCA of the evolving electricity grid mix up to year 2050, which focuses on the electricity generated and delivered domestically within the UK. LCA of the grid mix is an important environmental factor when considering the transition to BEV and forms the part of the section 6 later. The chapter is based on the following journal article: Raugei, M., Kamran, M., & Hutchinson, A. (2020). A prospective net energy and environmental life-cycle assessment of the UK electricity grid. Energies, 13(9), 2207. https://doi.org/10.3390/en13092207

Chapter 6: Carries out the LCA of the passenger vehicle fleet to assess the role of resource strategies on the overall environmental and energy trade-off of the transition to BEV. The boundary of assessment is expanded to the degree necessary to capture the interlinkages that have direct and indirect consequences of the life cycle impacts of transition to passengers EVs. The chapter is based on the following journal article: Raugei, M., Kamran, M., & Hutchinson, A. (2021). Environmental implications of the ongoing electrification of the UK light duty vehicle fleet. Resources, Conservation and Recycling, 174, 105818. https://doi.org/10.1016/j.resconrec.2021.105818

Chapter 7: Discussion on key research questions on the impact of battery circular strategy and shared mobility on consequences of resource use and environmental implications.

Chapter 8: Provides conclusions, outlines the novelty of the research, and presents directions for future research.

2 Literature Review

A systematic study of the literature was carried out to study the current issues related to challenges with the supply of raw materials for supporting the low carbon transition to answer the first research question. The second part of the literature review focuses on gathering insight on the current shared mobility strategy to develop scenarios for BEVs. The third part explores the evolving landscape of EV batteries and their end-of-life treatments. The fourth part of the literature review analyses the environmental impact and LCA approaches associated with the BEV transition and identifies gaps for developing a research framework for the methodology section. Key points arising from these separate parts are summarised at the end of this chapter.

2.1 Critical Chemical Elements for a Successful Energy Transition: a Systematic Review

The transition to a low-carbon energy future requires large amounts of many raw materials. Some of these materials are deemed critical in terms of their limited availability, concentrated supply chain networks, associated environmental impact, and various social issues. Acknowledging the significant dependency on raw materials for future energy scenarios, this section presents a systematic review of the existing literature to identify the barriers, solutions proposed, and the current research gaps associated with the supply of a range of critical chemical elements. Chapter 2, section 1 is based on Kamran et al (2023)[1].

[1] Kamran, M., Raugei, M., & Hutchinson, A. (2023). Critical Elements for a Successful Energy Transition: A Systematic Review. Renewable and Sustainable Energy Transition, 100068. https://doi.org/10.1016/j.rset.2023.100068

2.1.1 Systematic Review Process

Renewable electricity technologies (among which primarily wind and photovoltaics) and the electrochemical storage technologies (among which, currently, primarily lithium-ion batteries) which are required to buffer generation intermittency and to power EVs, are dependent on a supply of a range of raw materials, and as a result the global demand for the latter is expected to surge in the coming decades. The following groups of elements are recognized as essential for the transition energy technologies:

- Lithium, Cobalt, and Nickel – used in varying proportions in most cathode formulations for lithium-ion batteries (LIBs).

- Neodymium, dysprosium, and other "rare earth elements" (REE) – used in permanent magnets (PMs) for electric motors and wind turbines.

- Silver, Tellurium, Selenium, Gallium, Indium, and Cadmium – used in a range of photovoltaic (PV) technologies, including crystalline silicon (c-Si), and Cadmium Telluride (CdTe) and Copper Indium Gallium diselenide (CIGS) thin films.

- Platinum and other "platinum group metals" (PGM) – used in catalysts for water electrolysis and "green" hydrogen production.

- Copper – widely used in virtually all electrical applications.

The systematic review process was conducted to answer the first research question following the Preferred Reporting Items for Systematic reviews and Meta-Analyses (PRISMA) methodology (Page 2021). Details of the process are contained in Appendix B along with the statistical analysis. Two main search engines were selected and used to retrieve peer-reviewed journal papers, reviews, and editorial materials: Google Scholar and Web of Science. In addition, three publisher-specific search engines relevant to the field were also identified and used in parallel: Science Direct, Nature Publishing Group, and MDPI. Searches were done using the "topic" field where possible, which includes title, abstract and keywords. However, the Google Scholar and Nature Publishing Group search engines are limited to searches in the "title" or "article" fields only, and hence, for better comprehensiveness, the latter field was used in these cases. The MDPI search engine is instead limited to the "keyword" and "title" fields, both of which were employed. Table 2.1 synthesizes the literature collection and screening process, including the list of keywords used and the paper tallies per search engine, at each stage of screening.

Table 2.1: Literature identification/collection process and subsequent screening stages.

Search engine	Google Scholar	Web of Science	Science Direct	MDPI	Nature
Date of search (last updated)	13/01/22	14/01/22	14/01/22	14/01/22	14/01/22
Search fields → **Search keywords/phrases ↓**	Article	Topic	Topic	Keywords and Title	Article
"Critical Mineral*" OR "Critical Metal*" AND "Energy System" OR "Energy Transition"	365	21	18	20	6
"Strategic Mineral*" OR "Strategic Metal*" AND "Energy System" OR "Energy Transition"	167	0	4	1	0
"Key Mineral*" OR "Key Metal*" AND "Energy System" OR "Energy Transition"	71	3	5	37	2
"Mineral Supply*" OR "Metal Supply*" AND "Energy System" OR "Energy Transition"	495	3	7	6	7

“Mineral Availability” OR “Metal Availability” AND “Energy System” OR “Energy Transition”	151	0	5	3	1
“Rare Earth Element*” AND “Energy System” OR “Energy Transition”	450	25	2	4	2
Total results from searches	**1699**	**56**	**37**	**71**	**18**
First Screening (Titles and Abstracts only)	208	34	10	12	7
Second Screening (Full text)	179	32	7	12	7
Duplicate removal	161				
Third Screening (Papers that fall under the research question)	**100**				

It is acknowledged that the very choice of search keywords/phrases inevitably influences, to some degree at least, the results of the search in terms of the articles that are returned, and hence also of the breadth of the information covered therein. In order not to pre-emptively bias the process in favour of any particular resource, the choice was made to employ general search terms such as, e.g., “critical mineral”, “strategic mineral” and “key mineral” instead of specific named elements, metals or minerals (the sole exception to this rule was the inclusion of the phrase “rare earth element”, due to the fact that, despite the specific chemical meaning of this phrase, referring to Lanthanides, the phrase is sometimes also used loosely to refer to other scarce elements in the earth’s crust). Also, the choice was made to mandate the inclusion in each search of either of the terms “energy transition” or “energy system”, in order to guide the literature collection towards those articles that specifically dealt with these core aspects of the intended focus of this review and reduce out-of-scope bycatch. At the same time to reduce the irrelevancy of search results, filters were applied.

The initial results from the search engines were then subject to a three-stage screening process. The first stage entailed reading the abstracts and titles only, and discarding those documents that clearly did not deal with issues of resource criticality for the energy transition. The second screening involved removing duplicate entries which was done by using the dedicated tool in the EndNote software, and those results that were not journal articles were also removed, such as book sections, reports, and theses, which led to a total of 161 papers (listed in Appendix B). The third and final screening stage required skimming through the full text of the articles, aimed at discarding those papers that were found to be not relevant to supply issues.

2.1.2 Review of Critical Chemical Elements

A full review of battery chemical elements follows because of their importance in later chapters, followed by the summary of other critical chemical elements provided in Appendix B.3. A complete systematic review for other chemical elements can be found at (Kamran et al 2023).

2.1.2.1 Battery Elements (Lithium, Cobalt and Nickel)

With the transition to low carbon energy system and transport there will be considerable demand for battery metals. The metals discussed are lithium, cobalt and nickel which are considered critical for the development of lithium-ion batteries used for stationary storage applications and EVs.

2.1.2.1.1 Cobalt

Cobalt is extracted in around 14 countries; more than 70% of it is supplied from sedimentary deposits in the Democratic Republic of Congo (DRC), which represent almost 46% of the global reserves, followed by reserves in Australia, Indonesia, and Cuba (Petavratzi et al 2019). For clarification, the term "reserve" is the economically mineable mineral from discovered deposits, which depends on the technology used and the market value of the mineral, whereas "resource" is the estimated total amounts of discovered and yet-undiscovered deposits. Given that in most cases cobalt is a by-product of the extraction of copper or nickel, the processing is not optimised for cobalt recovery, therefore some of the cobalt ends up in tailings and slags after ore processing and refining. It is estimated that around 40% to 60% of cobalt content is lost during the concentration step, and specifically for the ores found in Australia, it is estimated that the recovery of cobalt is 40% (Petavratzi et al 2019). Cobalt, along with other battery chemical elements, may also potentially be sourced from deep sea mining. Deep sea mining in the Clarion-Clipperton Zone (a geological submarine fracture zone of the Pacific Ocean, with a length of around 7,000 km) could contain 5 times the cobalt reserves on land while potentially causing significantly lower carbon emissions per mass of metal extracted (Paulikas et al 2020; Levin et al 2020). However, the full extent of environmental implications of deep-sea mining are still unknown and the biodiversity in the zone is insufficiently assessed (Levin et al 2020).

Cobalt is considered one of the most critical elements for EV development in the automotive sector (Ortego et al. 2020; Elshkaki 2020). The need for high energy density batteries in recent years has led to an increased production of cobalt. However, there is large dependency on the centralised production of cobalt in Congo and its concentrated refinement in China, which creates geopolitical concerns over the supply of this element (McLellan 2019; Lee et al. 2020a). This is mainly due to the long history of instability in Congo, and the growing influence of China on the cobalt supply chain, allowing these minerals to be more susceptible to economic and political conflicts (Månberger & Johansson 2019). China is also a major manufacturer hub for BEVs and LIBs, representing 45% of global EV sales in 2020 (IEA 2022), followed by Europe and the USA, which also have a growing number of EV sales. These latter regions will be the largest consumers of cobalt without sufficient domestic production; hence it is important for these regions to implement recycling systems, not only to prevent wastage of material but also to diversify the production of cobalt (Seck et al. 2022).

When looking at the global picture, the future demand for cobalt may grow nine-fold from 2020 to 2050, by which time up to 64.5% of cobalt could be required by the transport sector (Seck et al. 2022). In terms of geological availability, studies that conducted supply vs. demand analysis for cobalt showed that, without considering the on-going reduction in cobalt content in batteries and the role of recycling, the future demand for cobalt would undoubtedly exceed its current reserve level before 2050 (Månberger & Stenqvist 2018; Watari et al., 2018; Klimenko et al., 2021; Seck et al. 2022). When considering the future reduction of cobalt content in batteries (up to NMC 811 cathode formulations), Seck et al. (2022), estimated in their scenarios that around 26% (350kt) of cobalt can be saved by 2050. According to their analysis, the yearly demand for cobalt could decrease by 13% by moving towards increase public and non-motorized transport; however, when considering the demand for cobalt from various end uses, the reserve is still expected to be exceeded by 2050, and 61.2% of cobalt resources in 2013 would be depleted by 2050 (Figure 2.1). Klimenko et al. (2021), examined the availability of cobalt by comparing future reserve estimates using historical trends. They analysed the requirement for cobalt considering both recycling, and the shares of cobalt-free and low-cobalt EV batteries. They found that the demand for cobalt in BEVs will hardly exceed a quarter of the prospective reserves by 2050, and by the year 2100 recycling will limit the demand to 55% of the prospective reserves, if the recycling rate for cobalt is improved to 50% by the middle of the century, from the current 30%. According to the same authors, the future availability of cobalt will not just depend on aggressive reductions in cobalt content, but also on a move towards sustainable mobility modes, development of mining technologies, exploration and increase in efficient recycling facilities.

Lèbre et al (2020) carried out a global assessment of environmental, social and governance (ESG) risks associated with energy transition metals. Their findings indicated that these risks are significantly higher for cobalt than for lithium, mainly due to the social impact associated with cobalt mining. The social concern for cobalt mainly stems from artisanal mining, which makes up 20 to 40% of cobalt production in Congo (USGS 2022; Petavratzi et al 2019). Without establishing responsible sourcing practices, artisanal mining can lead to compromising the general well-being of the workers for the short-term economic prosperity of the mining and trading industries. To prevent or reduce risks associated with mining health and safety and child labour, mining companies are required to formalise artisanal mining to provide standards for human security and thus ensure a more ethical and sustainable supply of cobalt (World Economic Forum, 2016). However, formalising artisanal mining has yet to ensure this (Rachidi et al. 2021; Calvão et al. 2021). In light of the two formalization projects of artisanal mining in Congo, namely Kasulo and Mutoshi, corporate engagement with artisanal miners increased their ability to source cobalt legally; however, these projects also shifted the risk of price fluctuation to artisanal miners, who are paid based on production output (Rachidi et al. 2021; Calvão et al. 2021). According to Jones et al. (2020), cobalt being a by-product of nickel and copper makes supply and prices more volatile; this can cause the number of artisanal miners registered with mining cooperatives to change dramatically depending on the market price of cobalt. Artisanal miners are paid lower incomes compared to trading professionals, and this is usually justified by the need to provide them with training, free personal protective equipment (PPE) and health system. However, it was pointed out that workers' safety is sometimes still not ensured, such as in the case of Kasulo. Also, in the event of lower cobalt prices, miners are mostly deprived of a fair price for their work (Calvão et al. 2021). Hence, such circumstances translate into lack of well-being and financial insecurity for artisanal workers, which still needs to be addressed. In terms of environmental and health risks, a survey conducted by Sovacool (2019) highlighted serveral issues, among which the contamination of rivers by washing of cobalt or waste dumping by artisanal miners, tailings from large mining site causing both air and water pollution, and the spread of diseases in mining camps due to lack of ability to maintain hygienic conditions.

2.1.2.1.2 Lithium

The major lithium resources are found in the so called "lithium triangle", which comprises regions of Bolivia, Argentina, and Chile that are rich in brine deposits, followed by regions with hard rock lithium deposits: Australia and China. In recent years, hard rock deposits have come to dominate the production of lithium, in the form of lithium concentrate which is then converted in a refinery plant to either lithium carbonate or lithium hydroxide (Petavratzi & Josso 2021). This was not the case a few years back, when lithium from brine deposits represented the primary source of lithium, commonly traded as lithium carbonate. Lithium hydroxide has higher lithium content over lithium carbonate; hence it is preferred by lithium-ion battery (LIB) manufacturers; however, converting lithium carbonate from brine into lithium hydroxide adds extra cost to the refining (Graham et al. 2021). China currently refines 75% of hard rock lithium from Australia and 25% of brine from lithium triangle countries. However, the shift in production from brines to hard rock, and the very concentrated refinement in China, entail an increased risk of supply chain disruption for other region planning on expanding their own battery manufacturing capacity (Hache et al. 2019; Heredia et al. 2020; Graham et al. 2021).

The demand for lithium for rechargeable batteries is expected to increase quite significantly in the coming years. Viebahn et al.(2015) estimated lithium demand for stationary storage and found that the demand is relatively low and not critical; however, the growing demand for lithium for BEVs may create shortages in the availability of lithium. Based on multiple studies on material demand projections, it was found that lithium production for BEVs may exceed the resource level by 2100, or the reserve and production level by 2050 (Junne et al. 2020; Hache et al. 2019; De Koning et al. 2018; Tokimatsu et al. 2018). This suggests the current production and reserves will not be sufficient to meet the growing demand for lithium. In the short term, lithium supply and demand could be matched by increasing the production rate and scale, thereby reducing the near future supply risk but new production start-up, which could take up to 10 years, and additional strategies would have to be implemented to cope with the long-term lithium demands (Hache et al. 2019, Greim et al. 2020). Authors examining scenarios for the future energy use of lithium suggest that key factors in balancing lithium supply and demand in the long term will be: developing an efficient recycling system, increasing material utilization efficiency, substituting demand for lithium by diversifying transportation technology such as developing new battery chemistries, and, lastly, limiting light duty vehicle stock growth by spatial planning and promoting improved public transportation and shared mobility (Greim et al. 2020; Junne et al. 2020; Jones et al. 2020; Klimenko et al. 2021) pointed out that the current lithium recycling rate is around 3%, and that it should be increased to at least 30% by the middle of the century to overcome lithium shortage based on their scenario projection. Watari et al. (2020) indicated that, by considering recycling and technology advancements, the divergence between lithium supply and demand can be reduced significantly.

In terms of environmental issues, concerns over the intensive water use for lithium brine extraction and purification processes has been raised by several authors (Hache et al. 2019; Lebre et al., 2020; Heredia et al. 2020; Mulvaney et al. 2021). Water use is also seen as a significant social issue due to the pre-existing water stress around salt lakes experienced by local and indigenous communities. The extraction of brine water from surface and underground deposits to fill the evaporation ponds has led to ongoing groundwater depletion; further concerns are the conversion from lithium brine to lithium carbonate, causing chemical leakage into the groundwater (Heredia et al. 2020; Mulvaney et al. 2021). Based on a social study on lithium brine extraction conducted by Liu & Agusdinata (2021), curbing mining water demand could significantly reduce the impact on local communities. Mulvaney et al. (2021) suggested mining industries must aim to eliminate the use of freshwater and waste discharges. Lithium activity developed within the lithium triangle states also raises some concerns addressed by local and indigenous communities in terms of access to safe drinking water, land rights of communities, access to a safe environment (Heredia et al. 2020; Liu & Agusdinata 2021). The increase in social stress may result in strikes that could have far-reaching impacts on the supply chain (Liu & Agusdinata 2021).

2.1.2.1.3 Nickel

Nickel reserves have increased by more than 10 Gt over the past 10 years (USGS 2022; USGS 2011). In terms of production, Indonesia is the largest producer of nickel, accounting for 37% of the total global nickel production in 2021, followed by the Philippines at 13% (USGS 2022). China, Korea, Australia, and Indonesia are found to be the most relevant countries for the nickel supply chain network (Tian et al. 2021). Currently, two-thirds of the nickel produced is used for stainless steel (Nickel Institute, 2016). Nickel is an essential element for battery technologies used in BEVs, and battery chemistries are moving to higher nickel content. Wind turbines will also demand nickel in large amounts, but less than for the growth of the EV sector (Calvo & Valero, 2021).

Some studies examined the demand for nickel based on energy technology or vehicle projection and found that nickel availability is not a constraint for the transition to a low-carbon energy system (De Koning et al. 2018; Bobba et al. 2020a). However, as for most materials with structural applications, the increase in demand for will not only depend on the energy transition but also on population and economic growth in developing countries (Henckens & Worrell 2020). Neglecting to take account of this may result in underestimating the demand for Nickel. Guohua et al. (2021) estimated the growth in nickel demand in China to 2050 for energy technologies, the vehicle feet and several applications based on historical trends in population and economic growth, and they found that the cumulative demand in China is expected to reach between 59% and 79%, or between 21% to 55%, of its global reserves in 2050, respectively without or with consideration of secondary supply (scrap and recycling). More than 50% of the global reserve being consumed by a single region clearly suggests that nickel is critical in terms of geological availability.

In terms of social and environmental impacts, nickel has higher land disturbance during mining, compared to lithium or cobalt (Lebre et al. 2020). Furthermore, the decreasing nickel ore grades are another significant concern, as these would mean more energy investment in the extraction process, and higher emissions and water use (Henckens & Worrell 2020). However, some authors have pointed out that thanks to an increasing secondary supply of nickel for energy transition technologies, the overall envionmenal impacts and water demand would decrease significantly (Harpprecht et al. 2020; Guohua et al. 2021). The water consumption for nickel production in China could be reduced by 31% to 50% when considering secondary supply. Currently, the end-of-life recycling rate for nickel is only 60%, where more than 95% of nickel is recycled in alloy form to produce stainless steels; however, such recycled nickel is not pure enough to be used in battery manufacturing (Henckens & Worrell 2020; International Nickel Study Group n.d.).

2.1.2.1.4 Summary of Battery Elements

With the transition to low carbon energy system and transport there will be considerable demand for battery metals. The metals discussed are lithium, cobalt and nickel which are considered critical for the development of lithium-ion batteries used for stationary storage applications and EVs. Figure 2.1 presents the demand projections for cobalt and lithium, according to different scenarios, vs. the respective reserve and resource estimates. With regards to the literature sources, it was found that battery recycling and sustainable transport modes can significantly reduce the associated geological risk for lithium and cobalt (Seck et al. 2022; Greim et al. 2020; Junne et al. 2020; Jones et al. 2020 Klimenko et al. 2021). Competing demand for nickel for the use of stainless steel from other sector will lead to considerable increase in future nickel demand (Henckens & Worrell 2020; Guohua et al. 2021). However, for the case of nickel few studies included the significant growth of nickel for stainless production. In terms of regional and geopolitical constrain for Cobalt and Lithium, it was found diversifying production by increasing recycling and investment in supply chain for consuming countries may reduce some of the supply risk (Seck et al. 2022). In term of social and environmental context, mining for lithium and cobalt show significant human right infringement, water stress, contamination of drinking water and significant pollution impacting the surrounding environmental and local communities (Hache et al. 2019; Lebre et al. 2020; Heredia et al. 2020; Mulvaney et al. 2021; Rachidi et al. 2021; Sovacool 2019; Calvão et al, 2021; Lebre et al. 2020). For nickel there hasn't been significant study on the social and environmental context, but it was found that secondary production of nickel from discarded scrap and recycling can significantly reduce some of the environmental impacts associated with nickel mining (Harpprecht et al. 2020; Guohua et al. 2021). Table 2.2 summaries these key literature findings.

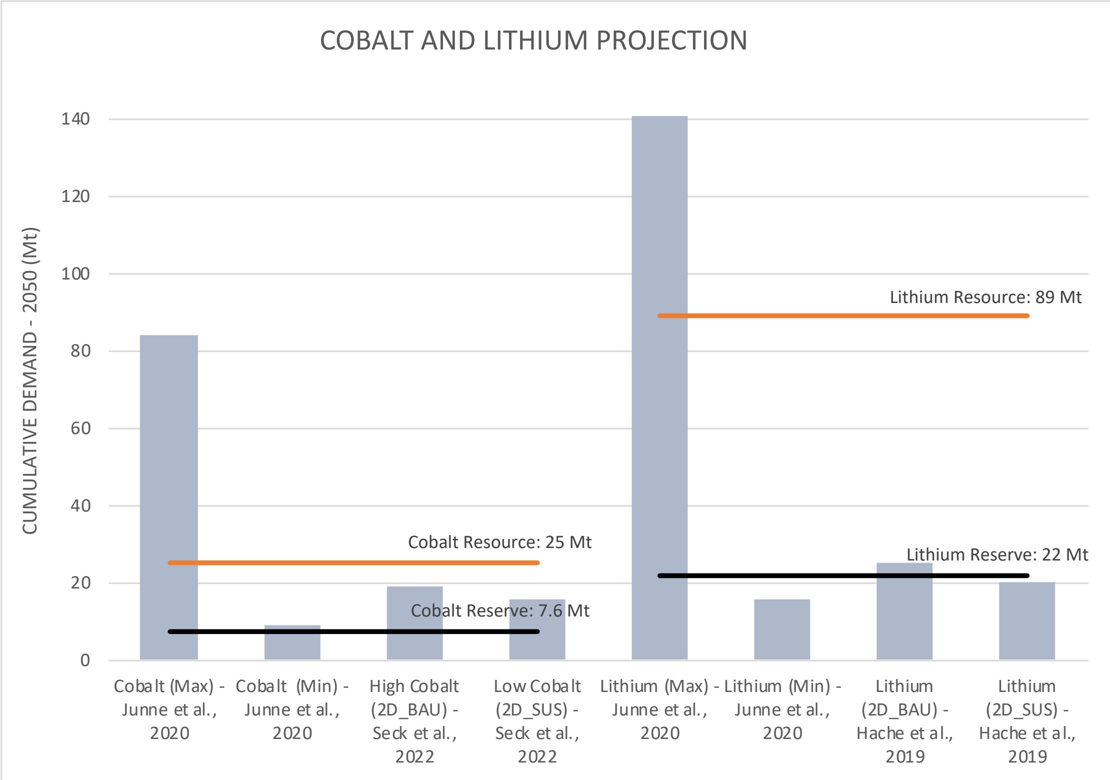

Figure 1.1: Resource and reserve estimates and cumulative demand projections up to 2050 for lithium and cobalt in all sectors, adopted from Junne et al., 2020 and Seck et al., 2022. 2D_BAU: 2-degree climate projection with "business as usual" and high mobility scenario; 2D_SUS: 2-degree climate projection with a shift to sustainable mobility and reduced number of private vehicles (both 2D BAU and 2D_SUS scenarios assume a recycling rate of 80.8% for cobalt used in EV batteries and no recycling for lithium).

Table 2.2. Summary of key barriers/challenges and suggested solutions for battery storage elements (cobalt, lithium and nickel).

Category	Issues	Elements	Potential Solutions	References
Geological Availability Risk	Insufficient reserves/ resource constraint	Cobalt	Increase recycling and shift towards sustainable transport modes. Reduce the use for cobalt in batteries, improve material efficiency. Increase exploration and development of mining technologies.	Seck et al. 2022, Klimenko, et al., 2021; Greim et al. 2020; Junne et al. 2020; Jones et al. 2020; Hache et al. 2019; De Koning et al. 2018; Tokimatsu et al. 2018; Takuma et al. 2018
	Reserves constraint	Nickel Lithium	Increase recycling and scrap supply. Shift towards sustainable transport mode.	Guohua et al., 2021
	Low recovery of cobalt during extraction	Cobalt	N/A	Petavratzi, 2019; Watari et al., 2022
Geopolitical and Regional Risk	Mining and/or refinery concentrated in a single region	Cobalt, Lithium	Increase and develop recycling in major consuming countries. Tailor trade strategies to reduce supply risk.	Seck et al. 2022; Lee et al. 2020a; McLellan, 2019; Månberger and Johansson 2019; Graham et al. 2021; Heredia et al. 2020; Hache et al. 2019

Environmental Risk	Contamination of water (lakes, rivers, or groundwater)	Cobalt, Lithium	Implement water management system such as water recycling process; aim to reduce wastewater discharge.	Mulvaney et al. 2021; Sovacool 2019; Lebre et al. 2020; Heredia et al. 2020
	Waste discharge to air and land	Cobalt, Lithium	Implement better tailings management; aim for waste reduction and recovery.	Mulvaney et al. 2021; Sovacool 2019
	Water scarcity and intensive water uses for brine extraction process	Lithium	Recycle water, minimizing waste products; improve recovery efficiently by alternative materials and technologies such as pre-concentration using ion exchanger.	Mulvaney et al. 2021; Lebre et al. 2020; Heredia et al. 2020; Hache et al. 2019
Social Risk	Health, well-being and safety risk of artisanal mining	Cobalt	Improve the provision of basic health and safety requirements at mining sites. Provide support and training for other livelihood incomes. Establish community benefit agreements and integrate artisanal and large-scale miners.	Rachidi et al., 2021; Mulvaney et al., 2021; Calvão et al., 2021; Lebre et al., 2020; Sovacool 2019
	Violation of local communities' rights: -Access to safe drinking water -Land rights of communities -Access to a safe environment	Lithium	N/A	Heredia et al. 2020

2.1.3 Discussion of the Systematic Review and Knowledge Gap

This systematic review of the literature on critical chemical elements for the energy transition allowed identification of the main challenges associated with them (which are summarised in Figure 2.2), and the strategies that have been proposed to maintain their reliable and secure supply, and to reduce environmental and social implications. Hence, certain other elements that have also previously been listed as critical (European Union, 2020; IEA, 2021) could not be reviewed, such as e.g., high-grade quartzite needed for c-Si PV, or natural graphite for LIB. The extent of information available from the various literature sources differs for each element as well. Therefore, some challenges could be discussed more in detail than others. The review also highlighted a number of knowledge gaps (summarised in Table 2.3).

2.1.3.1 Global Resource Availability and Recycling

Overall, it was found that copper, cobalt, platinum, and iridium could suffer in terms of availability. For example, all known global copper resources could be depleted by 2050 unless actions are taken to reduce this threat. Additionally, nickel, lithium, dysprosium, tellurium, indium and selenium could exceed current reserves by 2050, hindering the potential uptake of BEVs, FCEVs, CIGS and CdTe PVs, electrolysers and off-shore wind turbines if sufficient progress is not made in lowering the specific intensity of utilization of these elements in these technologies, together with investments in exploration and design for recycling, improvements in mining efficiency, and increased recovery and re-use of production as well as end-of-life scrap. In the case of copper, reduction in use needs to happen in other high consuming sectors as well, such as the building sector and grid networks (Grandell et al 2016; Bonnet et al 2019).

As green energy technology demand grows, so will the inevitable deterioration and reduction in resources, which will lead to an increase in the complexity of mining (Butterman et al 2004). Unlike fossil fuel sources, mineral and metals can be reused again and again with technological efforts. Thus, recycling and reuse provide a great opportunity to slow down the depletion of resources. Recycling of some metals will be more challenging than others such as recycling tiny amounts of platinum from fuel cells, compared to REEs from large permanent magnets (Watari et al. 2018). Some elements that are critical for thin film PVs are also used in very small quantities in various other applications and may be difficult to recover. Unlike fossil fuel sources, minerals and metals can, in principle, be reused again and again, with sufficient technological efforts. However, collection and recovery of these materials may be hampered by insufficient economic interest, as is the case for currently uncollected end-of-life copper cables and LCDs containing indium (van Oorschot et al. 2022; USGS 2022).

Policies could be implemented to provide economic incentive to encourage markets of secondary resources. Historical mine waste is also a potential source of accumulated by-product metals waiting to be exploited, for example Indium Corporation identified 15 kt of indium as residue reserves (Stamp et al 2014). In the long run, incorporating mine waste and recycling would turn accumulated harmful waste stockpiles into useful products, delay resource decline and the need to resort to more complex methods of extraction. However, at the same time, opening these mine waste sites also raises environmental and social concerns, and requires careful treatment and tailing management (Stamp et al 2014). Except for the case of indium, the challenges and benefits of waste mining have so far received very little attention in the literature. It was found that most studies that evaluated reserve constraints, focused on the current reserve for further growth, whereas only a few studies considered potential increase in reserves, e.g., for cobalt and nickel (Klimenko et al 2021; Guohua 2021). Also, when investigating supply risks for PV elements, PGMs and nickel, very few studies included other end uses, beyond energy technologies. This may sometimes lead to underestimating the future demand constraint, such as the potential increase in demand for indium for flat panel displays, or the significant use of platinum in other sectors, which collectively represent more than 50% of the total.

Deep-sea mining could provide an opportunity to address availability concerns related to copper, REEs, PGM and battery elements; however, the full extent of the associated environmental threats is still unknown (Levin et al 2020; Smith et al 2018; Lewicka et al 2021; Frenzel 2017). Furthermore, environmental regulation on deep sea mining could differ significantly from country to country, unless they are mined outside the exclusive economic zone (Jovanovic 2021), which could lead to a lack of proper monitoring and mitigating of environmental impacts, possibly to a worse extent than for terrestrial mining. For areas beyond exclusive economic zone, the International Seabed Authority is responsible for mining activities and protection of the ecosystem; however, mining in these areas could greatly impact many species that live on potential mining nodules, which could result in permanent loss of certain ecosystem functions of which the consequences are still unknown (Levin et al. 2020).

2.1.3.2 Geopolitical Risk

In terms of geopolitical risk, it was found that the supply of most of the elements reviewed is either concentrated at a single region or is limited in terms of trade networks, which makes other consumers of these elements dependent on a few selected countries. This may result in political instability, lack of regulations being followed, critical minerals being used for political strategy, or simply their production and refining to maximize regional economic interests, with lack of consideration for ethical sourcing. These considerations apply to cobalt, lithium, dysprosium and PGMs. For cobalt, this is due to its heavy concentrated mining and refining in the DRC and China respectively; for lithium, to its concentrated refining in China; for dysprosium, to its concentrated mining in China; and for PGMs, to their concentrated production in South Africa. Efforts could be made in diversifying supply, but for the case of dysprosium, the ore content is found to be less than 1% outside China. Therefore, reducing the actual dysprosium use in target applications would be the best strategy; this could be accomplished by moving to more efficient production processes requiring much less dysprosium, or by replacing dysprosium with terbium, as more of the latter becomes available thanks to the gradual phasing out of fluorescent lamps containing terbium in the coming years. It was also found that the extraction of most critical elements for thin film PVs are also concentrated in few regions, such as for tellurium, indium, and gallium, whose supply may also be limited by the production of the respective host elements (Helbig et al. 2016). Since these elements are by-products of zinc, lead, copper, and bauxite which are extracted globally, production can be diversified and expanded by building further refineries (Helbig et al. 2016). However, cheaper supply in other developing or undeveloped regions makes it difficult for mines in developed regions to operate, since labour and environmental cost are higher; this is the case for gallium in EU and REEs in the USA and Australia (Stegen 2015; Lewicka et al. 2021).

The increase in demand from energy technologies may likely attract mining investments, the long lead times of mining projects could pose short-term supply risks if such projects are not planned well ahead of time. Critical mineral recovery from mine wastes could also reduce the reliance on the few current producing countries. Global trade is an important aspect to consider in supply risk studies to prevent failures or disturbance in the supply chain. Geopolitical supply risk beyond the point of extraction was not considered in the reviewed literature, with the partial exception of battery metals only (Tian et al. 2021; Liu et al. 2021). The use of a GIS-based quantitative mapping tool such as the one recently introduced under the name "LAYERS" (Heidrich 2022) would be of significant value in estimating this extended risk.

2.1.3.3 Environmental and Social aspects

The booming demand for both niche and common elements open new opportunity for economic and social development in producing countries, but the same time it can have disastrous consequences unless social and environmental impact are managed properly. Only very few of the reviewed studies evaluated or discussed the environmental and social aspects of the supply of critical elements, with isolated exceptions for the cases of cobalt, lithium and REEs. None of the studies addressed post-mining scenarios, such as end of life management strategies of mining sites, or considerations of restoring communities and ecosystems (Maier et al. 2014). One reason could be due to the lack of transparency of mineral supply chains which makes it difficult to highlight both environmental and social impacts (Lee et al. 2021b). Authors that did investigate the environmental and social impacts associated with the extraction of these elements suggested the need for tailing, chemical leakage and water management, enforcement of safety regulations, increased use of wastewater and using alternative techniques in mining to reduce ecosystem contamination, water stress and harm to the local communities (Sovacool 2019; Lebre et al., 2020; Li et al. 2020; Wang et al. 2020; Mulvaney et al. 2021). Further issues have been identified for cobalt, HREEs and PGMs, to do with exploitation of workers, or lack of other livelihood incomes, which translates to unfair wages and leads to violence among workers, and insufficient safety and health provisions (Rachidi et al. 2021; Calvão et al. 2021; Li et al. 2020). To ensure stability in supply chains, environmental and societal costs should be internalised before starting mining projects. Efforts need to be made to also provide workers with better working conditions and expanding livelihood opportunity beyond mining in those mining countries (Sovacool 2019).

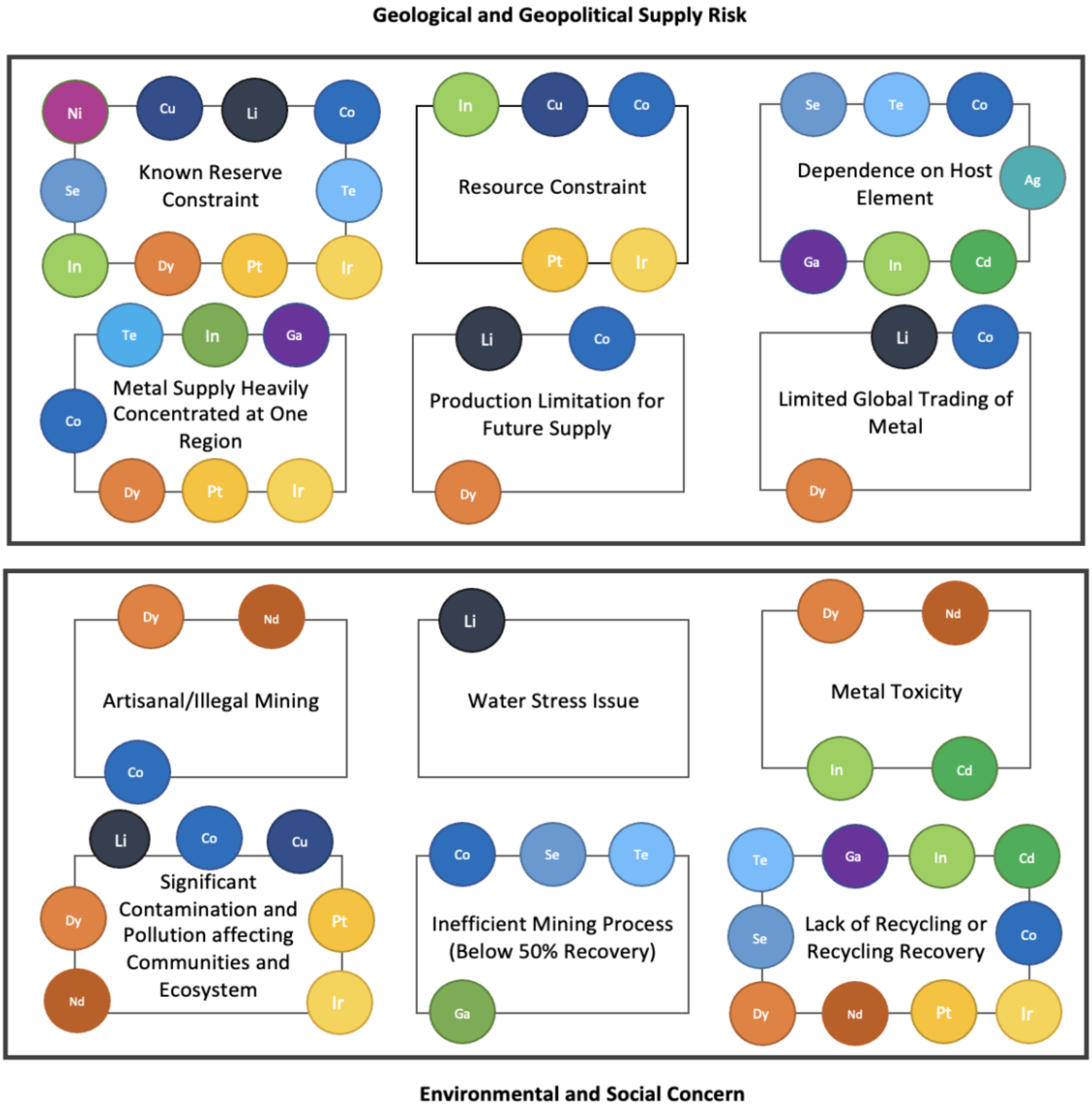

Figure 2.2: Issues highlighted for each element based on findings from the systematic literature review. Shades of similar colour represent common ore deposits.

Table 2.3: Identified knowledge gaps based on systematic review.

Category	**Knowledge gaps**	**Elements**
Global Supply Availability	Demand projection is mostly limited to energy sector only	PGMs, PV elements
	Outdated information on improvements made on element loading for low carbon energy technology	Selenium, indium

	Limited studies on potential mining from waste and historical mine sites	PV elements, PGMs
	Limited number of studies on supply and demand projections	Nickel, iridium
Geopolitical	Limited findings on geopolitical risk beyond mining	PGMs, REEs, PV elements, Copper
Environmental	Limited findings on environmental evaluation of extractive activities	PV elements, PGMs
	Limited findings on environmental benefits and challenges of using mine waste	All
	No discussion of restoring ecosystems at end of mining operations	All
	Limited findings on environmental impact of deep-sea mining	Cobalt, lithium, nickel, PGMs
Social	Limited findings on societal impacts of mining on local communities and workers	Nickel, REEs, PV elements, copper, PGMs

2.2 Shared Mobility Strategy

The total light duty vehicle traffic is expected to grow between 11% and 43% between 2015 and 2050 excluding the influence of TaaS (Department for Transport 2018). With a focus on reducing congestion, air pollution and emissions in the UK there is already a growth in shift from private light duty vehicles to user-based services. The concept of Transport as a Service (TaaS) is still at a very early stage but is seeing an increased attention already (Department for Transport 2021). Shared mobility vehicles with the objectives of reducing private vehicle are referred to as "TaaS vehicles" in this thesis. The potential value of TaaS is measured based on the reduction in private car usage, emissions and vehicle miles travelled (Zhao et al. 2021). Advances in autonomous vehicle are also expected to accelerate the movement towards TaaS (Habib & Lynn 2020; Hirst 2021). The integration of autonomous vehicle with shared mobility has significant potential to achieve better fleet management, improve vehicle efficiency and hence reduce emissions at the point of use (Roca-Puigròs et al. 2023; Morfeldt & Johansson 2020; Silva et al. 2020). Renault announced their first public testing of autonomous vehicle service in 2022 (Renault Group 2021). Recently a joint venture between Hyundai and Aptiv announced its first robo-taxis to launch in

2023, which can operate without a driver in most environment (Hyundai n.d.). However, the level of market penetration of a fully autonomous vehicle remains to be unclear due to technological advances, regulatory, ethical and cost barriers (Hirst 2021; Heineke et al. 2021).

Car-based shared mobility schemes under a TaaS ecosystem can be seen in two forms: trip sharing (ride sharing) or car sharing (Marsden et al 2019). Car sharing schemes allow consumers to access fleet of vehicles from a station or spots around the city that can be driven for a short period of time and returned at a nearby station or spots depending on the type of scheme (Finger et al. 2017). These include free-floating systems such as Car2Go and ZenCar, station-based system such as Cambio and peer to peer systems that allow private car owners to lease out their cars such as Turo, YourDrive and CarAmigo (Transport and Environment 2017). Trip sharing schemes allows different riders to book a trip at different spots usually heading the same way and share a portion or a full trip of a vehicle's journey known as ride- hailing (Finger 2017). Another form of trip sharing is carpooling schemes; this allows the owner of the private vehicles to share their vehicle with other riders, where the rider pays a portion of the cost of travel to the driver such as BlaBlaCar, Lifeshare and Waze carpooling (Finger et al 2017; Guyader et al. 2021). In terms of adoption, both form of mobility schemes is expected to rise in the future (Grosse-Ophoff et al. 2017; Dias et al. 2017).

2.2.1 Impact of Shared Mobility Scheme on Transport

Based on literature on car sharing schemes, studies have shown significant potential to reduce the overall number of private vehicles on the road and improve vehicle utilization (Firnkorn & Müller 2011; Baptista et al. 2014) . A survey conducted on shared mobility in London found that 37% of car sharing impacted their decision on to own a private vehicle whereas 11% sold their private vehicles within 3 months of using shared mobility schemes (le Vine & Polack 2017). According to a survey by Collaborative Mobility UK (CoMoUK), each car club vehicles in the UK has replaced 20 private cars in 2021 (CoMoUK 2022). However, in the case of Peer-to-Peer (P2P) car sharing, the impact on private vehicle is expected to be low (Marsden et al. 2019). There is very little information on P2P car sharing market in the UK, same is the case of ride hailing services in the UK.

In some ride sharing scheme, ride-pooling services such as UberPool Express and Lyft Shared Rides which have a designated pick-up point have shown to attract travellers who would otherwise use public transport resulting to an increase in vehicles miles travelled, based on the data for US (Schallar 2021). Similar findings were found by Hyun et al. (2021), where miles travelled per year showed negative relationship to ride hailing usage. Rayle et al. (2016), found ride hailing schemes are also much likely to replace trips made by taxi and traditional public transport.

Carlooping schemes such as Liftshare is one of the popular forms of ride sharing in the UK with over 650,000 members (Marsden et al. 2019). Carlooping case studies in the UK on Lifeshare and in the EU on BlablaCar have shown to result in increased vehicle occupancy and reducing number of private cars on the road (Clabburn 2019; BlaBlaCar 2019). Research by BlaBlaCar & Le Bipe for the case of EU, suggested that through ride-sharing, vehicle occupancy increased from 1.9 to 3.9 people per vehicle as well as led to a reduction of overall journeys made, particularly at peak times (Department of Transport 2022). TaaS will not only depend on car-based share mobility schemes, but other form of mobility such as bike sharing and traditional public transport (CoMoUK 2020a).

In Sydney, TaaS trials indicated a net reduction of car-based (shared and private car) journeys (Hensher et al. 2022). In Germany, Bremen mobility hub led to 5,000 fewer cars and 50% reduction in total distance travelled per household. Bergen mobility hub indicated significant decrease in parking permits sold (CoMoUK 2022). Whilst there has been reduction in congestion, it was found trips based on more environmentally friendly modes such as walking, cycling and public transport has also seen shift towards shared mobility, which may lead to negative environmental consequences (Hensher et al. 2022). Since, shared mobility is still at its early stage, travel behaviour may change over time and will largely impact on the key drivers and policy effectiveness (Keith et al. 2022). The main drivers found for adopting shared mobility schemes are financial, convenience and sustainability (Machado et al. 2018).

2.2.2 Environment Impact of Shared Mobility Scheme

For carsharing schemes, a study conducted on shared EV adoption for the case of Sweden estimated if just one car sharing vehicle successfully replace 10 owned cars, 41% reduction in carbon during travel could be achieved in 2050 (Morfeldt & Johansson 2020). A life cycle assessment on the use of electric car sharing showed 55% reduction in carbon emission when compared to a 20% reduction with fossil fuel-based sharing vehicles (Nong et al n.d.). Amatuni et al. 2019 conducted a life cycle assessment on the uptake of car sharing vehicles with different possible vehicle mileages and lifetime and concluded that car sharing could reduce 3 to 18% of life cycle GHG emissions for mobility; in the case of increasing occupancy was found to have the greatest effect on reducing CO_2 emission. In the case of ridesharing vehicles, Vilaca et al. (2022) estimated a 42% reduce in environmental impacts compared to a private vehicle, with highest potential reduced shown in human toxicity, mineral resource scarcity, marine and freshwater ecotoxicity (Vilaca et al. 2022). Keith et al. (2022), analysed the impact of copper usage and carbon emission with the diffusion of carsharing and ridesharing services and Zhao et al (2021), carried out a scenario analysis to understand TaaS impact on a large-scale transport system. Based on both their studies, TaaS could lead to significant carbon reduction from 2.5% up to 50% depending on vehicle occupancy and the use of low carbon resources to power the vehicles. Kawaguchi et al. (2019), carried out a study on the adoption of electric car sharing and ride sharing impact on copper usage and carbon emissions on the vehicle fleet, they found that although the shift of EV requires increase in copper usage, the diffusion of TaaS vehicle could have a significantly impact on reducing overall copper demand and CO_2 emission of the vehicle fleet with vehicle occupancy being the most sensitive parameter (Kawaguchi et al. 2019). Both trip-sharing and car-sharing vehicles are expected to have environmental benefits upon successful adoption (Jenn 2020; Ding et al. 2019; Gawron et al. 2019). Studies have shown the extent of environmental benefits of TaaS vehicles depends on the annual driving intensity of the vehicle, lifetime and how well do TaaS vehicles replace private vehicles (i.e., number of vehicles replaced and average occupancy) (Amatuni et al. 2019, Jenn 2020, Kawaguchi et al. 2019; Ding et al. 2019, Fernando et al 2020).

2.2.3 Shared Mobility Usage

Private vehicles typically have a high average lifetime of 14 years in the UK due to their low utilization (i.e., sitting idle for up to 95% of the time) mainly serving a single household or the driver (Keith et al. 2022). The calendar age of a vehicle becomes generally shorter with increasing annual driving intensity due to use related wear and tear (Morfeldt & Johansson 2022). Some authors suggest the relationship between vehicle lifetime and driving intensity may not be linear as some vehicle wear and tear are due to environmental exposure and in infrequent use (Keith et al. 2022). Others have found a close linear relationship between that the average vehicle lifetime reduction and increase in driving intensity based on a dataset for Swedish cars (Morfeldt & Johansson 2022). In general, TaaS vehicle are expected to have a higher mileage and lower lifetime compared to the private vehicle. Car sharing vehicle are also treated as rental cars that have an average age of 2 to 3 years after which the vehicle may end up in the market for second use (Mont 2004; Cho & Rust 2008; Mitropoulos & Prevedouros 2014; Guyon 2017).

There is a lack of data that quantify the usage of ride sharing in the UK (CoMoUK 2019; Angeloudis & Stettler 2019). Based on the studies for ride-hailing in US, a significant amount of increase in travel distance is expected due to the dispatch travel to the customers (Ding et al. 2019). Car sharing is also expected to have dispatch travel, this could include the extra journey to the returning point instead of the destination for station-based system or staring dispatch distance due to the random use and docking by the previous user in a free-floating system. Overall impact of car sharing vehicles in the UK have also shown to reduce the total travel distance, also known as vehicle miles travelled (VMT) which arises from a small proportion of users decreasing their VMT by a large amount (Wu et al. 2019).

In the UK, the total distance covered by car sharing vehicles (excluding Scotland) in 2019 to 2020 was around 6.4 million miles of which 74% were cars and 80% were vans aged less then 2 year and a small portion representing vehicles of 5 years or older. The average car sharing vehicle in the UK (1.5 years – car and 1.4 years– Van) are significantly newer than the average private vehicle age of 8.3 years (CoMoUK 2020b). Average car occupancy in the UK is 1.5, which is similar to TaaS vehicles in the UK, but this is projection could increase to 1.7 for TaaS vehicles in 2050 (Marsden et al. 2019). There isn't any clear information on VMT for TaaS vehicles.

The literature was found to vary significantly in terms of lifetime distance. Authors of either assumed similar lifetime distance for TaaS vehicles as private cars but with high utilization per year or increased lifetime distance. Some authors projected it may be economical for TaaS vehicles to have higher vehicle durability or modular components for easier replacement which allow for extended lifetime distance (Keith et al. 2022). Others assumed having electric powertrains should significantly improve lifetime distance as there are fewer moving parts than internal combustion engines, requiring less maintenance, as well as driving would be more efficient compared to a human-driven car which may also extend lifetime distance for the case of Autonomous Taas Vehicles (Arbib & Seba 2017). Furthermore, it is observed the replacement of private cars can differ between shared mobility schemes. Ding et al. 2019a, assumed a replacement of 7 private vehicles for free-floating scheme vs a replacement of 3 private vehicles for carpooling schemes, each having different number of passengers occupancy that is used for calculating VMT (Ding et al. 2019a). The occupancy rate for same type of TaaS vehicles differ from literature to literature as well and this also depends on geographical factors. For example, Ding et al., 2019a estimated occupancy rate of 3 for carpooling in Beijing, whereas Ferando et al. (2020) assumed 2.1 in their study for US. It is estimated that between 1 and 6.5 personal vehicles can be substituted by TaaS vehicles (Fleury et al. 2017). Table 2.4 provides the estimates made in literature on TaaS vehicle lifetime.

Table 2.4: Estimates on TaaS vehicle lifetime based on literature information.

Type	Vehicle lifetime mileage estimates (km)	Lifetime in years	Passengers' occupancy	Reference
Autonomous electric TaaS vehicles	804,672	5	-	Arbib & Seba 2017
Car Sharing	180,000 -348 000	12 – 15 (≈2 years first life)	-	Amatuni et al 2019
Carpooling	600,000	9.8	3	Ding et al., 2019a
free-floating	600,000	3.9	1.26	Ding et al., 2019a
Autonomous TaaS ICEV	321,869	2.1	-	Gawron et al 2019
Autonomous TaaS EV	321,869	2.3 -4.2	-	Gawron et al 2019
Electric Carpooling	200,639	-	2.1	Ferando et al 2017
Electric Car-sharing	401,278	-	1.86	Ferando et al. 2017

2.2.4 Summary

There is a clear understanding in the literature that the move towards TaaS can assist in reducing resources and GHG emissions, but the degree of environmental benefits of TaaS vehicles remains a question and is depended on various uncertainties. There is currently limited understanding on how shared mobility schemes and new autonomous technologies would evolve the future transport system (Department of Transport 2018; Innovate UK 2021; Mckinsey 2023). The impact of TaaS on road traffic is also difficult to predict given the on-going changes in transport system and technologies, there is wide range of uncertainty on how these changes will impact private ownership of vehicles.

2.3 Evolving landscape of EV batteries and end-of-life treatments

At present lithium-ion batteries (LIB) are the most common type of battery used in EVs. Battery reuse and recycling provides an opportunity to close the battery loop to prevent resource scarcity, the accumulation of toxic materials from end-of-life lithium-ion batteries, reduce reliance on the intensive mining and refining process for the battery elements as well as save cost on critical valuable elements such as lithium and cobalt (Beaudet et al. 2020). This section provides an overview of the battery roadmap, types of recycling and explores second life opportunities for lithium-ion batteries which will provide the groundwork for developing scenario in methodology chapter 3.

2.3.1 Battery Roadmap

A LIB consists of a cathode (positive electrode), anode (negative electrode), separator, and an electrolyte. The cathode and the anode are made of an intercalation compound that allows lithium-ion flow from the cathode to the anode and vice versa. The cathode of the battery represents the transition metal oxide or phosphate that has been lithiated during the construction of the battery to provide the supply of lithium ions. The anode consists of porous carbon, mainly graphite (sometimes graphene) Battery University (2022), the structure of graphite and its high conductivity are favourable for ensuring high-efficiency intercalation of lithium-ions in the anode (Heß & Novak 2013). The electrolyte is an ionic conductive insulating material that transports lithium ions between electrodes during discharge and charge cycling. The battery reaction is mainly the movement of the lithium-ions and the electrons in the battery from the cathode to the anode. When the battery is being charged, an oxidation reaction takes place in the cathode and a reduction reaction takes place at the anode, where the lithium-ion combines with graphite (Liu et al. 2016). During discharge, this process is reserved, and the lithium-ion is released from the graphite anode and combined with the cathode material (Satyavani et al. 2016). Figure 2.3 shows the battery components during the discharge process of LIB.

LIBs cathodes comprise different chemistries, which can be divided into different crystal structures, layered, spinel and olivine, which impacts the characteristics of the battery. The layered structures are used as cathode for high energy density systems, whereas spinel and olivine are considered in the case of high-power applications (Tran et al. 2021; Julien et al 2014). There are currently four main cathode chemistries of lithium-ion batteries widely available in commercial BEVs: (1) nickel manganese cobalt and lithium manganese oxide blend (NMC-LMO) - spinel (2) nickel manganese cobalt (NMC) - layered (3) nickel cobalt aluminium oxide (NCA) - layered and (4) lithium iron phosphate (LFP) – olivine. Each combination has distinct advantages and disadvantages in terms of performance, cost, safety, and other parameters, this is discussed in appendix A.

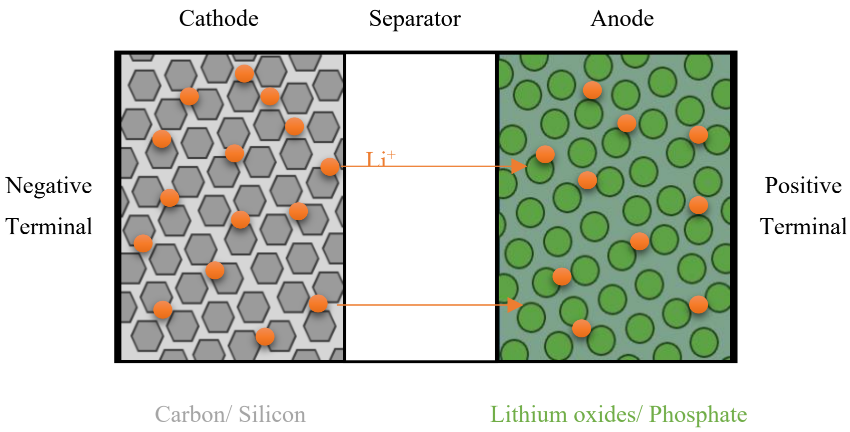

Figure 2.3: Shows the schematic of lithium-ion battery during discharge.

LIBs are mainly distinguished by their cathode type because the cathode material fundamentally determines many of the battery key characteristics (Manthiram 2020). LFP is mainly used by Chinese EV manufacturers and recently NMC cells for their high-performing vehicles (Greenwood et al. 2021; Mckinsey 2021). Recently Tesla started utilizing LFP cells as well in their standard EV models and Ford announced it will also introduce LFP cells in its Mustang Mach-E line supplied by CATL (CNBC 2022). LFP are also mainly used in electric bus and grid applications (Olivetti et al. 2017) and are expected to retain their potential for short-range vehicles, electric buses and electric bikes in the coming years (Pelegov & Pontes 2018; Pillot 2019; IEA 2023). NCA cells on the other hand is only known to be used by Tesla vehicles (Ding et al 2019b). The majority of registered EVs in the UK and globally use either NCA or NMC cathode types (Department for Transport 2022). NMC is the most widely adopted battery chemistry by most EV manufacturers and its market share is expected to increase further in the coming decades, alongside improved driving ranges (IEA, 2020; Ding et al., 2019b; Ortego et al., 2020). NMC accounted for 72% of batteries used in EVs in 2020 (excluding China) (Bhutada 2022). Table 2.5 represents information provided by Department for Transport (2022) on 2019 BEV market share in the UK and by EV Database (n.d.) and Battery University 2019 on battery capacity and driving range). Globally, the battery capacity of BEVs ranges from 18 kWh to 100 kWh (Ballinger et al. 2019; Elshkaki 2020). Different BEV make have different requirements for battery capacity and BEV manufacturers will produce BEVs with different mileage options for drivers, resulting in different capacity of batteries.

Table 2.5: Represents the market share of BEV in the UK in 2019 alongside their current battery capacity and driving range.

BEV Make	Market Share in UK (2019)	Cathode Chemistry	Battery (kWh)	Range (miles)
Nissan Leaf	17%	NMC	40	168
Nissan Leaf	17%	NMC	60	239
Bmw I3	9%	NMC-LMO	42	188
Volkswagen Golf	6%	NMC	32	125
Tesla Model 3	12%	NCA	73	285
Renault Zoe	12%	NMC-LMO	52	200
Tesla Model S	11%	NCA	95	315
Nissan E-Nv200	6%	NMC-LMO	40	124
Jaguar I-Pace	6%	NMC	85	230
Tesla Model X	6%	NCA	95	285

The market share of different cathode types is influenced by the development of battery technology in the future. There have been several roadmaps published indicating the possible future trends for EV batteries (Gifford 2022, Ding et al 2019b; Maisel et al. 2023, IEA 2023, Pillot

2019). Due to the high price of cobalt and its supply vulnerability issues there is a continued shift towards NMC types of lithium-ions that are higher in nickel content (IEA 2020; IEA 2023). From NMC 111, nickel, manganese and cobalt are all present in the same quantities, to a more advanced NMC 811 cathode that contains less cobalt and more nickel with a higher energy density (Fickling 2017). Current NMC cathode have been considered to move towards 622 ratio but alternative ratios with NMC 4333, NMC 532, NMC 721 also exist in the market (Volkswagen AG 2021; Ramanathan 2021; Maisel et al 2023; IEA 2023). Table 2.6 lists the average lithium, cobalt, manganese and nickel contents found in commercial NMC cathode of EV LIBs adopted from Olivetti et al., 2017.

Table 2.6: Average lithium, cobalt, manganese and nickel contents in EV, expressed as grams of metal per kWh of energy storage capacity adopted from Olivetti et al., 2017.

LIB Cathode Chemistry	Li	Co	Mn	Ni
NMC-111	139	394	367	392
NMC-622	126	214	200	641
NMC-811	111	94	88	750

It is expected in the future NMC 811 will have the highest market share compared to other NMC cathode composition (Karabelli et al. 2020; IEA 2020; Maisel et al. 2023). Improvement will also be seen in the other parts of the battery pack. Several BEV and BEV battery manufacturers have already announced the potential use of silicon anode instead of graphite in the near future to achieve higher energy density and better fast-charging capability (Porsche Newsroom 2021; Paul 2023; Wang et al. 2022). It expected pack-level energy densities could reach 275 Wh/kg based on further improvement in the anode and pack design (IEA 2020), reaching near its theoretical limit of ~300 Wh kg^{-1} (Thackeray et al. 2012). There has been much research on beyond Li-ion battery technology to achieve a higher theoretical limit for battery packs (Thackeray et al. 2012). Figure 2.4 represents a possible roadmap for future EV battery chemistry adopted from Ding et al. (2019b), IEA (2020) and Hill et at (2020).

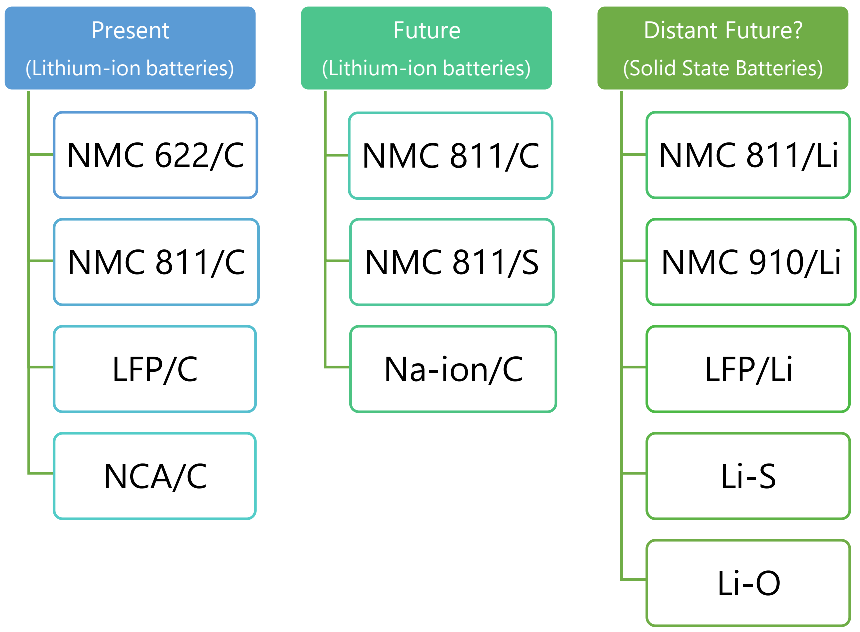

Figure 2.4 Represents possible roadmap for future EV battery, adopted from trends in Ding et al 2019b, IEA 2020 and Hill et at (2020).

Currently efforts on solid state battery designs are based on layered oxides such as NMC of LIBs with a lithium anode and use of electrolyte additive (Jung et al. 2016; Fan et al. 2018; Zhang et al. 2020; Ma et al. 2021). The use of lithium metal anode instead of graphite could allow for less anode material for the same amount of total energy, increasing the energy density of the cell (Ma et al. 2021). Other battery types include lithium-sulphur and lithium-air batteries, in the case of lithium-air there are several technical challenges which needs to be addressed before a practical battery could be developed, whereas lithium-sulphur struggles in achieving higher cycle life to make them commercially viable (Merrifield 2020). Hence, for solid state batteries, there are still several uncertainties involved on the working capability and viable commercialization in the future which makes it difficult to predict when these battery technologies will fully develop or be adopted.

There are also increasing efforts in sodium-ion battery due to the natural abundance and low-cost of sodium resources (Abraham 2020; Ma et al. 2021). Sodium-ion batteries are gaining increasing attention in recent years (Abraham 2020). Sodium-ion battery technology could complement LIB where energy density is not a primary factor, thus minimizing the fears of lithium shortage (Tarascon 2020). The CATL battery manufacturer announced its effort on using both sodium-ion cells and lithium-ion type cells in a single EV battery pack (NY Times 2023). Since, sodium-ion batteries have a lower energy density than LIBs, sodium-ion batteries are expected to replace markets of those similar LFP cathode type, such as short-range vehicles, electric bus and in stationary storage applications (Abraham 2020).

Furthermore, it is predicted that LIB with NMC cathode will continue to dominate the future battery market (Greenwood et al. 2021; Maisel et al. 2023). Therefore, the thesis will only focus on lithium-ion batteries technologies.

2.3.2 Current Recycling Methods

Recycling of Lithium-ion batteries can be categorized as either closed loop or open loop. The former is when the material recovered are in their high purity form to be used again in LiB production without requiring further processing; the latter involve recovery of the material which are assumed to replace primary supply of raw materials. However, the recovered materials would require further refining process before they can be used again in battery production (Rajaeifar et al. 2021). There are three main recycling processes for lithium-ion batteries, hydrometallurgical, pyrometallurgical and direct physical recycling (Zhou et al. 2020). Pyrometallurgy uses series of high temperatures to first evaporate the electrolyte without explosion below 300C, at 700C to burn the plastic component from the battery, at 1500C to smelt the battery into combination of alloys and slag which can be treated by the hydrometallurgical process to extract valuable metals (Zhou et al. 2020). The pyrometallurgical process avoids the need for crushing and other pre-treatment steps, the modules can be recycled after simple manual dismantling process (Rajaeifar et al. 2021). However, pyrometallurgical is a high energy intensive process due to the high temperature requirements and requires extensive effluent treatment to avoid the release of toxic gases in air. Other valuable elements such as manganese and lithium cannot be recovered and are lost as slag. At present, hydrometallurgy is typical recycling process to recover LIB metals, whereas direct recycling is more useful for recovering components from spent LIBs without using chemical processes (Zhou et al. 2020).

The hydrometallurgical process works by the leaching method, either by acid leaching or biological leaching, latter having a much lower recovery rate (Zhou et al. 2020). Battery material is dissolved and separated in the form of a solution. The hydrometallurgical process requires pre-treatment of used battery packs before dissolution, by various mechanical method which involves shredding, crushing, and grinding of the battery components (Chagnes et al. 2013). Many processing plants have incorporated mechanical treatment in combination with hydrometallurgy.

The battery cases, electrodes and electrolytes are treated separately to reduce cost and energy consumption and improve safety and recovery rate (Zhou et al. 2020). The use of mechanical methods makes the recovery process more complex compared to pyrometallurgy and furthermore the use of some organic solvent creates problems due to the solvent toxicity and corrosiveness in some cases (Fichtner et al. 2022; Zhou et al. 2020). The significant advantage of hydrometallurgy over pyrometallurgy is reduction in toxic gas emissions and energy use and increase in high purity recovery. Acid leaching is usually divided into two categories: inorganic and organic acids, the first being potentially more toxic solvent use and the latter is more benign method of leaching. Inorganic acids include hydrochloric acid, sulfuric acid, nitric acid, and phosphoric acid, whereas organic acids include citric acid, oxalic acids and tartaric acids (Zhou et al. 2020). Bioleaching works by dissolving electrode with metabolites excreted by microorganisms (bacteria and fungi) (Zhou et al. 2020). Currently, the use of inorganic acid is the most popular form of leaching with a recovery rate of 99% for Co, Mn and Li, and over 98% for Cu (Zhou et al. 2020). There is very little known in terms of recycling graphite from hydrometallurgy processes (Sommerville et al. 2019).

The concept of direct recycling is to restore the cathode capacity and property losses which occur through cycling, which requires the battery to be in good condition of recycling (Baum et al 2022; Neumann et al 2022). This involves mechanical, thermal, and chemical process to recover the battery components. The battery is disassembled to cell level, which is then treated by supercritical CO_2 to extract the electrolytes and restoring agent for the cell which involves heating and pressure treatment (Sloop et al. 2020). The cells are then broken down for collecting cathode materials to be reused. Direct recycling has the shortest recovery route, least energy intensive and is considered to be the most environmentally friendly method for recovering lithium-ion batteries (Zhou et al. 2020), however it cannot recover individual metals. Therefore, the recovered cathode may be obsolete by the time it is introduced to the market as battery technology are continuously evolving (Beaudet et al. 2020).

The pyrometallurgical process is the least favourable option as it cannot recover some of the key battery materials for reuse and incur significant energy cost for recycling which is a disadvantage for the production of low-cost LIBs (Zhou et al. 2020; Beaudet et al. 2020; Baum et al 2022). The different LIB cell chemistries pose another major challenge to current recycling systems (Neumann et al. 2022). Direct recycling requires a single cathode chemistry as input to achieve high recovery of high materials (Neumann et al. 2022), which is a major disadvantage of direct recycling since battery chemistries are evolving constantly which also makes standardising of chemistries and packaging type difficult. Hydrometallurgy, on the other hand, works well for a mixture of different cathode types (Neumann et al. 2022).

Overall, mechanical, and hydrometallurgical processes seem to be the dominating approaches for recycling facilities in Europe (Fichtner et al. 2022). There isn't any EV battery recycling facility operating in the UK at the moment, instead LiBs are exported to Europe for processing (Pillot 2019), however there is possibility of future recycling facility to open in the UK. Veolia announced it will open an EV battery recycling plant in West Midlands, UK, incorporating mechanical and hydrometallurgical recycling with the capacity to process 20% of the UK EoL EV batteries by 2024 (Veolia n.d.; Veolia 2022). Recently SER Group announced the launch of Cellcycle which will also incorporate a combination of mechanical and hydrometallurgical process to recycle EoL EV batteries and other battery types (Cellcycle n.d.). Both the companies aim at recovering aluminium, copper and black mass through mechanical processing, the black mass is then treated with hydrometallurgical process to recover lithium, cobalt and nickel along with other valuable elements.

Currently there are at least 32 established or planned LIB recycling facilities with estimated 322,500 tons of recycling capacity and 70,000 tons of planned recycling capacity. It is estimated in UK alone there could be 350,000 tons of EoL EV batteries by 2040 (Veolia 2022). With regards to the rising amount of EoL LIBs, the current number of recycling facilities are comparatively low. Several barriers still exist with safety, environmental concern, economic returns and closing the battery loop. Such as treatment of wastewater from hydrometallurgical process and safe discharge of EoL batteries (Baum et al. 2022). The highest value is currently obtained from cathode recycling, especially the recovery of cobalt, nickel and lithium which may be seen as a priority for efficient recycling (Chagnes et al. 2013; Baum et al. 2022).

2.3.3 Second-Life Applications for EV Batteries

There are now over 25 battery energy storage projects either in construction or operational in the UK, of which over 50 per cent of the storage projects since 2010 have used lithium-ion batteries (IRENA 2017). Figure 2.5 provides the share for energy storage application in the UK in 2017 adopted from IRENA (2017).

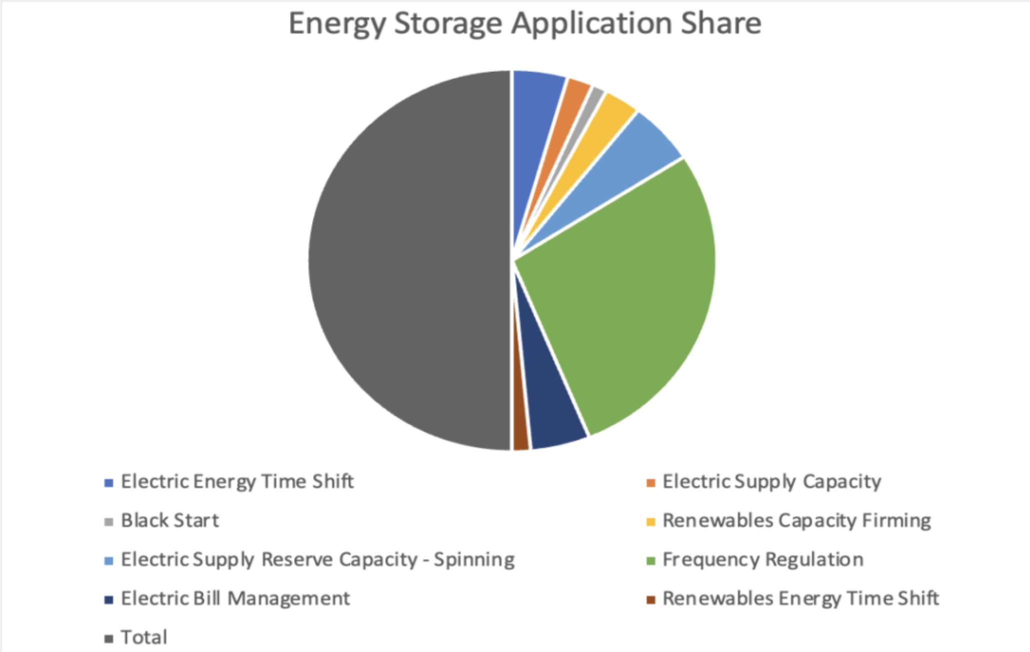

Figure 2.5: share for energy storage application in the UK in 2017 adopted from IRENA (2017).

Once EV batteries retire from BEV, batteries can be repurposed for second life applications which do not require intensive power from the battery, hence eliminating some of the environmental and energy impacts associated with manufacturing new batteries (Martinez-Laserna et al. 2018). EV battery also accounts for roughly 40% of the cost of an EV (Shahjalal et al. 2022), reusing batteries in second-life applications can optimize their economic value and resource utilization (Shahjalal et al. 2022; Pillot 2019). However, it could be argued that sophisticated LIB technology is not appropriate for static storage and that such batteries might be better recycled.

The batteries from the first life in BEV undergo several degradation mechanisms as discussed by Shahjalal (2022), which leads to either loss of cathode or anode active material, loss of lithium-ion or increased resistant causing capacity and power fade (Shahjalal et al. 2022). These repurposed pack have to meet the same performance and safety standard as new built battery packs to prevent risk of unregulated second-life battery market (Börner et al. 2022). Retired BEV LIBs need to undergo remanufacturing process for being repurposed for second life storage applications, which involves screening techniques to help detect damaged cells which can then be substituted with suitable cells. The battery including the BMS dismantled process and cell extraction needs to take place in a control environment to prevent oxidation at the cathodes and finally reassembly of the battery modules. Typically, batteries sent for repurposing aim to have limited disassembly and reassembly of the battery to save cost and being able to reuse as much of the previously parts of the battery. Since the repurposed battery pack could have a different configuration compared to its first life (Börner et al. 2022), they may require some changes or entirely new battery management system (BMS) to support the battery in its new application (Shahjalal et al. 2022).

Currently the largest second-life initiatives are: “Daimler Mobility House” with 13 MWh of second-life batteries used for compensating power fluctuations (Daimler 2015); “Advanced Battery Storage” launching this year with 60 MWh of second-life batteries to facilitate the integration of renewables by 2020 (Groupe Renault 2018); and the “SmartHubs Connected Energy” pilot project which is set to launch in 2021 with 14.5 MWh of second-life batteries to provide grid balancing services using 1000 second-life batteries (Connected Energy 2020). Furthermore, there are several micro-initiatives for building onsite management (Christensen et al. 2021). Building a large storage system would require sorting of batteries based on chemistry, pack type and capacity. Current initiatives are based on a single EV manufacturer making it simpler to track and trace batteries which may not always be the case. Combining cells from different EV manufacturers could lead to feasibility concern and complexities for repurposing second life batteries, since cells are not standardized, manufacturers will have multiple different battery designs, which will vary with in capacity, chemistry, and packing (cylindrical, prismatic, and pouch) (Pillot 2019; Börner et al. 2022; Engel et al. 2019). Additionally, EV manufacturers have access to battery ageing information based on their first life, which through data-driven estimation can provide valuable information on battery ageing, remaining lifetime, and appropriate decision regarding its use. Therefore, avoiding the disassembly of the battery packs and physical testing of cells (Zhu et al. 2021). However, currently only a handful of industry focus on state of health and BMS disclosure, which makes it difficult and costly to determine accurately the remaining lifespan for second life battery applications (Engel et al. 2019; Zhu et al. 2021).

The battery lifetime depends on the state of health, some studies assume a 60% state of health as a limit for stationary storage application (Casals et al. 2019). While stationary storage may be able to provide storage capabilities beyond this, it risks sudden acceleration of the ageing process (Casals et al. 2019; Martinez-Laserna et al. 2016). Based on the lifetime studies for second life batteries, the lifetime can vary between 5 to over 15 years, depending on the type of battery applications they participate in and their state of health (Smith et al. 2017; Hossain et al. 2019; Casals et al. 2019). Some of the applications second life batteries can participate in are such as frequency response, voltage support, renewable firming, peak-shaving, and grid balancing services (White et al. 2020; Haram et al 2021). In some cases, battery provide more than one services known as application stacking such as a combination of frequency response and grid balancing services (National Grid ESO n.d.). In such case, second life batteries have a shorter remaining life of 3 to 4 years (National Grid 2022). Battery energy efficiency also drops due to the power and capacity fade which causes a higher electricity consumption comparison to a new storage system (Philippot et al. 2022; Kamath et al. 2020). For a second life battery used in combination for domestic PV system, it was found there is a 2% drop in the round-trip efficiency per 20% capacity fade (Philippot et al. 2022). For frequency response it was found the battery energy efficiency could range between 92% (NCA) to 99% (NMC) depending on the cathode material (White et al. 2020). In some cases, energy efficiency of LIBs is considered to be 95% during EV use and 91% during second use (Tao et al. 2021). The impact of loss in energy efficiency for second life use was found to be minor. Several authors indicate that efficiency loss and short lifetime of second life can be compensated by preventing the production of new LiBs for energy storage applications due to the significant environmental benefits obtained using repurposed EV battery (Ahmadi et al. 2017; Cusenza et al. 2019; Faria et al. 2014; Philippot et al. 2022; Ioakimidis et al. 2019; Bobba et al. 2020a).

2.4 Life Cycle Assessment Method

This section begins with an overview of the LCA methodology, different types of LCA, and a description of the key input and output parameters. This is followed by literature studies undertaken for the life cycle assessment (LCA) of EVs in terms of analysing the environmental impact associated with EVs.

2.4.1 Overview of LCA Methodology

A life cycle assessment allows to assess the environmental impacts and resource use of a product system throughout its entire life cycle (Finnveden et al. 2009*)*, where a product system is a collection of activity that are needed to be performed to allow the specific functionality of a product, process or a service (Hauschild et al. 2018). Taking a lifecycle (cradle-to-grave) approach allows for a fair comparison between alternatives without shifting impact between life cycle stages, furthermore, LCA also covers a diverse range of environmental impacts and can include comparison across impact categories to assess trade-off (Hauschild et al. 2018). LCA is fundamentally based on elementary flows of energy and mass balance to assess the emission and waste of the product system. Generally, LCA are used for identifying the important environmental factors in product systems for product development, environmental management and policy making and is widely recognized as one of the effective decision support tools available. (Azapagic & Clift 1999; Tan & Khoo 2005).

The methodology of LCA is outline in ISO 14044:2006 which involves four stages (1) goal and scope (2) inventory analysis (3) Impact assessment and (4) Interpretation, as shown in Figure 2.6. The goal outlines the purpose of the LCA and intended audience, and scope defines the system boundary, level of detail of the LCA and the functional unit (FU) of the study (ISO 2006). The FU is a quantitative measure of the functions that the product system provides, which also allows for fair comparison between product, process, or service of the product system and forms the basis for LCA comparisons (Finnveden et al. 2009). The next step of the LCA is the life cycle inventory (LCI), which is a collection of elementary flow of input (energy and mass resource) and output (emission and waste) data of all activities involved in the product system (ISO 2006). Life cycle impact assessment (LCIA) aims to turn inventory results into aggregated information and involves classifying the elementary flow into a particular environmental category to measure the environmental impact of the entire system. The information is categorized by the characterization factors to either mid-point category or end-point category which depends on the aim of the LCA. The former measures environmentally related flows and are useful for identifying environmental performance, whereas latter measures potential impact of flows on the area of protection (i.e., human health, ecosystem and resource). Life cycle interpretation is the final phase of the LCA step, in which the results of an LCI and LCIA are evaluated according to the defined goal and scope to reach conclusions and recommendations (ISO 2006).

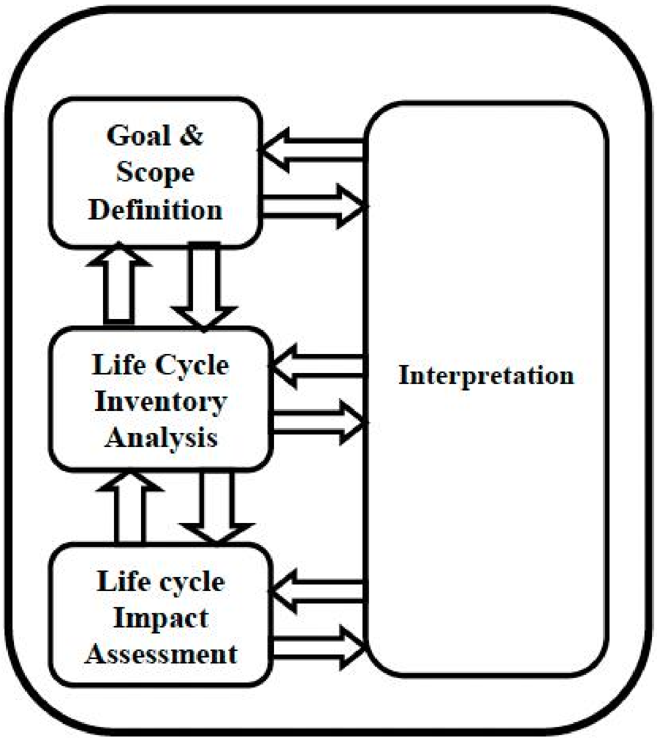

Figure 2.6: LCA framework, taken from ISO (2006).

LCA is further divided into two forms, attributional LCA (ALCA) and Consequential LCA (CLCA). The former is a steady state modelling approach which is focused on describing the environmentally relevant flows to and from a life cycle in which the parameters of the product system remain unchanged, data used in ALCA represent the average data. CLCA, on the other hand, is a changed oriented system which aims to study how environmentally relevant flows change in response to possible decisions (Finnveden et al. 2009). In CLCA marginal data is used to reflect the influence of changes instead of taking average data.

2.4.2 Literature Review on the LCA of EV

Dunn et al. (2012), carried out a cradle to gate attributional life cycle assessment of LiBs in plug-in hybrid electric (PHEV) and BEV that use the LMO cathode material. Their focus was on the impact of energy consumption and emission during different recycling methods for the manufacturing of battery materials.

Hawkins et al. (2013) carried out an attributional LCA of ICEVs and BEVs. The foreground inventory data for vehicles were taken from secondary sources (literature data), for the background processes the Ecoinvent dataset was used. The authors considered two battery types of BEV, this is lithium-ion batteries with LFP and NMC cathode type. The FU of the study was 1km driven under average EU conditions. Vehicle and battery lifetimes are assumed to be 150,000 km, for BEV it is assumed that the vehicle is powered by the average EU grid mix.

A similar study was also conducted by Bauer et al (2015) expanding the vehicle types to include compressed natural gas (CNG) and fuel cell vehicle (FCV) as well. Bauer et al (2015) assumed the vehicle mileage of 240,000km which is higher than the one assumed by Hawkins, et al. (2013), whereas the battery lifetime was taken to be 150,000km. The FU of the study was also one km driven by each vehicle. The focus of the study was to analyse how the impact would change when comparing vehicles at the time of study to the improved vehicles in future i.e., year 2030. The vehicle powertrain improvements considered in the study based on performance, energy use and vehicle mass. The average EU grid mix was taken for the use-phase.

Garcia et al. (2015) carried out a dynamic fleet-based life-cycle greenhouse gas assessment of the introduction of EVs in the Portuguese light-duty fleet from 1995 up to year 2030. The LCA integrated a dynamic material model (MFA) for the vehicle taking account of vehicle age, number of vehicles in the fleet, vehicle weight and fuel reduction, and type of vehicles in the fleet based on three technologies: ICEV diesel and gasoline and BEV. The LCA included the material acquisition, transportation, processing and assembly for the vehicle (including the battery) for the manufacturing phase and electricity grid mixes, fuel usage and maintenance for the use phase. The EoL emission of vehicle and hydrometallurgical recycling for battery was based on previously carried out studies.

Kim et al. (2016), carried out a cradle to grave attributional and process-based LCA of LiB batteries in EV. This included battery production and transportation, analysing the energy use and carbon emissions over the life cycle of EV. The EV considered in the study is compact size Ford Focus with a 24kWh LIBs of LMO/NCM cathode composition. The FU of this study was 1 kWh of battery energy capacity. The LCI is based on the primary data for the BOMs by battery cell and pack supplier and dataset from Greet 2014 model and Ecoinvent. For the EoL stage, the author included the recycling content by giving emission credits during production phase of battery materials.

Gemechu et al. (2017) carried out the supply risk assessment for the production phase of EV based on Hawkins, et al. (2013), without considering the environmental burden from the use-phase and end-of-life-phase. FU of the study was the production of one European standard small EV. The authors used a GeoPolRisk indicator, which was applied to the metals used in the life cycle of an EV.

Raugei et al. (2018) carried out a cradle-to-grave (excluding EoL) life cycle energy analysis of BEV for a range of electricity supply alternatives for the vehicle's use phase: (1) UK grid mix in 2035 (2) conventional combined cycle gas turbine (CCGT) (3) 85% wind and 15% PV generation. The life cycle includes material acquisition, processing, vehicle manufacturing (including the battery) for the manufacturing phase and maintenance, electricity grid mix and fuel supply for the vehicle use phase. The FU of the study was taken as 1 passenger vehicle with a service life of 150,000 km. The background inventory data, material sourcing and battery pack information was modelled using Ecoinvent dataset. The BOMs of the composition of the grid mix and vehicle production was based on previously carried out studies. The vehicle composition was based on the weighted average values for each segment type to represent the energy usage of a typical vehicle in the UK.

Wu et al. (2018), carried out an attributional LCA study to analyse the Global Warming Potential (GWP) of EV and conventional gasoline vehicles, taking account of the vehicle life cycle excluding EoL phase. The analysis was done for 2010, 2014 and 2020 based on different electricity grid mix compositions, including the adoption of combined head and power plant (CHP) for China. The authors took account of vehicle energy efficiency and light-weighting improvements. The study considered primary data for the for the raw materials, gasoline production, energy resources and manufacturing processes based on dataset developed by the China Automotive Technology & Research Centre. The FU of the study was over the lifetime of the vehicle taken to be 150,000km for mid-size passenger car. The battery type opted in this study was lithium-ion battery with LFP cathode which has the same lifetime as the vehicle. The study considered the weight reduction of battery from 2010 to 2020.

Marques et al. (2019) carried out a cradle-to-grave comparative life cycle assessment of two types of LIB chemistry – lithium manganese oxide (LMO) and lithium-ion phosphate (LFP) used in EV, addressing influence of operating conditions, battery capacity fade and location of manufacturing and charging influence. The study FU was 24 kWh EV battery capacity with a service life of 200 000 km. They study assumed the batteries at the EoL are dismantled and sent to pyrometallurgical and hydrometallurgical treatments.

Dolganova et al. (2020) carried out a review of LCA studies on BEV from 2009 to 2018 with focus on resource use assessment.

Hill et at. (2020) carried out a cradle to grave attributional (exception being the consequential elements of feedstock for fuel production) LCA of BEV and other alternative vehicles up to year 2050 with 10-year interval. Taking account of technological advancement (new battery chemistry and higher process efficiencies) and decarbonisation of electricity supply mix. The life cycle included the production of vehicles, maintenance, grid mix supply for the use-phase and EoL treatment of vehicles. The FU of the study was 1 km travelled by a single vehicle.

Bobba et al. (2020a) carried out a combined material flow analysis, life cycle assessment of GWP and supply risk assessment of nickel and lithium for LiB required by the EU fleet of passenger vehicles for year 2020, 2030 and 2050. The MFA took account of the uptake of Evs and change battery chemistries (the energy density of LIB was assumed to remain fixed throughout), linear increase in second life battery and recovery for LiB materials. Including the decarbonisation of the grid mix along time is considering. The life cycle assessment included the production of vehicles, grid mix supply for the use-phase and EoL treatment of vehicles.

Liao et al. (2021) carried out an economic and LCA of commercial of shared autonomous vehicle fleets and the role of V2G service, taking account of fleet life cycle and fuel use based on GREET model. The FU of the study was to meet the travel needs of 20,000 people through to 2050.

Franzò & Nasca (2021), carried out an LCA study of BEV, taking account of the entire vehicle lifetime, including the transportation phase. The focus of the study was mainly on the manufacturing and use phase, comparing the impacts of EV in different countries. The lithium - ion battery with LFP and NMC cathode type was chosen for the study. The battery and vehicle lifetime were assumed to be the same, i.e., 150,000 km. The transport phase included the transport of the vehicle components and the battery to the location of assembly and from assembly to the country in which vehicles were assumed to be driven. For the use phase, the grid mix of Italy was considered for the use phase, which represented 38% of renewable energy penetration in the grid.

Rajaeifar et al. (2021) carried out a life cycle assessment for EoL treatment of LiB used in BEV based on open loop and closed loop recycling of different pyrometallurgical technologies used in combination with hydrometallurgical recycling. The FU of this study was treatment of 1 tonne of LIB modules.

Wang & Yu (2021) carried of a cradle to grave LCA with the exception of use phase for LIB used in passenger BEV with the focus on the battery evolution from NMC 111 to NMC 811. The life cycle assessment included manufacture, transport, collect and recycle a LIB. The function unit was the average weight of BEV battery in China, 300kg. The study assumed at EoL, EV batteries are recycled using closed loop hydrometallurgical recycling processes to recover some of the battery materials (i.e, nickel, cobalt, manganese, aluminium, and copper) to be used again for the manufacturing of new EV batteries. Environmental credits were given to the material energy and emission avoided due to close-loop recycling.

Shafique et al. (2022) carried out a LCA for the manufacturing and use phase of BEV in ten countries based on current and future energy mixes. The LCA included raw material extraction, manufacturing of the vehicle (including battery), transportation for the manufacturing phase and electricity grid mixes and maintenance of BEV for the use phase. The FU of the study was selected for 1 passenger travelled over a distance of 1 km by the vehicle over the EV lifetime of 150,000 km.

Raugei et al. (2022) carried out a systematic review of LCA (cradle to gate) of carbon emission of BEV and ICEV passenger vehicles for year 2020 to future projection to year 2030 and 2050. Literature studies were harmonized based on the FU of transportation provided by one passenger vehicle over 1km for a lifetime of 225,000 km for typical vehicle in EU.

Vilaça et al. (2022) carried out a LCA of passenger vehicle fleet for central region of Portugal. The study investigated the comparison of private automated electric vehicles vs ride-sharing automated electric vehicles. The LCA considered manufacturing, operation (including different energy supply mix) and EoL (including hydrometallurgical recycling process of BEV battery). The FU of the study is an automated electric vehicle fleet with a lifetime of 150,000 km for a study duration of 5 years.

Barkhausen (2024) conducted a combined MFA and a cradle-to-grave LCA of carbon emissions and material requirement of LIBs for vehicle fleet up to the year 2050. The study examined recycling and reuse strategies for batteries with chemistries ranging from NCA and NMC 111 to NMC 811, along with variations in battery lifetimes from 10 to 12 years. The study assumed that at the end of life (EoL), EV batteries are recycled based on the proposed targets set by the European Commission (Regulation (EU) 2023/1542).

2.4.3 Discussion of Environmental Impacts and Key Findings from the Literature

Global Warming Potential

The outputs from LCA analyses are very dependent upon the assumed carbon intensity of the relevant electricity grid(s) for both the manufacturing phases and the use phases. The findings of older publications that provide data on GWP reductions may now be largely out of date.

From Hawkins, et al. (2013) results, EV were found to reduce the overall GWP up to 24% as compared to gasoline vehicles and 14% as compared to diesel vehicles. The main cause of EV GWP was the production phase. Similar finding was concluded by Bauera et al. (2015), the GHG emissions were mostly due to the vehicle production for BEV and FCV, results were between 0.2- 0.25 kg CO_2 kg/km for BEV in 2012, which was roughly slightly less than diesel and gasoline vehicles. According to Bauera et al. (2015) production phase results, the impact for vehicle powertrain (without battery) was almost the same for 2012 and 2030, when compared to other vehicle types. The increase in GHG impact from production phase of the vehicle was mainly due to the battery. Hawkins, et al. (2013) found the battery contributed to 35% - 41% of GWP in the EV production and electric engine contributed to around 7-8%. Raugei et al. (2023) study indicated 50% of GHG emissions of vehicle was found to be associated to the battery packs. Furthermore, it was found the choice of battery also have impact of the GHG emissions (Marques et al. 2019). Dunn, et al. (2012), suggests that approximately half the impact is contributed by aluminium and copper used in the battery assembly. Using 100% recycled aluminium can reduce total energy consumption during BEV production by 33%.

Furthermore, Wu et al. (2018) results indicated, that the GHG emissions during BEV production increased by 13.2% in 2020, as compared to 2014, due to the use of lightweight materials in place of conventional materials (i.e., steel). Whereas, in 2020 electricity grid mix (621.0 gCO2 e/kWh) was assumed to have 18.9% lower emission compared to 2014, the total life cycle GHG emissions of an EV in 2020 (25.7 tCO_2 e/vehicle) were over 16% lower than those in 2014 and 13.4% reduction compared to ICEV in 2020. In the case of prospective scenarios, Bauera, et al (2015) results for year 2030 for GWP was calculated to be 0.09 kgCO2eq/km for BEV, which was mostly due to a more significant reduction in GHG (< 300 gCO2eq/kWh) from the power source i.e., the electricity grid, which is less than half of conventional vehicles. A similar result was shown by Franzò & Nasca (2021), an increase of RES penetration by 17% would reduce the EV CO2 emissions associated to the vehicle use phase by 27%, and an increase by 27% would reduce the EV CO2 emissions associated to the vehicle use phase by 43% (Franzò & Nasca 2021).

According to the results by Franzò & Nasca (2021), during the life cycle of an EV, the use phase is considered to play a major role on the GWP impacts, regardless of the vehicle size, followed by the vehicle and battery manufacturing phases. It was noted the variations in energy requirements for battery manufacturing, the vehicle consumption, and the CO_2 emission levels associated to energy used show huge impact on the total CO_2 emissions of an EV. Moreover, Franzò & Nasca (2021), found moving from small sized vehicles to larger vehicles (from Segment A to D), the CO_2 emissions increase. This is due to the changes in battery size, vehicle weight and battery energy consumption.

Garcia et al. (2015), analysed the impact on overall fleet with the uptake of BEV. They found that although integration of BEV would improve the emission of average distance travelled, but in terms of overall fleet emission this may not be the case due to the increase in fleet vehicle and distance travelled. Liao et al. (2021) compared shared autonomous ICEV and BEV and found BEV can save up to 8 times GHG emission when participating in V2G service and up to 35.8% in GHG emissions excluding V2G service. The author also concluded the GHG emissions could decrease significantly due to the reduced fleet size as more distance is travelled by a single shared vehicle regardless of the impact by increase in battery capacity (24kWh to 74kWh battery pack) (Liao et al. 2021).

Barkhausen (2024), analysed the GWP for LIBs required by EU fleet up to 2050. They found battery materials represent the highest impact followed by use-phase regardless of the change in cathode type of LIBs and improvement in energy density, however they did not consider the evolving grid mix impact on the use-phase. Similarly, Bobba et al. (2020a) analysed the battery GWP for the EU vehicle fleet. They found NMC 111 has the highest battery manufacturing impact per kWh of provided energy. They also found improvement in battery chemistry from NMC 111 to NMC 811 coupled with decarbonisation of the grid mix can lead to significant decrease in GWP. This is decreasing GWP by 22% in 2030 and 31% in 2050 compared to current NMC manufacturing. Whereas the highest reductions were found for the use-phase up to 56% in 2050. However, the authors did not consider the improvement energy density would have on energy consumption, therefore these benefits may be higher.

In terms of EoL, Rajaeifar et al. (2021) analysed different combination of pyrometallurgical and hydrometallurgical stages and found pre-treatment of battery can significantly reduce GHG emissions from the pyrometallurgical furnace usage by removing carbon containing materials. Furthermore, the authors found, open-loop recycling for this combination of recycling would emit more GHG emission without pre-treatment of carbon containing materials. Whereas a closed-loop recycling indicated to be the most beneficial option in terms GHG emissions and recovery of materials.

Toxicity Potential

According to Bauera et al. (2015), toxic substances are mainly released by coal mining and metal mining activities (mainly nickel, copper, platinum, and aluminium, which lead to the increase in human toxicity potential (HTP) in vehicle glider and battery manufacturing, as well as the power transmissions and distribution grids. Shafique et al (2022) discovered most environmental indicator represented impact mainly due to the production lithium-ion battery. Vilaça et al. (2022) found that shared mobility systems can reduce the environmental impact of passenger electric vehicle fleets by 20% to 42%, with HTP showing significant potential for reduction. According to Hill et al. (2020), the use of copper in the battery anode current collector had the most significant contribution to these impacts, to a lesser extent (<20%) due to the use of copper in wiring and the motor. Similar finding was concluded by Hawkins, et al. (2013), the toxicity impacts were mainly due to production phase followed by use phase (similarly for freshwater ecotoxicity potential and eutrophication potential). The production phase HTP were due to additional copper requirements for BEV (vs ICEV), in the case of NMC type LIB, due to nickel requirements, which stems from mine tailings accounting for 75% of HTP, whereas the rest of HTP was due to coal mining activities. Overall, the BEV were found to be 180% to 290% higher in HTP vs ICEV (Hawkins et al. 2013). Considering the improvement in the current grid mix of

2020 in EU, Hill et al. (2020) found that 97% of the total lifetime impacts were dominated by materials used in vehicle and battery manufacturing.

Marques et al. (2019) discovered that the choice of battery type significantly influences the life cycle production and EoL impact, in comparison to operational performance, when comparing LFP and LMO batteries. The authors found that although LFP batteries require fewer replacements during the vehicle's service life, they have a higher production impact across most impact categories and a higher overall life cycle impact for the same battery capacity.

Particulate Matter Formation

The direct PM2.5 emissions are similar for all powertrain types, as they are primarily influenced by significant contributions from brake, tyre, and road wear, rather than exhaust emissions due to the implementation of particulate filters in new vehicles (Hill et al. 2020). In the case of particulate matter formation potential (PMFP), nickel, copper, and aluminium were found to be the main source of emissions from the production phase and electricity mix containing higher proportion coal and lignite combustion to power Evs (Hawkins et al. 2013). PMFP were found to be dominated by SO_2 emission presenting 35% to 46% of impact. For PMFP, LIB followed by vehicle assembly, vehicle chassis and electronic controller presented most of the production impacts (Shafique et al. 2022). A similar finding was found by Hill et al. (2020), that EV have a higher impact due to the manufacturing of batteries but is still lower than ICEVs over their lifetimes.

Photochemical Oxidation Formation Potential

In terms of photochemical oxidation formation potential (POFP), releases of nitrogen oxides are the predominant cause of impact mainly due to ICEV combustion activities followed by mining activities. China EV made the greater contribution to ozone formation due to its high reliance on coal for electricity generation, whereas of Sweden, Norway and France showed the lowest contributions (Shafique et al. 2022). Hawkins et al. (2013) found that EV perform better in comparison to ICEV in terms of POFP. Dunn et al. (2013) also suggests copper and aluminium are the key contributors to emissions of SO_x and NO_x in the manufacturing phase of the battery life cycle. Based on Hill et al. (2020), BEV have a greater effect on POCP compared to PHEVs and FCEVs. However, the impact of BEVs is still lower than ICEVs. These findings can be attributed to the manufacturing process of batteries, which contributes to the higher overall impact of BEVs in terms of POCP.

Metal Depletion

Metal depletion is a common concern with EVs due to their reliance on critical and rare earth metals. According to Hawkins et al. (2013) analysis the metal depletion of EVs is roughly three times than that of ICEVs (ReCiPe method used for metal depletion in this study, does not include the characterization factors for lithium).

Gemechu et al. (2017) study shows the global impact per EV are dominated by copper, aluminium, and steel, whereas magnesium and neodymium are most relevant to the geopolitical-related supply risk. Further, both magnesium and neodymium have 'low environmental impact contribution' due to their low mass requirements in EVs. The biggest impact was found to be due to the manufacturing of the batteries, specifically metal supply for cell production. The total depletion impact of metals in the powertrain of EV would be to larger due to the increase mass of the battery, to the lesser extend caused by electric motor and inverter, and sensitivity of the materials it contains, which is mostly copper and aluminium. According to Gemechu et al. (2017), metals such as Fe, Al, and Cu are used in large quantity during the life cycle of EV, but do not display supply risks. This is because their production is widely distributed around the world. However, the systematic review carried out in section 2.1 suggest although copper may be found in large quantity, it is still expected to face supply risk challenges due to the demand of copper from developing nations.

Based on a resource assessment review carried out by Dolganova et al. 2020, BEVs were found to show higher metal depletion vs ICEV and in the case of batteries this was mainly due to lithium, manganese, copper, and nickel (Dolganova et al. 2020), which contradicts the finding from Gemechu et al. 2017. The authors suggested battery chemistries that do not rely on cobalt, nickel, or copper are considered to be advantageous because the reduction of these materials can minimize overall resource criticality. Wang et al 2023, considered the evolution of LIB (NMC 111 to NMC 811) and found the resource depletion decreased from 12.7 kg to 6.82kg for the battery. They found almost all the resource depletion happened during the manufacturing process of battery raw materials in their study.

MFA conducted by Bobba et al. (2020a) indicated the second use of LIB could delay the availability of lithium and nickel for recycling and hence reduce secondary content.

Energy Consumption

According to Hawkins et al. (2013) analysis, fossil depletion potential could decrease by 25% to 36% with electric transportation, when EVs are powered by the average European electricity grid mix. Dunn et al. (2013), suggest that using 100% recycled aluminium can reduce total energy consumption during BEV production by 33%. The authors also analysed the impact of recycling which suggests, in case of a closed-loop recycling scenario with direct recycling process, almost half of the total manufacturing energy consumption of the battery, made from the raw materials are conserved when cathode material, aluminium, and copper are recycled.

Raugei et al. (2018) result found the higher efficiency of the electric power train contributes significantly to reducing non-renewable cumulative energy demand (nr-CED) of BEV vs ICEV even when BEV is powered by 100% non-renewable generation. Based on the results nr-CED was found to be 34% lower for compact vehicle and 36% lower for an average vehicle in the UK based on the 2016 UK electricity grid mix, when compared to similar size ICEV. In the case of 100% renewable generation to power BEV, material input for the manufacturing phase were found to be a significant impact for nr-CED of the vehicle life cycle.

Wang & Yu (2021), found the nr-CED decreases from 42,400MJ to 39,100MJ due to the shift from NMC 111 to NMC 811, which lead to the reduction in energy consumed during the manufacturing process of battery raw materials from 64% to 61%. The authors found comparison to the battery evolution, the impact of hydrometallurgical battery recycling is more significant, such that when considering recycling of LIB, the non-renewable energy consumption was found to reduce further to 23,500MJ.

2.4.4 LCA Literature Review Results

The literature review indicated that GWP was the most common impact indicator considered for the BEV LCA, followed by energy analysis and metal depletion. The literature largely points out that EVs, coupled with a low carbon electricity grid mix, offer potential for reducing GWP, Figure 2.7 shows the life cycle GHG emission of BEV based on evolution of electricity grid mix. However, from the literature it was concluded that the adoption of BEVs could lead to the increase in higher human toxicity potential, freshwater eco-toxicity and metal depletion impacts (Hawkins, et al., 2013; Bauera, et al., 2015; Marques et al 2019; Hill et al 2020). It was noted that the problem arises due to metal requirements of EV manufacturing (Gemechu et al. 2017; Hawkins et al. 2013; Bauera, et al. 2015). Hence, the need for recycling and reuse may be seen to reduce some of these impacts (Dunn et al. 2012; Wang et al. 2021).

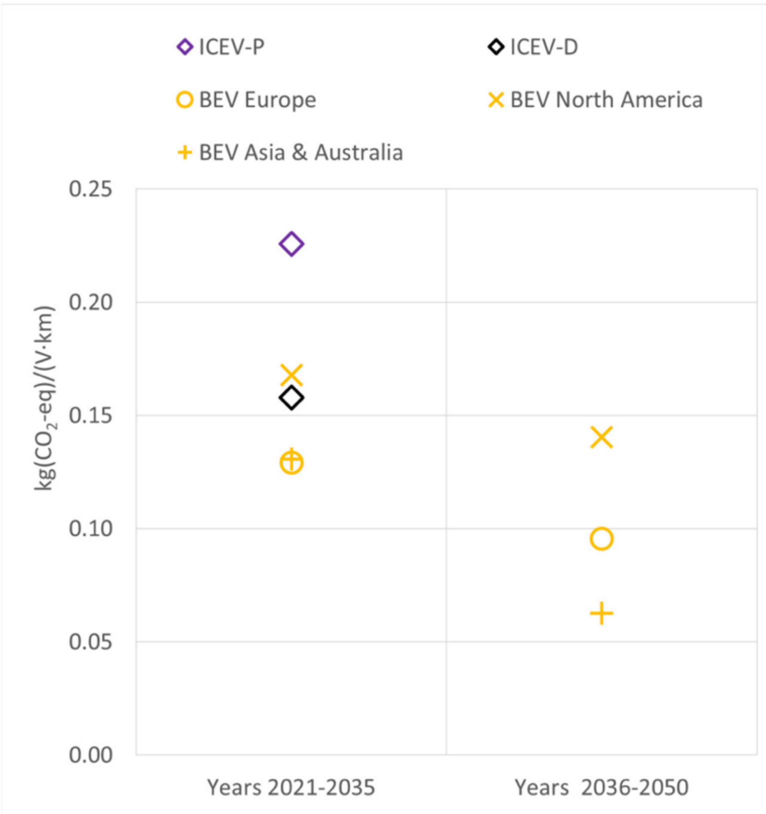

Figure 2.7: Life cycle GHG emissions of ICEV and BEV based on dynamic evolution of grid mix used to power BEV batteries over their lifespan (figure adopted with permission from Raugei et al 2022b).

It was observed during the review of literatures that EV results on GWP are sensitive to assumptions made regarding the penetration of renewables in the electricity grid, vehicle lifetime and use-phase energy consumption. The electricity grid mix is one of the most important parameters for the LCA calculation. It was noted that using the electricity grid mix based on increased penetration of renewable sources, leads to a significant reduction in GWP from 12%-24% to more than 60% reduction in GWP (Hawkins et al. 2013; Bauera et al. 2015; Hill et at. 2020; Raugei et al. 2023).

Some studies considered technological improvements of vehicle such as performance, EV energy consumption and mass as well as battery pack weight reduction. Regardless of the improvements, it was noted that HTP was still significantly high as compared to ICEVs (Bauera, et al. 2015; Wu, et al. 2018), and the benefits were seen in terms of GWP and nr-CED (Raugei et al. 2018; Franzò & Nasca 2021). It was noted that the increase in vehicle size could lead to higher GWP and nr-CED, but were still found to be less compared to ICEVs of the same vehicle size (Franzò & Nasca, 2021; Raugei et al. 2018). Authors that looked into the evolution of the battery chemistry found that recycling would play a bigger role of the life cycle impact of the vehicle. Hill et al. (2020) GWP results indicated the biggest impact was due to the cathode production (presenting almost 50% of the impact). The impact of anode was shown to increase but still less than the cathode as more solid-state battery presented the major mix of battery technology.

2.4.5 Knowledge Gaps

The LCAs conducted by the authors mentioned above were mainly focused on individual ownership of a BEV. Very few studies examine the possible impact on the mass adoption of EVs. Some authors only focus on LIB required by the fleet (Bobba et al. 2020a; Barkhausen 2024). None of the studies captured the real time evolution of the vehicle fleet, except for Bobba et al (2020a) and Barkhausen 2024. The studies took account of the battery evolution and the uptake of BEVs, but their study was limited to life cycle impact of LIB required by the fleet. Additionally, in the case of Barkhausen (2024), LIB reuse was not incorporated into the LCA, which means the benefits of battery circularity are not fully realized. There is clear indication that the use of BEV is beneficial in terms of GHG emission. However, this may not necessarily be the case in terms of overall fleet emission due to the possibility of the increase in fleet vehicle and distance travelled (Garcia & Freire 2017). When considering shared mobility BEVs, the benefits of reducing fleet size showed a positive impact on GHG emissions (Liao et al. 2021) and an improvement in the environmental performance of the fleet production phase (Vilaça et al. 2022). Overall fleet studies were found to have limited consideration of TaaS, technology improvements, the impact of evolving grid mix and rise in mobility needs coupled with the transition to BEV on GHG emissions and the impact on other environmental indicators such as HTP and metal availability.

Furthermore, metal depletion is one of the major concerns regarding BEV production. Battery and vehicle powertrain recycling can conserve some of the metal depletion in close-loop recycling as well as contribute to significant reduction in energy consumption during BEV production (Dunn, et al. 2012; Rajaeifar et al. 2021). However, there is significant delay between the times when batteries are manufactured and when they are available for the recycling after BEV EoL, the impact of which needs to be analysed in order to quantify the total benefits of batteries recycling on the adoption of BEVs.

Generally, attributional LCA is well suited for products that are already offered on the market and where changes in production do not result in any large-scale consequences. When decisions are being analysed that may result in large scale changes of an entire system, a consequential LCA approach might be needed. In a consequential LCA, activities are linked to include all aspects that are expected to change as a consequence of a demand for the specific product in a system.

Despite the growing body of scientific literature looking into the environmental performance of present and future electricity grid mixes, and of electric vs. conventional vehicles, a need remains for overarching, fully integrated consequential environmental analyses of the complex landscape of scenarios that may soon unfold as a result of the intricate interplay between the co-evolving energy and transport sectors.

2.5 Key Points Arising from Published Literature

- The acceleration to a low carbon energy transition requires large amount of critical raw materials which are critical in the sense of their availability capacity, concentrated supply chain networks, environmental impact, and social issues. The issues for each critical elements for energy transitions were discussed and summarised in section 2.1 along with the knowledge gap. It was found the focus has been mainly dominated by evaluating supply risk in terms of raw material availability and mining concentration. The social and environmental studies were least discussed among the literature.

- The literature findings highlight that the transition to a low carbon energy system is possible but requires efforts to address supply concerns and requires strategic planning of mix of energy technologies. These include achieving circularity in the near future due to the growth mainly from the transport sector for cobalt in LIBs, platinum used in fuel cell and electrolyser, iridium used in electrolyser and dysprosium used in permanent magnets. Copper was found to be the most concerning due to the expected demand from developing nations in addition to the demand for energy transition. The geopolitical, social, and environmental risk for lithium, cobalt, REE and PGM could also act as hinder to the reliability and security for future supply as demand for these elements continue to grow.

- There is a clear understanding in the literature that the move towards TaaS can assist in reducing resources and GHG emissions through reducing the number of passenger vehicles on the road. However, currently there is very limited data on certain types of shared mobility in particularly ridesharing for the UK. The impact of TaaS on road traffic is also difficult to predict given the on-going changes in transport system and technologies. There is wide range of uncertainty such as occupancy for types of TaaS vehicles, TaaS vehicle mileage and EoL for TaaS vehicles, and how these changes will impact private ownership. An average mobility service for TaaS vehicles will be considered to displace private ownership. TaaS will be introduced in the Methodology, Material Analysis and CLCA in Chapters 3, 4 and 6 respectively.

- Mechanical and hydrometallurgical processes currently dominate battery recycling technologies. These processes enable recovery of critical battery elements. Assumptions for the recycled material are used in Chapters 4 and 6 respectively.

- Battery lifetimes vary considerably with vehicle applications and usage. Chapter 6 employs data and assumptions associated with a variety of scenarios.

- The outputs from published LCA analyses are very dependent upon the assumed carbon intensity of the electricity grid. The carbon intensity affects aspects of manufactured components of the vehicles and their batteries, but more specifically the use phase of the vehicle (lifetimes considered, total distance travelled and assumed energy consumption).

- In consideration of the carbon intensity of the relevant electricity grid, which of course varies between countries, LCA studies that are more than, say, five years old are far less valuable than more recent studies. This is because the general trend is for a considerable carbon intensity reduction of the grid over time in several developed countries. This aspect is reviewed and discussed in Chapter 5.

- LIBs plays a critical role on the various environmental indicators. Almost 50% of the impacts of LIBs are attributed due to the cathode productions, which will be modelled using a closed-loop recycling strategy. Copper and Aluminium for the battery were also found to be a main contributor to various environmental indictors, and these are considered to remain in open-loop recycling as they are used in various applications in comparison.

3 Methodology

The focus of the research outlined in Chapter 1 is on the passenger light duty vehicle (4-wheeled cars and small vans) fleet which represents the major contribution to GHG emissions from the transport sectors. The research framework takes account of the changes in the UK mobility sector and the evolution of the UK electricity grid mix, as well as a circular strategy for electric mobility batteries and shared strategy for mobility services. The chapter starts off by describing the research framework which encompasses all the components and their relationship to each other. The chapter then describes two methodological approaches that make up the overall research framework.

3.1 Overview

In a global context, the main concerns regarding batteries are the potential environmental impacts and supply risk of cathode materials. The latter concerns arise from expected reserve constraints and concentrated supply, which may lead to supply chain disruptions as demand for electric vehicle batteries increases and hinder potential markets of battery manufacturing (Månberger & Johansson 2019; Hache et al. 2019; Heredia et al. 2020; Junne et al. 2020; Graham et al. 2021; Seck et al. 2022). Completing markets for nickel used in production of stainless steel from other sectors is expected to result in a considerable increase in future nickel demand (Henckens & Worrell 2020; Guohua et al. 2021). Additionally, the shift towards high-nickel-content batteries further exacerbates this concern.

The project is part funded by Faraday Institute’s Recycling and Reuse of EV Lithium-ion Batteries (ReLIB) project, with an interest to understand the demand for lithium-ion battery cathode materials and to what extend battery recycling and reuse and help enhance overall efficiency of the supply chain in the UK and achieve environmental sound management of the materials contained in the battery pack.

The mobility and electricity sector are undergoing various evolutions of decarbonisation. The vehicle fleet is expected to grow over-time to meet the future mobility needs. Furthermore, there is an expected increase in electric mobility and all ultra-low emission vehicles by the end of 2050 following the government policy objectives. Hence, the resources required for the vehicle fleet will also vary over time. The UK electricity grid is also undergoing significant changes to run on low carbon energy resource by 2035 (National Grid 2023) and to achieve the net zero GHG emission target by 2050. The on-going electrification of various sectors including transport to achieve the UK decarbonisation target is expected to substantially increase the future electricity demand. Hence, there will be significant investment of energy and resource in storage systems, renewable and other low carbon energy technologies including grid network lines to support their integration and the additional demand due to on-going electrification.

Batteries play a vital role in both the energy transition of both the transportation and electricity sector by enabling the electrification of transportation and supporting the increase of renewables in the electricity mix, hence there is an expected increase in demand for batteries. Whilst the lithium-ion battery (LIB) is the main and mature technology for both sectors, the chemistry of lithium-ion batteries is still evolving towards better energy density for battery electric vehicles (BEVs) and reduced reliance on critical elements. Hence, the resource required for battery elements is expected to vary over-time due to change in battery chemistry and co-evolution of transport and electricity sectors.

Capturing the shift in future trends allows understanding on the how the resource use and environmental trade-off will vary over time to meet future decarbonisation objectives. The three principal elements, vehicle fleet, electricity grid and battery technology are captured in Figure 3.1 in the form of bubbles expanding or shrinking in mass and energy flow as the technologies and their resource requirement for these elements change overtime, dictating the overall environmental interaction.

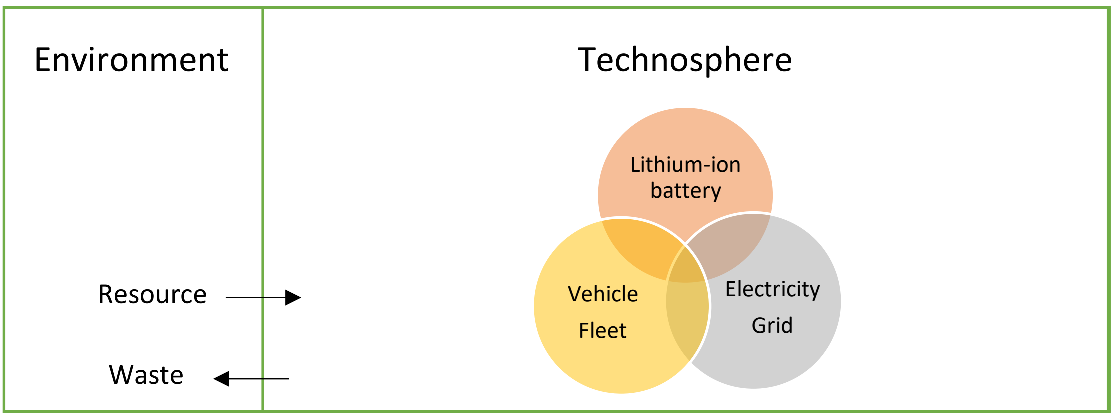

Figure 3.1: Captures the principal elements varying over-time and their interaction with the ecosystem.

The transition to electric mobility has several interlinkages operating at multiple levels. To assess the resource use and environmental impacts of the future vehicle fleet, a whole system perspective is taken along with potential mitigating strategies, discussed in this section, which includes:

- Uptake of Electric Mobility
- Evolution of the Electricity Grid Mix
- Evolution of the battery chemistry
- Battery Reuse and Recycling Opportunity
- Transition from Vehicle Ownership to Transportation as Services

Figure 3.2 represents the influence diagram comprising of interlinkages between the principal elements. Uptake of shared mobility has a potential to improve vehicle utilization and reduce the number of vehicles on UK road, and associated vehicle resources and environmental impacts. Additionally, a more widespread adoption of transport as a service (TaaS) would also make it more practical to implement vehicle-to-grid (V2G) energy storage schemes. Since the shared mobility concept in the UK is still at its early stage and represent significant lack of data and uncertainty on mobility services provided by various types of shared mobility schemes, an average of all TaaS vehicle will be considered in this thesis to represent mobility services. Furthermore, LIB evolution is also taken into account, as it plays a critical role on various environmental indicators (see section 2.4). A closed-loop recycling and reuse scenario is considered for the active battery cathode material (lithium, nickel, cobalt and magnesium), as these metals are used in large quantities in BEVs and play a very vital role in the evolution of the mobility sector. Battery reuse and recycling represent an opportunity to mitigate some of the resource and environmental impact concerns by (1) extending the battery life in less power-intensive electricity grid applications to support the uptake of low carbon resources once EV batteries have reached End-of-Life (EoL) in BEVs; (2) recovering some of the critical battery materials by closed-loop recycling to be reused for manufacturing of new batteries.

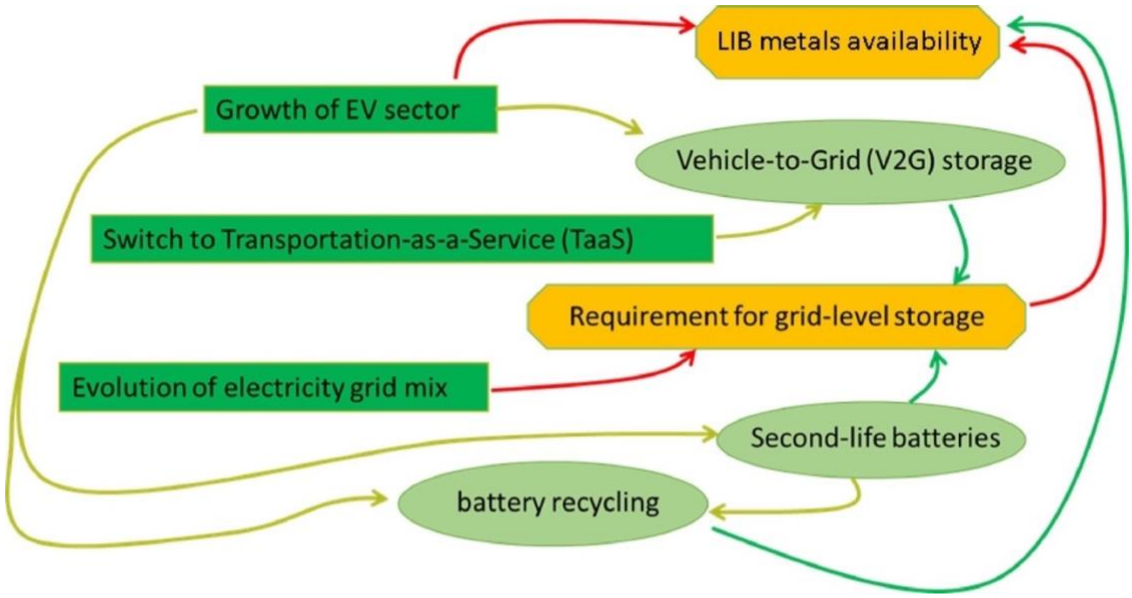

Figure 3.2: Influence diagram showing three on-going transformative transitions (decision nodes = dark green rectangles), the associated challenges for the future ("value" nodes = orange octagons), and the possible technological solutions (uncertainty nodes = light green ovals). Light green arrows indicate direct dependence; dark green arrows indicate a probabilistically conditioned reduction in severity for the challenge they point to; red arrows indicate a probabilistically conditioned increase in severity.

3.2 Framework outline

There are three main approaches for future scenario planning (1) predictive (what will happen) (2) explorative (what can happen) and (3) normative (how to reach a specific target) outlined in Borjeson et al (2006). Predictive and explorative scenarios look forward into the future, while normative scenarios start from a point in the future and look into potential pathways to achieve the desired outcome from the present (Fauré et al 2017). Explorative scenarios provide framework for the assessment of policies and strategies by assessing their consequences.

The aim of the thesis is to investigate the trade-off of different BEV pathways by incorporating resource strategies in this case being the uptake of shared mobility and battery recycling and reuse, and hence an explorative strategic approach was taken for scenario building. The aim of scenarios is to describe a range of consequences on the demand for key battery materials to meet the expected adoption of BEVs and the environmental consequences associated with evolution of the light duty vehicle (LDV) and how these consequences will unfold depending on future trends of transport and energy sectors. The three scenarios are as follows:

(1) **“Worst-case” scenario**: Taking account of future trends but excluding resource strategies.

(2) **“Baseline” scenario**: Taking account of future trends and collection of EoL EV batteries for closed-loop battery recycling and second life use.

(3) **“TaaS” scenario**: Taking account of future trends, collection of EoL EV batteries and uptake of TaaS vehicles. (TaaS vehicles being a mix of carsharing and ridesharing vehicles).

The defined explorative scenarios will influence the requirement for LIBs and their materials, and the third scenario will also influence the number of BEV vehicles on the road, consequently, the environmental trade-off of the defined scenarios. The “worst-case” scenario includes the on-going decarbonisation trend of passenger fleet from ICEVs to BEVs and the evolving grid, but does not consider any resource strategy, therefore serves as a benchmark to helps in contrasting the effects of implementing resource strategies in other scenarios. By assessing this scenario, allows to understand the extent of impact that the absence of resource strategies has on the demand for LIBs and their materials. Since, this scenario does not align with the current UK policy objectives to secure critical materials or improve vehicle occupancy (DTF 2021), it is limited to material analysis only. “Baseline” scenario includes on-going UK decarbonisation trends and the collection of end-of-life (EoL) electric vehicle (EV) batteries. It highlights the impact of a circular economy approach, which involves UK based recycling and reuse of batteries. The scenario helps quantify the benefits of resource strategies on the demand for battery materials and provides insights into the effectiveness of current and potential policies aimed at encouraging battery recycling and reuse, informing future legislative directions. “TaaS” scenario assesses the implications of emerging transportation models like TaaS on LIB demand and environmental sustainability. This scenario provides a comprehensive evaluation of both resource strategies and new transportation models, offering a holistic view of their combined impact. The latter two scenarios are aimed at to inform and guide policy towards achieving a more environmentally sustainable future, considering both resource availability and ecological impact. The development of the scenarios is discussed in more detail in Chapter 4. A combination of material flow analysis

(MFA) and life cycle assessment (LCA) methodologies was applied to quantitatively evaluate the resource and environmental trade-off of the defined scenarios.

The two approaches are part of the industrial ecology tool which are widely applied individually to provide quantitative information about the environmental impacts, materials, and energy flows (Ayres 2002). Traditionally, both these methods are used in a static way for analysing historical and existing system. A classic LCA is used to analysing the environmental performance of the life cycle of product systems to help identify areas for improving environmental burdens (Hauschild et al. 2018). A classic MFA on the other hand provides a snapshot at certain point in time of the materials and energy flows, which allows to identify improvements for reducing waste and optimizing these elementary flows (Brunner & Rechberger 2016). However, due to the limitation that a static approach has in capturing future scenarios and consequences of transformation strategies, the methods of LCA and MFA are continuously evolving to include dynamic and prospective approaches to enabling a deeper understanding of society's future resource, energy use and the environmental impacts (Clift et al. 2015). Furthermore, this includes integrating different ecological approaches to develop new framework for better and more comprehensive understanding the role of new technologies or strategies to help identify potential hotspots and improvement opportunities. Such studies enable a more holistic approach to capture various components and processes within the technosphere and their interactions with the environment. This includes the development of hybrid LCA approaches (combination of CLCA and ALCA), dynamic MFA and combination of MFA with process-based LCA (Clift et al. 2015).

Several authors recommended combining the latter approach to assess resource strategies of future scenarios to allow for a more holistic assessment of the environmental impacts and resource efficiency (Bobba et al. 2020a; Bobba at al. 2020b; Barkhausen et al. 2023; Pinto et al. 2019). Most studies have been focused on the static analysis which does not to take account of the temporal effects of technological changes and strategies. Furthermore, the environmental impacts are aggregated as if they are as if they occurred at the same time. This means the environmental consequences associated with each life cycle stages over a period are evaluated as if they occurred simultaneously (Garcia & Freire 2017). Therefore, this does not allow for a comprehensive understanding of resource and waste flows over-time.

A few studies attempted a dynamic MFA and LCA, to name, Pinto et al. (2019), Sevigne Itzia et al. (2015); Modaresi et al. (2014). The studies outlined the benefits of insight gained by conducting an in-depth temporal coverage to observe variability over time and determine possible changes in trends of energy and mass flows. With all this information it is possible to quantify how changes in material production and recycling systems affect certain transformation strategy. It also allows for a high level of detail and consistency of elementary cycles and flows of the analysed system to assess and quantify the future environmental and resources consequences over-time (Brunner & Rechberger 2016).

This thesis aims to capture the dynamic MFA and LCA framework to assess the environmental trade-off through real-time and capture how the environmental impacts will change as the magnitude for the mass flows various temporally. Figure 3.3: depicts the methodological framework for assessing the demand for key battery materials and environmental trade-off associated with evolution of the light duty vehicle fleet. MFA is a quantitative systematic assessment for illustrating and determining the flow of materials of the analysed system defined in space and time (Kaufman 2012). A dynamic MFA allows to study the behaviour of material over-time, it provides valuable insights into the patterns of material consumption. A dynamic MFA approach is used to capture the demand and track the various mass and energy flows up to the year 2050. The changes in the mass and energy flow will dedicate the LCA inventory for production, use and end of life. The environmental impacts of the system described by the MFA were assessed using a hybrid LCA approach based on a combination of attributional and consequential LCA. As the focus is to assess the environmental benefit of resource strategies, a consequential LCA approach is taken in this case to analysis the consequences of battery recycling and reuse and uptake of shared mobility of the analysed system. The two methodological approach is further explained in section 3.2.1 and 3.2.2.

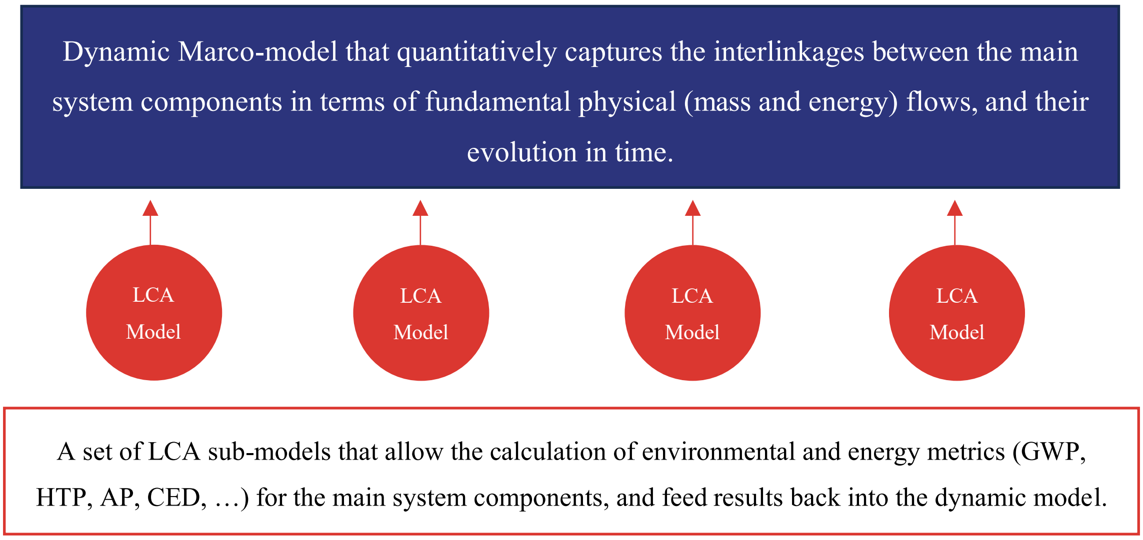

Figure 3.3: Methodological framework for integrated MFA assessing the demand for key battery materials and environmental trade-off associated with evolution of the light duty vehicle fleet.

3.2.1 Material Flow Analysis

In prospective (change-orientated) dynamic studies, scenarios are developed to understand how identified drives such as technology, policy, economics, and consumer behaviour will influence the stock and flow of materials over time (Lanau et al. 2019; Jaar et al. 2022). A bottom-up approach is applied by integrating a time-dimension to account of future trends, change in stock, and related flows.

The main aim of the dynamic MFA is to capture the changes in resource flows (consequential elements of the LCA) of the analysis system, i.e., the demand for passenger vehicles and lithium-ion batteries and to quantify the net demand for battery cathode elements to understand the implication of resource strategies. Due to the expected large growth in battery consumption and changes in transport and grid technologies, the amount and composition of battery resources will continue to vary in the future. As explored in literature Chapter 2.1 of the literature review, battery cathode materials such as cobalt, nickel and lithium are critically scarce. Additionally, cobalt and lithium are also subject to geopolitical, social and environmental risk. Being able to track the net demand for these materials can provide understanding on different approach taken to mitigate their risk. Figure 3.4 shows the main system blocks captured by the material flow diagram.

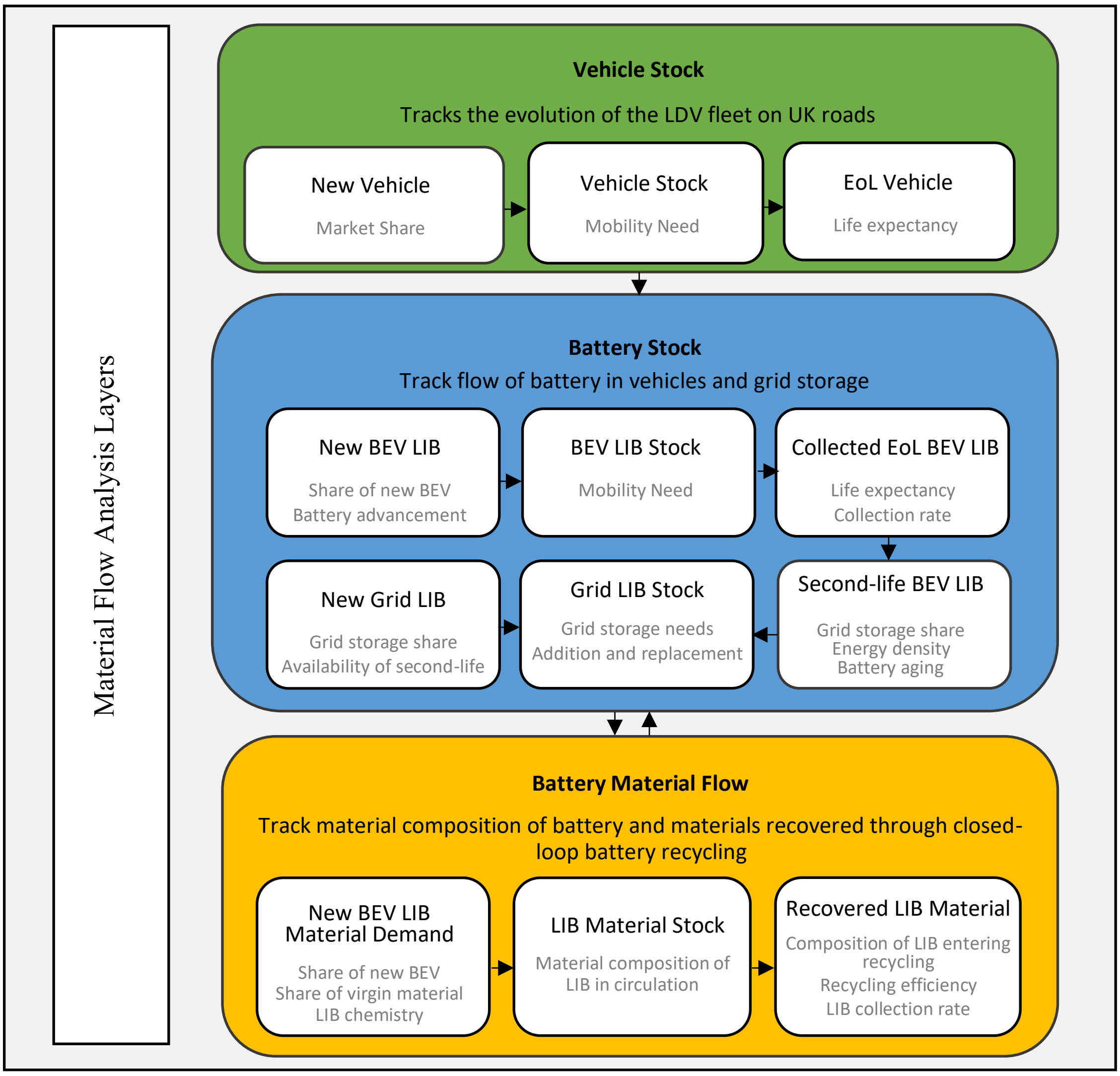

Figure 3.5 Shows the main system blocks captured by the material flow analysis.

MFA model is dynamic in a way that it captures the temporal changes (year-by-year updates) on vehicle stock, trends in battery chemistry, grid storage demand and recycling flows, as well as incorporates scenario-based variations of policies and resource strategies. Although the MFA captures the temporal and changes within the system due to market evolution and policy, it does not dynamically adjust the system structure or relationships between variables (Barlas 2009). Therefore, the MFA is not fully dynamic, since the system boundaries and core relationships remain constant.

Table 3.1 presents the scenario narrative. Excel model was developed for future UK case scenario analysis of material flows and stock levels for the vehicle fleet and battery materials (more detail on the structure of MFA, see Chapter 4). The underpinning model equations are derived based on methodology of MFA models to calculate material demand from dynamic stock flow (Muller et al. 2014; Chen et al. 2015; Garcia et al. 2015; Bobba et al. 2019).

The vehicle fleet tracks the circulation and substitution of different types of vehicles based on the mobility needs, battery evolution tracks the composition changes of battery element over-time, battery recycling tracks the number of batteries entering and batteries elements recovered through recycling, grid storage requirement tracks the number of batteries met through second life, which ultimately provides information on the net demand of battery materials. The evolution of LiB consists of improvement in the cathode material (NMC 622 to NMC 811) including the future reduction in battery pack size, which is expected to reduce resource constraint on the battery production and improve LiB energy density. The grid storage requirement is based on National Grid's FES 2021 future energy scenarios for up 2050. The uptake of EV and number of vehicles on the road was based on the UK road traffic forecast up to 2050 and dictated by UK government plans to ban all sales of new petrol and diesel cars by 2030 and the mobility service provided TaaS. Battery collection is based on the available data and estimated figures for BEV batteries and EoL vehicles (European Commission, 2019a). Key parameters for the MFA scenarios are dictated by the literature Chapter 2 and UK government policies presented in figure 3.4.

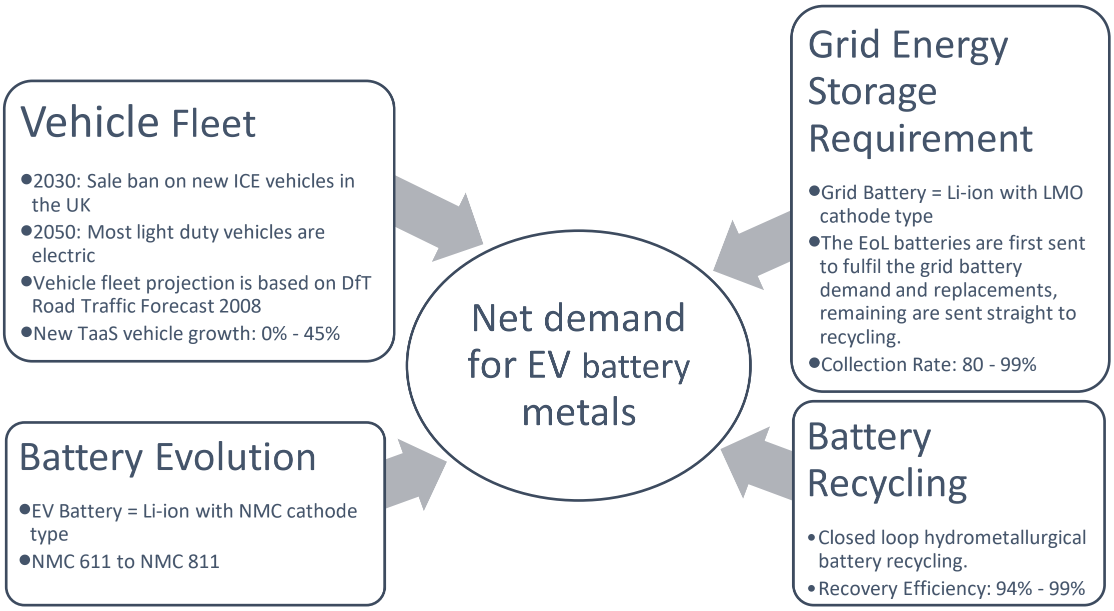

Figure 3.4: Represents the components of MFA, impacting the net demand for BEV batteries and their elements.

Table 3.1: Overview on the three scenarios for the passenger vehicle fleet.

Scenario	**Worst – Case:** Benchmark Scenario	**Baseline:** Battery Circularity	**TaaS:** Battery Circularity & uptake of Transport as a Service (TaaS) Vehicles
Study	Passenger Light Duty Vehicle Fleet		
Analysis	MFA	Integrated MFA and LCA	
Scenario Narrative	Transition to BEV, but no collection of batteries at end of life.	Transition to BEV, batteries are collected for second-life use in grid storage and closed-recycling for BEV battery cathode materials.	Transition to BEV, increase in shared mobility vehicles (car sharing and ride sharing) and batteries are collected same as "Baseline" scenario.
Timeframe	2020 – 2050		
Policy	Sale of fossil fuel vehicles ban by 2030 Zero emission vehicles by 2050		
Mobility Service Need	Forecast growth by Department of Transport		
Grid Storage Need	Forecast growth by National Grid Future Energy Scenario		
Vehicle Types	Private diesel, petrol and electric 4-wheel cars and small vans.		Private diesel, petrol and private and shared electric 4-wheel cars and small vans.
Battery Chemistry	Lithium– ion battery with nickel manganese cobalt (NMC) material composition from NMC 622 to NMC 811		

BEV Battery Mass	Average weighted battery mass by vehicle segment and BEV market share		
Battery Collection	0%	80% (2020) – 99% (2055)	
New TaaS vehicles	0%		0% (2020) – 45% (2050)

3.2.2 Life cycle assessment

A Life Cycle Assessment (LCA) can be used to assess the long term environmental and energy sustainability of the analysed system. A prospective Consequential (CLCA) and Attributional (ALCA) model of the transport-energy system will be developed to assess overall the environmental impact. ALCA considers the internal flow of the product system without considered the effect the system has on its final flows (Sevigné-Itoiz et al. 2015), it is the share of impact linked with a product life cycle (Schaubroeck et al. 2021). CLCA is a type of LCA which aims to describe how the physical flows can change as a consequence of an increase or decrease in demand for the product in a system under study, it is the change in impact induced by a decision and its consequences (Schaubroeck et al. 2021).

A CLCA approach is taken to analyse the effect of battery recycling and reuse and uptake of shared mobility, would have on the environmental indicators. CLCA expands the scope of analysis to the total change in the larger encompassing system arising from the product or process being investigated, in this case the impact of EoL EV on second life grid storage, battery recycling on the battery manufacturing or the uptake of shared mobility on the vehicle manufacturing and use-phase. CLCA uses marginal data to quantify changes within the boundary of the system resulting from the displacement or substitution of these components captured by the dynamic MFA. These components in other words are identified as marginal technologies and are sensitive to supply and demand change over-time (Weidema et al. 2009). The integration of dynamic MFA allows us to determine the change in elementary flows over time of these marginal technologies, which makes it possible to assess the consequences of previous year. The rest of the system is based on the average data. The combination of both types of LCA makes it a suitable method to understand the environmental consequences of different prospective scenarios in UK while also evaluating the overall environmental impact of the analysed system.

All the input and output flows of the various process steps within the analysed and the background inventory are used in order to keep track of all the indirect raw material and energy requirements and emissions that are associated to each ‘foreground’ system are considered in the LCA. The final LCA model included the following foreground:

- The electricity grid mixes to capture the impacts during the operation of EVs.
- Vehicle manufacturing, maintenance, and end of life.
- Li-ion battery manufacturing and closed – loop recycling.

The electricity grid mix plays a critical role in decarbonising the transition to BEVs (explored in chapter 2). This is therefore modelled separately in chapter 5, and the electricity grid is based on FES 2020 “leading the way” (National Grid, 2020). Six impact categories were investigated: greenhouse gas emissions, toxicity, abiotic depletion, acidification potential, abiotic depletion potential, human toxicity potential.

Figure 3.5 shows the structure of the prospective hybrid LCA model. The vehicle fleet LCA model takes account of the manufacturing of LDVs that are newly registered in each year of the analysis, the use phase and decommission of LDVs that reach their EoL in the same year of the analysis, for the operation of the entire LDV fleet. The consequential elements of the LCA are the battery supply chain, second life in grid storage, recycling of battery metals, i.e., Lithium, Cobalt, Nickel and Magnesium and uptake of shared mobility. Five impact categories were investigated: non-renewable energy consumption, greenhouse gas emissions, toxicity, abiotic depletion, photochemical ozone creation potential and human toxicity potential. The life cycle analysis was done using Gabi software and the inventory data collection was based on Eco-invent 3.5 datasets, GREET battery inventory and literature sources.

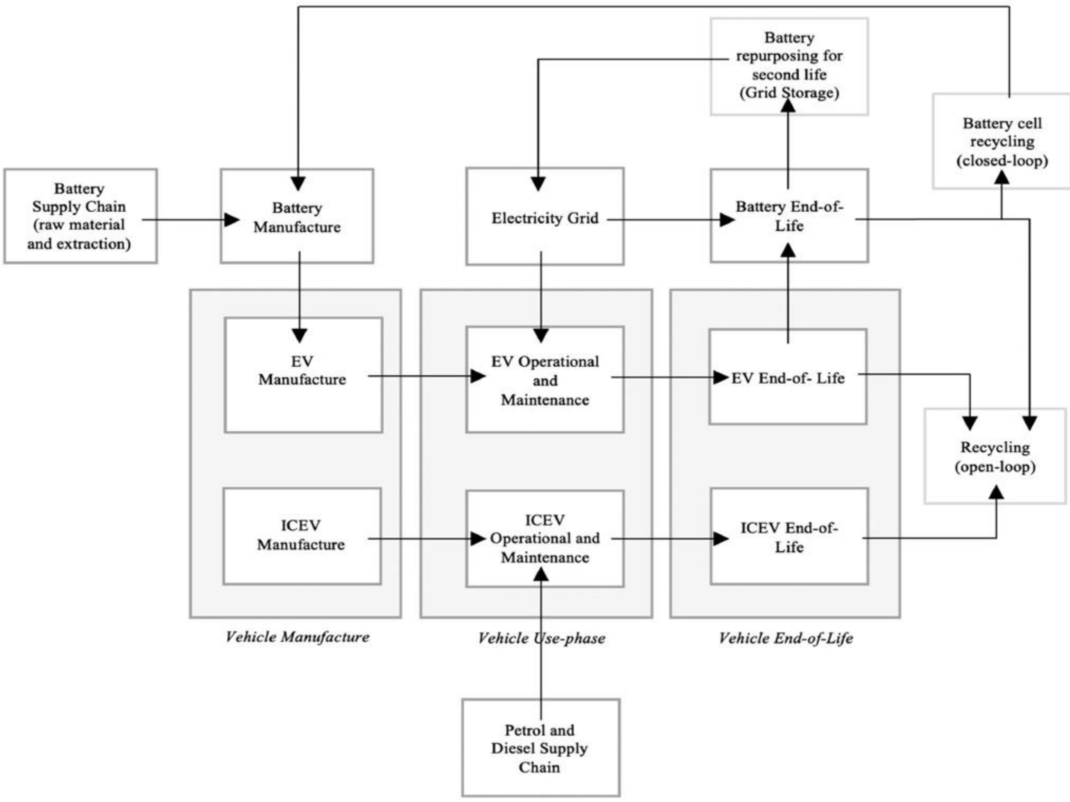

Figure 3.5: Structure of the prospective hybrid LCA model, with identification of the individual sub-models used for each of the key processes comprising the analysed system.

3.2.3 Model Integration

A dynamic fleet-based scenario was previously carried out by Garcia et al. (2015) for the case of Portuguese to examine the GHG impact of fleet transition from ICEV to BEV. The thesis expands the framework to analyse material flow, product stock, and environmental impact for vehicle fleet scenarios to understand the implications of shared mobility and battery circularity. This kind of framework is also known as integrated MFA which incorporate additional evaluation tool (Jaar et al. 2022). Figure: 3.6: Shows the model integration overview for the fleet level environmental assessment.

Traditional LCA are often static, providing a snapshot of environmental impact at a single point in time. Integrating a dynamic MFA introduces a temporal element to track stocks and materials and enables feedback loops within the product system, such as the effects of material recycling and reuse or an increase in shared mobility on future stocks and materials. This allows to account for changes in resource consumption over time which is a crucial feature in assessing the environmental trade-off of a technology transition (Garcia and Freire 2017). Furthermore, in most life cycle studies, impacts are aggregated into a single impact as if they occurred at the same time, when in reality that is not the case. Although aggregated impacts could allow for a simplified comparison between different scenarios, it may also overlook the benefits of strategies and need for staged intervention that might lead to suboptimal policy recommendations. The need for temporal distribution of impacts and analysing transient effects on the scale and timing of adopting new strategies and technologies has gained significant attention in assessing future transitions (Garcia and Freire 2017). The framework for environmental assessment is used to assess the trade-off of "Baseline" and "TaaS" scenarios.

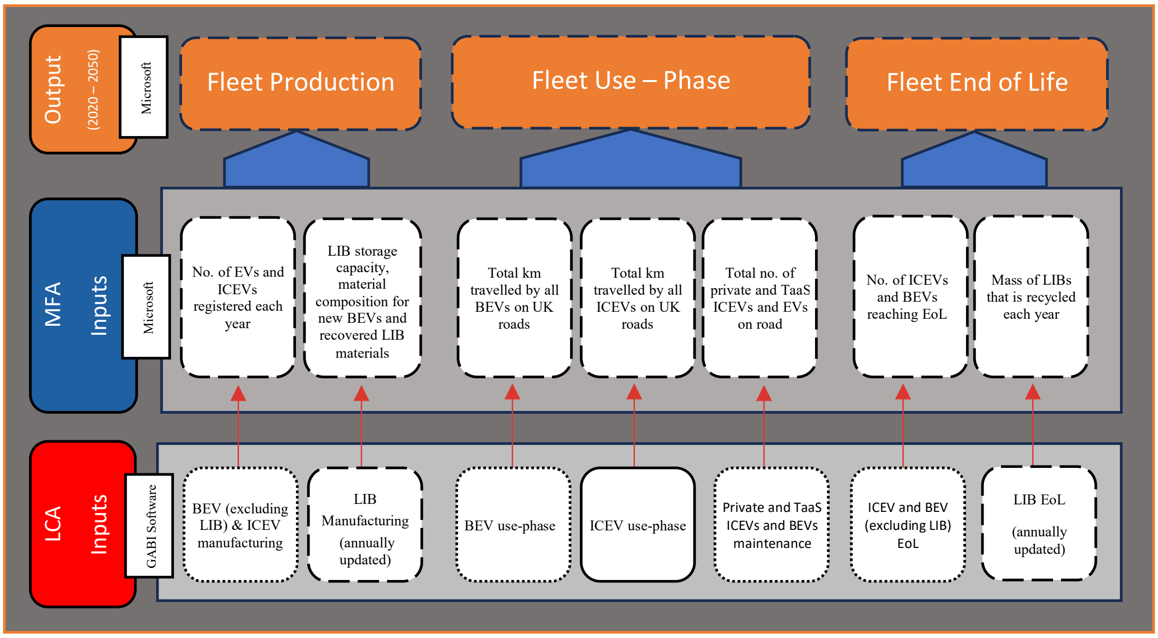

Figure 3.6: Shows the model integration framework for the fleet level environmental assessment. Solid line represents fixed inputs, dash lines represent inputs that vary for each year up to 2050 and dotted line presents inputs varying due to the evolving UK electricity grid mix only for each year up to 2050.

LCA takes a prospective hybrid approach which includes attributional and consequential elements. To assess the environmental benefit of resource strategies, a consequential LCA approach is taken in this case to analysis the consequences of battery recycling and reuse and uptake of shared mobility of the analysed system which is described by the MFA stock and material flow. The transition from ICEV to BEV and the grid mix evolution are considered as prospective elements of the LCA and are dictated by market projection captured by MFA vehicle stock model and the National Grid Future Energy Scenario projection respectively. Whereas all other elements are taken as attributional. The results of environmental impact analysis for each LCA sub-model are linked to material flow analysis developed for each scenario to generate environmental impact at fleet level from 2020 to 2050. This is done by setting up a holistic macro level model for the fleet level LCA results which combines the data extracted from each LCA sub-models and material flow analysis.

This methodology presents several challenges. One primary challenge is that the fleet-level LCA results depend on each sub-LCA model developed in different software, requiring careful data extraction and integration. Additionally, most sub-LCA models operate independently of the material flow analysis, with the exception of the electricity grid mix and battery manufacturing models. These models rely on data from the material flow analysis to determine the share of recovered materials used in battery manufacturing and to track end-of-life (EoL) batteries sent to second-life grid energy storage each year. This dependency adds a layer of complexity to the modelling framework, necessitating coordination and data management across different platforms.

4 Structure of the Material Flow Model

This chapter[2] aims to contribute to laying the foundations for a prospective life-cycle assessment of the co-evolution of the transport and energy sectors in the UK over the next three decades, by illustrating and discussing the results of a dynamic mass flow analysis of passenger vehicles, battery requirement and all the key lithium-ion battery (LIB) metals: lithium (Li), cobalt (Co), manganese (Mn) and nickel (Ni).

4.1 Framework

In prospective dynamic studies, scenarios are developed to forecast the evolution of stocks and related flows under different conditions. Drivers identified in retrospective studies can be used to develop these scenarios and model possible future developments of material cycles, such as their demand, production and availbiliy of secondary resources. (Lanau et al. 2019)

The chapter is structured into four separate but interdependent subsections, which illustrate the data sources, assumptions and calculation approaches used in this study to model: (1) the UK vehicle fleet; (2) the technical evolution of LIBs used in the battery electric vehicle (BEV) sector; (3) LIB solutions to satisfy the future expected demand for grid-level energy storage (including considerations of vehicle to grid (V2G) and second-life BEV battery applications); and (4) LIB recycling. Figure 4.1 represents the material flow with respect to battery materials.

[2] The chapter is based on the article: *Kamran, M., Raugei, M., & Hutchinson, A. (2021). A dynamic material flow analysis of lithium-ion battery metals for electric vehicles and grid storage in the UK: Assessing the impact of shared mobility and end-of-life strategies. Resources, Conservation and Recycling, 167, 105412.*

Three main alternative scenarios as described in the methodology, namely:

1) **"Worst case" scenario:** Transport as a service (TaaS) achieves no penetration in the LDV fleet, and EoL EV LIBs are not collected to be re-used in second life grid storage applications or recycled.

2) **"Baseline" scenario:** TaaS achieves no penetration in the LDV fleet, but widely available and steadily increasing EoL EV LIB collection rates and subsequent second life and recycling.

3) **"TaaS" scenario:** same as (II) but assuming a high penetration of TaaS.

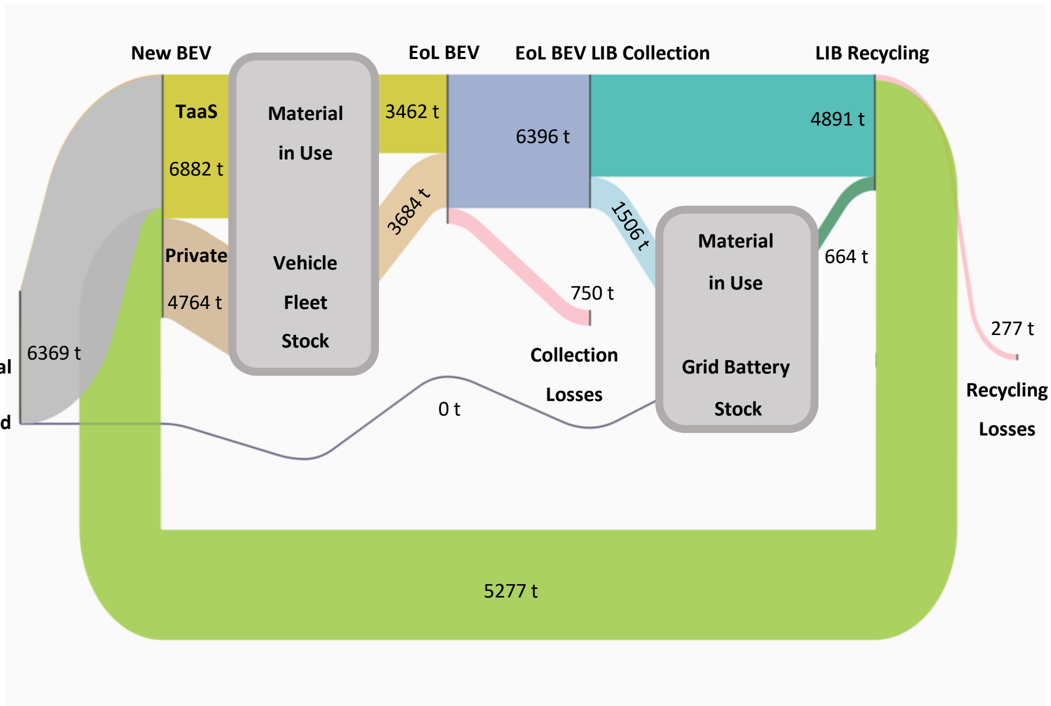

Figure 4.1: Simplified Sankey diagram for the mass flow of battery material though passenger vehicle fleet and electricity grid storage. The diagram presents the flow of lithium in tonnes for year 2035 of the 'TaaS' scenario.

4.2 Vehicle Fleet

The vehicle fleet model tracks the evolution of the passenger light duty vehicles (LDVs) on the UK road, i.e., 4-wheeled passenger cars and small vans. In this study, the "worst case" and "baseline" scenarios assume zero penetration of TaaS, and the "TaaS" scenario assumes a high penetration of TaaS (up to 45% of all new vehicle registrations in 2050). Although in reality, TaaS vehicles already represent a very small percentage in the current LDV fleet, the choice was made to settle on two such clearly defined scenarios in order to explore the clear impacts associated with the uptake of TaaS vehicles, all of which are assumed to be electric. Also, in light of the discussions on the planned ban of petrol and diesel LDV sales, all scenarios are set for a linear progression in the sales of new EVs, to reach 100% in 2030. The total annual distance travelled (DT) by all LDVs on UK roads has been growing at a moderate rate, with slight fluctuations and a clear change in rate of increase at around the year 1990, as illustrated in Figure. 4.1 (Department for Transport, 2018c). The growth in distance travelled is dictated by DfT Road traffic forecasts 2018 "reference" scenario (Department for Transport, 2018c).

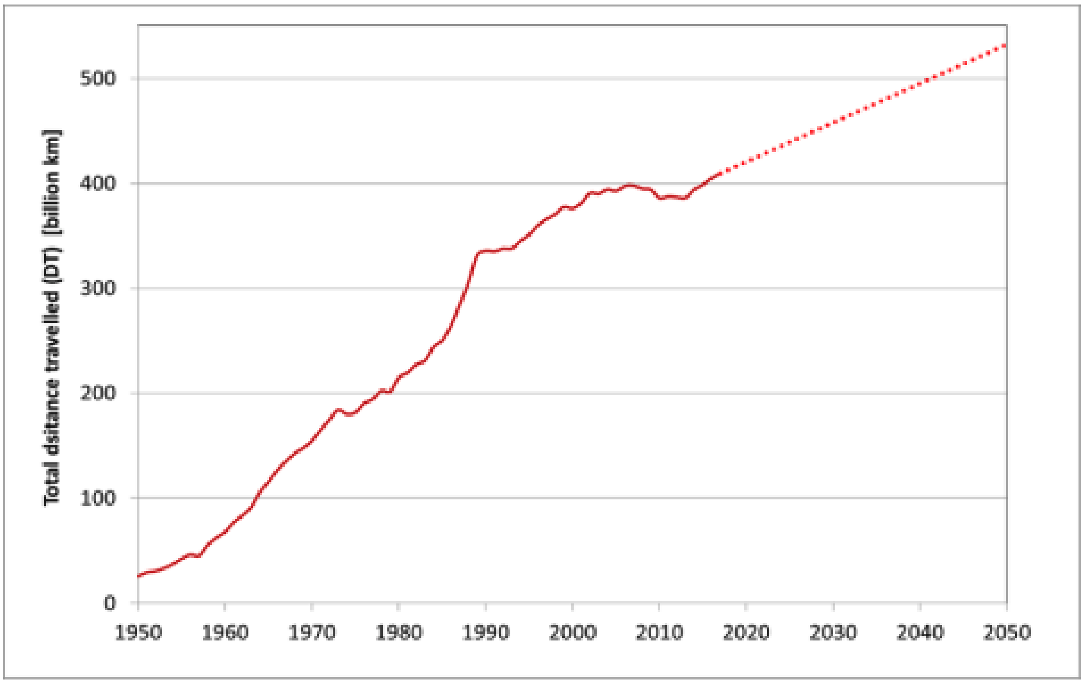

Figure 4.2: Total annual distance travelled (DT) by light duty vehicles on UK roads. Continuous line = historical data (Department for Transport 2018c); dotted line = linear extrapolation to 2050 for UK Road Forecast Scenario 1 (Department for Transport 2018a).

Average ICEVs in the UK have a lifetime millage of around 190,000 km, which is a weighted average for diesel and petrol vehicles (Ricardo-AEA 2015). EVs have fewer powertrain components and moving parts as compared to ICEVs, leading to reduced wear and tear, and hence potentially extended vehicle lifetime mileage (Arbib & Seba 2017). However, this is less likely to be the case in practice, as the service costs, including specifically battery replacement, may make purchasing a new EV a more attractive option, even more so when considering that the EV sector is still young and undergoing rapid evolution. It is assumed that EVs would have the same lifetime mileage as ICEVs.

The numbers of private cars and taxis (including private hire vehicles) on UK roads in 2018 were 30 million and 280 thousand respectively (Department for Transport 2019), and for simplicity, the total DT by all cars on UK roads in 2018 is considered to be covered by private cars. This leads to the average yearly mileage for private vehicles equal to 13,708 km/year, a 14-year vehicle lifespan for a lifetime mileage of 190,000 km; this calculation is in good agreement with the average reported lifespan of vehicles in the UK (Department for Transport 2018b).

For BEVs, it is assumed that the batteries will reach their end of life (EoL) simultaneously with the vehicle after 14 years. This is longer than the average battery warranty of 8 years provided by BEV automakers, it is still consistent with the actual life expectancy for LIBs, reported to range from 10 to 20 years (EDF 2020; Xu et al. 2016).

Due to the lack of data, the yearly mileage for TaaS is taken as the combination of ride-sharing and car-sharing mileages reported in the literature (see chapter 2 section 2), to represent a broad range of TaaS vehicles, i.e., 64,000 km/year, which, combined with a lifetime mileage of 190,000 km, results in a lifetime of 3 years. The number of new private and TaaS vehicles each year is determined by the percentage of new vehicle that are considered to be TaaS. New TaaS vehicle introduced in the fleet is assumed to grow linearly up to 45% by 2050, as shown in Figure 4.3.

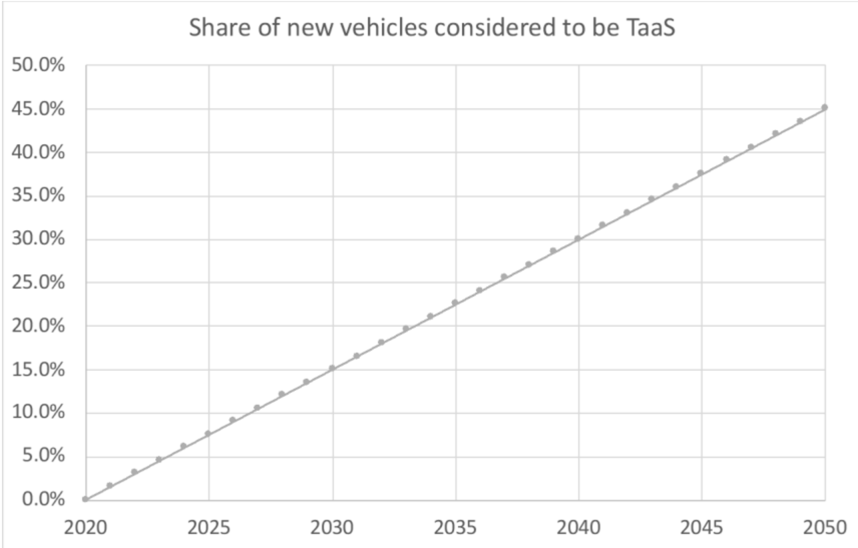

Figure 4.3: Share of new vehicles assumed to be TaaS accounts for the number of TaaS light duty vehicles on the UK road each year for the "TaaS" scenario.

The vehicle fleet model takes account of the vehicles that were already part of the fleet before the starting year of analysis (2018); the decommissioning of such initial vehicle fleet was modelled on the basis of available information on past vehicle registrations for up to 14 years in the past (i.e., from 2003); 2.5% of the total 2018 vehicle fleet could not be accounted for was then assumed to be spread over the course of the first 10 years of analysis (i.e., 2018–2028). Figure 4.4 represents the number of registered light duty vehicles on UK road is taken from 2003 to 2017.

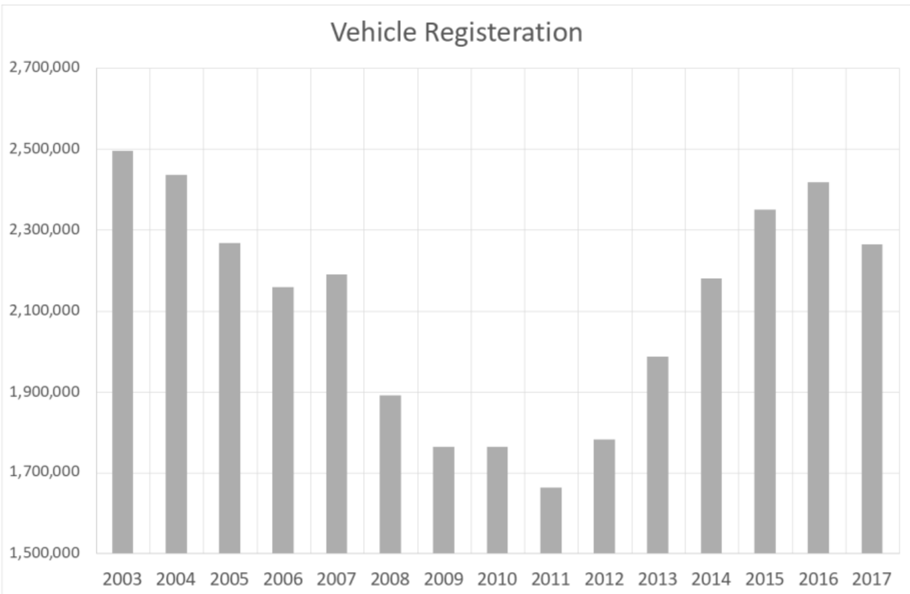

Figure 4.4: The number of registered light duty vehicles on UK road is taken from 2003 to 2017 to take account of the light duty vehicles end-of-life and number of new light duty vehicles introduced.

In the model, the balancing condition is then set that each year, the cumulative decommissioned service amount, i.e., the cumulative km no longer available because of old vehicles reaching their EoL, be exactly compensated for by new vehicle registrations (while considering that each private vehicle is expected to cover 13,708 km/year, and each TaaS vehicle 64,000 km/year). The vehicle fleet model also takes account of the vehicles that were already part of the fleet before the starting year of analysis (2018); the decommissioning of initial vehicle fleet was modelled based on available information on past vehicle registrations. Table 4.2 shows the basic parameters of the vehicle fleet model.

Table 4.1: Parameters and assumptions for EV and LIB.

EV and LIB parameters	Description	Formula	Starting (t=0) /default value	Units	Refs/notes/ assumptions
DT	Total distance travelled by all cars on UK roads each year		409.4 million	km/years	(DfT 2018a)
V_0	Total number of private cars on UK roads in 2017 (year 0)		29,865,900		(DfT 2018a)
M	Vehicle lifetime mileage		190,000	km	LIB 1st lifetime (in EV) assumed = M (Ricardo AEA 2015)
MV_p	Private vehicle yearly mileage	$\frac{DT}{V_0}$	13,708	km/years	
T_p	Private vehicle lifetime	$\frac{M}{MV_p}$	14	years	
MV_t	TaaS vehicle yearly mileage		64,000	km/years	

T_t	TaaS vehicle lifetime	$\frac{M}{MV_t}$	3	years	
VSC	EV LIB pack mass		323	kg	Based on current EV data weighted by vehicle segment, 2017 UK passenger vehicle registration data (Battery University, 2020; Raugei et al., 2018)
VEC	EV electricity consumption (use phase)		0.2	kWh/ km	Based on EV data weighted by vehicle segment (Raugei et al., 2018)

By tracking the evolution of the number of new, in-service and EoL LDVs (subdivided into private ICEVs, private Evs and TaaS Evs) on UK roads with yearly resolution, the model is thus able to predict the associated cumulative demand for LIB materials in the LDV sector, and the annual outflow of EoL EV LIBs. EV battery size and mass varies with vehicle segment. Evs are currently still in an early stage of adoption, and therefore in the LDV fleet model the expected segment composition of the EV fleet as a whole is projected on the basis of the current (2017) ICEV segment composition for all registered vehicles, and since the latter has not changed significantly in recent years, it is then assumed to remain constant through to 2050 for both EV and ICEVs. Thus, the average LIB pack mass is calculated at around 323 kg (see Appendix C), equating to a usable storage capacity of approximately 50 kWh, based on current LIB technology (this latter value is then expected to change in the future with improved in battery technology) (Electric Vehicle Database, 2020; Battery University 2020; Raugei et al., 2018). The detail on structure and modelling assumption are represent in Appendix D.

4.2.1 Vehicle Fleet Model Equations

The vehicle stock flow equation in the MFA model is derived from the similar approach used by Garcia et al. (2015), which calculates vehicle stock based on projected mobility needs.

- Time-dependent Variables are indicated in *italics* using the *V(t)* notation, where *(t)* indicates the running year.
- Fundamental assumed fixed Parameters are indicated in **bold** font, e.g., **P**.

The total number of vehicles on the road:

$$V(t) = ICEVt(t) + EVt(t) + ICEVp(t) + EVp(t) \quad \textbf{(1)}$$

Where,

$V(t)$ = Number of vehicles on the road

$ICEV_t(t)$ = Number of Taas ICEV on roads

$EV_t(t)$ = Number of TaaS EV on roads

$ICEV_p(t)$ = Number of private ICEV on roads

$EV_p(t)$ = Number of private EV on roads

Demand for new service amount (in terms of cumulative km) due to old V decommissioning and net increase in yearly demand for total distance travelled in UK roads:

$$\varDelta DT(t) = \left[eol_{ICEVt(t)} + eol_{EVt(t)}\right] \times \boldsymbol{MYt} + \left[eol_{ICEVp(t)} + eol_{EVp(t)}\right] \times \boldsymbol{MYp} + [DT(t) - DT(t-1)] \quad (2)$$

Where,

$\varDelta DT(t)$ = Required cumulative distance to be met by vehicles (km)

$eol\_ICEV_t(t)$ = Number of TaaS ICEV that have reached their end of life
w
$eol\_EV_t(t)$ = Number of TaaS EV that have reached their end of life

$\mathbf{MY_t}$ = TaaS vehicle yearly mileage (km/year)

$eol\_ICEV_p(t)$ = number of Private ICEV that have reached their end of life

$eol\_EV_p(t)$ = number of Private EV that have reached their end of life

$\mathbf{MY_p}$ = Private vehice yearly mileage (km/year)

$DT(t)$ = Total requirement for distance travelled (overall service) by light duty vehicles on UK road (km)

$DT(t-1)$ = Total requirement for distance travelled (overall service) in the previous year by light duty vehicles (km)

Number of new private and TaaS vehicles:

$$nV_p(t) = \frac{\varDelta DT(t)}{\left[\frac{SnV_t(t)}{[1 - SnV_t(t)]}\right] \times \mathbf{MYt} + \mathbf{MYp}} \quad (3)$$

$$nV_t(t) = \frac{nV_p(t) \times SnV_t(t)}{[1 - SnV_t(t)]} \quad (4)$$

$$nICEVp(t) = nVp(t) \times [1 - SnEVp(t)] \quad (5)$$

$$nEVp(t) = nVp(t) \times SnEVp(t) \quad (6)$$

$$nICEVt\ (t) = nVt(t) \times [1 - SnEVt(t)] \quad (7)$$

$$nEVt\ (t) = nVt(t) \times SnEVt(t) \quad (8)$$

Where,

SnV_t (t) = Share of new vehicles that are TaaS

$SnEV_t\,(t)$ = Share of new TaaS vehicles that are Evs in a given year (this is set to 1 through to 2050 in all analysed scenarios)

$SnEV_p(t)$ = Share of new private vehicles that are Evs in a given year

$nICEV_p(t)$ = Number of new Private ICEV registrations

$nEV_p(t)$ = Number of new Private EV registrations

$nICEV_t\,(t)$ = Number of new TaaS ICEV registrations

$nEV_t\,(t)$ = Number of new TaaS EV registrations

<u>Number of new vehicle registrations:</u>

$$nV\,(t) = nV_t\,(t) + nV_p\,(t) \quad (9)$$

Where,

$nV(t)$ = Number of new vehicle registrations

$nV_t(t)$ = Number of new TaaS V registrations

$nV_p(t)$ = Number of new Private V registrations

<u>Vehicles reaching end-of-life:</u>

$$eol\_V\,(t) = eol\_ICEV_p(t) + eol\_EV_p(t) + eol\_ICEV_t(t) + eol\_EV_t(t) \quad (10)$$

$$eol\_ICEV_p(t) = nICEV_p(t - \mathbf{Tp})^{3} \quad (11)$$

$$eol\_EV_p(t) = nEV_p(t - \mathbf{Tp}) \quad (12)$$

$$eol\_ICEV_t(t) = nICEV_t(t - \mathbf{Tt}) \quad (13)$$

$$eol\_EV_t(t) = nEV(t - \mathbf{Tt}) \quad (14)$$

Where,

$eol\_V(t)$ = Number of vehicles that have reached their end-of-life

Tp = Private vehicle lifetime

Tt = TaaS vehicle lifetime

4.3 EV Battery Evolution

The EV battery evolution model tracks the electrode composition of future generations of lithium-ion batteries, based on the expected battery technology advancement in the automotive sector (discussed in the section 2). The battery composition allows determination of the amount of metals required per battery in each year. This information is then fed to the vehicle fleet model to quantify the annual demand for key battery metals required by the LDV sector. A typical LIB cell is composed of a graphite anode, a metal oxide or phosphate cathode and a liquid electrolyte. The type of cathode formulation dictates the characteristics and the performance of the LIB, and therefore it plays a vital role in its improvement (Liu et al. 2016).

The trends for the amounts of four key metals per kWh of battery pack (Olivetti et al. 2017), lithium (Li), nickel (Ni), manganese (Mn) and cobalt (Co), are tracked in the model in order to

[3] Vehicles which are registered before 2003 or when the date of original first registration is unknown; those are spread over the first 10 years of the analysis.

quantify their corresponding overall demand. The model assumes a linear progression in the improvement in the battery energy density to 2035, due to a gradual hand-over from NMC-622 to NMC-811. After that, a further linear improvement in energy density is expected to take place due to the reduction in the weight of the battery pack casing and ancillary systems. It is further assumed that all energy density improvements will be exploited to achieve increased vehicle driving range, while the overall mass of the battery pack is considered to remain constant. Table 4.2 summarizes the expected bi-modal trends in LIB energy density and key metal contents from 2020 to 2050.

Table 4.2. Summaries the basic parameters regarding the battery evolution adopted from (Element Energy, 2016; IEA, 2020; Olivetti et al. 2017).

Year	**Cathode type**	**kWh(LIB)/ kg(LIB)**	**kg(Li)/ kWh(LIB)**	**kg(Ni)/ kWh(LIB)**	**kg(Mn)/ kWh(LIB)**	**kg(Co)/ kWh(LIB)**
2020	NMC-622	0.15	0.126	0.641	0.2	0.214
2035	NMC-811	0.25	0.111	0.75	0.088	0.094
2050	NMC-811	0.275	0.111	0.75	0.088	0.094

4.3.1 EV Battery Equations

The EV battery equations were derived to track battery improvements over time.

<u>Legend:</u>

- Time-dependent Variables are indicated in *italics* using the *V(t)* notation, where *(t)* indicates the running year
- Fundamental assumed fixed Parameters are indicated in **bold** font, e.g., **P**.

<u>Storage capacity improvement of the battery for each year:</u>

$$C(t) = \rho(t) \times \mathbf{m_V} \quad (15)$$

Where,

$\mathbf{m_V}$ = EV LIB pack mass per vehicle (kg). This is assumed to be constant through to 2050.

$\rho(t)$ = Specific Energy (kWh/kg)

C (t) = Storage capacity (kWh)

Metal "M" mass content in the battery pack for each year[4]:

$$m_{M/LiB}(t) = \rho(t) \times c_M(t) \quad (16)$$

Where,

$m_{M/LiB}(t) =$ Mass content of metal "M" (where "M" = Li, Co, Mn, Ni) per unit mass of battery pack (kg/kg)

$C_M(t)$ = Mass content of metal "M" (where "M" = Li, Co, Mn, Ni) in the battery pack per kWh (kg / kWh)

4.4 Grid Battery Storage

This section examines the potential for second-life batteries to meet the demand for grid storage. The "Grid Battery Storage" model tracks the additional and residual required storage capacity that will be met by both second life and purpose-built LIBs stacks in each year, consistently with the overall yearly requirement for battery storage projected by FES 2020 "Leading the way"

[4] "M" is used here to represent Li, Ni, Mn and Co

Scenario (National Grid, 2020). Figure 4.6 presents the total installed battery storage capacity for each year.

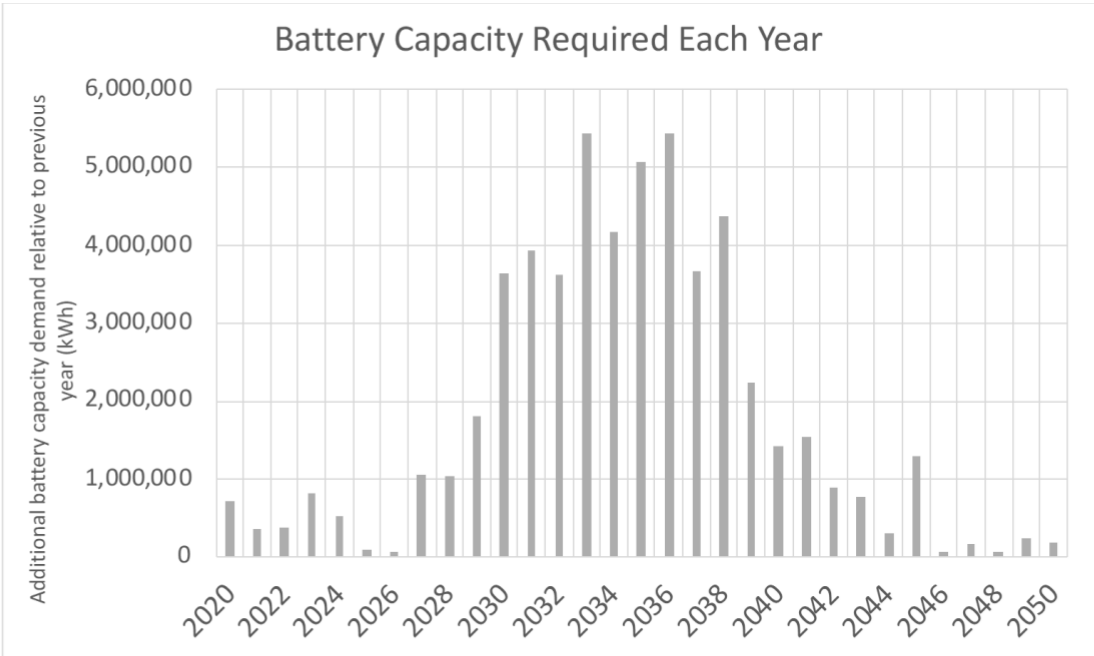

Figure 4.6: Grid battery storage capacity relative to previous year (kWh), based on "Leading the Way" future energy scenario from National Grid (2020).

With regards to the three overall scenarios, in scenario I ("worst case") there is zero collection of EoL EV LIBs, and so the entire battery demand for grid storage is only met by purpose-built LIBs. In scenario II ("baseline") the collected EoL EV LIBs are from private EVs, whereas in scenario III ("TaaS") the collected EoL EV LIBs are from a mix of private and TaaS EVs. Figure 3.7 presents the collection rate assumed for the BEV batteries at the EoL from 2020 to 2050.

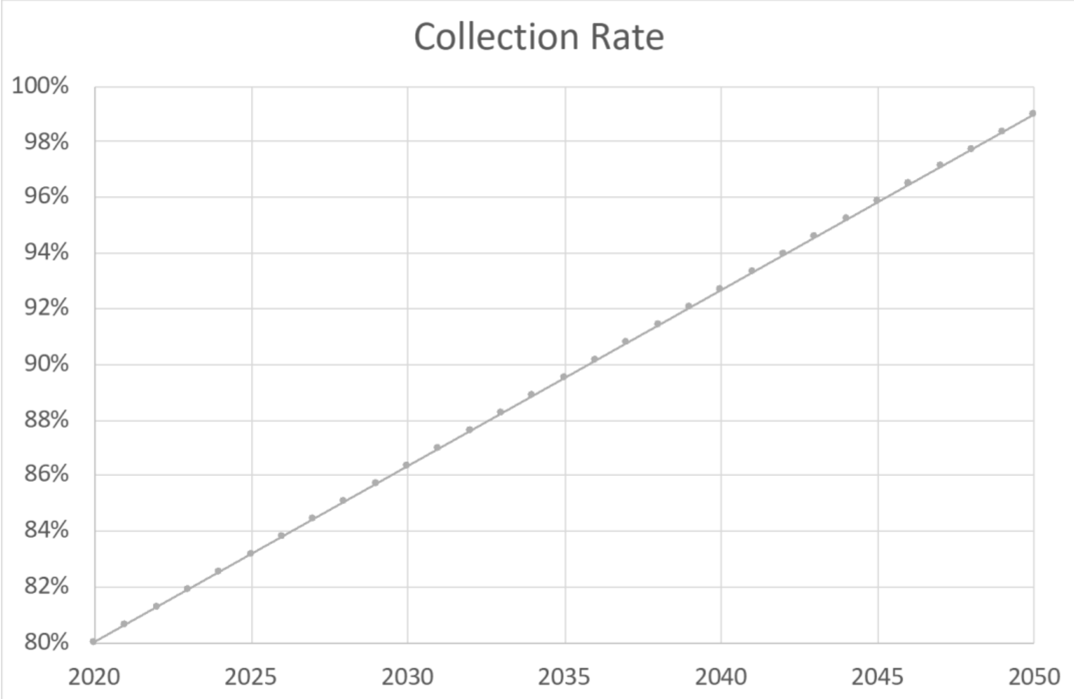

Figure 4.7: The collection rate is assumed to grow linearly from 80% in 2020 to 99% by 2050. The "Worst-Case" assumes zero collection rate of end-of-life BEV batteries.

LIBs in EV applications undergo both power and capacity fade, the latter affecting more significantly the EV driveability performance (Saxena et al. 2015). The capacity fading of EV LIBs at the end of their first life is determined by the combination of cycle aging and calendar aging. Aging due to the number of times the battery is discharged and recharged is known as cycle aging, whereas calendar aging is the natural aging process of the battery, independent of charge and discharge cycles (Xu et al. 2016). Based on expected EV mileage, the trends of cycle aging and calendar aging were extrapolated here from De Gennaro et al. 2020), leading to estimations for the capacity fade at EoL equal to 20% of the initial usable capacity for TaaS EVs, and 30% for Private EVs; in the latter case, the most significant effect was found to be calendar aging (Redondo-Iglesias et al. 2018).

V2G is expected to come into play by year 2026, according to the projections of the FES "Leading the way" scenario, from 22GWh of storage capacity in 2030 to 270GWh in 2050. According to the FES assumptions, a growing share of all BEVs would end up participating in V2G services, up to 45% in 2050 (National Grid 2020). When assuming such V2G storage capacities and engagement ratios in this dynamic material flow model, the additional average yearly load on each EV battery (relative to the drive cycle consumption) remains below 1%. Therefore, it seems unlikely that V2G would significantly impact the average expected LIB lifetimes in the EV fleet as a whole. Although this can be a good opportunity to bring potential value to second life batteries and V2G, there are still technical and regulatory barriers which need to be overcome (Catapult 2020; PricewaterhouseCoopers 2019).

Unlike BEV batteries, stationary grid-level energy storage does not require the high energy density provided by NCA and NMC technologies. Therefore, the model assumes all purpose-built LIBs for grid energy storage are of LMO cathode composition. Energy storage batteries can participate in various grid applications such as reducing peak demand, smoothing the power output from renewable generation, controlling the ramp rate and maintaining the grid frequency (Hesse et al. 2017). The vast number of applications that LIBs can participate in makes determining their remaining lifespan quite uncertain. Based on the lifetime studies for purpose-built grid-level energy storage in various applications, a lifetime of 10 years is assumed for dedicated first-life LIBs (Thorbergsson et al. 2013; Zhang et al. 2019). For second life batteries, studies suggest that batteries retired from their first life after 8 to 10 years could be re-used for an additional 5 to over 10 years, depending on the type of battery applications they participate in (Hossain et al. 2019; Smith et al. 2017; Casals et al. 2019). Some studies have even assumed a 20 year total lifetime, in which second life is determined by subtracting the first life from the total lifetime (Greim et al. 2020; Hossain et al. 2019). In this thesis, a conservative assumption of a 5-year lifespan is made for second-life grid applications. Table 4.3 summarizes the basic parameters and assumption made for second-life NMC LIBs, and grid-dedicated LMO LIBs.

The yearly available storage capacity of EoL EV LIBs is based on the collected amount of EoL LIBs tracked in the "Vehicle Fleet" model and the residual storage capacity of EoL EV LIBs in that particular year. The collection rate for EoL EV LiBs is increased linearly from a conservative 80% in 2020 to 99% in 2050m this based on the collection figures on EV batteries and EoL vehicles (European Commission, 2019a). The residual battery storage capacity at EoL depends on the cathode composition tracked by the "Battery Evolution" model and the capacity fade of the battery in its first life. For EoL private and TaaS BEVs, the composition of the battery is based on the year the EV was introduced in the vehicle fleet, which is 14 years and 3 years before, respectively.

The "Grid Battery Energy Storage" model quantifies the total storage capacity met each year by second-life batteries and purpose-built LIBs up to 2050. Based on the expected installed battery storage capacity (kWh) through to 2050 (National Grid, 2020), the additional demand for storage capacity required each year is calculated. The latter is set to be preferentially met by second-life

LIBs. When the available storage capacity of EoL EV LIBs is not sufficient, the residual demand is then met by purpose-built LIBs in that particular year. As both the second-life and purpose-built LIBs reach the end of their remaining life and need replacement, the model then assumes that the additional storage capacity demand created by such replacement is again preferentially met by EoL EV LIBs.

Table 4.3. Basic second-life and purpose-built LIB parameters and assumptions.

Grid LIB and 2nd life Parameters	Default value	Units	Refs/notes/assumptions
Energy Density of LMO LIB	0.114	kWh/ kg	(Notter et al. 2010)
Li content in LMO LIB	0.006	kg(Li)/ kg (battery pack)	Based on Ecoinvent foreground inventory data on LMO production (Ecoinvent 2020; Notter et al. 2010)
Mn content in LMO LIB	0.099	kg(Mn)/ kg (battery pack)	Based on Ecoinvent foreground inventory data on LMO production (Ecoinvent, 2020; Notter et al., 2010)
Average 1st life lifetime for LIBs used in grid-level applications	10	years	Assumption based on literature studies (Thorbergsson et al. 2013; Zhang et al. 2019)
Average 2nd life lifetime	5	years	Assumption based on literature studies (Hossain et al. 2019; Casals et al. 2019; Smith et al. 2017)
LIB capacity fade at EoL for private Evs	30%		Extrapolated based on (De Gennaro et al. 2020)
LIB capacity fade at EoL for TaaS Evs	20%		Extrapolated based on (De Gennaro et al. 2020)

4.4.1 Grid Battery Storage Equations

The grid battery storage equation in the MFA model is derived to track the demand for grid battery and second-life battery flow over time.

<u>Legend:</u>

- Time-dependent Variables are indicated in *italics* using the *V(t)* notation, where *(t)* indicates the running year
- Fundamental assumed fixed Parameters are indicated in **bold** font, e.g., **P**.

<u>Collected end-of-life LiB for each year:</u>

$$eol_{EVLiB_{coll}}(t) = eol_{EVLiB_{p_{coll}}}(t) + eol_{EVLiB_{t_{coll}}}(t) \tag{17}$$

$$eol_{EVLiB_{p_{coll}}}(t) = eol_{EVLiB_{p}}(t) \times R_{coll}(t) \tag{18}$$

$$eol_{EVLiB_{t_{coll}}}(t) = eol_{EVLiB_{t}}(t) \times R_{coll}(\mathrm{t}) \tag{19}$$

Where,

$eol_{EVLiB_{coll}}(t)$ = Total end-of-life EV batteries collected

$eol_{EVLiB_{p_{coll}}}(t)$= End-of-life private EV batteries collected

$eol_{EVLiB_{t_{coll}}}(t)$ = End-of-life TaaS EV batteries collected

Capacity fade at End-of-life of EV[5]:

$$C_{nameplateEVLiB}(t) = [eol_{EVLiB_{c_{coll}}}(t) \times \rho(t - \mathbf{Tp})] + [eol_{EVLiB_{t_{coll}}}(t) \times \rho(t - \mathbf{Tt})] \quad \textbf{(20)}$$

$$C_{fade}(t) = \frac{[eol_{EVLiB_{c_{coll}}}(t) \times \rho(t - \mathbf{Tp})] \times [eol_{EVLiB_{t_{coll}}}(t) \times \rho(t - \mathbf{Tt})]}{C_{nameplateEVLiB}(t)} + \frac{\mathbf{C_{p_{fade}}} \times \mathbf{C_{t_{fade}}}}{C_{nameplateEVLiB}(t)} \quad \textbf{(21)}$$

Where,

$C_{nameplateEVLiB}(t)$ = Total nameplate storage from EoL Evs (kWh)

$C_{fade}(t)$ = Average capacity fade of private and TaaS end-of-life EV combined (%)

$\mathbf{C_{p_{fade}}}$ = Capacity fade of private EV at the end-of-life (%)

$\mathbf{C_{t_{fade}}}$ = Capacity fade of TaaS EV at the end-of-life (%)

End-of-life EV battery capacity available each year for second-life:

$$C_{p_{eol_{EVLiB}}}(t) = eol_{EVLiB_{p_{coll}}}(t) \times \rho(t - \mathbf{Tp}) \times [1 - C_{fade}(t)] \times \mathbf{RS2} \quad \textbf{(22)}$$

$$C_{t_{eol_{EVLiB}}}(t) = eol\_EVLiB_{t\_coll}(t) \times \rho(t - \mathbf{Tt}) \times [1 - C_{fade}(t)] \times \mathbf{RS2} \quad \textbf{(23)}$$

[5] Weighted average of TaaS and private end-of-life EV battery is taken to simplify how many EV batteries can be repurposed for second-life in grid storage applications.

$$C_{eol_{EVLiB}}(t) = C_{p_{eol_{EVLiB}}}(t) \times C_{t_{eol_{EVLiB}}}(t) \tag{24}$$

Where,

$C_{p_{eol_{EVLiB}}}(t)$ = Storage capacity of end-of-life private EV batteries each year assumed to be available for repurposing (kWh)

$C_{t_{eol_{EVLiB}}}$(t) = Storage capacity of end-of-life TaaS EV batteries each year assumed to be available for repurposing (kWh)

$C_{eol_{EVLIB}}(t)$ = Total available storage capacity of end-of-life EV batteries for repurposing (kWh)

RS2= Share of end-of-life EV batteries sent to second-life relative to optimal (fixed at 1 in the model)

<u>Battery storage demand initially met by purpose-built and EoL EV LIBs[6]:</u>

$$C_{SL}(t) = C_{eol_{EVLiB}}(t) \text{ , if } C_{eol_{EVLiB}}(t) \leq C_{d_g}(t) \tag{25}$$

$$C_{SL}(t) = C_{d_g}(t) \text{ , if } C_{eol_{EVLiB}}(t) \geq C_{d_g}(t) \tag{26}$$

$$C_{PB}(t) = C_{d_g}(t) - C_{SL}(t) \tag{27}$$

Where,

[6] The first three equations are the initial calculation steps before the additional demand created by battery replacements is considered.

$C_{d_g}(t)$= Additional battery capacity demand relative to previous year (kWh), this is based on FES 2020 "Leading the Way" scenario

$C_{SL}(t)$= Storage capacity in end-of-life vehicle batteries initially repurposed for second-life to fulfil $C_{d_g}(t)$ (kWh)

C_{PB}(t) = Grid initial battery storage demand met by purpose-built grid batteries (kWh)

<u>Additional battery storage demand created by the need for battery replacements each year[7]:</u>

$$C_{Add_{SL}}(t) = C_{SL}\ (t - T\mathbf{sl}) \tag{28}$$

$$C_{Add_{PB}}(t) = C_{new_g}\ (t - \mathbf{Tpb}) \tag{29}$$

$$C_{Add}(t) = C_{Add_{SL}}(t) + C_{Add_{PB}}(t) \tag{30}$$

$$C_{RA}(t) = C_{eol_{EVLIB}}(t) - C_{SL}(t) \tag{31}$$

Where,

$C_{Add_{SL}}$(t) = Additional demand for storage created by 1st replacement cycle of second-life batteries (kWh)

Tsl = Assumed second-life battery lifetime (year)

$C_{Add_{PB}}$(t) = Additional demand for storage created by 1st replacement cycle of purpose-built batteries (kWh)

Tpb = Assumed dedicated (purpose-built) battery lifetime (year)

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[7] The following equations are repeated for additional demand created by the required number of grid battery replacement cycles.

$C_{Add}(t)$ = Total additional demand created by 1st replacement of battery (kWh)

$C_{RA}(t)$ = Residual availability of battery storage from end-of-life Evs (kWh)

<u>Total demand for grid battery storage capacity (kWh) met:</u>

$$C_{Total_{SL}}(t) = C_{SL}(t) + \sum C_{Add_{SL}}(t) \quad \textbf{(32)}$$

$$C_{Total_{PB}}(t) = C_{PB}(t) + \sum C_{Add_{PB}}(t) \quad \textbf{(33)}$$

Where,

$C_{Total_{SL}}$ (t) = Total capacity of end-of-life EV batteries repurposed for grid battery storage (kWh)

$C_{Total_{PB}}$ (t) = Total capacity of dedicated (purpose-built) batteries for grid battery storage (kWh)

<u>Share of end-of-life EV batteries that are repurposed for second-life in grid applications:</u>

$$S2_p(t) = \frac{C_{Total_{SL}}(t)}{C_{peol_{EVLIB}}(t) \times \mathbf{RS2} \times C_{fade}(t)} \quad \textbf{(34)}$$

$$S2_t(t) = \frac{C_{Total_{SL}}(t)}{C_{teol_{EVLIB}}(t) \times \mathbf{RS2} \times C_{fade}(t)} \quad \textbf{(35)}$$

$$S2(t) = S2_p(t) + S2_t(t) \quad \textbf{(36)}$$

Where,

$S2_p(t) =$ Share of end-of-life private EV batteries repurposed for second-life (%)

$S2_t(t) =$ Share of end-of-life TaaS EV batteries repurposed for second-life (%)

$S2(t) =$ Total share of end-of-life EV batteries repurposed for second-life (%)

4.5 Battery Recycling

The battery recycling model tracks the retired EV batteries from the vehicle fleet entering recycling after their first and second lives. It also keeps track of the amounts of battery metals (Li, Ni, Mn and Co) respectively required by the vehicle fleet and available after recycling each year. The collection of EoL purpose-built grid batteries has not been considered because there may not be enough incentive for it. It is then assumed in the model that once the battery metals are recovered in a particular year, they are sent straight back to the manufacturing of BEV batteries to meet the demand for BEVs in the same year, as calculated in the vehicle fleet model. Although realistically not all materials may end being reused for BEV LIBs, it has been considered to assess the full benefits of material reuse.

Considering the higher efficiency of metal recovery that is typical of hydrometallurgical recycling processes (explored in section 2), the model assumes that in the future all EV batteries entering recycling will undergo hydrometallurgical treatment. Table 5 summaries the expected recovery efficiencies for the four considered metals using hydrometallurgical recycling (Sheret & Santen, 2007; Chen & Zhou, 2014; Melin 2019; Greim et al. 2020).

Table 4.4. Recovery efficiencies of metals in hydrometallurgical recycling process.

Metals	Recovery Efficiency	Reference
Li	95%	(Greim et al. 2020)
Ni	99%	(Cheret & Santen 2007)

Mn	95%	(Chen & Zhou 2014; Melin 2019)
Co	94%	(Cheret & Santen 2007)

4.5.1 Battery Recycling Equations

The battery recycling equations in the MFA model is derived to track battery entering recycling and battery materials recovered over time.

<u>Legend:</u>

- Time-dependent Variables are indicated in *italics* using the *V(t)* notation, where *(t)* indicates the running year
- Fundamental assumed fixed Parameters are indicated in **bold** font, e.g., **P**.

<u>Batteries required by the light duty vehicle fleet each year (kg):</u>

$$m_F(t) = \left(nEV_t + nEV_p\right) \times \ \mathbf{m_v} \quad \textbf{(37)}$$

$$m_{FM}(t) = m_{EVLiB}(t) \times \ m_{M/_{LiB}}(t) \quad \textbf{(38)}$$

Where,

$m_F(t)$ = Batteries required by the vehicle fleet (kg)

$nEV_p(t)$ = Number of new private EV registrations

$nEV_t(t)$ = Number of new TaaS EV registrations

$\mathbf{m_V}$ = EV battery pack mass (kg), this is assumed to be constant through to 2050

$m_{FM}(t)$ = Metal "M" (where "M" = Li, Co, Mn, Ni) content in light duty vehicle fleet batteries each year (kg)

$m_{\frac{M}{LiB}}(t)$ = Mass content of metal "M" (where "M" = Li, Co, Mn, Ni) per mass of the battery pack (kg)

<u>End-of-life EV batteries each year (kg):</u>

$$m_{pE}(t) = \mathrm{eol}_{\mathrm{EV}_p}(t) \times \mathbf{m_v} \quad \textbf{(39)}$$

$$m_{tE}(\mathrm{t}) = \mathrm{eol}_{\mathrm{EV}_t}(\mathrm{t}) \times \mathbf{m_v} \quad \textbf{(40)}$$

$$m_{pEM}(\mathrm{t}) = m_{pE}(\mathrm{t}) \times \; m_{\frac{M}{LiB}}(t - \mathbf{Tp}) \quad \textbf{(41)}$$

$$m_{tEM}(\mathrm{t}) = m_{tE}(\mathrm{t}) \times \; m_{\frac{M}{LiB}}(t - \mathbf{Tt}) \quad \textbf{(42)}$$

Where,

$m_{pE}(t)$ = End-of-life batteries from private EV (kg)

$\mathrm{eol}_{\mathrm{EV}_p}(t)$ = Number of private EV that have reached their end of life

$m_{tE}(t)$ = End-of-life batteries from TaaS EV (kg)

$\mathrm{eol}_{\mathrm{EV}_t}(\mathrm{t})$ = Number of TaaS EV that have reached their end of life

m_{pEM} = Metal "M" (where "M" = Li, Co, Mn, Ni) content from end-of-life private EV batteries (kg)

$\mathbf{Tp}$ = Private vehicle lifetime

$m_{tEM}(t)$ = Metal “M” (where “M” = Li, Co, Mn, Ni) content from end-of-life TaaS EV batteries (kg)

$\mathbf{Tt}$ = TaaS vehicle lifetime

<u>End-of-life EV batteries send to recycling directly (kg):</u>

$$m_{pR1}(t) = m_{pE}(t) \times R_{coll}(t) \times \left[1 - S2_p(t)\right] \quad \textbf{(43)}$$

$$m_{tR1}(t) = m_{tE}(t) \times R_{coll}(t) \times [1 - S2_t(t)] \quad \textbf{(44)}$$

$$m_{pMR1}(t) = m_{pR1}(t) \times m_{\frac{M}{LiB}}(t - \mathbf{Tp}) \quad \textbf{(45)}$$

$$m_{tMR1}(t) = m_{tR2}(t) \times m_{\frac{M}{LiB}}(t - \mathbf{Tt}) \quad \textbf{(46)}$$

Where,

$m_{pR1}(t)$ = End-of-life private EV batteries send to recycling directly (kg)

$R_{coll}(t)$ = Collected rate of end-of-life EV batteries (%)

$S2_p(t)$ = Share of end-of-life private EV batteries repurposed for second-life (%)

$S2_t(t)$ = Share of end-of-life TaaS EV batteries repurposed for second-life (%)

$m_{pMR1}(t)$ = Metal “M” (where “M” = Li, Co, Mn, Ni) content from end-of-life private EV batteries send to recycling directly (kg)

$m_{tMR1}(t)$ = Metal “M” (where “M” = Li, Co, Mn, Ni) content in end-of-life TaaS EV batteries sent to recycling directly (kg)

End-of-life EV batteries (kg) send to recycling via second-life (kg) each year:

$$m_{pSL}(t) = m_{pE}(t) \times R_c(t) \times S2_p(t) \quad \textbf{(47)}$$

$$m_{tSL}(t) = m_{tE}(t) \times R_c(t) \times S2_t(t) \quad \textbf{(48)}$$

$$m_{pMSL}(t) = m_{pSL}(t) \times m_{\frac{M}{LiB}}(t - \mathbf{Tp}) \quad \textbf{(49)}$$

$$m_{tMSL}(t) = m_{tSL}(t) \times m_{\frac{M}{LiB}}(t - \mathbf{Tt}) \quad \textbf{(50)}$$

$$m_{pR2}(t) = m_{pSL}(t - \mathbf{Tsl}) \quad \textbf{(51)}$$

$$m_{tR3}(t) = m_{tSL}(t - \mathbf{Tsl}) \quad \textbf{(52)}$$

$$m_{pMR2}(t) = m_{pMSL}(t - \mathbf{Tsl}) \quad \textbf{(53)}$$

$$m_{tMR2}(t) = m_{tMSL}(t - \mathbf{Tsl}) \quad \textbf{(54)}$$

Where,

$m_{pSL}(t)$ = End-of-life private EV batteries repurposed for grid storage applications (kg)

$m_{tSL}(t)$ = End-of-life TaaS EV batteries repurposed for grid storage applications (kg)

$m_{pMSL}(t)$ = Metal "M" (where "M" = Li, Co, Mn, Ni) content from end-of-life private EV batteries send to second-life (kg)

$m_{tMSL}(t)$ = Metal "M" (where "M" = Li, Co, Mn, Ni) content from end-of-life TaaS EV batteries send to second-life (kg)

$m_{pR2}(t)$ = End-of-life private EV batteries send to recycling via second-life (kg)

$m_{tR3}(t)$ = End-of-life TaaS EV batteries send to recycling via second-life (kg)

$m_{pMR2}(t)$ = Metal "M" (where "M" = Li, Co, Mn, Ni) content from end-of-life private EV batteries send to recycling via second-life (kg)

$m_{tMR2}(t)$ = Metal "M" (where "M" = Li, Co, Mn, Ni) content from end-of-life TaaS EV batteries send to recycling via second-life (kg)

Tsl = Assumed second-life battery lifetime

Total EV batteries (kg) entering recycling:

$$m_R(t) = m_{pR1}(t) + m_{tR1}(t) + m_{pR2}(t) + m_{tR2}(t) \quad \textbf{(55)}$$

Where,

$m_R(t)$ = Total end-of-life EV batteries sent to recycling (kg)

4.5.1.1 Net Demand for Lithium-ion Battery Materials

The equations below are developed to calculate the net demand for virgin metals for BEVs and grid storage batteries.

Legend:

- Time-dependent Variables are indicated in *italics* using the *V(t)* notation, where *(t)* indicates the running year
- Fundamental assumed fixed Parameters are indicated in **bold** font, e.g., **P**.

Net demand for virgin metal "M" for BEVs:

$$m_{Mnet1}(t) = m_{MF}(t) - [m_{pMR1}(t) + m_{tMR1}(t) + m_{pMR2}(t) + m_{tMR2}(t)] \times \mathbf{E_M} \quad \textbf{(56)}$$

Where,

$m_{Mnet1}(t)$ = Net demand for virgin metal assuming recovered end-of-life EV battery metals are used to fulfil the demand of required virgin metals for light duty electric vehicles

$\mathbf{E_M}$ = Recycling efficiency of metal "M" (where "M" = Li, Co, Mn, Ni) (%)

Net demand for virgin metal "M" for BEVs and grid battery storage

$$m_{Mnet2}(t) = m_{Mnet1}(t) + \left[C_{Total_{PB}}(t) \times \mathbf{M_{PB}}\right] \quad \textbf{(57)}$$

Where,

$m_{Mnet1}(t)$ = Net demand for virgin metal for BEVs and grid battery storage (kg)

$\mathbf{M_{PB}}$ = Mass content of metal "M" (where "M" = Li, Co, Mn, Ni) in LMO battery (kg/kWh)

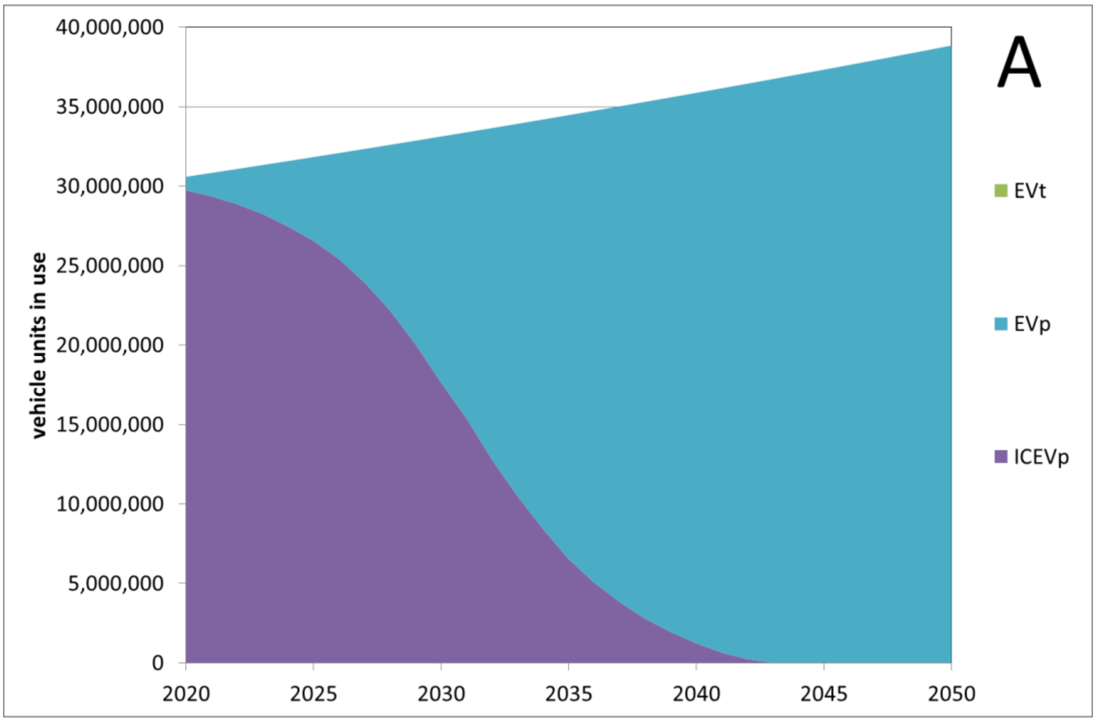
$C_{Total_{PB}}(t)$ = Total capacity of dedicated (purpose-built) batteries for grid battery storage (kWh)

4.5 Results

Figure 4.8 illustrates the modelled development of the total UK LDV fleet, from 2020 to 2050, respectively when assuming: (A) no penetration of TaaS ("worst case" and "baseline" scenarios), and (B) a relatively rapid success of various TaaS schemes, eventually cumulatively accounting for up to 45% of all new LDV registrations in 2050 ("TaaS" scenario). In all scenarios the entire

UK fleet is expected to be essentially rid of conventional ICEVs by the early years of the 2040 decade. While this might appear as a rather extreme outcome over a relatively short time scale, it is in fact entirely consistent with the current government plans to mandate the complete phase out of all new ICEVs by 2030, combined with an average lifetime of 14 years for private LDVs, as discussed in Section 2.1. The reduction in demand for total vehicle units in the "TaaS" scenario (-13% by 2050) is the direct result of the improved efficiency with which TaaS vehicles can deliver the same unit of service (in terms of km travelled per year) when compared to privately-owned vehicles. Figure 4.9 then illustrates a direct overlay of the expected new vehicle registrations in each year, under the "worst case" and "baseline" *vs*. "TaaS" scenarios assumptions.

Figure 4.10 presents the different quantities of EoL EV LIBs that are collected and either sent directly to recycling or repurposed for second-life grid storage. In the "worst case" scenarios, no collection is assumed to take place, and as a result all of the EoL EV batteries are "lost". In the "baseline" scenario, EoL LIB material flows begin in the year 2032, when the first private BEVs reach their EoL. Virtually all collected LIBs go to second life in the first year, due to the demand for grid storage initially almost absorbing all available EoL LIBs; however, the share of EoL BEV LIBs that are sent straight to recycling then quickly increases, due to the much more rapid growth in BEV numbers *vs*. grid storage requirements, and it eventually reaches almost 90% in 2050. In the "TaaS" scenario, the shorter-lived TaaS Evs can already be seen reaching their EoL as early as in 2024, but until 2029, the majority of the collected EoL LIBs go straight to recycling, since the demand for grid storage is still small. Then, after 2032, the arrival of larger numbers of EoL private BEVs leads to a similar trend as in the "baseline" scenario, in terms of a preponderance of direct recycling over second life. Over the three decades under consideration, from 2020 to 2050, the impact of second-life applications on LIB recycling rates can be expected to be minimal.



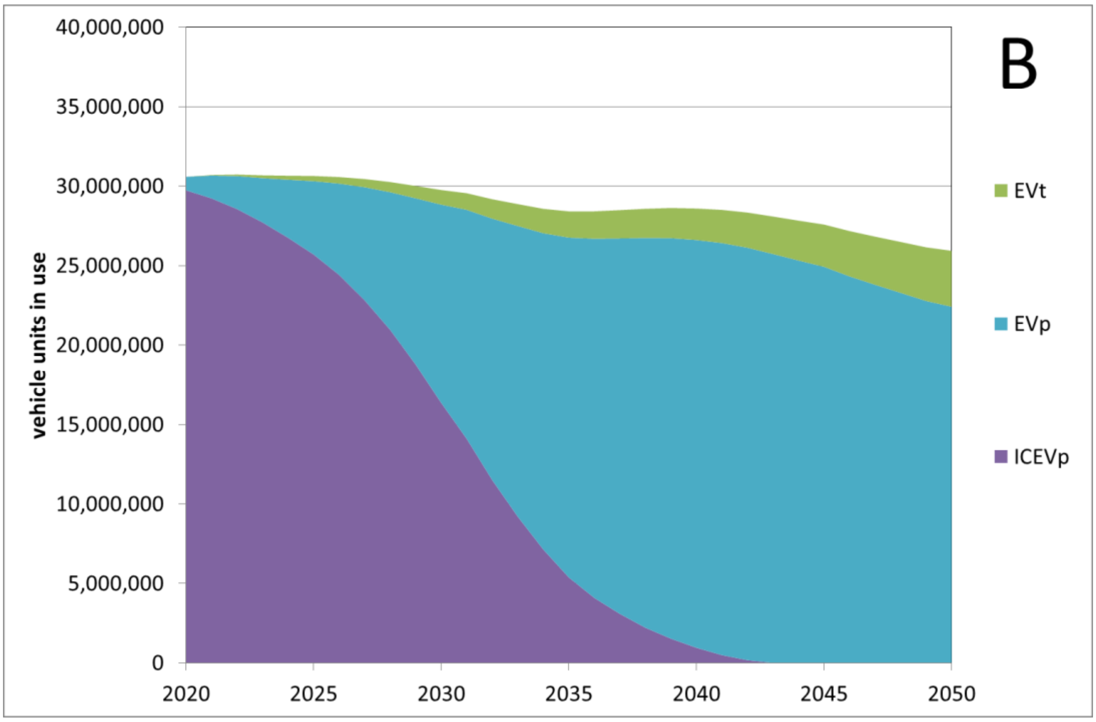


Figure 4.8: Projections for the total light duty vehicle fleet in the UK. A = "worst case" and "baseline" scenarios; B = "TaaS" scenario. Evt = electric vehicles used for transport-as-a-service; Evp = privately owned electric vehicles; ICEVp = privately owned internal combustion engine vehicles.

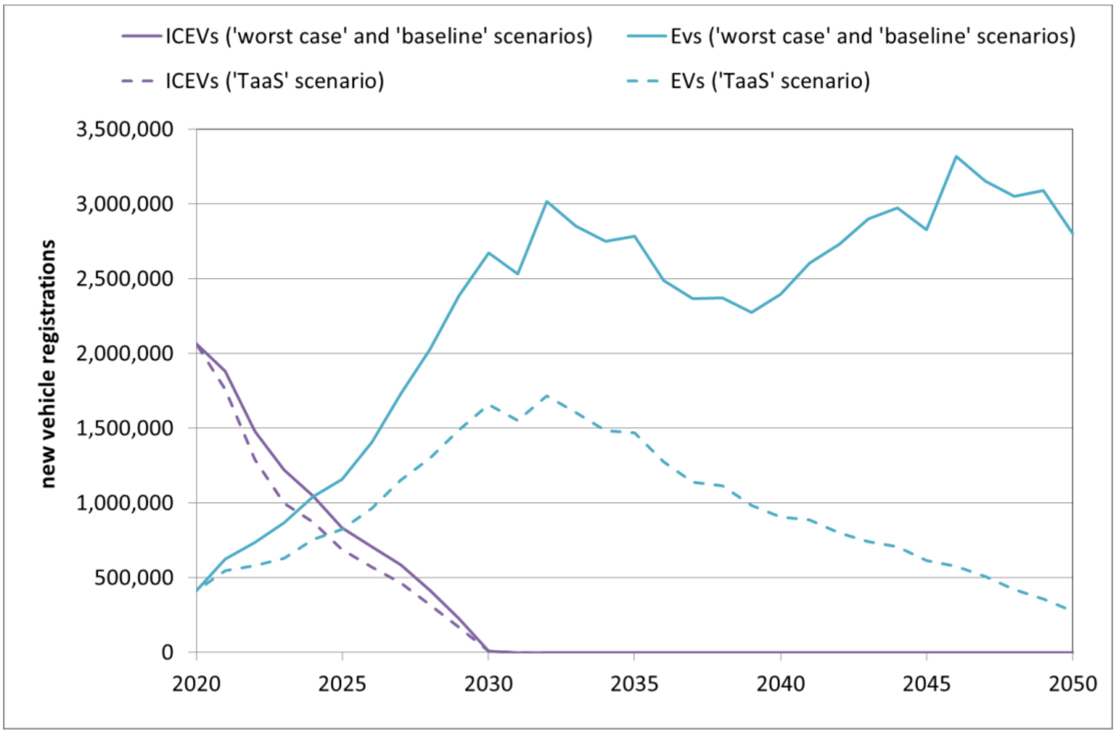
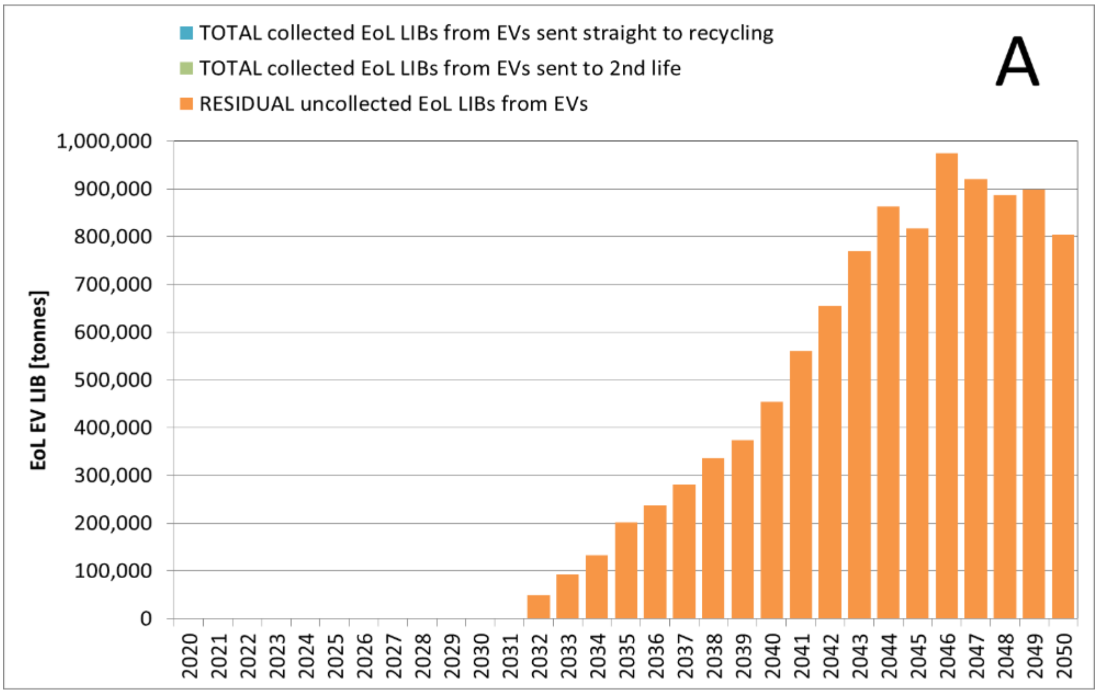
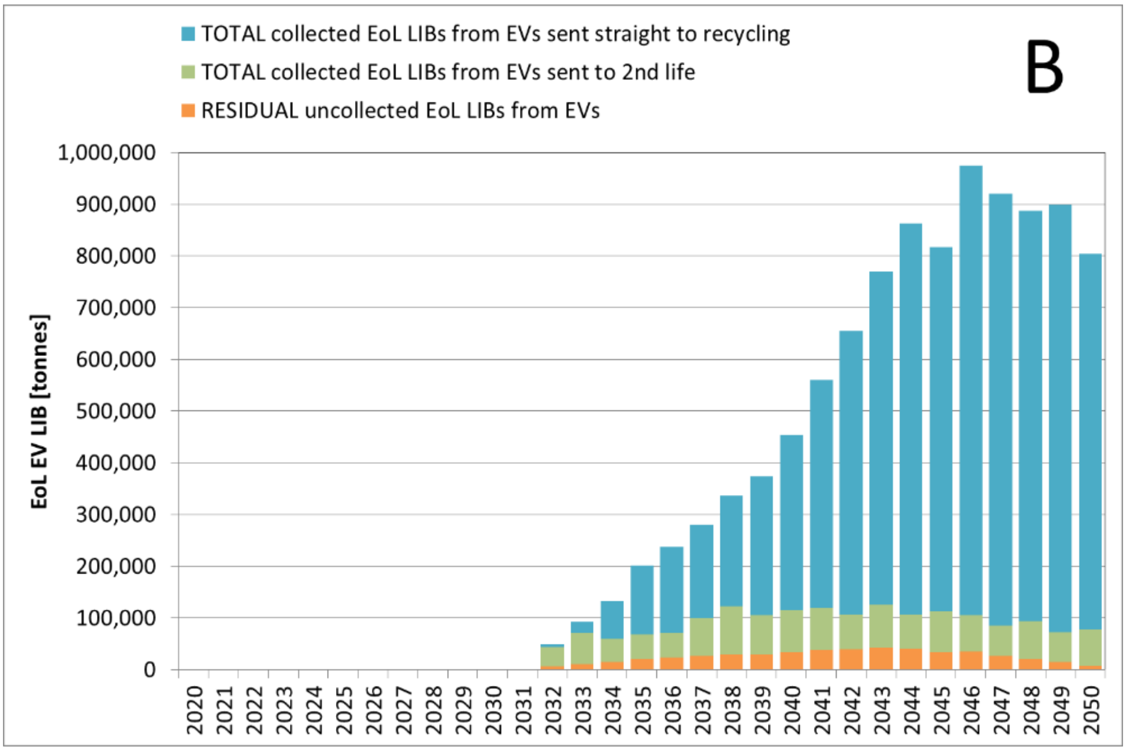


Figure 4.9: Projections of new vehicle registrations per year, "worst case" and "baseline" *vs.* "TaaS" scenarios.





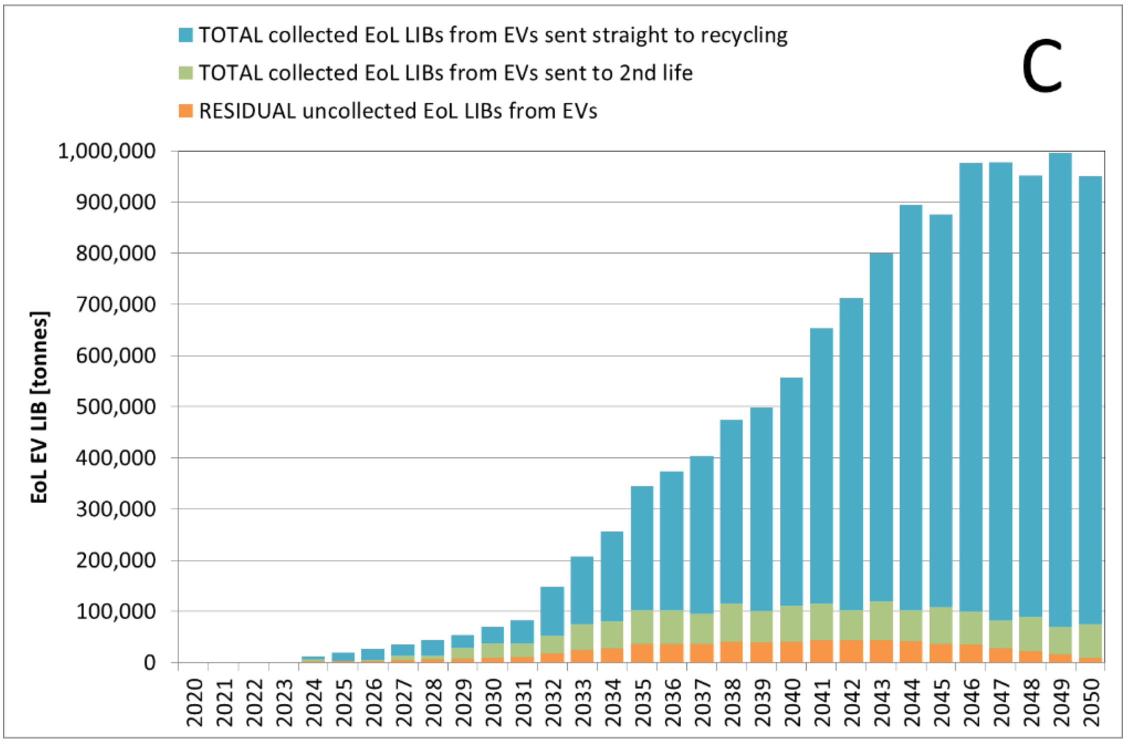


Figure 4.10: Quantities of EoL EV LIBs that are respectively: (i) uncollected; (ii) collected and sent straight to recycling; and (iii) collected and repurposed for second-life grid storage applications. A = "worst case" scenario; B = "baseline" scenario; C = "TaaS" scenario.

Figure 4.11 shows the results of the grid battery storage model, where the blue line indicates the expected incremental demand for installed LIB storage capacity relative to the previous year (this information is provided in National Grid's "Leading the way" FES projections (National Grid, 2020), and applies equally to all three scenarios), and the three green lines indicate the results of three model calculations for the resulting net demand for purpose-built stationary (LMO) LIBs, respectively in the "worst case", "baseline" and "TaaS" scenarios, after accounting for: (i) the need to replace LIB units when they reach their expected end of service life, and (ii) the availability of additional storage capacity provided by second-life EV LIBs. The results clearly differ for the three scenarios under consideration. In the "worst case" scenario the yearly demand for purpose-built LIB storage for the grid rises to over 5 GWh by the early 2030s and then essentially fluctuates between 5 and 7 GWh/year, due to the need to keep replacing the batteries that reach their EoL. In the "baseline" scenario, instead, the availability of second-life LIBs from the LDV sector significantly curbs the demand for purpose-built LIB storage after 2030, bringing it down to zero by 2032. In other words, after 2032 the availability of EoL BEV LIBs is expected to exceed the total demand for LIB storage capacity by the grid, even when considering the limited lifespan of both the originally installed LMO batteries and the second-life NMC batteries coming from EoL BEVs, and the multiple replacements required. The "TaaS" scenario then further improves on these results, because the more rapid turn-over of TaaS vehicles in the LDV fleet effectively brings forward the availability of a sufficient quantity of EoL EV LIBs to be re-used in second-life applications. In this third scenario, therefore, the demand for purpose-built LMO batteries for grid storage is essentially brought down to zero as soon as in 2024.

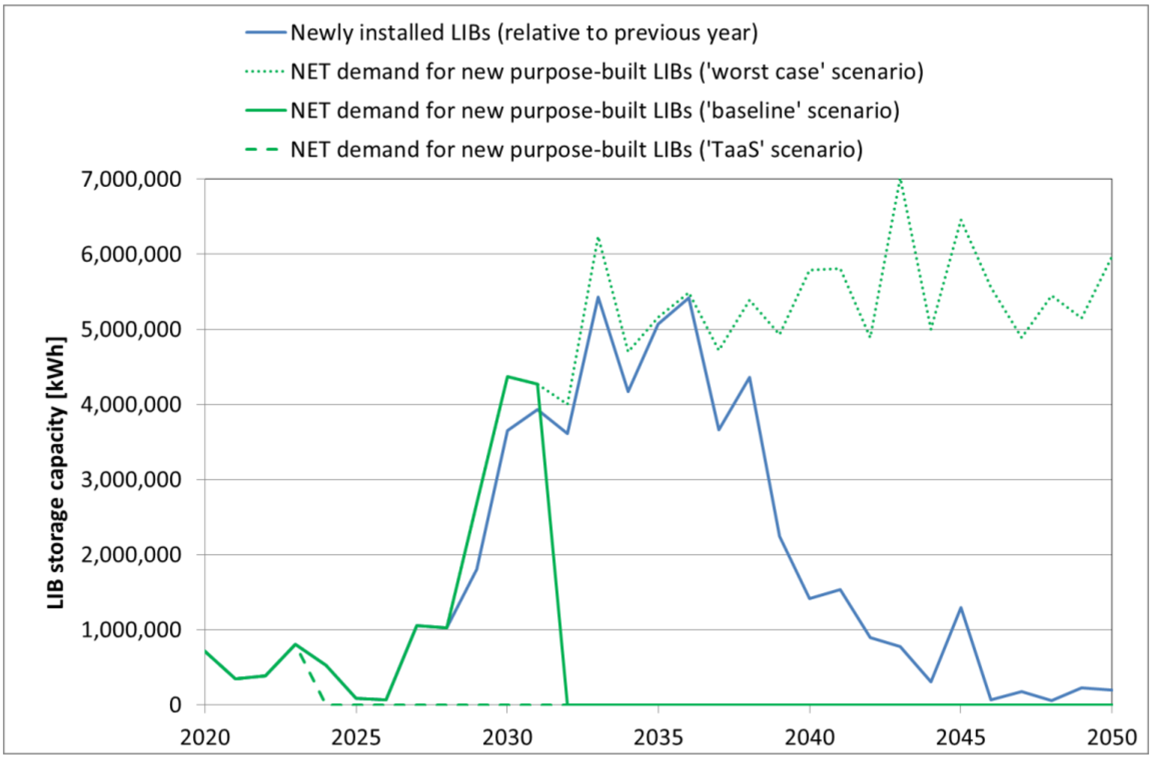
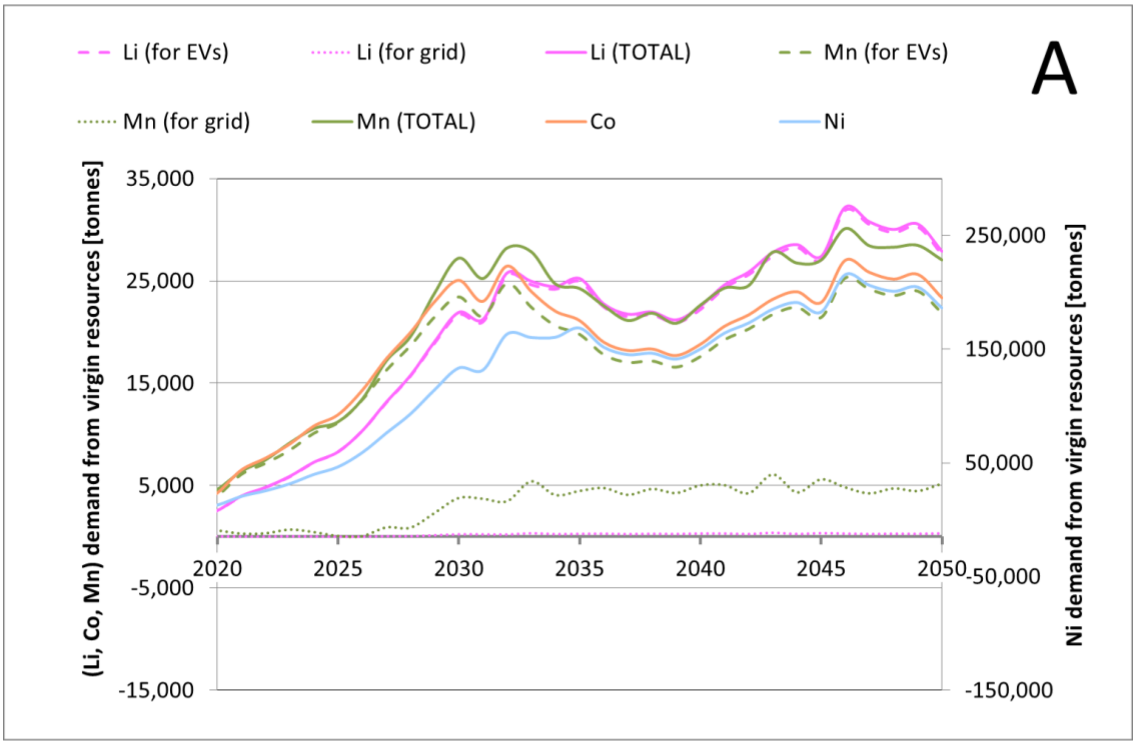


Figure 4.11: Projections of demand for purpose-built LIB storage capacity for the UK electricity grid.

Figure 4.12 presents the results of the complete material flow analysis of the net demand for virgin LIB metals (Li, Mn, Co and Ni) in the UK, when considering the link between the transport and energy sectors resulting from the second-life reuse of EoL EV LIBs, and the effect of LIB recycling. These are presented for three scenarios: A ("worst case"), B ("baseline") and C ("TaaS"). The individual demands for Li and Mn by the LDV fleet and the electricity grid are also reported separately (respectively, using dashed lines and dotted lines). A second vertical scale is used to plot the demand for Ni, since the latter is generally one order of magnitude larger than those for the other metals.

Comparing the results for the "baseline" and "worst case" scenarios highlights the key roles that recycling and to a lesser extent, second life are poised to play in reducing the demand for virgin LIB metals. Specifically, in the "worst case" scenario the demands for all metals first peak in the early 2030s, respectively at around 25,000 tonnes/year (Li, Co and Mn) and approximately 160,000 tonnes/year (Ni), then dip by 20-30% by 2040, before rising even higher towards 2050. The dynamics of these demand curves are primarily dictated by the growth of the BEV fleet. Instead, in the "baseline" scenario a steadily growing collection rate for EoL BEV LIBs, from 80% in 2020 to 99% in 2050, is enough to not only prevent a second peak in demand for all key

LIB metals, but to effectively drive the need to source them from raw resources back down to present levels by 2050, thereby almost "closing the loop" on the LIB sector and potentially staving off the concerns related to any long-term shortage of supply. In the case for "TaaS" scenario, the projected contraction in the total LDV fleet size, combined with the more rapid turn-over of EoL LIBs afforded by the widespread deployment of TaaS vehicles, were found to drive the net demand for virgin LIB metals to negative values after 2040. If the conditions for this scenario were met, there would be a net surplus of Li, Mn, Co and Ni availability coming from the combined throughput of LIBs in the LDV and electricity grid sectors in the UK in the last decade of the considered time frame, from 2040 to 2050.



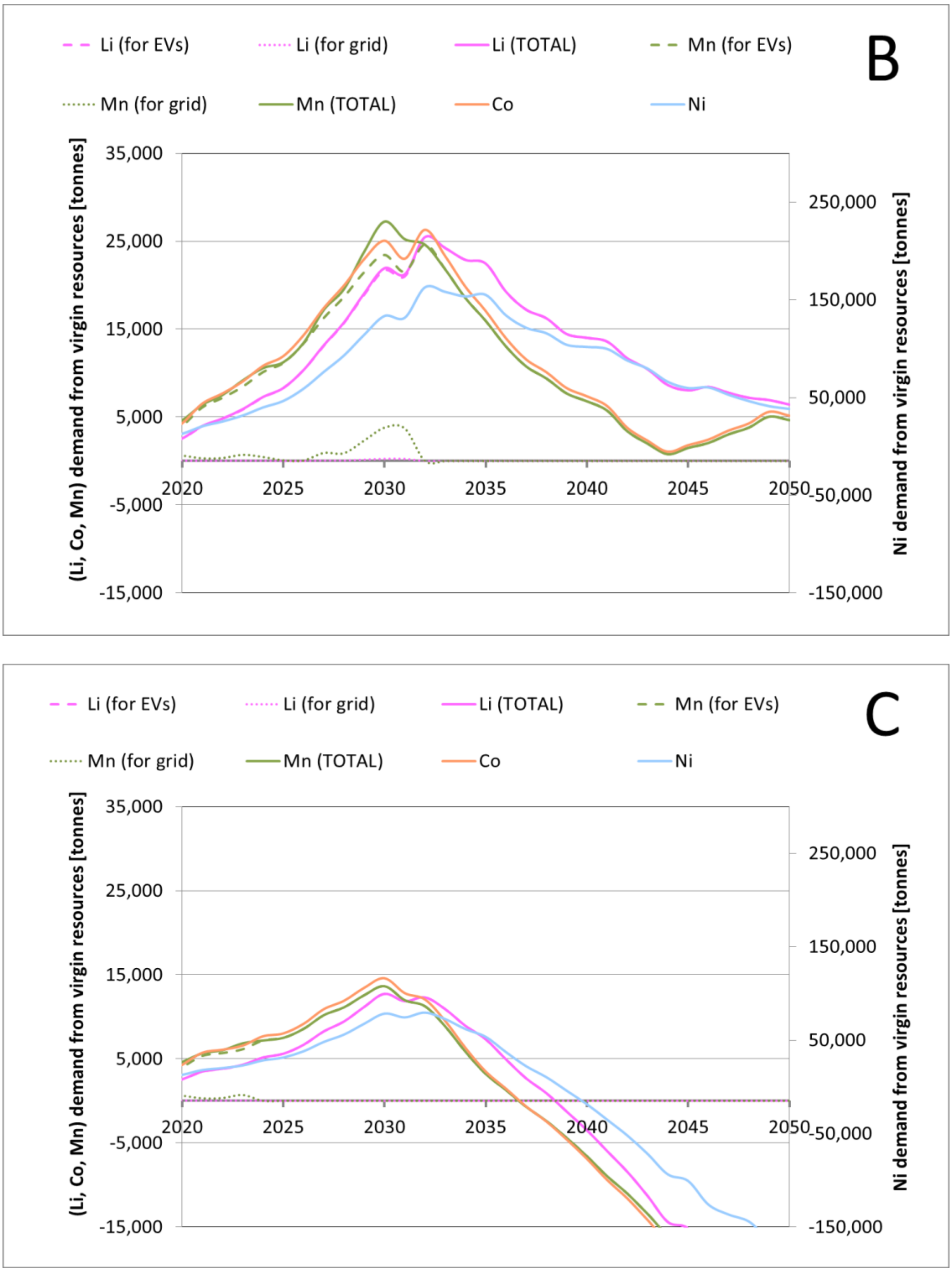


Figure 4.12: Projections of total demand for virgin LIB metals in the UK. A = "worst case" scenario; B = "baseline" scenario; C = "TaaS" scenario. Note separate vertical scale for Ni demand.

5 UK Electricity Grid Mix

This chapter[8] investigates the life-cycle environmental impacts of the UK electricity grid mix evolution over time, specifically to understand the changes that may be expected when moving from the current UK grid mix to a future one featuring larger amounts of variable renewable generation and associated energy storage.

5.2 Current and Future (projected) UK Electricity Grid Mix

LCA of the grid mix is an important environmental factor when considering the transition to BEV. Although the new generation technologies deployed in the UK are often considered to be 'zero carbon' at the point of consumption, the same is not true for their manufacturing, nor are they necessarily 100% environmentally sound in all respects. To get a realistic sense of the environmental impact associated with the grid mix use, this chapter considers all the power plant manufacturing, operation and decommissioning (including transmission and storage) for each year through to 2050, with a 10-year interval.

National Grid Electricity System Operator's "Future Energy Scenarios (FES)" provides projections on how the UK's electricity and gas networks might evolve. The FES scenarios project the need for grid flexibility in response to increased electrification across various sectors, particularly transport, heating, and industry (National Grid 2020). The National Grid (Future Energy Scenario) FES "leading the way" scenario projects electricity generation capacities for all the grid mix technologies and the associated annual electricity generation for the next 30 years. (National Grid 2020). Table 5.1 reports the assumed percentages of grid generation contributed

[8] The chapter is based on the following journal article: Raugei, M., Kamran, M., & Hutchinson, A. (2020). A prospective net energy and environmental life-cycle assessment of the UK electricity grid. Energies, 13(9), 2207. https://doi.org/10.3390/en13092207

by each technology up to 2050, with 5-year resolution, based on the FES 2020 "leading the way" scenario (National Grid 2020).

Figure 5.1: The assumed percentages of grid generation contributed by each technology, based on the FES 2020 "leading the way" scenario.

Generation technology	2020	2025	2030	2035	2040	2045	2050
Biomass	9.6%	8.8%	1.2%	0.5%	0.4%	0.3%	0.3%
Waste	3.4%	4.3%	3.4%	2.4%	1.6%	1.0%	0.5%
Coal	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
NGCC	31.4%	6.3%	4.9%	1.2%	0.1%	0.0%	0.0%
Biomass + CCS	0.0%	0.0%	5.3%	10.5%	10.4%	11.4%	10.5%
Nuclear (PWR – current)	21.9%	16.1%	9.2%	7.7%	7.0%	5.8%	5.3%
Nuclear (SMR – future)	0.0%	0.0%	0.0%	0.9%	0.7%	0.4%	0.3%
Hydro	2.1%	2.2%	1.8%	1.5%	1.4%	1.2%	1.1%
Marine (tidal)	2.3%	3.7%	3.5%	2.2%	1.5%	1.1%	1.0%
Wind (onshore)	11.7%	18.8%	17.2%	16.9%	17.9%	16.5%	16.4%
Wind (offshore)	13.2%	33.5%	45.8%	47.0%	49.0%	52.4%	54.3%
Photovoltaic	4.4%	6.3%	7.6%	9.2%	10.1%	9.9%	10.4%

National Grid's "Leading the way" scenario considers the deployment of five energy storage technologies: pumped hydro storage (PHS) – in large part relying on existing dammed hydro installations located overseas and tapped via interconnectors, compressed air energy storage (CAES), dedicated lithium-ion batteries (LIB), vehicle-to-grid (V2G) storage provided by the existing battery electric vehicle (BEV) fleet, and liquid air energy storage (LAES). The first two technologies are generally capable of providing longer-duration storage than the latter three. Specifically, LAES is a fairly new technology that is projected to ultimately provide 4% of the total storage capacity; due to the very limited information available on its supply chain, associated storage capacity was instead lumped together with that provided by CAES. The resulting specific assumptions on storage technologies for the future of the UK grid mix are summarized in Table 5.2.

Table 5.2: Grid-level energy storage output adopted from National Grid FES 2020.

Technology	Storage Generation [TWh] 2020	Storage Capacity [GWh] 2020	Storage Generation [TWh] 2050	Storage Capacity [GWh] 2050	Round-trip storage efficiency
PHS	0.9	38.4	4.6	98.6	80% (U.S. Grid Energy Storage Factsheet 2018)
CAES	0	0	3.0	32.5	67% (Kaldmeyer et al. 2016)
LAES	0	0	2.2	15.6	-
LIB	0.7	1.6	18.5	56.2	80% (U.S. Grid Energy Storage Factsheet 2018)
V2G	0	0	0.3	-	80% (U.S. Grid Energy Storage Factsheet 2018)

The overall electricity generation output is estimated to increase by 49% in 2035 and 116.53% in 2050, relative to 2020, in response to the increasing demand partly due to the electrification of transport and heating. Figure 5.1 illustrates the UK grid mix composition in terms of total electricity generated in the year 2020 up to year 2050. More specifically, the yellow line in Figure 5.1 shows that approximately 82% of total generation in 2050 is expected to be provided by VRE (variable renewable energy) technologies (i.e., wind, solar and tidal), which, as the phrase implies, are inherently intermittent. As the cumulative share of these technologies increases over the next decades, the grid will thus require increased levels of energy storage to provide the required flexibility to match the energy demand profile. Storage technologies will therefore increasingly be deployed alongside VRE, to provide such flexibility, as shown by the red line in Figure 5.1.

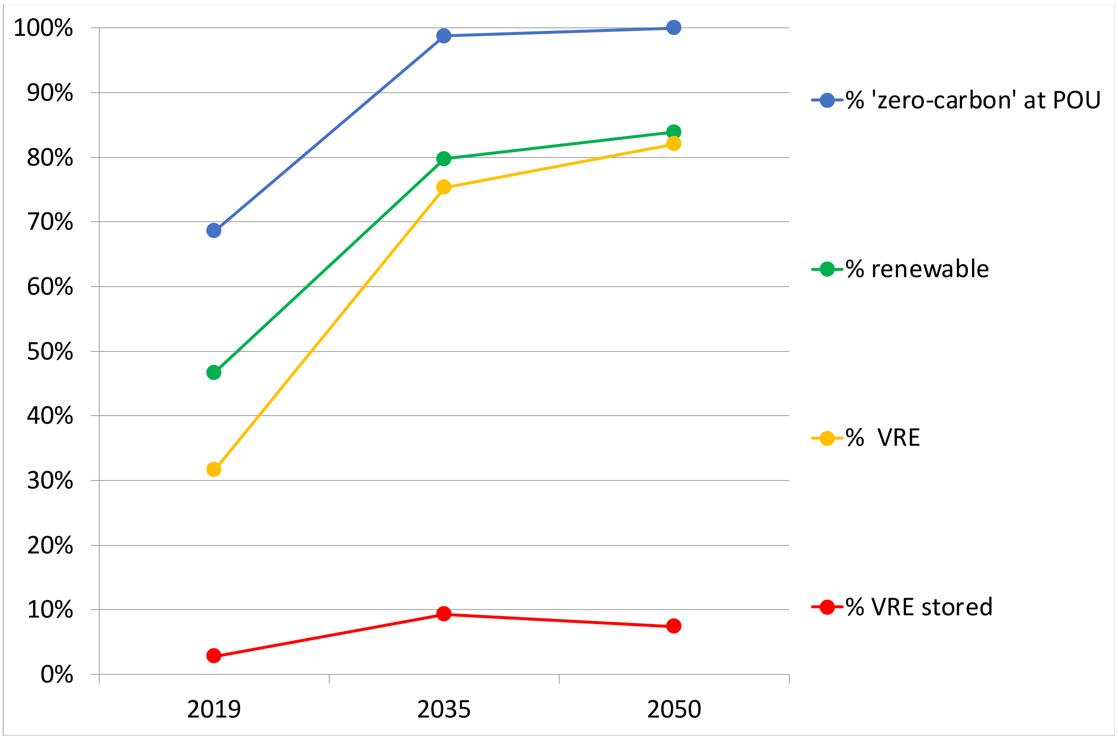


Figure 5.1: Expected trends (under "Leading the way" scenario assumptions) for, respectively: % of total generation that is considered 'zero-carbon' at point of use (i.e., nuclear, biomass, waste, hydro, tidal, wind and solar); % of total generation that is considered 'renewable' (same as above but excluding nuclear); % of total generation that is classified as 'Variable Renewable Energy' (VRE; i.e., wind, solar and tidal); % of VRE generation that is sent to energy storage (as opposed to used directly).

5.3 Electricity Generation and Storage Technologies

The LCA focuses on the electricity generated and delivered domestically within the UK, disregarding all electricity exchanged via the interconnectors (with the sole exception of a relatively small share of the future variable renewable electricity that is assumed to be sent to be stored in pumped hydro facilities located on the continent). Life cycle inventory information is provided below:

Coal: Currently, there are six coal-fired power stations in UK; all are expected to shut down by 2025, for which there are plans to convert its units to biomass- and gas-fired generation in the near future (Coal Countdown n.d.). The Ecoinvent LCI database "GB" (Great Britain) hard coal electricity production model was adopted for the life cycle inventory of coal-fired electricity generation in the UK.

Natural gas combined cycles: UK natural gas combined cycle (NGCC) inventory was based on the corresponding Ecoinvent GB database model. The natural gas output volume was reduced by 19% to account for the difference in Ecoinvent natural gas input value (0.16 m^3) vs. reported natural gas energy input (0.19 m^3) per kWh of electricity output at the power plant stage, as well as the actual average heating value of the UK gas feedstock based on digest of UK energy statistics (DUKES) 2019 report.

Biomass: The biomass used for electricity generation in the UK mainly consists of wood pellets from North America forestry, domestic wood chips from UK forests, and a small percentage of domestic residues (DECC 2020; National Statistic 2019). According to DUKES 2019 report, based on the 2018 renewable flow-chart, 45% of the biomass share was domestically sourced and 55% was imported (DUKE 2019). The GB heat and power co-generation model from the Ecoinvent database was selected to represent biomass electricity generation. However, this model does not include mixed inputs of woody biomass feedstocks, and it was therefore modified to account for such. Specifically, the Ecoinvent "RER" (regional European) wood chip model was used to represent the share of wood chips used (since there is not one available specifically for the UK), and two additional processes were added to the model to account for the wood pellet imports. The first process accounts for wood pellet production, and the second one for the transportation of the wood pellets from North America by freight ship. The transport distance was assumed to be 5,300 km (Online Freight Marketplace 2020) and the mass of the dry pellets was multiplied by a factor of 1.2 to account for the average moisture content in the transported pellets (Woody Biomass n.d.).

Biomass combined cycles plus carbon capture and storage (future technology): Biomass energy with CCS (BECSS) is potentially one of the few options for 'negative emissions'. The combination of biomass and CCS in energy conversion technologies has many technological similarities with CCS applied to fossil fuel conversion; however, there are also several differences such as biomass fuel typically has other combustion/gasification properties, lower energy density and greater variation between biomass types.

At present there is no detailed inventory information available for biomass combined cycles with carbon capture and storage (Biomass-CCS). Literature data on the carbon capture process was adopted for 90% CO_2 capture and incorporated in the adjusted model for conventional Biomass plants. The electricity output of the plant was reduced by 15.9% to account for the CO_2 capture unit and compression process (Singh et al, 2011; Fadeyi et al, 2013). The life cycle inventory for the transportation of CO_2 captured and stored is excluded from the study due to the uncertainty involved in the location of the plant and possible storage size. Tables 5.3 and 5.4 provides the construction and operational inventory quantities and the adjusted emissions per kWh of electricity generated.

Table 5.3: Foreground inventory of life-cycle inputs for carbon capture and storage (CCS) technology. All values are per kWh of electricity generated.

Item	Quantity	Unit
Activated Carbon	$3.2 \cdot 10^{-5}$	kg
Concrete	$2.1 \cdot 10^{-7}$	kg
Electricity[9] (for CO_2 compression)	$4.7 \cdot 10^{-2}$	kWh
Monoethanolamine (MEA)	$1.8 \cdot 10^{-4}$	kg
Polyethylene, high density (HDPE)	$7.1 \cdot 10^{-7}$	kg
Sodium hydroxide (NaOH)	$5.5 \cdot 10^{-5}$	kg
Steel (low alloyed)	$7.7 \cdot 10^{-5}$	kg

Table 5.4: Foreground inventory of use-phase emissions per kWh of electricity generated by the NGCC + CCS adopted for Biomass + CCS system.

Item	Quantity	Unit
Carbon dioxide (CO_2)	47	g
Nitrogen oxides (NO_x)	$1.7 .10^{-1}$	g
Sulphur dioxide (SO_2)	$3.8 .10^{-3}$	g

[9] Accounted for by deduction from plant output.

Particulate matter (PM)	2.2 .10^{-3}	g
Formaldehyde (HCHO)	1.1 .10^{-1}	g
Acetaldehyde (CH_3-CHO)	7.0 .10^{-2}	g
Ammonia (NH_3)	1.5 .10^{-2}	g
Monoethanolamine (MEA)	2.6 .10^{-2}	g

Waste: Electricity generation from waste incineration is also a co-product of a multi-output process. According to ISO recommendations (Environmental Management, 2006), in this case system expansion was adopted in preference to allocation, since waste incineration with energy recovery represents an almost textbook example in which: (i) a primary function is clearly identified (i.e., getting rid of the waste), and (ii) a comparable alternative process exists which delivers only one of the two outputs (i.e., an incinerator without energy recovery). Consequently, the energy recovery process and associated electricity generation was calculated to have negligible emissions assigned to it, since the only additional up-front inputs required vs. the incinerator without energy recovery are those for the boiler and turbine system, while the use-phase emissions at the stack are virtually the same.

Nuclear: There are 15 operating nuclear reactors in UK, 14 of which are advanced gas-cooled reactors (AGR) which are expected to shut down before 2035, and one is a large, pressurized water reactor (PWR) which was initially also expected to shut down in 2035, but whose operation may be extended for 20 more years (World Nuclear Association 2020). Out of the currently operating nuclear reactor technologies in the UK, life cycle information in Ecoinvent was only available for PWRs, and therefore the latter was used to model the life cycle inventory associated to nuclear electricity generation.

Nuclear (small modular reactors – future technology): Small Modular Reactors (SMR) are factory-built nuclear reactors of less than 300MWe installed power, inspired by the current large nuclear power plants (Pannier & Skoda, 2014). They are also known as integral PWR since their main components, such as the stream generator, reactor and pressurizer, are all located in one vessel. SMRs offer the opportunity to add nuclear generating capacity with a smaller capital cost and thus reduce construction risks. They can be categorized in two groups: (1) Generation III water-cooled SMR based on existing large nuclear plants but on a smaller scale, and (2) Generation IV SMRs based on the use of novel fuels and coolants, which can provide other services such as heat for industrial processes (Advanced Nuclear Technologies 2019). Generation IV small modular reactors are not expected to achieve commercial maturity until 2030 onwards (Carless et al. 2016), while Generation III SMR are considered to be more mature technologies as they are based on the existing large nuclear plants concept. According to the world nuclear association there are currently two potential SMR projects, one with NuScale and other with Rolls Royce both based on light-water pressurised SMR designs (World Nuclear Association, 2020).

At the time of writing there is no existing LCI database model for SMR technologies, therefore, in this study the Ecoinvent model for PWRs was adopted as a basis for the life cycle inventory of light-water pressurised SMR. The latter are the scaled-down version of large PWR which utilize the same working concepts, but instead of having pumps and coolant loops for directing the flow of water, they utilize natural circulation to direct the cool water to the reactor core after going through the steam generator to turn the turbine to generate electricity (Godsey, 2019). The main components of both systems are considered to be the same, and both are expected to have the same lifetime of 60 years. The Ecoinvent model was adjusted to account for the reduction in efficiency and improvement of the capacity factor (CF) for SMRs compared to large PWRs, which lead to an overall reduced output (-15% in relative terms) (Carless et al, 2016).

Hydroelectric: Hydropower electricity generation in UK consist of 24% run-of-river and 76% reservoir (DUKE, 2019). The Ecoinvent model for GB run-of-river hydroelectricity model was adopted, and the Ecoinvent “DE” (Germany) model for hydro- reservoirs was used as a proxy, as the database does not contain a corresponding GB model.

Marine tidal (future technology): Tidal energy is generated through the rise and fall of tides, due to the interaction of gravitational pull of moon and to a lesser extend the sun on the ocean and the rotation of the Earth (Hammons, 1993). There are three types of tidal technologies: lagoon, barrage and stream turbines. National Grid’s FES 2019 scenarios assume that the target tidal capacities will be met primarily using tidal lagoons, and secondarily stream turbines. However, there is mounting uncertainty on the future development of tidal technologies, with on the one hand, the recent cancellation of one large tidal lagoon project (BBC News 2019), and on the other hand, new upcoming developments and installed projections on stream turbine (Noonan 2019).

In this study, the assumption was therefore made that the electricity generated by tidal will be harnessed by tidal stream turbines.

There is no model in the Ecoinvent database for this technology. Therefore, all life-cycle inventory and technical information for use in this study was sourced from the published scientific literature on an OpenHydro tidal stream turbine (Walker et al. 2013). The inventory information includes energy inputs for the installation, manufacturing and maintenance of the system and the material inputs for the construction of the device, power cabling and foundation. The system as described was expected to have a lifetime of 20 years and was rated at 2MW. The average capacity factor (CF) for the stream turbine tidal plant was taken as 5.5% from the DUKES 2019 report, and all energy and material inputs were duly scaled to 1 kWh of electricity generated over the lifetime of the system. The resulting foreground inventory information is provided in Table 5.5.

Table 5.5: Foreground inventory of life-cycle inputs for stream turbine tidal electricity generation. All values are per kWh of electricity generated.

Item	Quantity	Unit
Cast Iron	$1.5 \cdot 10^{-6}$	kg
Cement	$2.5 \cdot 10^{-5}$	kg
Copper	$3.2 \cdot 10^{-6}$	kg
Electricity (for plant construction)	$1.9 \cdot 10^{-2}$	kWh
Glass fibre reinforced plastics (GRP)	$9.4 \cdot 10^{-6}$	kg
Polyethylene (PE)	$4.7 \cdot 10^{-7}$	kg
Steel (low alloyed)	$1.6 \cdot 10^{-4}$	kg

On-shore wind: The Ecoinvent electricity production model for GB 1-3 MW onshore wind turbines was used to represent the total onshore wind electricity generation in UK. The model assumes a 20-year lifetime for all moving components and a 40-year lifetime for all the stationary components of the wind installation.

Off-shore wind: The Ecoinvent electricity production model for GB 1-3 MW offshore wind turbine was used to represent the total offshore wind electricity generation in UK. The model assumes the same lifetimes as for onshore wind turbines.

Photovoltaic: National Grid's "leading the way" scenario considers distribution-connected and micro-connected solar capacity and accordingly in this study the assumption was made that most of the solar photovoltaic (PV) generation will come from roof-top mounted systems; additionally, the Fraunhofer Institute for Solar Energy report (2019) confirms that multi-crystalline silicon (mc-Si) continues to be the leading PV technology by far in terms of global annual production. In order to limit the complexity of the model, a single Ecoinvent process (GB roof-top mounted mc-Si PV) was therefore adopted as the basis for the assessment of solar PV electricity generation in UK. However, since PV systems are still on a continuously and rapidly improving trend, the model was adjusted to reflect the current and expected future mc-Si module efficiencies, respectively reported at 17% in 2020 (Fraunhofer Institute for Solar Energy Systems, 2020), and estimated at 25% in 2050 (Frischknecht et al., 2016). This information was used to adjust the area of solar panels required to produce 1kWp of installed power in the model.

An average insolation of 1,000 kWh/(m^2·yr) was then assumed (Global Solar Atlas, 2020), which combined with a performance ratio (PR) of 80% (Frischknecht et al. 2016), led to a calculated capacity factor (CF) of 9.1%. Finally, the expected lifetime of the PV modules was kept at 30 years before 2035 (Frischknecht et al. 2016), and then increased to 35 years through to 2050 (IEA 2015).

Pumped Hydro Storage (PHS): Pumped hydro storage (PHS) uses electricity to pump water into the high-elevation reservoirs during high generation and low demand, and then releasing the water to generate electricity at peak demand. Since PHS for the UK is projected to utilize pre-existing hydro reservoir systems (mainly located overseas and accessed via the interconnectors), which were built for the primary function of generating hydroelectricity, and since the electricity used for pumping the water uphill would otherwise have to be curtailed, the life-cycle impacts associated with PHS were taken to be zero, thus avoiding any double-counting.

Compressed Air Energy Storage (CAES): Compressed Air Energy Storage (CAES) systems store excess electricity by compressing air to high pressure in underground reservoirs such as pre-existing salt mines. The stored air is then heated and expanded to drive a turbine to generate electricity when the electricity demand is high and the generation is low (Kaldmeyer et al. 2016). Currently there are two CAES systems operating worldwide, one is in Huntorf, Germany since 1969 and the other is in McIntosh, United States since 1991 (Budt et al. 2016). Both work by burning natural gas to provide heat for the expansion of air to drive the turbine generator.

However, the FES 2020 "Leading the way" scenario expects CAES systems to take off from 2030 onwards, and therefore in this study the assumption was made that by that time the UK's CAES installed capacity will be of the more advanced adiabatic type (A-CAES). This type of CAES works by retaining and storing the heat generated during the compression of the air using a thermal energy storage (TES) system, and then reusing the stored heat for the expansion process instead of burning natural gas. There has been a lot of on-going research in A-CAES over the last decade, including the planned EU-based research Project "ADELE" (ADELE 2010), the Storelectric project planning to build large-scale A-CAES in Holland (Storelectric 2018), and a recently completed demonstration project by Hydrostor in Toronto Island, Canada (hydrostor, n.d.).

Life cycle inventory (LCI) information on A-CAES is not available in the Ecoinvent database. The technical data were therefore adopted from the available literature; specifically, the maximum number of storage cycles was taken to be 10,000 (Venkataramani et al. 2016), and the cycle efficiency of the plant was taken to be 67% (Kaldmeyer 2016). It was also assumed that the compressed air will be stored in pre-existing underground caverns. The information on material and energy inputs for plant construction, compression unit, heat expander and thermal energy storage system was adopted from the published literature (Bouman et al. 2016) and rescaled linearly in terms of storage capacity. The inventory information for the input quantities per kWh of electricity storage capacity is provided in Table 5.6.

Table 5.6: Foreground inventory of life-cycle inputs for adiabatic compressed air energy storage (A-CAES) technology, including plant construction, compressors, thermal energy storage (TES) and heat expanders. All values are per MWh of electricity storage capacity.

Item	Quantity	Unit
Aluminum	$4.4 \cdot 10^{-1}$	kg
Cast Iron	48	kg
Concrete	$5.2 \cdot 10^{2}$	kg
Copper	4.0	kg
Diesel (burnt in building machines)	$9.1 \cdot 10^{2}$	MJ
Electricity (for plant construction)	18	kWh
Foam Glass	3.2	kg
Heavy fuel oil (burnt in industrial machines)	$9.1 \cdot 10^{2}$	MJ
Insulation (rock wool)	19	kg
Limestone	4.6	kg
Lubricating oil	$2.5 \cdot 10^{3}$	kg
Polypropylene (PP)	$6.3 \cdot 10^{-1}$	kg
Sand-lime brick	24	kg
Steel (high alloyed)	91	kg
Steel (low alloyed)	$1.3 \cdot 10^{2}$	kg

Steel (unalloyed)	$1.1 \cdot 10^2$	kg

Lithium-ion Battery Storage (LIB): Dedicated grid-level lithium-ion battery (LIB) storage was modelled on the basis of the Ecoinvent model for lithium manganese oxide (LMO) technology. LMO is among the most mature options for LIBs, and although it lags behind some of the other cathode formulations in terms of energy density (Placke et al. 2017; Zubi et al. 2018), this was deemed relatively unimportant for dedicated stationary applications, and counterbalanced by its comparatively long cycle life, its overall stability, and its reliance on abundant and eco-friendly materials (Zubi et al. 2018). The lifetime of purpose-built LIB was selected as 10 years and is dictated by the requirement for residual grid battery which was not met by second life EV battery (carried out in Chapter 4). Table 5.7 provides a breakdown of purpose-built grid battery required for grid energy storage.

Table 5.7 provides a breakdown of grid energy storage requirement met by second life BEV LIB and of purpose-built grid battery.

Year	Required LiBs storage capacity [GWh]	Installed purpose-built LIBs [GWh]	Installed 2nd-life LIBs [GWh]
2020	1.6	1.6	0.0
2025	3.8	3.8	0.0
2030	11.4	11.4	0.0
2035	33.6	13.5	20.1
2040	50.7	4.3	46.4
2045	55.5	0.0	55.5
2050	56.2	0.0	56.2

Vehicle-to-grid Storage (V2G): For each year of analysis, vehicle-to-grid (V2G) storage schemes rely on the Li-ion batteries already installed in the existing electric vehicle (EV) fleet, when connected to the network of charging points, to provide short-duration storage for grid support. In order to avoid incurring in double-counting, energy storage made available through V2G is therefore regarded to have zero impacts assigned to it, since the primary function of vehicle batteries is to provide electricity storage for transportation.

Hydrogen Electrolysis: National Grid's "Leading the way" scenario considers dedicated wind turbine coupled with electrolysis to meet hydrogen production demand of which 8% of hydrogen is used in electricity generation in 2050. Rest of the hydrogen supply is used for transport and gas heating purposes (National Grid 2020). Given a small share of hydrogen supply for electricity use, the choice was made to leave hydrogen out of the analysis.

Electricity transmission and interconnectors: The current HV transmission lines were modelled using the UK-specific life-cycle inventory (LCI) information provided in the Ecoinvent database, and then scaled for 2035 and 2050 by simple linear extrapolation based on the projected total gross electricity generation. In terms of energy balance, transmission losses were conservatively set at 5% (ESO 2019).

5.4 Life Cycle Impact Assessment (LCIA)

The functional unit (FU) of this study was set as 1 kWh of electricity delivered by the UK grid as a whole, including energy storage and transmission. The main data source used for the life cycle inventory (LCI) analysis was Ecoinvent version 3.5 database (Ecoinvent n.d.; Moreno Ruiz et al. 2018) complemented where appropriate and acquired by a range of other literature sources as described in detail in Section 5.2. The whole analysis was carried out using the latest release of the dedicated LCA software package GaBi (ThinkStep n.d.). The life cycle impact assessment results are based on 5 midpoint impact categories using CML method for characterization factors published by the IPCC.

Global Warming Potential (GWP) represents the contribution to climate change caused by the emission of different greenhouse gases (GHGs) (such as CO_2, CH_4, CO) expressed in terms of an equivalent emission to CO_2. This is the extent a given emission of a unit mass can absorb infrared radiation (the amount of heat trapped) compared to a mass unit of CO_2 (Heijungs et al. 1992).

GWP is calculated over a 100-year time horizon to assess cumulative effect for GHGs over long term.

There are two alternative accounting rules for GWP, one including biogenic emissions, which are those that arise from the combustion of biomass (wood chips and pellets, and biogas derived from the anaerobic degradation of organic matter). Second excluding biogenic emissions, this assumes that the same amount of carbon emission absorbed from the atmosphere during the biomass growth phase (such as the trees used for the wood chips and pellets) is released back either naturally decomposed or burned, therefore resulting in net zero carbon emissions (Wiloso et al. 2016). This may not always be the case, such as the wood chips coming from domestic forestry residues may often be closer to being net zero than wood pellets imported from overseas, furthermore, the biomass that was harvested would require a well-managed short-rotation forestry that leads to net zero standing biomass change over time. As a result, the real-world net carbon emissions of biogenic feedstocks will potentially be higher than zero, with considerable uncertainty. To reflect such uncertainty, the choice was made to calculate and report GWP under both assumptions, i.e., respectively including and excluding biogenic carbon emissions.

Acidification Potential (AP) represents the acidifying chemicals such as sulphur oxides (SO_x), nitrogen oxides (NO_x) and ammonia (NH_x) which contributed to acidification of soils, water, biological organism, environment, and buildings (such as decrease in soil nutrients and increased corrosion of materials) (Guinée 2002). AP is measured in terms of hydrogen ions (H^+ ions) produced (causing acidifying effect) by a given emission in relative to SO_2 and is expressed is expressed in terms of kg SO_2 equivalent (Guinée 2002).

Human Toxicity Potential (HTP) represents the contribution to impact on human health caused by the toxic substances present in the environment. This is calculated based Uniform System for the Evaluation of Substances (USES-LCA) which represents human toxicological classification factors of 180 substances (Guinée 2002). HTP is measured relative to kg 1,4-dichlorobenzene (1,4-DB) equivalents for an infinite time horizon. HTP results have a greater level uncertainty than those for all other impact categories, due to the methodological difficulty of comparing and combining into a single indicator the individual toxicity potential of wide range of organic and inorganic emissions. The uncertainty is especially large in the case of metal emissions (Apeldoorn 2004).

Abiotic Depletion Potential (ADP) – Elements represents the depletion of natural resources found on Earth. This is measured by comparing each raw materials used with its ultimate reserve (resource) and rate of extraction in reference to the ultimate reserves and current extraction rates of element antimony (Sb). This indicator is calculated excluding the contributions of all energy inputs (such as fossil fuels and uranium). It should be noted that abiotic depletion is an impact category that is still frequently the object of methodological discussion, and alternative approaches exist to the quantification of the associated impact, often reflecting differences in problem definition (Guinée 2002; Schulze et al. 2020).

Non-renewable Cumulative Energy Demand (nr-CED) represents all nonrenewable primary energy used directly and indirectly from the environment such as natural gas, crude oil and uranium. This is measured by applying characterization factors to all non-renewable primary energy and energy investment inventoried expressed as Joules of crude oil equivalent the UK grid mix (including energy storage and transmission) (Frischknecht 2015), the result for this is represented in Appendix E.

5.5 Results

Figure 5.2 represents the Global Warming Potential (including biogenic carbon) per unit of electricity delivered by the UK grid mix. Considering the total grid mix results from 2020 to 2050, GHG emissions may be expected to drop by 167% (i.e., from 0.22 to -0.07 kg CO2-eq/kWh). This is mainly due to the phase out of natural gas making up 31.4% of electricity generation in 2020 towards higher percentage of low carbon sources and biogenic carbon capture and storage. Latter being responsible for achieving substantial negative carbon emissions. However, in the case where CCS is not present (from 2020 to 2025), biomass generation presents the highest impact after coal per unit of electricity delivered (impact of each energy source is present in Appendix E). The share impact from biomass generation represents 30 – 57% of the impact whilst only representing 9.6 – 8.8% of total electricity delivered. This is mainly due to the inefficient process of burning biomass. Furthermore, the carbon released by decomposing biomass such as forestry residues and mill residues are expected to decay over a long period of time. Whereas in the case of biomass energy generation, CO_2 emissions of biomass are released in one time (Liu et al. 2017), leading to a higher rate of carbon sequestration. In the case of BECSS, the 84% of biogenic carbon from biomass generation which would otherwise be released back into the atmosphere is captured and stored, creating a carbon debt (taking account of electricity for injecting carbon in storage and to run CCS unit). Post 2030, GWP are expected to be mainly driven by offshore wind and PV generation, followed by biomass, onshore wind, and transmission upgrades for meeting the increase electricity supply. Both storage system, ACAES and LIB are expected to have an insignificant share of GWP <0.1% compared to other grid mix technologies without accounting for Biomass with CCS. For LIB is mainly due to most storage requirements being met by second EV batteries, and almost complete from 2040 onwards. In the case where LIB requirement was not met by second life storage, LIB impact is still expected to be insignificant compared to other grid mix technologies (Raugei et al. 2020).

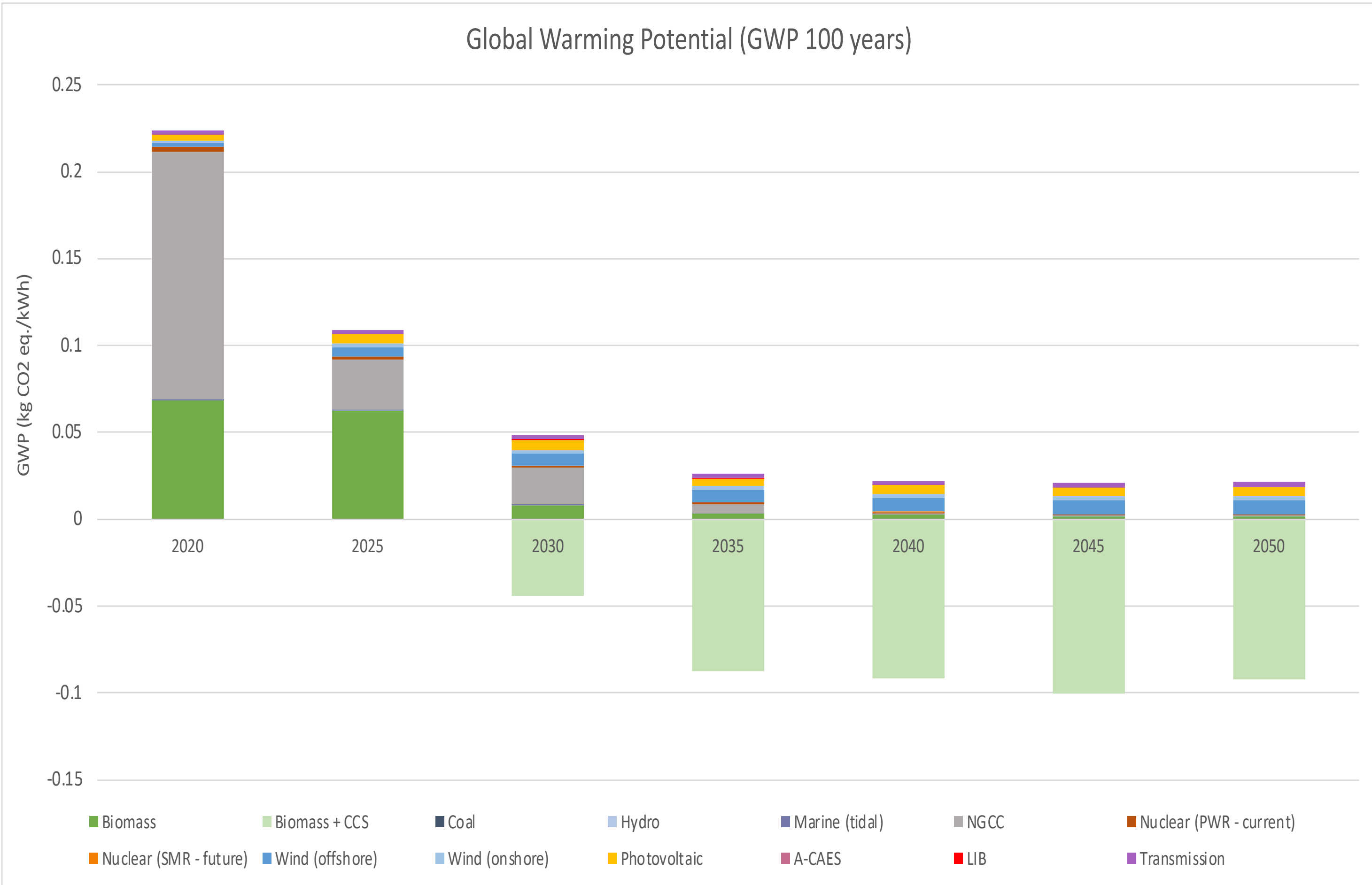


Figure 5.2: Global Warming Potential per unit of electricity delivered by the UK grid mix (including energy storage and transmission), expressed as relative to the 2020 value (100% = 170 g CO_2-eq/kWh).

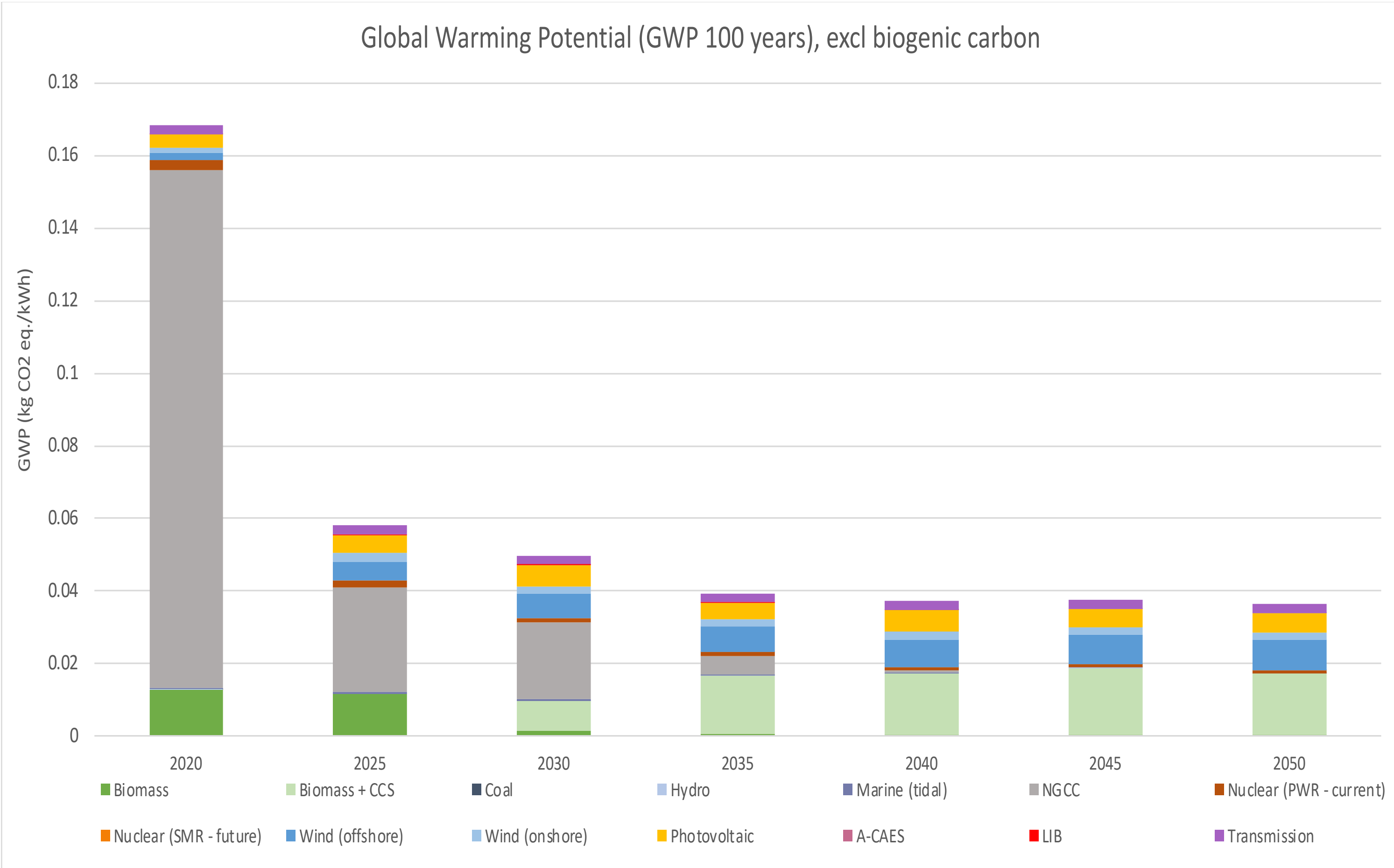


Figure 5.3: Global Warming Potential (excluding biogenic carbon) per unit of electricity delivered by the UK grid mix (including energy storage and transmission), expressed as relative to the 2020 value (100% = 170 g CO_2-eq/kWh).

In terms of AP (Figure 5.4), there is an overall expected net increase by 12% from 2020 to 2050 per unit of electricity delivered. The increase in impact is mainly due to the use of BECSS (11% in the grid mix), contributing to 71% of the impact per electricity delivered, followed by the photovoltaics and offshore wind. Latter due to high contribution in the grid mix. Whereas photovoltaics generation represents the same share of in the grid mix as BECSS in 2050, but with less than 84% impact compared to BECSS. Both biomass and BECSS were found to have the highest AP impact per kWh delivered. The main factor being the acquisition of the wood pellet and chips, and in the case of BECSS, the additional impact due to the chemicals required for the CCS unit (Yang et al. 2019). The lowest impact was found to be due to Nuclear (PWR), Hydropower and Tidal energy technology including storage. Furthermore, the impact from storage was almost negligible.

For human toxicity potential (HTP) the total impact is found to also increase going from 2020 to 2050, by 33% (Figure 5.5). This is mainly due to the increase share of wind generation followed by BECSS. The total impact are significantly different than for GWP or AP, and for the first time even those technologies that are conventionally regarded as the 'greenest' (i.e., wind and solar PV) end up being responsible for sizeable shares of the total impact. These results are due to a combination of these technologies' comparatively large demand for heavy metals (mainly copper, aluminium and nickel) per functional unit (also corroborated by previous independent studies (Hertwich et al. 2015); Kleijn et al. 2011), and the toxic emissions associated to the respective metal supply chains (mainly at the mining and beneficiation stages) (Jacobson et al. 2017; CML 2016). For the same reasons, electricity transmission lines and LIBs are also non-negligible contributors to this impact category. The HTP of nuclear electricity is likewise significant in the mix, almost entirely due to the emissions arising from the uranium supply chain.

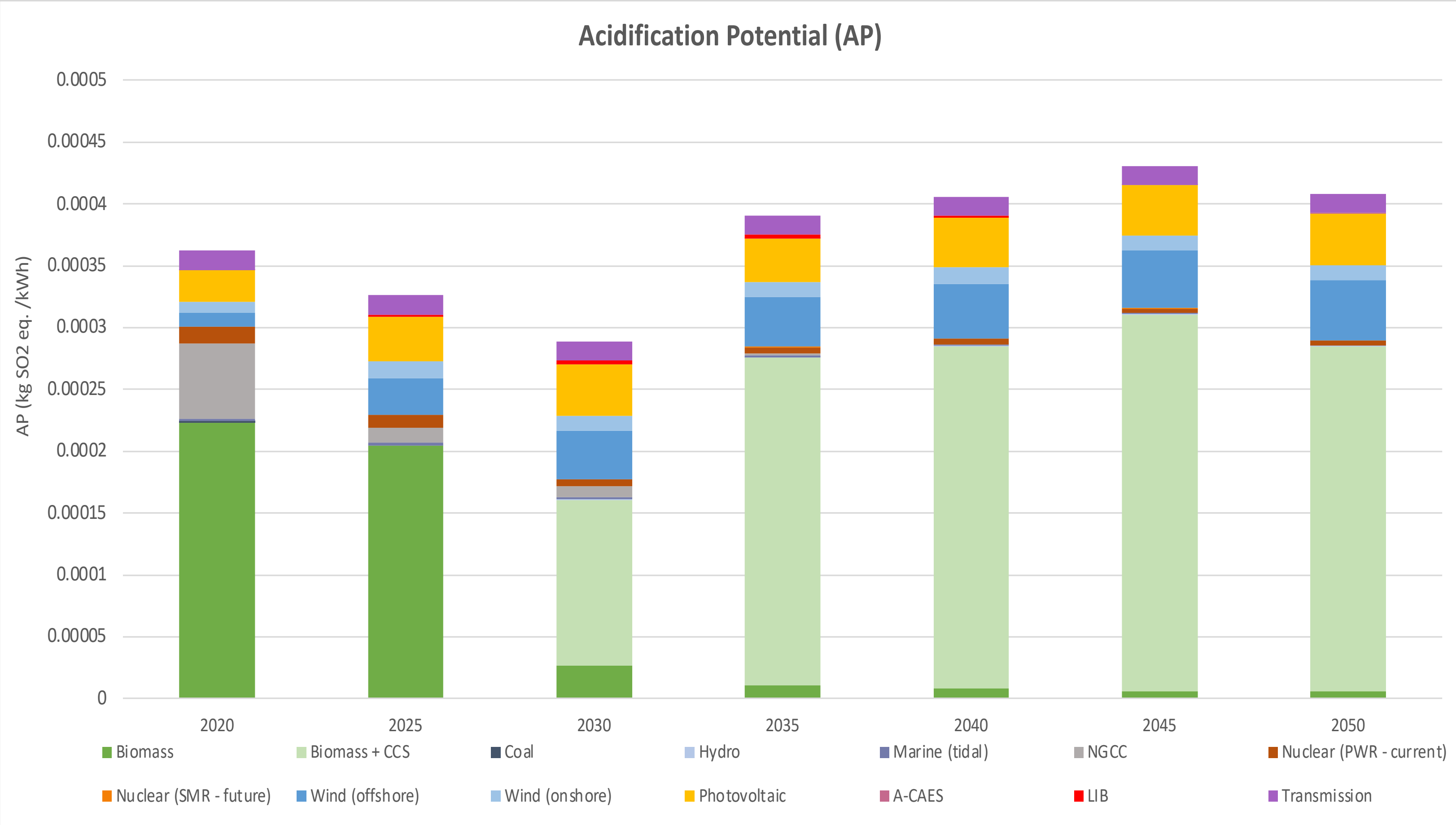


Figure 5.4: Acidification Potential per unit of electricity delivered by the UK grid mix (including energy storage and transmission), expressed as relative to the 2020 value (100% = 2.1 g SO_2-eq/kWh).

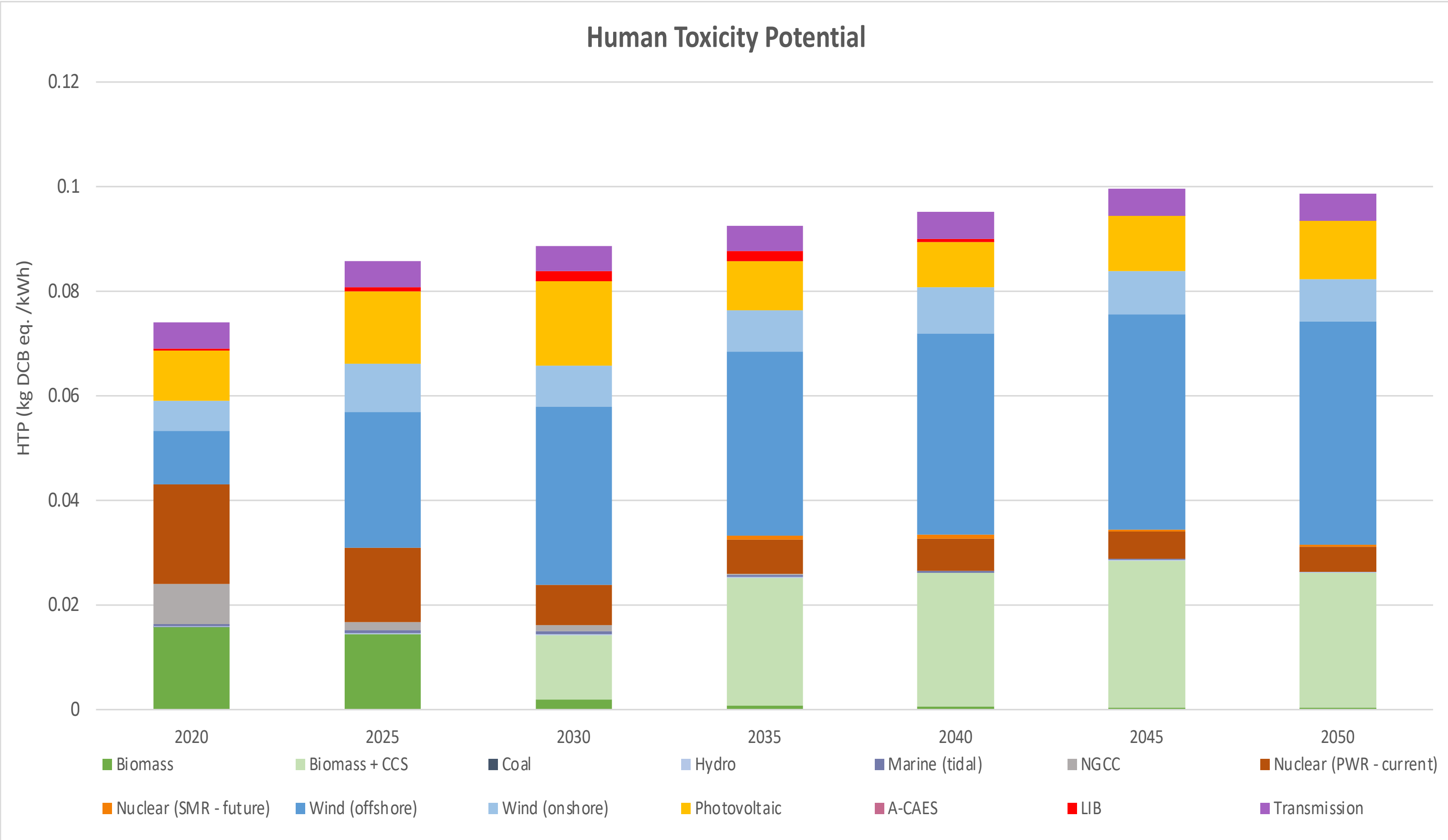


Figure 5.5: Human Toxicity Potential per unit of electricity delivered by the UK grid mix (including energy storage and transmission), expressed as relative to the 2020 value (100% = 83 g 1,4-DB-eq/kWh).

The abiotic resource depletion results (‘elements’, i.e., excluding energy resources such as fossil fuels and uranium) are even more striking, in that they point to a significant increase of the total grid mix impact (Figure 5.6). Even more so than for HTP, these results are mainly driven by wind and PV’s increased demand for metals (mainly copper) per unit of electricity delivered, and LIB and transmission lines once again play a non-negligible role. The ADP from LIB represents a significant share in 2030 and 2035, however due to the increase in second life battery use for grid storage, the demand for new LIB reduces dramatically by 2050 and its contribution to ADP impact.

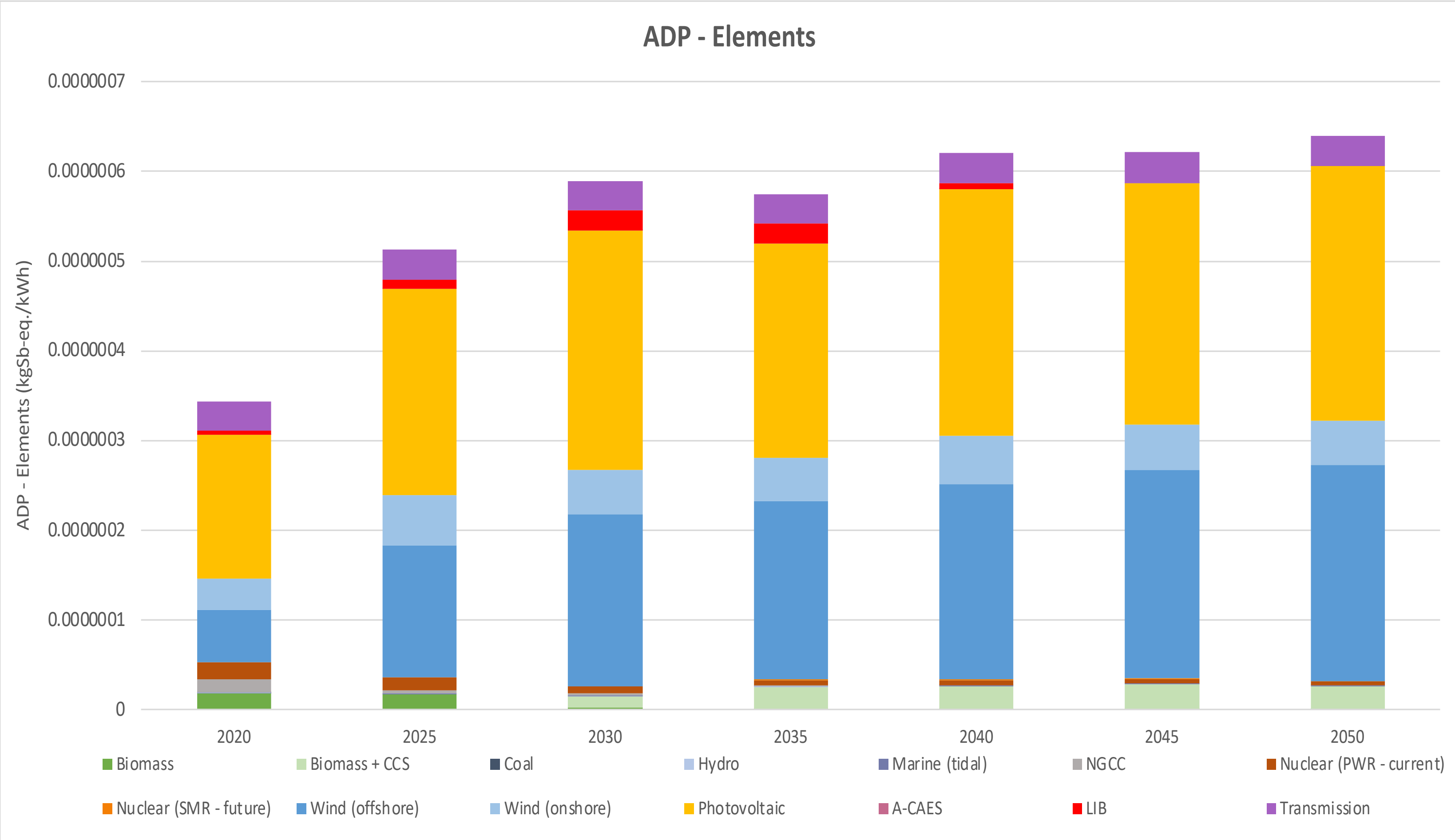


Figure 5.6: Abiotic Depletion Potential (elements) per unit of electricity delivered by the UK grid mix (including energy storage and transmission), expressed as relative to the 2019 value (100% = $3.6{\cdot}10^{-4}$ g Sb-eq/kWh).

6 Prospective LCA of the Light Duty Vehicle Fleet Evolution

The purpose of this chapter[10] is to assess the role of resource strategies (discussed in the chapter 3) on the overall environmental impacts of the evolution of the light duty vehicle (LDV) fleet in the UK over the next three decades (i.e., 2020 to 2050). The chapter is based on the following journal article:

6.1 Framework

Two resource strategies are considered in this chapter with regards to the transition to electric mobility, battery circular strategy and the increase in shared electric mobility. Battery circular strategy takes account of the reuse of battery in grid energy storage application and the reuse of some of the battery materials (i.e., active cathode material) coming directly from end of life (EoL) battery electric vehicles (BEVs) or from their second EoL from grid energy storage application. Whereas shared mobility strategy takes account of the increased mileage service covered by an average mix electric car sharing and ride sharing vehicles to replace the requirement of private vehicles. Two different pathways considered in this chapter, first one called "Baseline" scenario only consider the former strategy and the second called "TaaS" scenario considering both.

The boundary of assessment is expended to the degree necessary to capture the interlinkages highlighted in Figure 6.1, that have direct and indirect consequences of the life cycle impacts of transition to passengers BEVs. These include the expected major shift from internal combustion engine (ICE) to electrical power trains, the gradual increased penetration of renewable energy

[10] The chapter is based on the following journal: Raugei, M., Kamran, M., & Hutchinson, A. (2021). Environmental implications of the ongoing electrification of the UK light duty vehicle fleet. Resources, Conservation and Recycling, 174, 105818.

into the grid mix, the role of battery second life in grid energy storage and closed-loop recycling and the possible large-scale uptake of shared mobility.

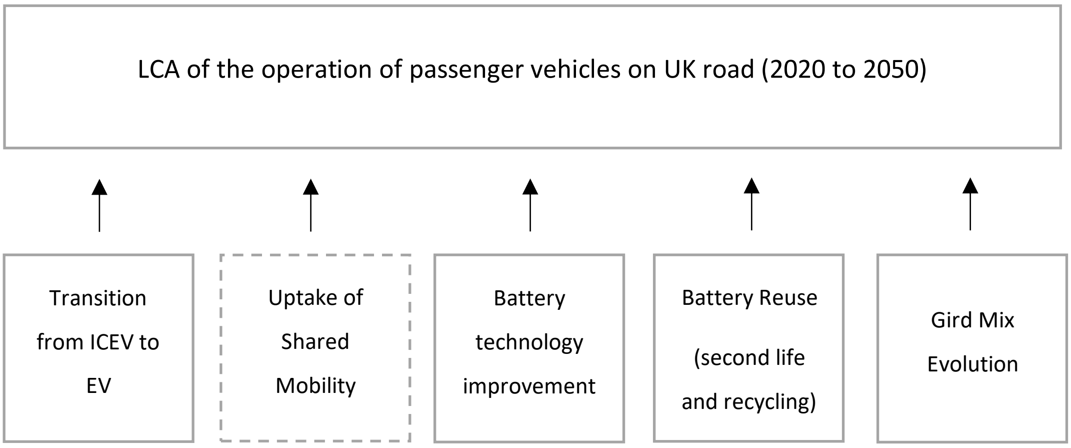


Figure 6.1: The interlinkages captured to assess the LCA of passenger vehicles. The solid line represents the "Baseline" scenario, and the inclusion of dotted line represents the "TaaS" scenario.

The functional unit (FU) of the study was set as the operation of, and the net changes to, the whole LDV fleet in the UK over the course of one year, including: (1) manufacturing of those LDVs that are newly registered in the year of analysis; (2) use phase of all LDVs on UK roads in the year of analysis, including their maintenance and the supply chains of the required energy carriers (i.e., petrol and diesel for ICEVs and electricity for BEVs); (3) decommissioning of those LDVs that reach their EoL in the year of analysis. This is taken together with the changes in the BEV lithium-ion battery (LIB) supply chains, BEV LIB recycling and the grid mix supply for each year, due to technology advancement, second life and recycling influence of batteries. One the other hand, the number of vehicles on the roads are dictated by the assumption on total passenger mobiliy mileage of each year and the uptake of shared mobility impacting the total number of vehicles on the road.

A dynamic MFA approach was used in Chapter 4 to capture the demand and track the changes in various mass flows for each year up to the year 2050, which dictate the LCA inventory for production, use and end of life for passenger vehicles in each year. LCA takes a prospective hybrid approach which includes attributional and consequential elements. The transition from ICEV to BEV and the grid mix evolution are considered as prospective elements of the LCA. The improvement in battery technology and battery reuse has a direct consequence on the battery supply chain for each year of which the impacts are captured in a consequential way. Furthermore, the uptake of shared mobility leads to a gradual reduction in the total number of vehicles required to meet the same overall demand for personal mobility (i.e., total distance travelled per year) which has a direct consequence on the number of vehicles and their batteries required by the fleet, this is also captured in a consequential way. Whereas all other elements are taken as attributional. Figure 6.2: Presents the LCA components of the foreground system, all the input and output flows of the various process steps within the analysed and the background inventory in order to keep track of all the indirect raw material and energy requirements and emissions that are associated to each 'foreground' system are considered in the LCA.

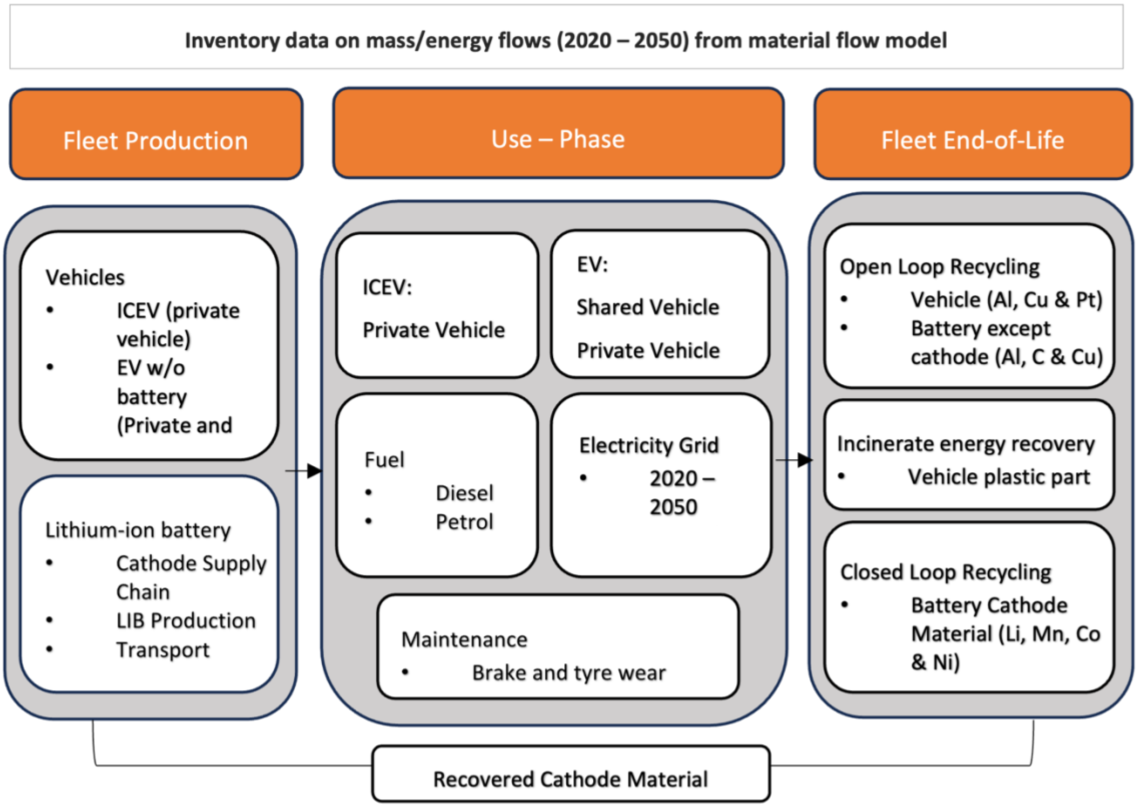


Figure 6.2: Structure of the prospective LCA passenger fleet model, with identification of the individual sub-models used for each of the key processes comprising the analysed system.

Two main scenarios are assessed in this chapter, respectively named "Baseline" and "TaaS" defined in Chapters 3 and 4, to assess the impact of mitigating strategies of battery reuse and shared mobility on the LDV. The focus was put entirely on battery electric powertrains (thereby disregarding hybrid vehicles as they are likely playing only a relatively minor and temporary role in the fleet, over the first few years of the considered time frame. Since the main focus of the thesis is exploring impact of TaaS and BEV batteries. It was decided to adopt a pre-existing vehicle model excluding the battery. The grid mix model is adopted from Chapter 5.

The growth in the total distance travelled each year is based on the 2018 UK department of transport road traffic "reference" scenario, assuming a 11% growth up to 2050. Specifically, TaaS vehicles are expected to cover 64,000 km/year and remain in service for 3 years, assuming a mix of car sharing and ride sharing vehicles. The two scenarios are otherwise the same, and both assume a linear increase in total distance travelled (from 420 billion km/year in 2020 to 530 billion km/year in 2050), a gradual phase-out of ICEVs (consistently with the UK government's target to ban sales of new light-duty ICEVs by 2030), and a linearly increasing collection rate for EoL BEV LIBs, destined to second life and recycling. Tables 6.1 and 6.2 report the key parameters defining the two scenarios, respectively, carried out in Chapter 4.

Table 6.1: Key model parameters for "Baseline" scenario (vehicle numbers rounded to nearest 10,000).

Year	New ICEVs	New BEVs	% new BEVs for TaaS	EoL ICEVs	EoL BEVs	TOTAL ICEVs in circulation	TOTAL BEVs in circulation	EoL BEV LIB collection rate
2020	2,060,000	410,000	0%	2,230,000	0	29,740,000	850,000	80%
2025	830,000	1,160,000	0%	1,740,000	0	26,550,000	5,280,000	83%
2030	9,000	2,670,000	0%	2,420,000	0	17,630,000	15,500,000	86%
2035	0	2,780,000	0%	1,880,000	630,000	6,520,000	27,950,000	89%
2040	0	2,400,000	0%	710,000	1,400,000	1,230,000	34,640,000	93%
2045	0	2,830,000	0%	0	2,530,000	0	37,330,000	96%
2050	0	2,800,000	0%	0	2,490,000	0	38,850,000	99%

Table 6.2: Key model parameters for "TaaS" scenario (vehicle numbers rounded to nearest 10,000).

Year	New ICEVs	New BEVs	% new BEVs for TaaS	EoL ICEVs	EoL BEVs	TOTAL ICEVs in circulation	TOTAL BEVs in circulation	EoL BEV LIB collection rate
2020	2,060,000	410,000	0.0%	2,230,000	0	29,740,000	850,000	80%
2025	690,000	820,000	7.5%	1,740,000	60,000	25,700,000	4,940,000	83%
2030	7,000	1,660,000	15.0%	2,420,000	215,000	16,360,000	13,400,000	86%
2035	0	1,470,000	22.5%	1,760,000	1,070,000	5,380,000	23,040,000	89%
2040	0	910,000	30.0%	570,000	1,720,000	960,000	27,640,000	93%
2045	0	620,000	37.5%	0	2,710,000	0	27,580,000	96%
2050	0	270,000	45.0%	0	2,940,000	0	25,930,000	99%

6.2 Vehicle Manufacturing

This section describes the construction, use phase and end of life (EoL) of Internal Combustion Engine Vehicle (ICEV) and Battery Electric Vehicle (BEV). The models for both vehicles are adopted from a pre-existing model developed for a compact (C-segment) passenger car with a kerb mass of 1100kg (excluding battery) and service life of 150,000 km (Raugei et al. 2015). The vehicle manufacturing phase comprises the following sub-assemblies, body and chassis (consisting of all steel components), powertrain, electrical system and trim. The model assumes EoL recycling and recovery based on the average mix of the technologies that were displaced, which is the recommended recycling crediting approach for in attributional LCAs (Raugei et al. 2015). The specification for both the model is presented in Figure 6.3 for the vehicle glider and the vehicle powertrain components (excluding battery system).

Table 6.3: Vehicle specification and assumptions for the vehicle LCA model adopted from Raugei et al. (2015).

Vehicle Specification		
	ICEV	**BEV**
Engine	Volkswagen petrol engine, C segment	Renault Fluence Z.E, C segment
Kerb mass	1370 kg	1370 kg
Fuel	8 litres/100 km	19 kWh/ 100km
Service life	190,000	190,000
Maintenance		
Maintenance every 30,000 km	Replacement of tyres, brake pads, vehicle lubrication, 5% of worn-out trim.	Replacement of tyres, brake pads, vehicle lubrication, 5% of worn-out trim.

Maintenance over lifetime	one lead-acid battery replacement, 10% outer body panel replacement.	10% outer body panel replacement.
EoL Open Loop Recycling		
Steel	85% mass	85% mass
Al alloys (engines and wheels)	75% mass	75% mass
Pt (catalytic converters)	100%	-
Copper	100%	100%
plastic parts	Incinerated with energy recovery credits	

The vehicle model was scaled up to represent the average LDV fleet on UK roads, the fleet-average kerb mass was taken as 1370 kg. Whereas the service life was set to 190,000 km to represent the average vehicle lifetime mileage in EU (Ricardo-AEA 2015). The fuel consumption for ICEV was set to be 8 litres/100 km to represent the mix of petrol and diesel fuel (assuming 50% petrol and 50% diesel) (Raugei et al. 2018). The electricity consumption for BEV was taken as 19 kWh/100 km representing the average vehicle segment electricity consumption in 2017 based on the following literature (Raugei et al. 2018).

In recent years, the LDV fleet on UK roads has comprised a mix of vehicles manufactured domestically and in Europe, with the latter mainly represented by Germany (Department for Transport 2020; European Automotive Manufacturers Assocation 2019). However, considering the upcoming change regarding the motor vehicle trade between the UK and EU, also in light of Brexit, it is likely that the share of domestic vehicles will increase in the future, and hence for the sake of simplicity vehicle manufacturing was modelled as taking place in the UK.

The recycling, use-phase and maintenance electricity use was taken to be based on the average UK grid mix from 2020 through to 2050 (modelled in section 5), with 5 years interval. Tables 6.4 and 6.5 represents the life cycle assessment impact for ICEV and BEV (excluding the battery and the electricity grid supply inputs for the main processes), adopted from (Raugei et al. 2015 and Raugei et al. 2018).

Table 6.4: The environmental impact of BEV life cycle excluding electricity input and battery manufacturing.

BEV				
Impact	Manufacture (excluding electricity input and battery)	Use (excluding electricity input)	Maintenance (excluding electricity input)	EoL (excluding electricity input)
CED	68,200	0.0	22,100	32,600
nr-CED	63,700	0.0	21,600	31,800
GWP (kg CO_2 eq.)	3,270	0.0	1,280	2,200
GWP – excl. bio C (kg CO_2 eq.)	3,300	0.0	1,280	2,200
POCP (kg Ethene eq.)	2.7E+00	3.4E-09	4.6E-01	8.7E-01
ADP (kg Sb eq.)	2.2E-01	0E+00	1.1E-02	4.2E-02
HTP – cancer (CTUh)	1.3E-03	1.4E-11	1.3E-04	1.3E-03
HTP – non cancer (CTUh)	1.1E-02	9.9E-09	5.1E-04	3.1E-03

Table 6.5: The environmental impact of ICEV life cycle, excluding electricity input.

ICEV				
Impact	Manufacture (excluding electricity input)	Use (1km travelled)	Maintenance (excluding electricity input)	EoL (excluding electricity input)
CED	83,300	3.45	22,000	34,200
nr – CED	78,600	3.44	21,500	33,400
GWP (kg CO_2 eq.)	2,530	0.23	1,350	2,540
GWP – excl. bio C (kg CO_2 eq.)	2,30	0.23	1,350	2,540
POCP (kg Ethene eq.)	8.7E-01	7.0E-05	4.7E-01	8.7E-01
ADP (kg Sb eq.)	2.6E-02	3E-08	6.7E-02	2.6E-02
HTP – cancer (CTUh)	1.5E-03	1.0E-09	1.4E-04	1.5E-03
HTP – non cancer (CTUh)	2.8E-03	2.2E-08	5.3E-04	2.8E-03

6.3 Lithium-ion Battery Manufacturing

The EV battery manufacturing model comprises of two parts, (1) supply chain for the battery cathode material and (2) manufacturing of LIB battery. The supply chain model takes account of the production and refining process for all four metals up to the chemical forms in which they are fed to the LIB manufacturing industry, i.e., respectively, lithium hydroxide (LiOH), cobalt sulphate (CoSO4), manganese sulfate (MnSO4) and nickel sulfate (NiSO4). It is then assumed these metals are transport to Europe reflecting reflect the same relative shares of global production capacity where possible.

The primary supply chains of the key cathode metals Lithium (Li), Cobalt (Co), Manganese (Mn) and Nickel (Ni) were modelled as follows:

Lithium: In 2022, the three main global suppliers of primary lithium were Australia (47% of global supply, from spodumene rocks), China (14.6%, mix of spodumene rocks and Li brine), Chile (30%, from Li brines) and Argentina (4.7%, from Li brines) (US Geological Survey, 2023). Extraction from brine is done by pumping lithium brines to the surface and concentrated by series of solar evaporation ponds which usually takes roughly 9 to 12 months (BGS 2016). The brines producers commonly processed to lithium carbonate through a series of solar evaporation and chemical processing (Lusty et al. 2022) which are generally used in low nickel content LiBs, whereas hard rock mines are processed as lithium concentrate, both are later refined to produce lithium hydroxide for the use in high nickel content lithium-ion batteries (Lusty et al. 2022). Currently more than 70% of lithium (concentrate and carbonate) is refined in China. The model, therefore, focuses on the production taking place in those four countries and transported to China for refinement to reflect share of global product capacity. All processes are based on Ecoinvent 3.6 dataset except for concentrated spodumene to lithium hydroxide. Currently dataset does not have a direct process for concentrated spodumene to lithium hydroxide which is preferred route for future spodumene supply chain. Table 6.6 provides the inventory for spodumene conversation was taken from (Chordia et al. 2022), a full material inventory for the production of LiOH is also provided in Appendix F.

Table 6.6: Inventory data for 1kg of LiOH from concentrated spodumene, adopted from (Chordia et al, 2022).

Items	Quantities	Units
Spodumene Concentrate	5.81	kg
Calcium hydroxide	1.33	kg
Electricity, medium voltage	43.92	MJ
Sodium hydroxide	0.112	kg
Carbon dioxide liquid	0.084	kg

Nitrogen liquid	0.4	kg
Heat, natural gas	42.5	MJ
Hydrochloric acid	0.072	kg
Soda (sodium carbonate)	1.58	kg
Inert material landfill facility	1.15	kg
Municipal incineration	5.73	kg

Cobalt: Cobalt is extracted mainly from mining of copper-cobalt ores and, to a lesser extent, nickel-cobalt ores. The Republic of Congo is currently the most prominent producer of Cobalt by far, contributing to 70% of global supply (US Geological Survey, 2023), and so Cobalt production was modelled as coming from there, with subsequent transport to China for refining as cobalt sulphate (Dai et al. 2018b). Europe represents significant refining capacity in Finland, Belgium and Norway (Lusty et al. 2022), however these are considerably small compared to 75% of global refinement occurring in China (Idoine 2023). Therefore, the model considered mining in Congo and transported to China for refinement to reflect share of global product capacity, all based on Ecoinvent dataset.

Manganese: Manganese is relatively abundant in the Earth's crust, and is extracted by mining a range of ores, the most important of which is pyrolusite (MnO2). In 2022, the three main producing countries were South Africa (36%), Gabon (23%), Australia (16%) (USGS, 2023). Therefore, the focus of the model was on production taking place in those five countries and refinement in China representing 92% of the global refined production all based on Ecoinvent dataset.

Nickel: In 2022, the largest shares of global nickel production were evenly distributed among a relatively large number of countries, among which foremost were the Indonesia (48%) followed by Philippines (10%), rest representing less than 10% for Russia, Canada, Australia and New Caledonia. The two main sources of Nickel are sulfide mines, which is characteristic of Canada and prevalent (>60%) in Australia, and lateritic deposits (USGS 2021). The latter is gradually becoming more prevalent, but unfortunately, detailed life-cycle inventory data for laterite mining is yet unavailable. Also, Canadian production is expected to drop after the planned phasing out of Thompson Mines' operations in Manitoba, Newfoundland and Labrador. Therefore, Australian production of raw Ni from sulfide mines was selected for the model. Shared of global refined production is mainly from 25% Indonesia and 29% China and 23% other (Lusty et al. 2022). However, refinement in Indonesia is mainly for lateritic nickel ore, whereas in the case of China it is a mixture for sulfide and lateritic refinement, hence for consistency, Australian production was modelled to be transported to China for refinement.

The FU of the metal supply chain was taken as 1kg of active cathode material (LiOH, CoSO4 MnSO4 and NiSO4) transported to Europe. Figure 6.3 presents the flow diagram of supply chain for the battery cathode material.

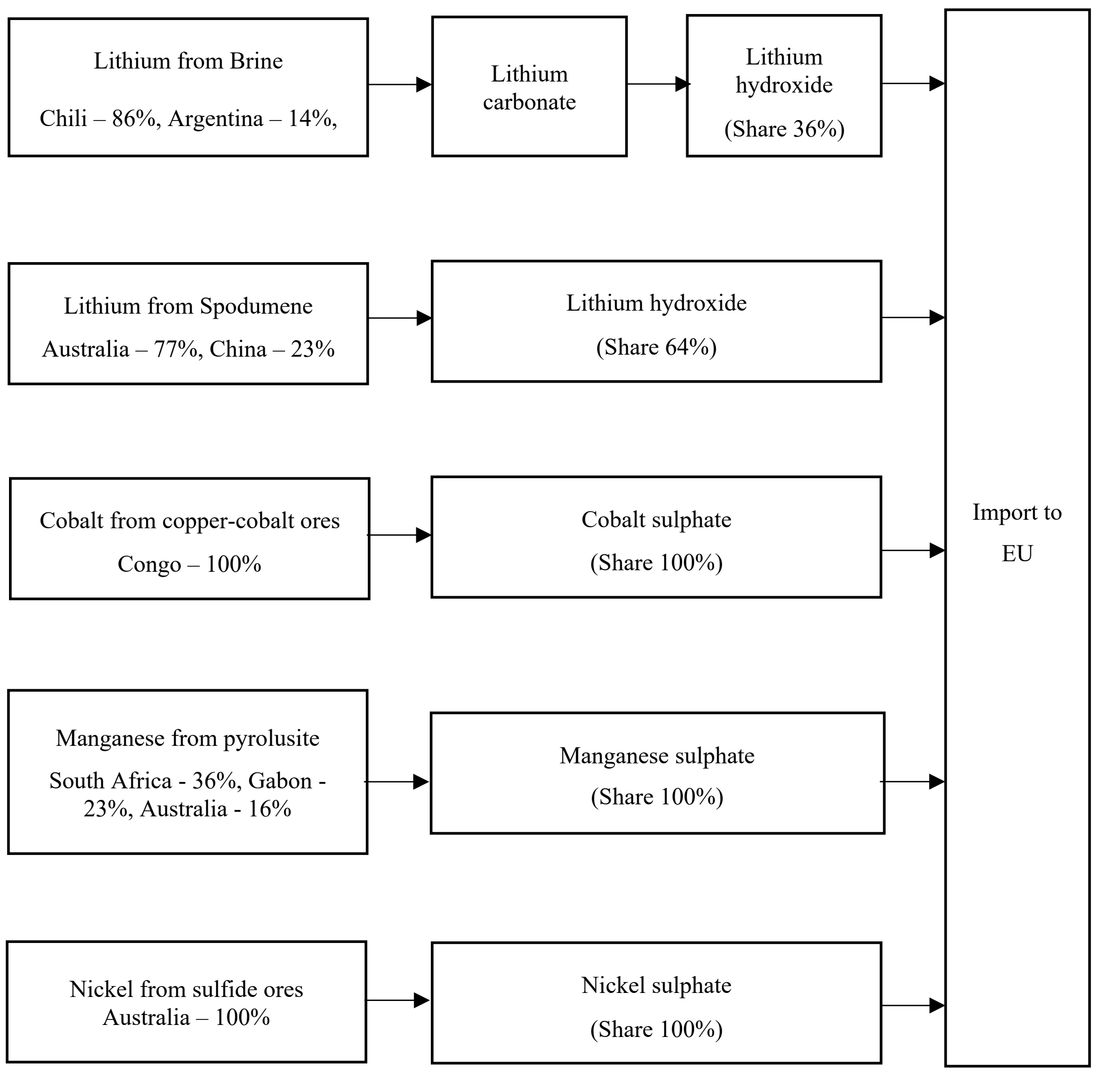


Figure 6.3: represents the active cathode material supply chain to EU for manufacturing.

Currently China, South Korea and Japan represent the major shares of battery manufacturing worldwide (Eddy et al. 2019). However, there is growing demand for battery production to be close to car manufacturers. Therefore, considering the recent uptake of BEVs and the planned ramp-up of battery production capacity in Europe, it was assumed that the cell components and battery packs for the UK fleet will all be manufactured and assembled in Europe. Specifically, LG Chem battery production in Poland is scheduled to increase to 65-70 GWh/year (Reiserer 2019), SK Innovation production is to reach 30 GWh/year in Hungary (Inside Evs 2021), and Northvolt further plans production of up to 32 GWh/year in Sweden (Phillips 2020). These represent the largest battery manufacturing capacities in Europe, and thus a suitably weighted combination of the current Polish, Hungarian and Swedish grid mixes from Ecoinvent database was used in the model to estimate the impacts from the electricity use during battery manufacturing up to the year 2050.

The battery manufacturing model assumes an average battery pack mass of 323kg, which initially equates to a usable energy storage capacity of 50kWh per vehicle, based on current EV battery technology used. The battery pack mass is then assumed to remain constant, while the usable capacity is expected to change in the coming years as the technology improves. A linear increase in energy density is assumed up to year 2035 due to the expected shift from NMC 622 to NMC 811 (summarised in Chapter 4, section 4.2).

To take account of the reuse of active cathode materials (LiOH, CoSO4, MnSO4 and NiSO4) due to closed loop recycling, the ratio of virgin content is used to represent the impact of 1kg of active cathode material production. The consequential approach was adopted for the battery supply chain model, such that the masses of the recovered metals from recycled EoL BEV batteries and from second-life batteries from grid storage, results in reducing the quantities of the same metals that are sourced from primary supply chains for the manufacturing of new BEV batteries. The number of batteries that are recycled is determined by the number of EoL BEVs, the collection rate, and by how many batteries are repurposed for second life (represented in Chapter 4), and all these parameters are estimated dynamically and vary per year. Table 6.7: represents improvement in the active cathode material and the share of virgin material content resulting by closed-loop recycling in the “Baseline” and “TaaS” scenarios.

Table 6.8: represents improvement in the active cathode material and the share of virgin material content through to 2050 for “Baseline” and “TaaS” scenario.

Material Content (kg) per kg of active cathode material	2020	2025	2030	2035	2040	2045	2050
LiOH	0.28	0.33	0.38	0.42	0.42	0.42	0.42
CoSO4	0.37	0.36	0.33	0.27	0.27	0.27	0.27
MnSO4	0.36	0.36	0.32	0.26	0.26	0.26	0.26
NiSO4	1.11	1.42	1.77	2.15	2.15	2.15	2.15
Share of virgin cathode material (Baseline Scenario)							
vir_Mn_share	1.00	1.00	1.00	0.81	0.39	0.07	0.21
vir_Co_share	1.00	1.00	1.00	0.81	0.39	0.08	0.22
vir_Li_share	1.00	1.00	1.00	0.90	0.63	0.30	0.23
vir_Ni_share	1.00	1.00	1.00	0.92	0.68	0.33	0.21
Share of virgin cathode material (TaaS Scenario)							
vir_Mn_share	1.00	1.00	0.94	0.31	0.00	0.00	0.00
vir_Co_share	1.00	1.00	0.94	0.31	0.00	0.00	0.00
vir_Li_share	1.00	1.00	0.95	0.56	0.00	0.00	0.00
vir_Ni_share	1.00	1.00	0.95	0.60	0.00	0.00	0.00

Further improvements in batteries assumed to be due to the reduction in weight of the battery pack casing and ancillary, this reduction is used to increase the energy density further while keeping the mass constant though to 2050 as to achieve an increasing vehicle driving range. The foreground material and energy use inventory for cell and battery pack production was informed by a recent report by Argonne Laboratory (Dai et al. 2020), and the average LIB cell-to-pack mass ratio was taken as 0.80. Bill of materials and the battery manufacturing model can be found in the Appendix F. Whereas the flow diagram of battery manufacturing is shown below in Figure 6.4. As for transportation impacts, it was assumed that the batteries will be transported over an average distance of 2,000 km by large (16-32 tonnes) lorries, from the EU to the UK. Table 6.8 and 6.9 represents the environmental impact results for 1 kg of LIB manufactured for "Basline" and Taas" scenario respectively.

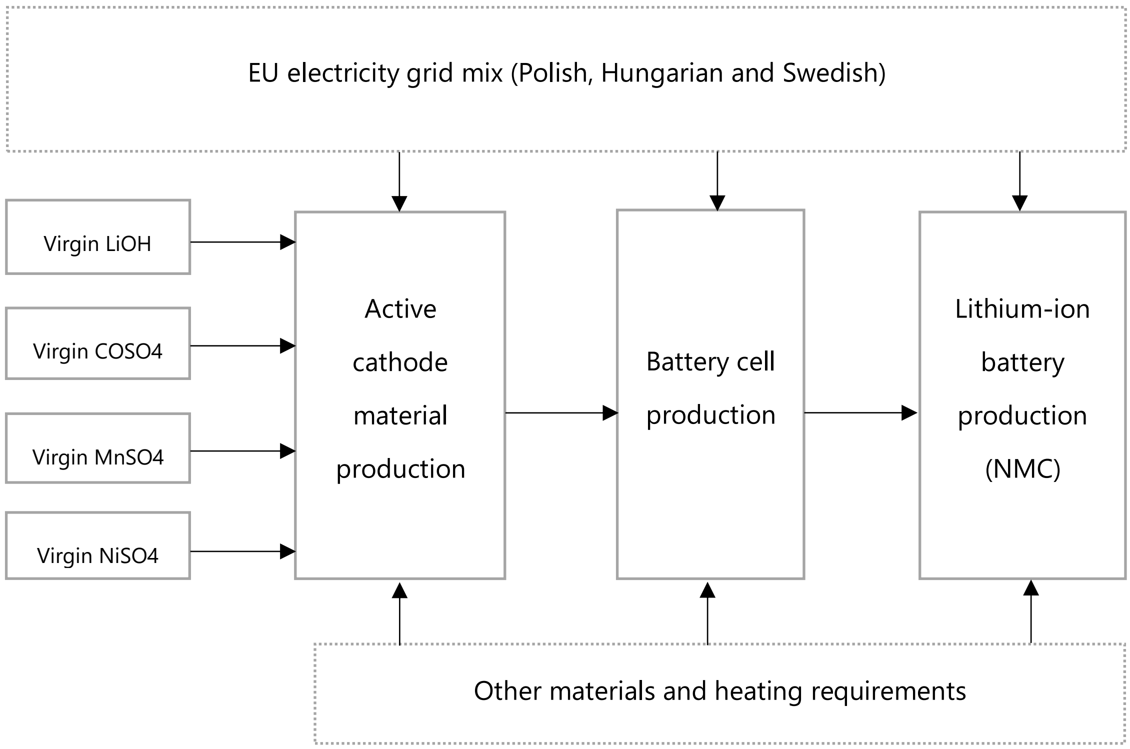


Figure 6.4: Represents the flow diagram of LIB manufacturing process in the EU. The solid lines represent the elements of manufacturing process that vary for each year due to change in required virgin materials and improvement in battery technology, whereas the dotted lines represent the static process.

Figure 6.8: Impact for 1kg of battery manufactured and transported to UK, baseline scenario.

Impact **1 kg of LIB pack**	**2020**	**2025**	**2030**	**2035**	**2040**	**2045**	**2050**
CED (MJ)	221	240	258	265	229	181	168
nr-CED (MJ)	204	222	240	246	212	167	155
GWP (kg CO_2 eq.)	13.3	14.6	16.0	16.6	14.0	10.5	9.6
GWP – excl. bio C (kg CO_2 eq.)	13.2	14.5	15.9	16.4	13.9	10.5	9.6
POCP (kg Ethene eq.)	1.7E-02	2.0E-02	2.3E-02	2.5E-02	2.0E-02	1.2E-02	9.9E-03
ADP (kg Sb eq.)	7.1E-04	7.5E-04	7.9E-04	8.2E-04	7.5E-04	6.5E-04	6.2E-04
HTP – cancer (CTUh)	3.0E-06	3.3E-06	3.6E-06	3.7E-06	3.3E-06	2.6E-06	2.3E-06
HTP – non cancer (CTUh)	2.6E-05	2.8E-05	3.0E-05	3.2E-05	2.8E-05	2.3E-05	2.1E-05

Figure 6.9: Impact for 1kg of battery manufactured and transported to UK, TaaS scenario.

Impact 1 kg of LIB pack	2020	2025	2030	2035	2040	2045	2050
CED (MJ)	221	234	252	218	138	138	138
nr-CED (MJ)	204	217	234	202	126	126	126
GWP (kg CO_2 eq.)	13.3	14.3	15.6	13.2	7.4	7.4	7.4
GWP – excl. bio C (kg CO_2 eq.)	13.2	14.2	15.5	13.1	7.4	7.4	7.4
POCP (kg Ethene eq.)	1.7E-02	1.9E-02	2.2E-02	1.8E-02	5.4E-03	5.4E-03	5.4E-03
ADP (kg Sb eq.)	7.1E-04	7.4E-04	7.8E-04	7.3E-04	5.6E-04	5.6E-04	5.6E-04
HTP – cancer (CTUh)	3.0E-06	3.2E-06	3.5E-06	3.1E-06	1.9E-06	1.9E-06	1.9E-06
HTP – non cancer (CTUh)	2.6E-05	2.7E-05	3.0E-05	2.7E-05	1.7E-05	1.7E-05	1.7E-05

6.4 Lithium-ion Battery Recycling

Battery pack dismantling was excluded from the model due to the lack of information on the associated processes at scale, but all the non-cell parts of the battery pack were assumed to be treated as scrap, and their recycling was modelled under the same assumptions as for the other metal parts of the EoL vehicle (section 6.2), this is the recovery of copper and aluminium. Currently the most mature process for recycling LIBs is pyrometallurgical recycling. However, due to its high energy demand and low metal recovery efficiency (Arambarri et al. 2019), and the fact that a shift to hydrometallurgical recycling is already underway in Asia and Europe, all future LIB cells were assumed to undergo hydrometallurgical recycling in the EU, using inorganic acid leaching to recover the key battery materials: Li, Ni, Mn and Co.

The foreground material and energy use inventory for the recycling process was informed by Argonne Laboratory report (Dai & Winjobi, 2019). It was assumed that the recycling plants will be based in the UK; accordingly, the UK grid mix model (represented in chapter 5) was used to track the evolving impacts of electricity use over time, from 2020 to 2050. It was further assumed that electricity will be used for industrial furnaces and all machinery. Table 6.10 represents the impact of recycling 1kg of LIB battery pack.

Figure 6.10: The impact of recycling 1kg of LIB battery pack.

Impact (1 kg of input LIB pack to recycling)	2020	2025	2030	2035	2040	2045	2050
CED (MJ)	24.7	24.4	24.3	24.3	24.4	24.4	24.3
nr – CED (MJ)	23.0	22.5	22.3	22.3	22.3	22.2	22.2
GWP (kg CO_2 eq.)	1.2	1.2	1.2	1.2	1.2	1.2	1.2
GWP – excl. bio C (kg CO_2 eq.)	1.2	1.2	1.2	1.2	1.2	1.2	1.2
POCP (kg Ethene eq.)	5.9E-04	5.9E-04	5.9E-04	6.0E-04	6.0E-04	6.0E-04	6.0E-04
ADP (kg Sb eq.)	1.3E-05	1.3E-05	1.3E-05	1.3E-05	1.3E-05	1.3E-05	1.3E-05
HTP – cancer (CTUh)	1.4E-07	1.4E-07	1.4E-07	1.4E-07	1.4E-07	1.4E-07	1.4E-07
HTP – non cancer (CTUh)	3.6E-07	3.6E-07	3.6E-07	3.6E-07	3.6E-07	3.6E-07	3.6E-07

Finally, it was assumed that all the recovered cathode metals will be reused directly for the manufacturing of new LIB cells (i.e., closed-loop recycling), which is already captured in section 6.4. Accordingly, only the net surplus of recycled metals, after the annual demand for new LIBs to equip newly registered BEVs has been satisfied, was accounted for towards the calculation of EoL impact credits, this is represented in Table 6.11. This was calculated based on the displacement of the respectively primary supply chains (up to chemical form in which the metals are fed to the LIB manufacturing industry). For credits of materials used excluding cell, Ecoinvent process for recovering aluminium and copper was considered, the assumption was taken as similar to recovering aluminium and copper from vehicles, i.e., 75% of aluminium mass recycling and 100% of copper recycled. Table 6.12: represents the impact and credits of recycling and recovering copper and aluminium materials from 1 kg of LIB.

Table 6.11: Credits due to surplus of cathode material with respect to the demand for materials for BEV LIBs.

Impact – 1 kg of recovered material (CREDITS for surplus CATHODE material w.r.t. demand for new LIBs)	Li	Ni	Mn	Co
CED (MJ)	-303	-341	-54.3	-155
nr – CED (MJ)	-281	-322	-51	-126
GWP (kg CO_2 eq.)	-20.2	-24.7	-1.95	-8.8
GWP – excl. bio C (kg CO_2 eq.)	-20.3	-23.7	-1.97	-8.93
POCP (kg Ethene eq.)	-9.2E-03	-6.8E-02	-3.4E-03	-7.8E-03
ADP (kg Sb eq.)	-3.7E-04	-8.9E-04	-2.2E-05	-1.4E-04
HTP – cancer (CTUh)	-2.3E-06	-6.3E-06	-1.9E-07	-6.0E-07
HTP – non cancer (CTUh)	-9.4E-06	-5.1E-05	-8.0E-07	-3.8E-06

Table 6.12: Impacts and credits due to recycling and recovering copper and aluminium materials from 1 kg of LIB.

Impact (1 kg of input LIB pack to recycling)	IMPACTS (excl. credits)	CREDITS due to recovered Al and Cu
CED (MJ)	8.69	-29.44
nr – CED (MJ)	8.18	-24.93
GWP (kg CO_2 eq.)	0.472	-1.632
GWP – excl. bio C (kg CO_2 eq.)	0.474	-1.632
POCP (kg Ethene eq.)	0.000219	-0.00142
ADP (kg Sb eq.)	3.33E-05	-0.00014
HTP – cancer (CTUh)	5.21E-08	-7.5E-07
HTP – non cancer (CTUh)	1.2E-06	-7.6E-06

6.5 LCA Model Equations

The LCA data is exported from Gabi to the Excel, along with MFA data on stock and material flows to calculate the environmental impact at fleet level.

- Time-dependent Variables are indicated in *italics* using the *V(t)* notation, where *(t)* indicates the running year

Total ICEV manufacturing Impact each year:

$$I_{ICEV_{manu}} = nICEV(t) \times i_{ICEV_{manu}} \tag{58}$$

Where,

$I_{ICEV_{manu}} = ICEV\ manufacturing\ impact\ for\ 1\ unit\ of\ ICEV$

$nICEV(t) = new\ ICEV\ registrations$

<u>Total ICEV use phase (incl. maintenance) impact each year:</u>

$$Dist_{ICEV(t)} = (ICEVp(t) \times Myp) + (ICEVt(t) \times Myt) \tag{59}$$

$$Service_{ICEV(t)} = \frac{ICEVp(t)}{Tp} + \frac{ICEVt(t)}{Tt} \tag{60}$$

$$I_{ICEV_{usephase}}(t) = \left(Service_{ICEV} \times i_{ICEV_{maint}}\right) + \left(Dist_{ICEV} \times i_{ICEV_{use}}\right) \tag{61}$$

Where,

$Dist_{ICEV(t)} = Distance\ travelled\ by\ all\ ICEVs\ on\ UK\ roads$

$Service_{ICEV(t)} = full\ service\ life\ eq. maintenance\ units\ for\ ICEVs$

$i_{ICEV_{maint}} = 1\ ICEV\ unit\ maintenance\ (excl. electricity\ input)$

$i_{ICEV_{use}} = 1\ km\ travelled\ by\ ICEV$

Total ICEV EoL impact each year:

$$I_{ICEV_{eol}}(t) = eol_{ICEV(t)} \times \ i_{ICEV_{eol}} \quad \textbf{(62)}$$

Where,

$I_{ICEV_{eol}}(t) = \ ICEV\ EoL\ impact\ each\ year$

$eol_{ICEV(t)} = number\ of\ ICEV\ decommissionings$

$i_{ICEV_{eol}} = \ ICEV\ end\ of\ life\ \ impact\ for\ 1\ unit\ of\ ICEV$

Total EV manufacturing (excl. LIB) impact for each year:

$$I_{EV_{manu}}(t) = \ nEV(t) \times \ i_{EV(exc.LiB)} \quad \textbf{(63)}$$

Where,

$I_{EV_{manu}}(t) = Total\ impact\ for\ EV\ manufacturing\ (excluding\ battery)$ each year

$nEV(t) = \ new\ EV\ registrations$

$i_{EV(exc.LiB)} = Impact\ of\ manufacturing\ 1\ unit\ of\ EV\ (excluding\ battery)$

Total LIB manufacturing for new BEVs impact for each year:

$$I_{Lib_{manu}}(t) = \ mv \ \times nEV(t) \times \ i_{LiB_{manu}}(t) \quad \textbf{(64)}$$

Where,

$mv = mass\ of\ battery$

$i_{LiB_{manu}}(t) = for\ 1\ kg\ of\ LIB\ pack\ (incl. transport\ from\ EU)$

$I_{LiB_{manu}}(t) = Total\ impact\ of\ battery\ manufacturing$

Total EV use phase (incl. maintenance) impacts each year:

$$Dist_{EV(t)} = (EVp(t) \times Myp) + (EVt(t) \times Myt) \tag{65}$$

$$Service_{EV(t)} = \frac{ICEVp(t)}{Tp} + \frac{ICEVt(t)}{Tt} \tag{66}$$

$$I_{EV_{usephase}}(t) = \left(Service_{EV} \times\ i_{EV_{maint}}\right) + \left(Dist_{EV} \times\ i_{EV_{use}}\right) \tag{67}$$

Where,

$Dist_{EV(t)} =\ Distance\ travelled\ by\ all\ ICEVs\ on\ UK\ roads$

$Service_{ICEV(t)} =\ full\ service\ life\ eq. maintenance\ units\ for\ EVs$

$i_{ICEV_{maint}} =\ 1\ EV\ unit\ maintenance\ (excl. electricity\ input)$

$i_{ICEV_{use}} =\ 1\ km\ travelled\ by\ EV$

Total EV EoL (excluding battery) impact each year:

$$I_{EV_{eol}}(t) = eol_{EV(t)} \times \ i_{EV_{eol}} \tag{68}$$

Where,

$I_{EV_{eol}}(t) = \ EV\ EoL\ (excluding\ battery) impact\ each\ year$

$eol_{EV(t)} = number\ of\ EV\ decommissionings$

$i_{EV_{eol}} = \ EV\ end\ of\ life\ \ impact\ for\ 1\ unit\ of\ EV\ (excluding\ battery)$

Total Lithium-ion battery EoL impact each year:

$$I_{LiB_{eol}}(t) = \ mR(t) \times \ i_{cell_{rec}}(t) \times \ i_{rec_{LiB}(exc\ cell)}\ (t) \tag{69}$$

Where,

$mR(t) = \ LIBs\ entering\ recycling\ each\ year$

$i_{cell_{rec}}(t) = \ 1\ kg\ of\ input\ LiB\ pack\ to\ recycling$

$i_{rec_{Lib}(exc\ cell)}\ (t) = \ 1\ kg\ of\ input\ LiB\ pack\ to\ recycling\ excluding\ cells$

$I_{LiB_{eol}}(t) = \ LIB\ impact\ for\ each\ year$

Total ICEV EoL Credits from recycling:

$$C_{ICEV_{rec}}(\mathrm{t}) = \ eol_{ICEV(t)} \times \ c_{ICEV_{rec}} \tag{70}$$

Where,

$C_{ICEV_{rec}}(\mathrm{t}) = \ Total\ credit\ ICEV\ recycling\ for\ a\ given\ year$

$c_{ICEV_{rec}} = \ credits\ for\ recycling\ 1\ unit\ of\ ICEV$

Total EV EoL CREDITS (excl. LIB CREDITS):

$$C_{EV_{rec}}(\mathrm{t}) = \ eol_{EV(t)} \times \ c_{EV_{rec}} \tag{71}$$

Where,

$C_{EV_{rec}}(\mathrm{t}) = \ Total\ credit\ from\ EV\ recycling\ for\ a\ given\ year$

$c_{EV_{rec}} = \ credits\ for\ recycling\ 1\ unit\ of\ EV$

Total LIB EoL CREDITS:

$$C_{LiB_{rec}}(t) = (mR(t) \ \times \ c_{cell_{rec}}(t) \ \times \ c_{rec_{LiB}(exc\ cell)}\ (t)) + \ (M_{excess} \times \ c_M) \tag{72}$$

Where,

$C_{LiB_{rec}}(t) = \ Total\ credits\ from\ LiB\ recycling$

$c_{cell_{rec}}(t) = \ credits\ for\ 1\ kg\ of\ LiB\ pack\ to\ recycling\ of\ cells$

$c_{rec_{LiB}(exc\ cell)}\ (t) = \ credits$ for $1\ kg\ of\ input\ LiB\ pack\ to\ recycling\ excluding\ cells$

$c_M = \ credits\ for\ 1\ kg\ surplus\ active\ cathode\ metal\ recovered$

$M_{excess} = surplus\ demand\ for\ active\ cathode\ metal$

6.6 Life Cycle Impact Assessment (LCIA)

A range of environmental impact categories were also considered, namely: global warming potential (GWP), photochemical ozone creation potential (POCP), abiotic depletion potential (ADP), and human toxicity potential (HTP), to identify and discuss any potential trade-offs and impact shifting arising from the planned transition to electrical mobility. GWP, POCP and ADP were calculated using the widely adopted CML method (University of Leiden 2021). HTP was calculated using the USETox method (Hauschild et al. 2008; Rosenbaum et al. 2008), which is widely reputed to be the most sophisticated, up-to-date and accurate method to estimate potential toxicity impacts in LCA (UNEP-SETAC 2021). Details on GWP, ADP and HTP is discussed in chapter 5.

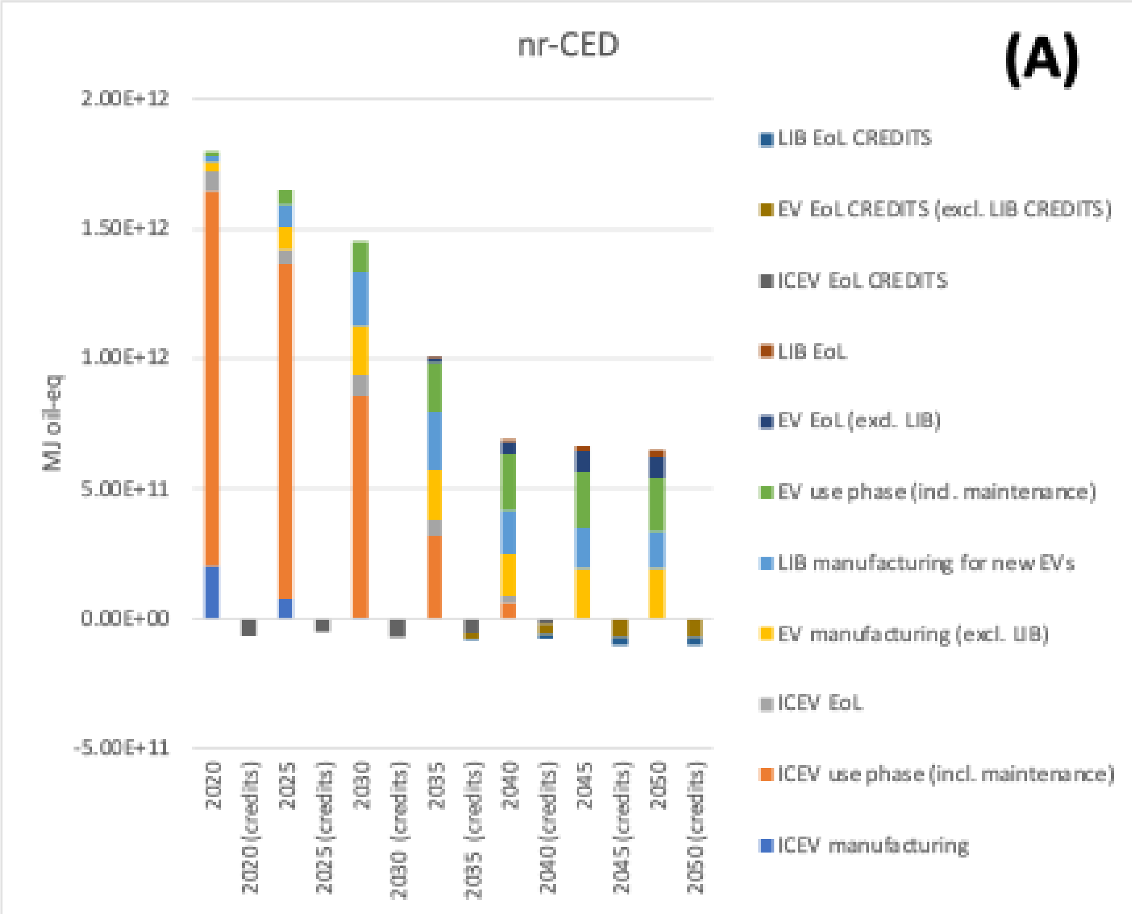
Photochemical ozone creation is a type of impact that takes place at the local/regional scale. Its effects on human health are more severe when the emissions take place in densely populated areas (such as in cities), as opposed to remote or primarily industrial locations (such as, for instance, at mineral mining sites or at metal processing and battery manufacturing facilities). Consequently, when calculating the POCP results, the decision was made to restrict the boundary of the analysis to the vehicle use phase only, to provide a clearer indication of the evolution of the LDV fleet's impact in terms of local air pollution and potential for photo-smog formation in urban centres and along motorways in the UK.

Finally, although not an LCIA indicator in the strict methodological sense, the non-renewable cumulative energy demand (nr-CED) is also reported, thereby providing an indication of the total non-renewable primary energy directly and indirectly harvested from the environment, expressed in units of crude oil equivalent (Frischknecht et al., 2015).

6.7 Results

Figures 6.5 and 6.6 respectively illustrate the projected UK LDV fleet's overall demand for non-renewable primary energy and greenhouse gas emissions. As expected, the trends for these two indicators are very similar, with a clear overall reduction of impact as Evs gradually displace ICEVs over time. The ICEV use phase initially represents the major contribution to CED, mainly due to the consumption of petrol and diesel; this then gradually decreases over time, as the number of ICEVs dwindles. In both scenarios, the total number of Evs surpasses the number of ICEVs on the roads after the year 2030. With the rise in Evs, there is a clear steady increase in nr-CED and GWP due to the EV manufacturing phase and use phase for the baseline scenario regardless of the UK grid mix evolution. This is mainly due to the increase in demand for number of vehicles. The same trends can be seen for LIB manufacturing (it is worth reminding that in the model, it is assumed that the batteries are manufactured outside the UK, where the grid mix is taken as a background process and is assumed to remain static over time). On a per-unit basis, the energy demand and carbon emissions associated with the manufacturing of new Evs (and specifically the battery packs) are significantly higher than those for ICEVs. However, this is overcompensated by the positive effect of low-carbon electricity replacing petrol and diesel as the energy carrier used to power the vehicles during their use phase, coupled with the intrinsically higher tank-to-wheel efficiency of electric *vs*. internal combustion power trains (typically, approximately 85% vs. 25%).

The improvement in nr-CED and GWP is even larger in the "TaaS" scenario because of an overall reduction in the total fleet size. Relative to the "baseline" scenario, there is thus a clear decrease in the impacts of the EV and LIB manufacturing stages, despite the fact that the shared mobility vehicles are replaced every 3 years. A further differentiator between the two scenarios is that in the "TaaS" scenario the batteries become available for recycling much earlier on.


nr-CED
(A)
MJ oil-eq
2.00E+12
1.50E+12
1.00E+12
5.00E+11
0.00E+00
-5.00E+11
2020
2020 (credits)
2025
2025 (credits)
2030
2030 (credits)
2035
2035 (credits)
2040
2040 (credits)
2045
2045 (credits)
2050
2050 (credits)
LIB EoL CREDITS
EV EoL CREDITS (excl. LIB CREDITS)
ICEV EoL CREDITS
LIB EoL
EV EoL (excl. LIB)
EV use phase (incl. maintenance)
LIB manufacturing for new EVs
EV manufacturing (excl. LIB)
ICEV EoL
ICEV use phase (incl. maintenance)
ICEV manufacturing

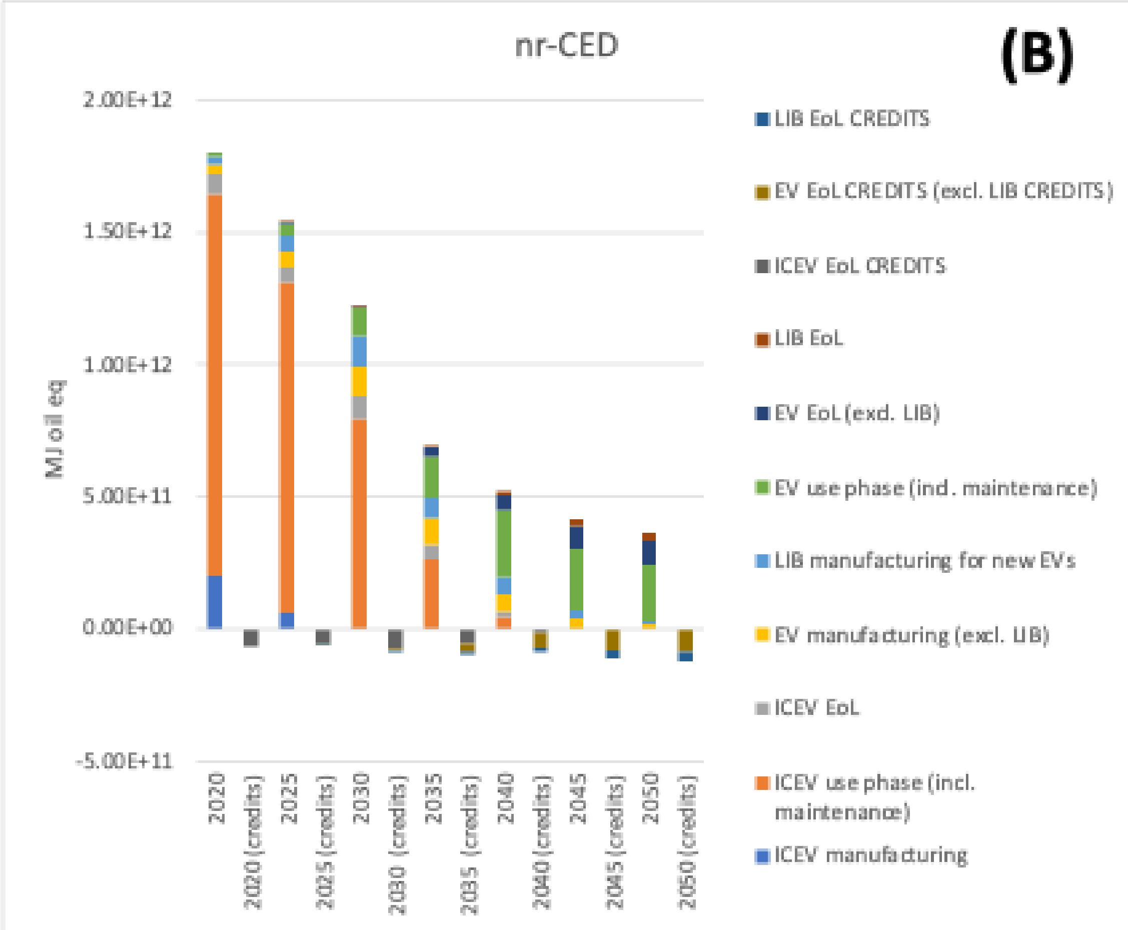
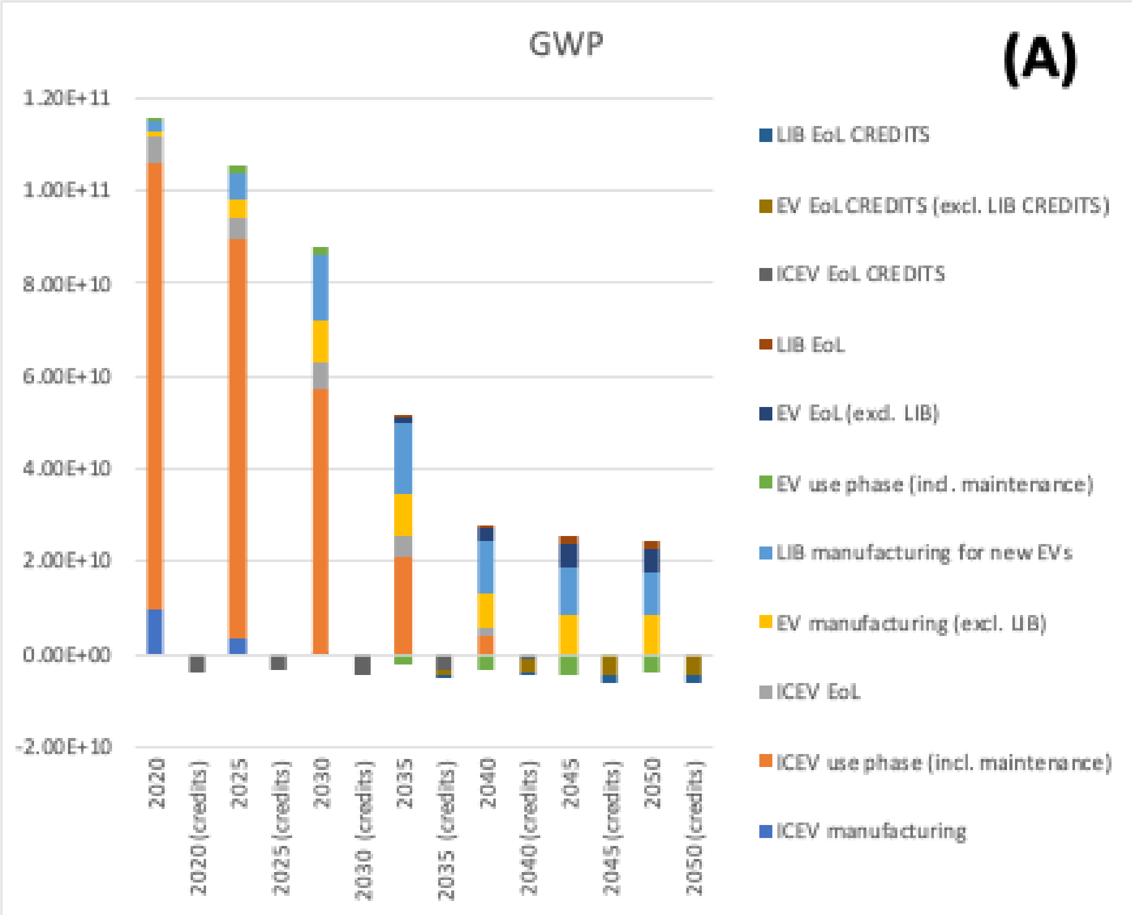


Figure 6.5: Non-renewable Cumulative Energy Demand (nr-CED) of the UK light-duty vehicle fleet. (A) = "Baseline" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (B) = "TaaS" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (C) comparison of overall net impact (= total impact – credit) for "Baseline" vs. "Taas" scenarios.



GWP
(A)
1.20E+11
1.00E+11
8.00E+10
6.00E+10
4.00E+10
2.00E+10
0.00E+00
-2.00E+10
2020
2020 (credits)
2025
2025 (credits)
2030
2030 (credits)
2035
2035 (credits)
2040
2040 (credits)
2045
2045 (credits)
2050
2050 (credits)
LIB EoL CREDITS
EV EoL CREDITS (excl. LIB CREDITS)
ICEV EoL CREDITS
LIB EoL
EV EoL (excl. LIB)
EV use phase (incl. maintenance)
LIB manufacturing for new EVs
EV manufacturing (excl. LIB)
ICEV EoL
ICEV use phase (incl. maintenance)
ICEV manufacturing

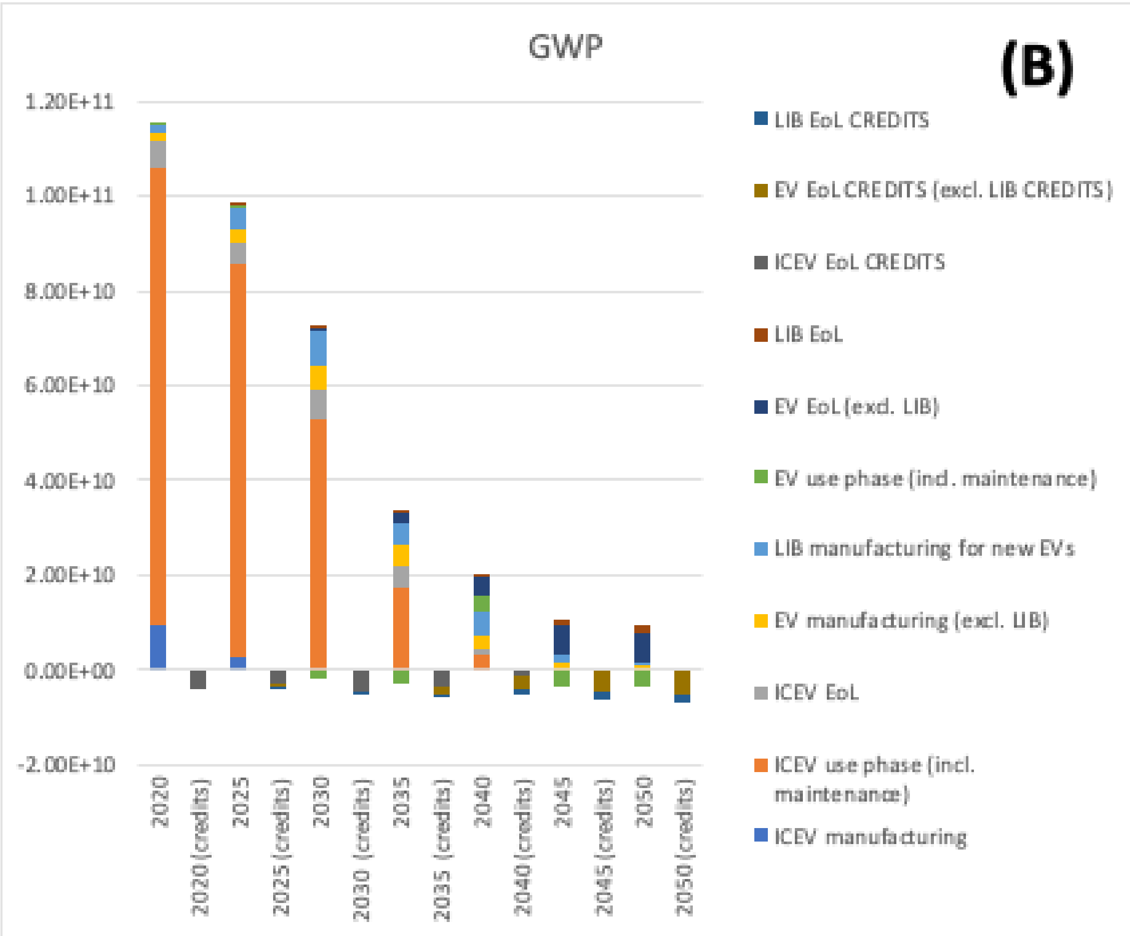
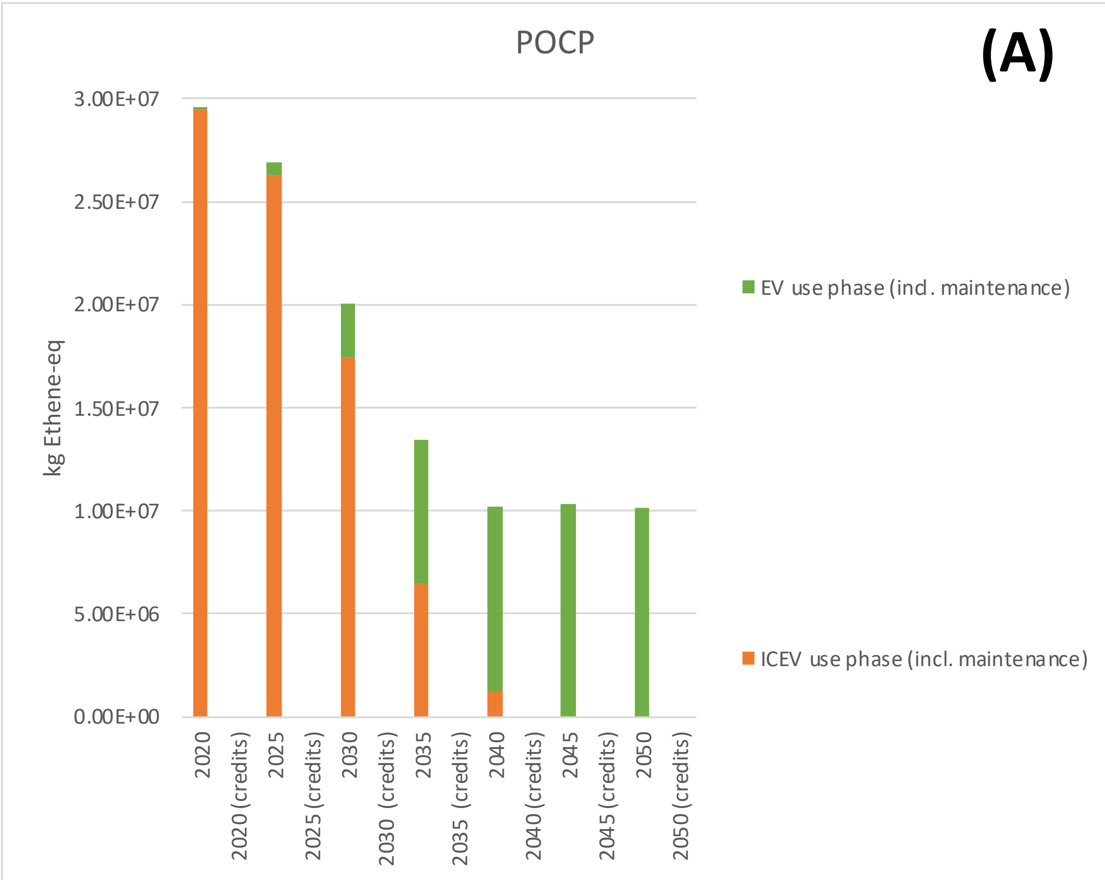


Figure 6.6: Global warming potential (GWP, excluding biogenic C) of the UK light-duty vehicle fleet. (A) = "Baseline" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (B) = "TaaS" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (C) comparison of overall net impact (= total impact – credit) for "Baseline" vs. "Taas" scenarios.

Moving on to the Photochemical Ozone Creation Potential (POCP) results for the use phase of the LVD fleet (Figure 6.7), a greater than 85% reduction in impact is found in both scenarios. This is a strong indication that the phasing out of internal combustion engines is a clear benefit in terms of local air pollution, and completely dominates over all other impacts resulting from non-tailpipe emissions, including those due to the provision of electricity to BEVs. It should be noted that the POCP indicator only captures the impacts arising from the photochemical oxidation of gaseous emissions (leading to secondary respiratory irritants such as ozone and peroxy-acyl nitrates). Thus, it may fail to highlight the contribution to local air pollution caused by particulate matter (PM) emissions, both from vehicle tailpipes and from tyre, brake-pad and tarmac wear

(Emissions Analytics 2020). However, diesel engines are known to be a major source of PMs which are considered to be similar to gasoline direct injection engines (Awad et al. 2020), and therefore phasing them out may be expected to be beneficial in this regard, too. Also, the regenerative braking systems on BEVs suggest that brake-pad emissions are going to be reduced (Society of Motor Manufacturers and Traders 2020).



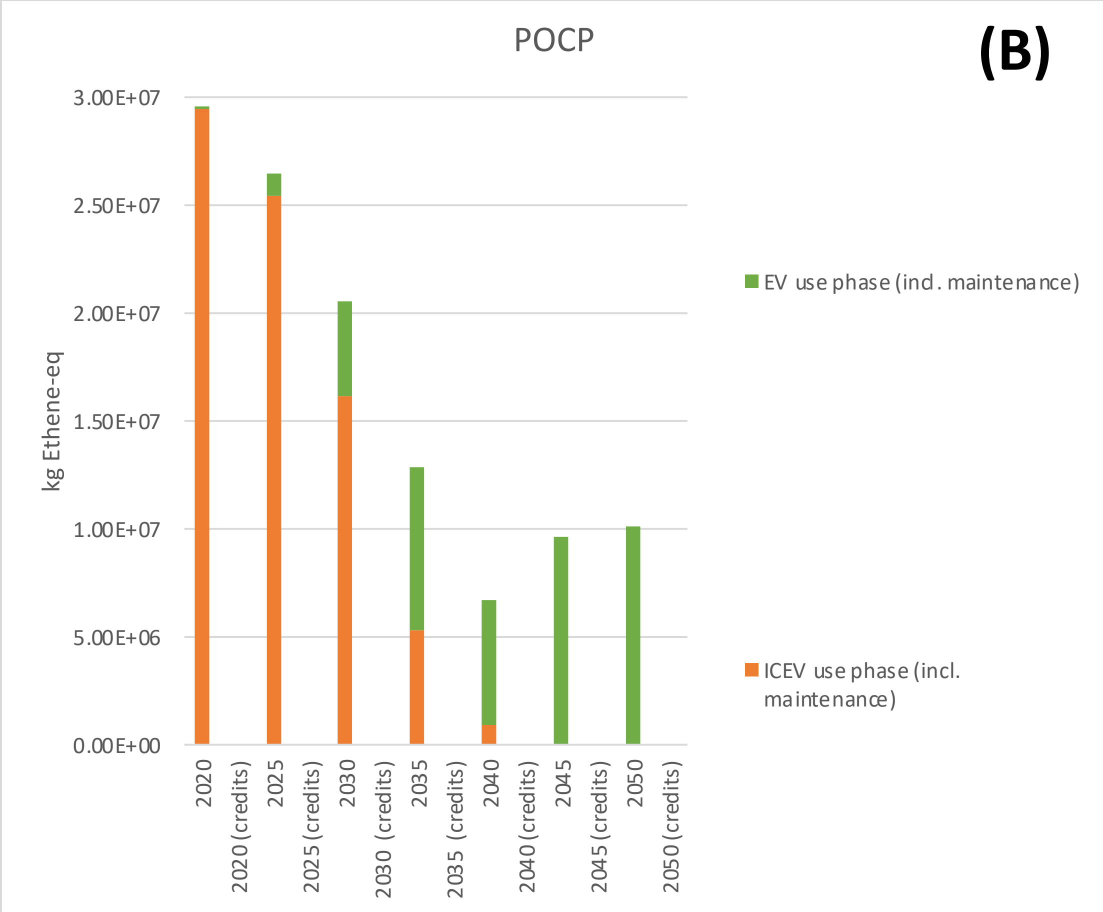


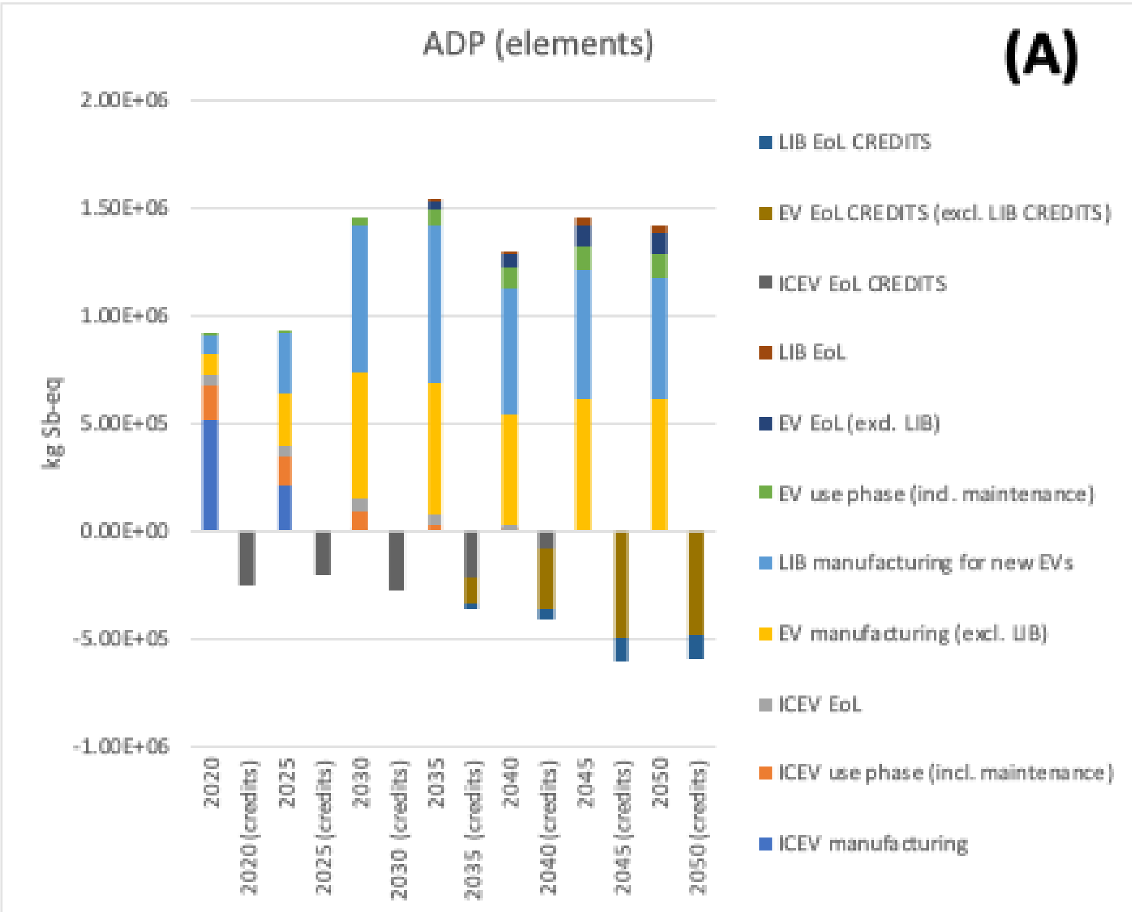
Figure 6.7: Photochemical ozone creation potential (POCP) of the UK light-duty vehicle fleet, use phase only (which exclude credit that might arise during the EoL phase). (A) = "Baseline" scenario, with total impact broken down by vehicle type (ICEV and EV); (B) = "TaaS" scenario, with total impact broken down by vehicle type (ICEV and EV); (C) comparison of overall impact (= ICEVs + BEVs) for "Baseline" vs. "Taas" scenarios.

Finally, Figures 6.8, 6.9 and 6.10 show that in the "Baseline" scenario, the demand for Li, Co, Ni, Mn and Cu for the EV power trains (including for the motor as well as for the LIBs) causes a progressive increase in ADP and HTP impacts, in spite of the positive role played by EoL recycling of the LIB packs. These findings point to a potentially critical trade-off between reduced energy and climate impacts on one side, and increased resource depletion and toxicity impacts on the other side. However, results for the "TaaS" scenario show that such trade-off could be resolved by the widespread adoption of shared mobility, whereby a smaller overall EV fleet could be sufficient to satisfy the growing demand for personal mobility. In so doing, the total net demand for critical metals, and the associated depletion and toxicity impacts, could be kept in check, leading to an overall reduction, rather than increase, in total ADP and HTP over the next three decades.

The increase in ADP values is mainly due to the larger use of critical and non-critical metals in EV and LIB manufacturing. In the “baseline” scenario, the slight decrease in the total ADP in year 2045 vs. the year 2040 is due to a reduced demand for new Evs in that particular year. For the “TaaS” scenario, instead, the number of new Evs peaks between year 2030 and 2035, after which there is a gradual decrease in the total number of new Evs each year due to the increase in shared mobility services. Such decrease in EV units in turn causes a decrease in battery production, and hence a significant decrease in the total ADP values 2035 onwards.

The human toxicity impacts are initially largely due to the manufacturing and decommissioning of ICEVs. Beyond 2030, in the “baseline” scenario, as the number of Evs start to increase significantly, the main share of the toxicity impacts is then shifted to the manufacturing of Evs and LIBs. In particular, the associated metal supply chains are responsible for greater toxicological impacts greater than the emissions from the generation of the electricity required during the vehicles’ use phase.

In the “TaaS” scenario, as the demand for new BEVs is reduced thanks to the increase in shared mobility services, the major contributor to HTP is the EoL phase of BEVs. Also, since in this scenario there is an oversupply of recycled LIB metals after the year 2035, this results in not only a reduction in the impact associated with the extraction of raw materials for new battery manufacturing, but also in additional net HTP credits.


ADP (elements)
(A)
2.00E+06
1.50E+06
1.00E+06
5.00E+05
0.00E+00
-5.00E+05
-1.00E+06
kg Sb-eq
2020
2020 (credits)
2025
2025 (credits)
2030
2030 (credits)
2035
2035 (credits)
2040
2040 (credits)
2045
2045 (credits)
2050
2050 (credits)
LIB EoL CREDITS
EV EoL CREDITS (excl. LIB CREDITS)
ICEV EoL CREDITS
LIB EoL
EV EoL (excl. LIB)
EV use phase (incl. maintenance)
LIB manufacturing for new EVs
EV manufacturing (excl. LIB)
ICEV EoL
ICEV use phase (incl. maintenance)
ICEV manufacturing

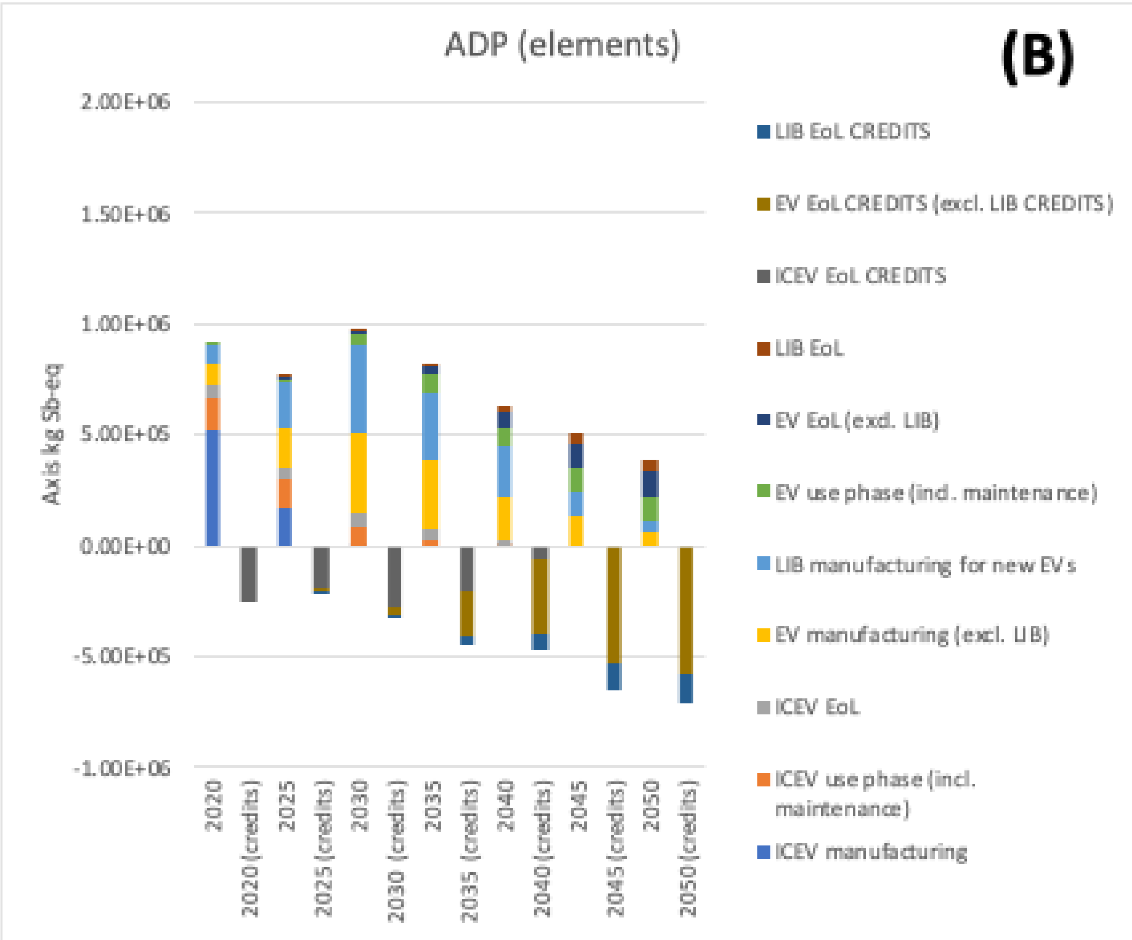
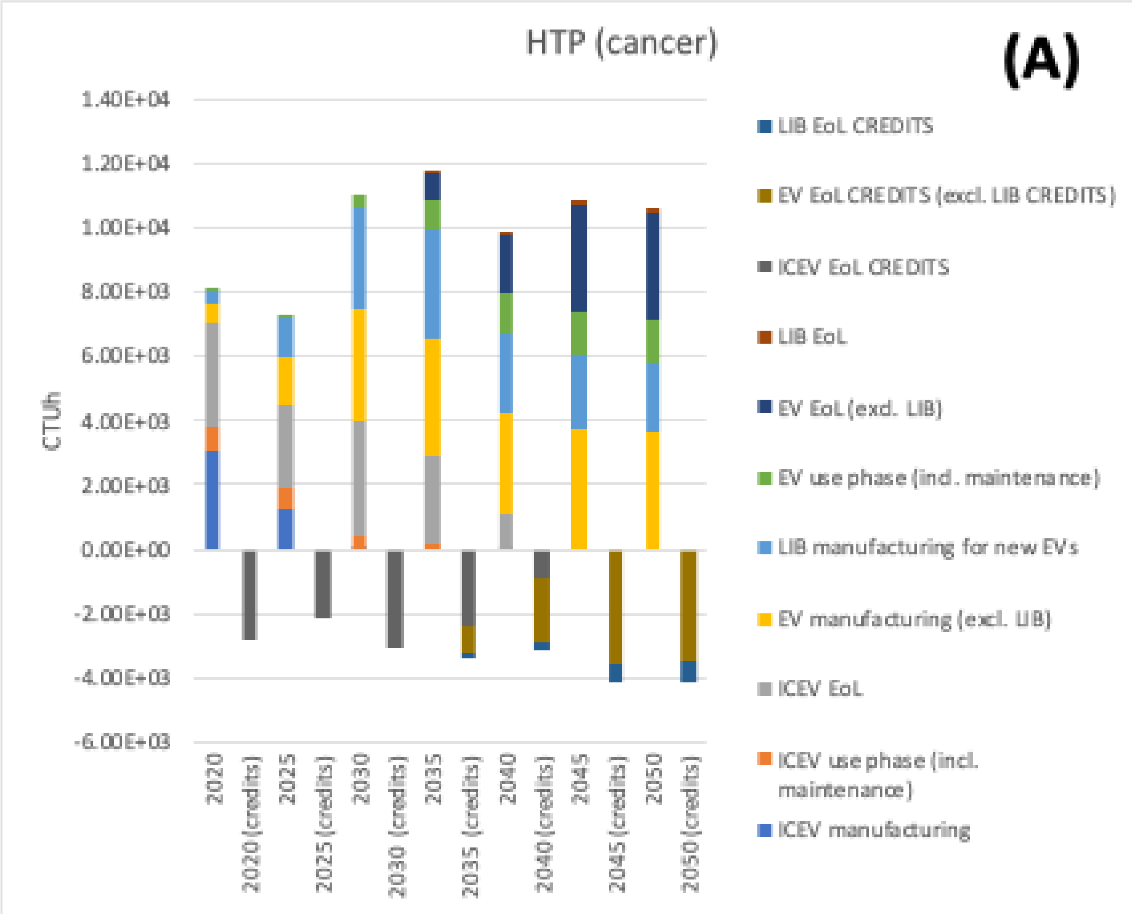


Figure 6.8: Abiotic depletion potential (ADP, elements) of the UK light-duty vehicle fleet. (A) = "Baseline" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (B) = "TaaS" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (C) comparison of overall net impact (= total impact – credit) for "Baseline" vs. "Taas" scenarios.



HTP (cancer)
(A)
1.40E+04
1.20E+04
1.00E+04
8.00E+03
6.00E+03
4.00E+03
2.00E+03
0.00E+00
-2.00E+03
-4.00E+03
-6.00E+03
CTUh
2020
2020 (credits)
2025
2025 (credits)
2030
2030 (credits)
2035
2035 (credits)
2040
2040 (credits)
2045
2045 (credits)
2050
2050 (credits)
LIB EoL CREDITS
EV EoL CREDITS (excl. LIB CREDITS)
ICEV EoL CREDITS
LIB EoL
EV EoL (excl. LIB)
EV use phase (incl. maintenance)
LIB manufacturing for new EVs
EV manufacturing (excl. LIB)
ICEV EoL
ICEV use phase (incl. maintenance)
ICEV manufacturing

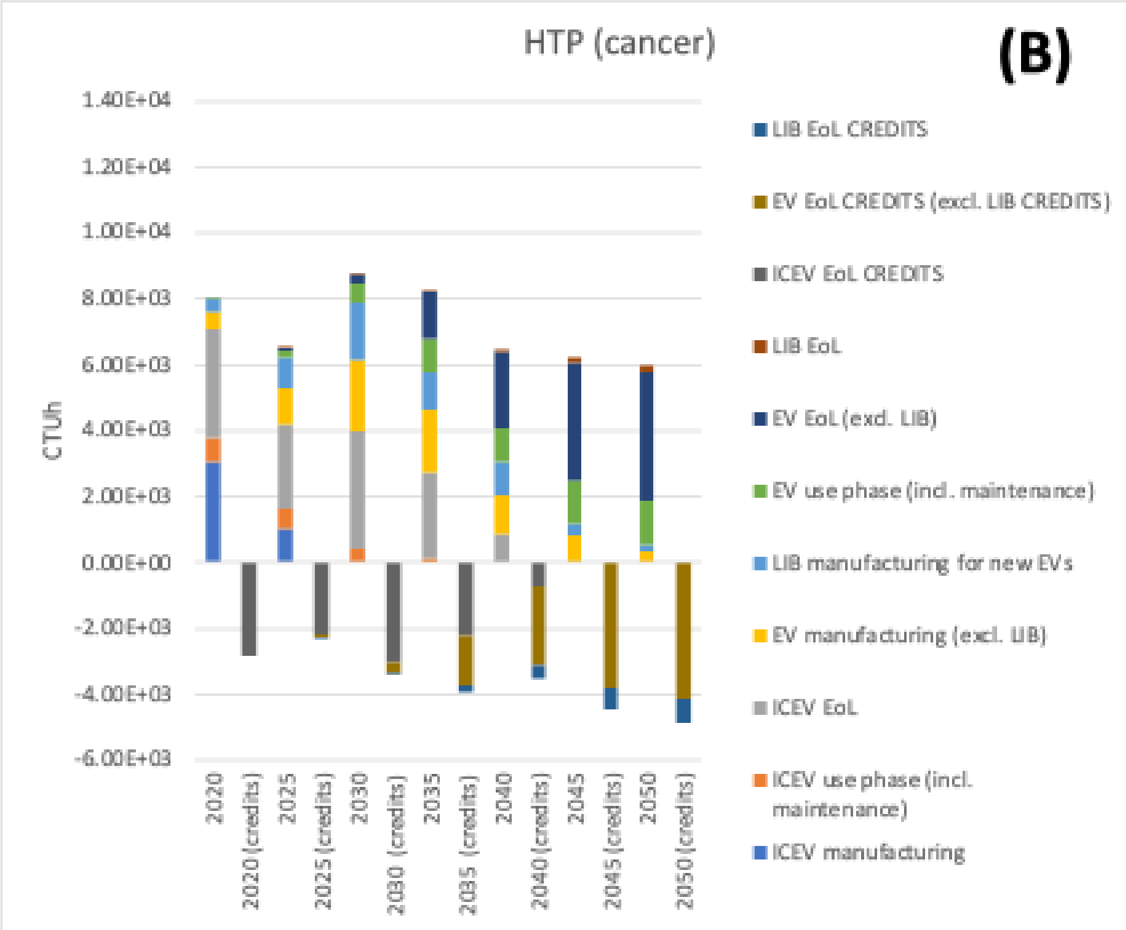
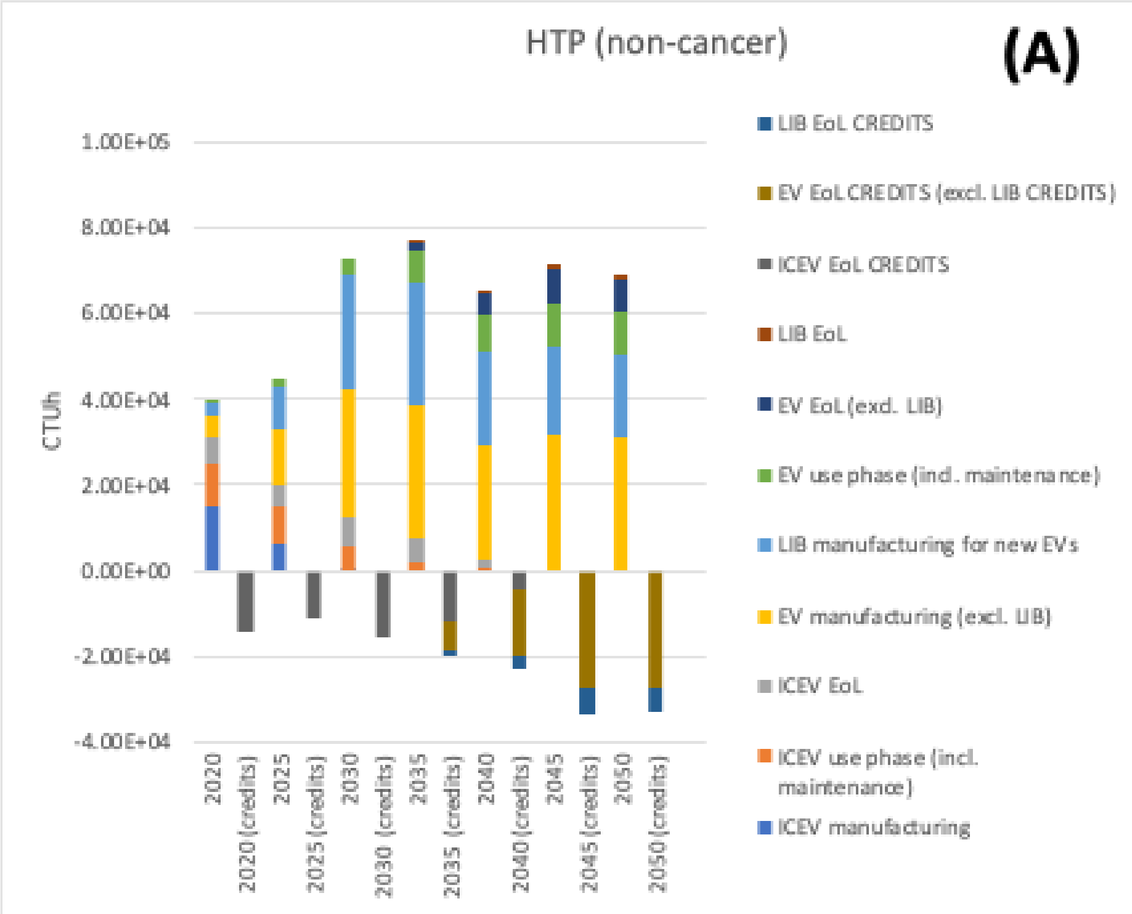


Figure 6.9: Human toxicity potential, cancer (HTP, cancer) of the UK light-duty vehicle fleet. (A) = "Baseline" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (B) = "TaaS" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (C) comparison of overall net impact (= total impact – credit) for "Baseline" vs. "Taas" scenarios.



HTP (non-cancer)
(A)
CTUh
1.00E+05
8.00E+04
6.00E+04
4.00E+04
2.00E+04
0.00E+00
-2.00E+04
-4.00E+04
2020
2020 (credits)
2025
2025 (credits)
2030
2030 (credits)
2035
2035 (credits)
2040
2040 (credits)
2045
2045 (credits)
2050
2050 (credits)
LIB EoL CREDITS
EV EoL CREDITS (excl. LIB CREDITS)
ICEV EoL CREDITS
LIB EoL
EV EoL (excl. LIB)
EV use phase (incl. maintenance)
LIB manufacturing for new EVs
EV manufacturing (excl. LIB)
ICEV EoL
ICEV use phase (incl. maintenance)
ICEV manufacturing

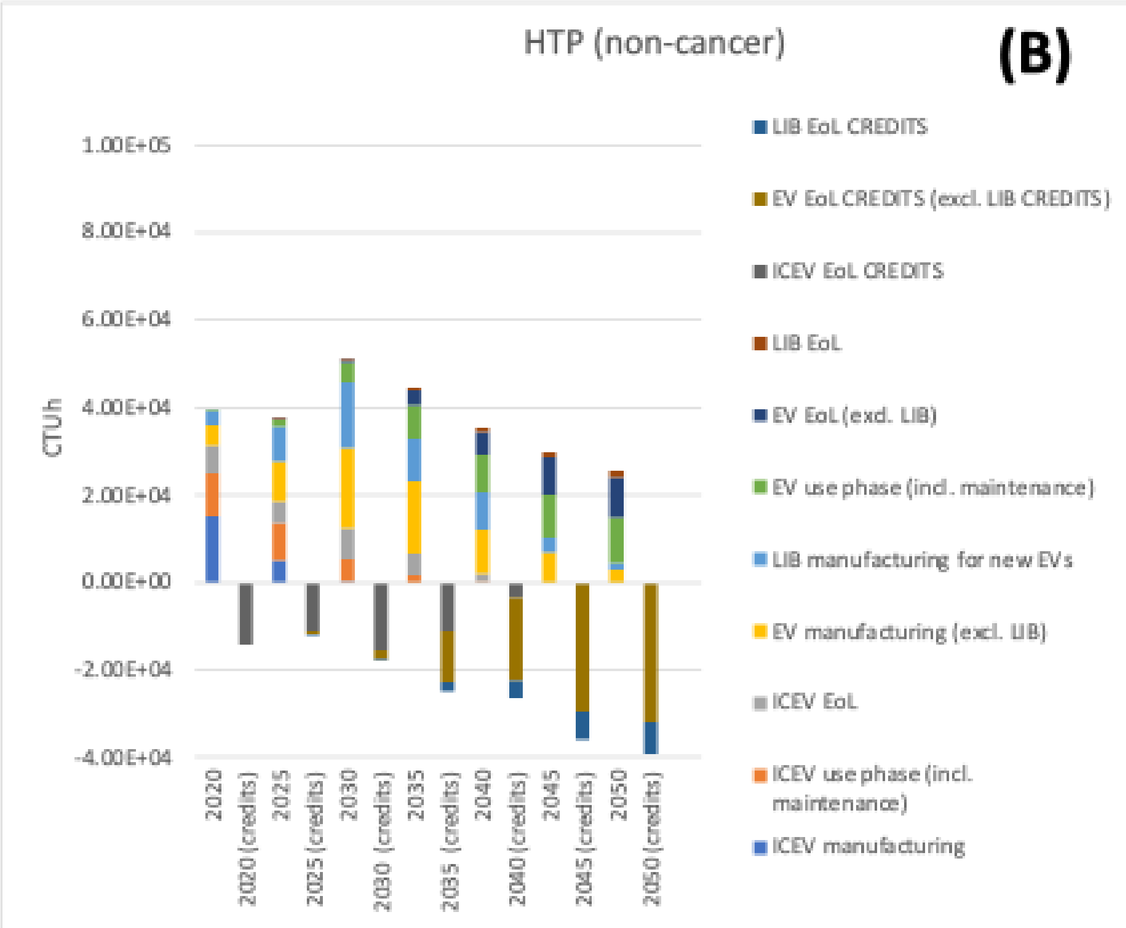


Figure 6.10: Human toxicity potential, non-cancer (HTP, non-cancer) of the UK light-duty vehicle fleet. (A) = "Baseline" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (B) = "TaaS" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (C) comparison of overall net impact (= total impact – credit) for "Baseline" vs. "Taas" scenarios.

7 Discussion

Battery electric vehicles (BEVs) coupled with low carbon electricity grid play a vital role in supporting the low carbon transition of transport operation. This requires not only the transformation and expansion electricity system to enable the mass uptake of BEVs, but also developing a circular strategy to secure critical materials for battery supply chains, with the aim of making them 95% recyclable by 2035 (DTF 2021). Additionally, there is a need to increase average road vehicle occupancy by 2030 by integrating transport as a service (TaaS) (DTF 2021). This Chapter reiterates the research questions highlighted Chapter 1 followed by a discussion of this thesis' findings in addressing each question. Lastly, this Chapter discusses the limitation of the study.

1. **The overall supply challenges of critical raw materials required in the transition to Evs and low carbon electricity grid mix.**

Research question 1 was addressed by carrying out a systematic review which focuses on identifying current challenges, proposed solutions and research gaps in the existing literature related to the supply of critical raw materials used in energy transition technologies. Four aspects are considered: (1) global raw material availability (2) geopolitical and regional considerations (3) environmental impact and (4) social considerations. This discussion brings together work carried out in different areas with regard to supply issues of raw materials in section 7.1.

2. **Does reusing retired EV batteries in second-life grid storage delay when battery raw materials become available from recycling? What implication may this have on the raw material requirement for electric mobility batteries?**

Research question 2 was addressed by carrying out a dynamic material flow analysis (MFA) up to year 2050 to track and quantify EV battery and their raw materials requirement in the combined evolving electricity grid and transport sectors, in particularly the battery cathode material due to its criticality. Two case scenarios were considered respectively, with and without battery second life and recycling to understand the implication on raw material requirement for electric mobility for the case of UK passenger vehicle fleet. This discussion is addressed in section 7.2.

3. **When considering the effect of minimizing the use of private vehicles through uptake of TaaS, what implication may this have on the raw material requirement for BEV batteries and battery recycling opportunities?**

Research question 3 was addressed by means of a similar methodological framework to research question 2, with the addition of transport as a service (TaaS). This is characterised by a gradual shift from traditional private vehicle ownership to shared mobility schemes. This is discussed in section 7.3.

4. **What is the overall energy and environmental trade-off of the transition to BEVs?**

Research question 4 was addressed by carrying out a holistic prospective life cycle assessment to understand and quantify the energy and environmental trade-off for the transition of ICEVs to Evs for the passenger vehicle fleet through to year 2050, whilst simultaneously considering the various changes occurring within the transport and electricity grid system, the evolution of lithium-ion battery chemistry, electricity grid mix and uptake of shared mobility. Two cases scenarios were considered, drawing up from research question 2 and 3, and discussed in section 7.4.

7.1 Systematic Review of Critical Elements

Overall, this systematic review has indicated that most critical elements have the potential to meet the demands of the transition to a global low-carbon energy system, but doing so requires considerable efforts to address supply concerns and a careful, strategic planning of the mix of energy technologies to be deployed. For instance, based on the findings of this review, it does not appear likely that silver will represent a significant constraint to the growth of c-Si PV; however, it is acknowledged that there is still significant uncertainty on this particular point, which primarily stems from the wide range of projections on future PV growth overall. Conversely, the competing demand of indium and selenium will probably hamper the large-scale uptake of Copper-indium-gallium-selenide (CIGS) PV. Instead, the indium requirement for battery electric vehicles (BEVs) and nuclear power plants is unlikely to be an issue, as these technologies only require very small amounts of this critical element. Gallium is also used in small quantities in BEVs, and moreover there is potential to expand bauxite refinery for gallium production. For offshore wind and electric motor technologies, there should be significant efforts to reduce the dysprosium content and increase circularity in the permanent magnets market. Significant improvements will also need to be made in general for rare earth elements (REEs) in terms of environmental safety regulation, the lack of which has been shown to hinder further investments in their supply chains.

To support the mass transition to BEVs, on-going improvements will need to continue in reducing or eliminating the cobalt content in batteries and improve circularity for both lithium and cobalt. This is where developments in future battery chemistries that use more abundant materials, like lithium iron phosphate and sodium ion formulations, may be significant. There is also a growing consensus in the literature to recommend shifting light duty ICEVs to BEVs first, followed by heavy duty internal combustion engine vehicles (ICEVs) to fuel-cell electric vehicles (FCEVs), to reduce long term supply risk and meet most of the early demand for platinum through EoL ICEVs. Improvements also need to be made on reducing significant losses of valuable electrical materials in the EoL collection of vehicles and improving recycling, especially for PGMs in developing nations. Indeed, the co-location of battery recycling facilities with battery manufacturing would significantly enhance the potential for recovery of valuable materials for re-use. The mining industry requires vast investments, and producing regions are likely to focus on maximizing economic gains, which is prone to lead to both social injustice and lack of enforcement of environmental regulations, both of which can be seen currently in the case of PGMs, cobalt and lithium. These pressures, coupled with the political instability in those regions where extraction is concentrated, conspire to make the supply of elements for BEVs and FCEVs more vulnerable to disturbance. Therefore, social and environmental impacts need to be made a primary focus of attention to ensure a reliable and sustainable supply of these critical elements, as well as to avoid creating new impacts in the pursuit of reducing GHG emissions.

7.2 Battery Material Impact of Recycling and Second-Life of BEV Batteries

The findings point out the collection and recycling of end-of-life (EoL) BEV battery packs is not only critical in pursuing resource circularity in the supply chain for lithium-ion batteries, but also very effective in achieving such a goal within the next three decades in the UK. This has been shown by a well-integrated and internally consistent dynamic material flow analysis of a tightly coupled light duty vehicle and electricity sectors.

Results point out repurposing EoL BEV batteries for grid storage applications is clearly a recommendable strategy, in that it allows dramatically curbing the requirement for purpose-built battery storage deployment, while only marginally affecting the demand for virgin materials for the EV sector. This is due to a combination of: (i) the order-of-magnitude larger expected throughput of battery storage capacity in the transport vs. the electricity sectors, and (ii) the relatively short time lag assumed as 5 years imposed by second life on the eventual recovery of the battery metals by recycling. Assuming priority is given to the second-life reuse of retired BEV batteries for grid energy storage (i.e., EoL BEV batteries are primarily directed to meet grid storage demand), it is estimated that 28%, 21%, and 10% of retired EV batteries will be utilized in grid storage applications in 2035, 2040, and 2050, respectively (excluding uncollected EoL BEV LIBs). This contribution would fulfil the total projected demand for grid battery storage in those years.

The decreasing share of end-of-life BEV batteries used in grid energy storage as grid demand rises is primarily due to the significant increase in retired BEV batteries, which is projected to grow by a factor of three from 2035 to 2050 (i.e., 169,000 batteries in 2035 vs. 683,000 in 2050). As more end-of-life BEV LIBs become available, the grid storage requirement increases only marginally in comparison (including the battery replacement and additions). Whilst the grid battery storage figures are based on National Grid 2020 most ambitious FES scenario (leading the way), the current storage requirement in the most recent 2023 scenario is not too far off (i.e., 33.6 GWh vs 42 GWh in 2035; 50.7 GWh vs 49 GWh in 2040; 56.2 GWh vs 63 GWh in 2050).

Figure 7.1 shows the net demand for BEV LIB cathode materials without and with repurposing for grid storage and recycling EoL BEV LIB. The net demand for battery materials (lithium, nickel, cobalt, manganese) for the adoption of EV in the UK sees an 83% to 86% decrease in virgin metal in 2050 when considering battery recycling and reuse (i.e., 22.2 kilotonnes to 3.9 kilotonnes for lithium, 18.7 kilotonnes to 3.0 kilotonnes for cobalt). The most significant decrease is seen for the nickel demand (i.e., 148.5 kilotonnes to 21.5 kilotonnes), which is at least nearly 7 times higher compared to lithium, cobalt and manganese.

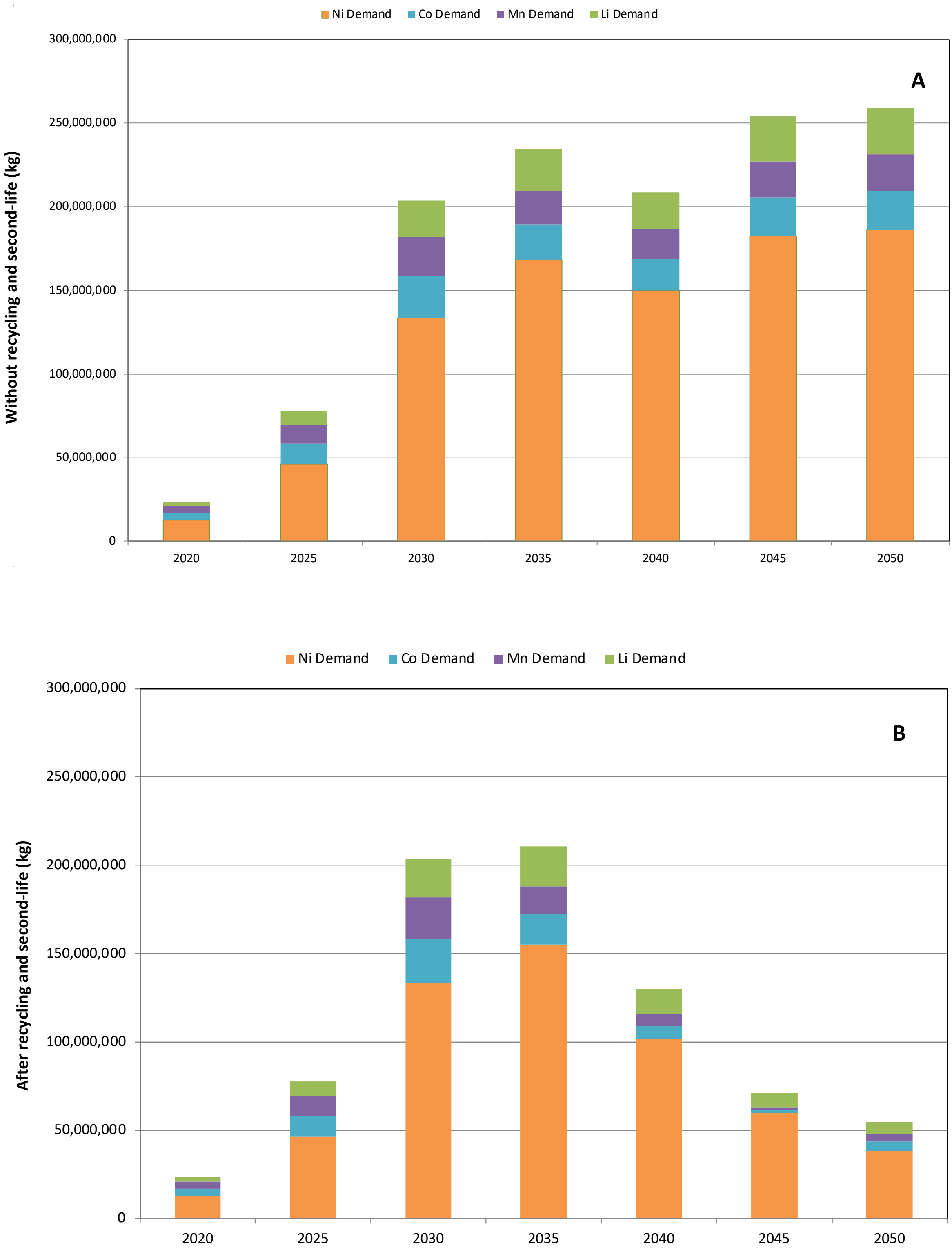


Figure 7.1: Lithium, nickel, manganese and cobalt net demand for passenger BEV fleet without (A) and with (B) recycling and repurposing EoL BEV LIBs. (80-99% linear increase in collection rate by 2050).

In terms of limitation, the lifetime of EV batteries and number of BEVs on road plays a significant role as to when batteries are available for recycling. In this case study a 14-year assumption was taken for the lifetime, a 2030 ban on new ICEVs was assumed, and the number of vehicles was dictated by the Department of Transport forecast model, which indicates the effect of recycling as significant from 2040 onwards, when more than half (i.e., 59%) of EoL EV batteries meet the requirement for new BEVs in the given year. In reality, this may not truly be the case, as the lifetime of BEVs and the overall number of vehicles on the roads are dependent on various factors and can be somewhat unpredictable, and hence marginally impact the quantities of EV batteries required and available for recycling.

Nevertheless, this does not take away the main findings of this thesis on the sheer quantities of BEV batteries required and importance of supporting large scale closed-loop recycling of BEV batteries to limit the demand for raw material. Furthermore, repurposing BEV batteries for grid applications is undoubtedly shown to be a viable option in terms of material flows. However, there are still barriers that would need to be overcome to support the reuse of EV batteries at scale, such as design to disassembly to lessen cost regarding the testing and repurposing EoL EV batteries.

7.3 Battery Material Impact of Shared Mobility

A projected large shift in behaviour change, from the conventional vehicle ownership model to a shared mobility model with a focus on successful adoption of transport as a service, could further reduce the demand for virgin battery metals, to the point where the UK could actually become a net supplier of lithium, cobalt, manganese and nickel to the battery industry by 2040. This is mainly due to the service amount covered by shared mobility vehicles displacing significant number of private vehicles on the road despite shorter 3 years lifetime of the TaaS vehicle, as shown in Figure 7.2.

Figure 7.2: Number of vehicles registered (A) and reaching EoL (B) for Baseline (steady increase in number of vehicles overtime) and TaaS scenarios (uptake of TaaS vehicles, leading to a reduction of private vehicles required to meet mobility needs)

The number of total vehicles on UK road reduces by 39% by 2050, when assuming 0% to 45% linear increase in the percentage of new vehicles to be considered TaaS vehicles from 2020 to 2050. The EoL vehicles in TaaS scenario are slightly high compared to Baseline, this is mainly due to the shorter lifetime, mainly due to the shorter lifespan. Nevertheless, due to high mileage covered by TaaS vehicles and the increase in percentage of vehicles considered to be TaaS vehicles, overall fewer vehicles are required on the UK road. This drastically reduces the number of LIBs required in TaaS scenario (Figure 7.3).

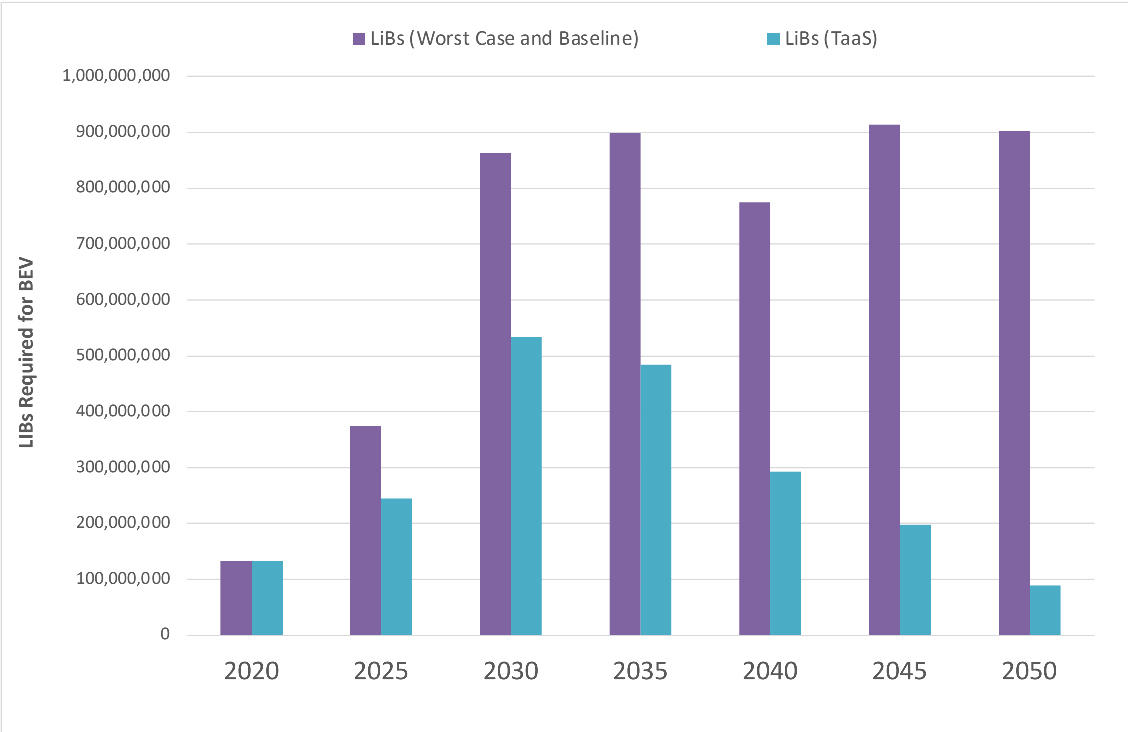


Figure 7.3: Number of batteries required for Worst-case (no collection of EoL batteries), Baseline (collection of EoL batteries, second use in grid service and closed loop recycling of active cathode material) and TaaS scenarios (same as baseline scenario with the addition of meeting share of passenger mobility needs through TaaS vehicles).

Due to the constant decrease in number of vehicles on road, BEV batteries from recycling outnumber requirement for new BEV batteries, hence, this leads to an oversupply of recycled metals 2035 onwards, as shown in Figure 7.4. This oversupply of recycled metals cannot then be expected to last indefinitely, and supply and demand would eventually balance out again in the farther future, when the penetration of TaaS reaches a plateau and the overall vehicle fleet size stabilizes again. However, attempting to draft quantitative scenarios that extend significantly beyond three decades is fraught with increasingly large uncertainties, not only about vehicle use trends, but also about Other potential energy storage technologies that might become commercially viable and partly displace LIBs as the preferred option. Finally, further degrees of uncertainty are also tied to potential future exports of recycled LIB metals and to the possible mass penetration of LIBs into other markets beyond the transport sector; however, these considerations fall outside of the scope of this thesis and do not detract from the general validity of the results presented here, in terms of the effectiveness of recycling and shared mobility.

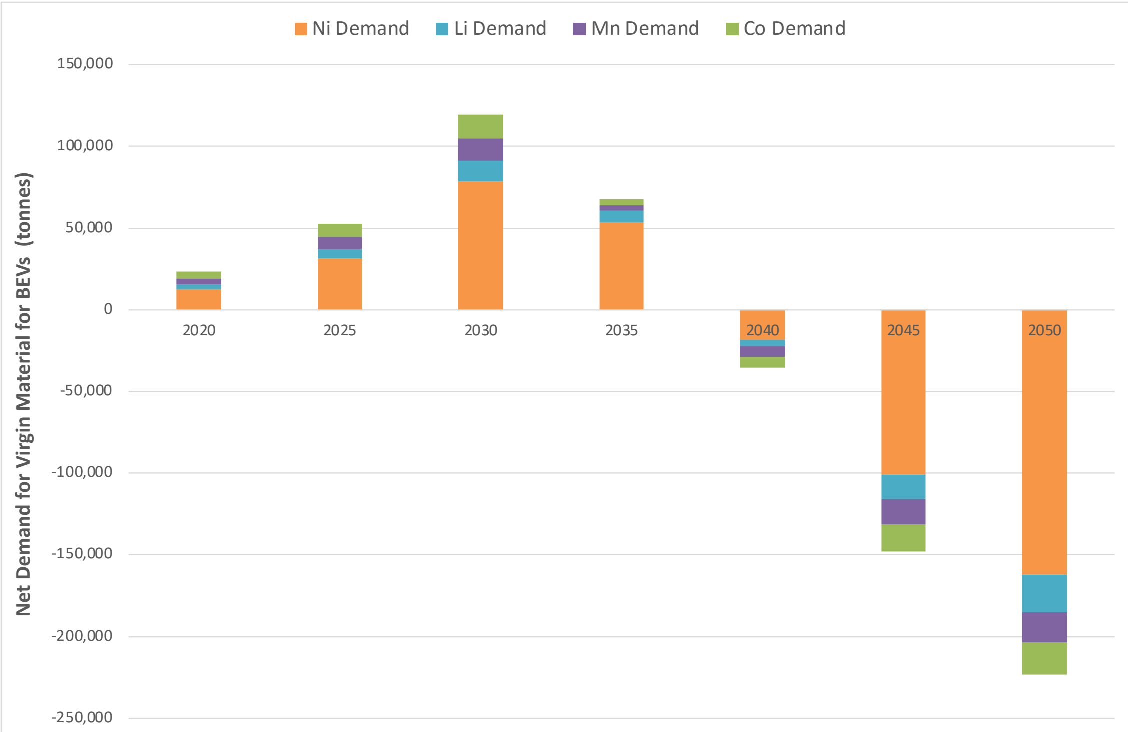


Figure 7.4. Lithium, nickel, manganese and cobalt net demand for passenger BEV fleet for TaaS scenario. (80-99% linear increase in collection rate by 2050).

7.4 Environmental Trade-off

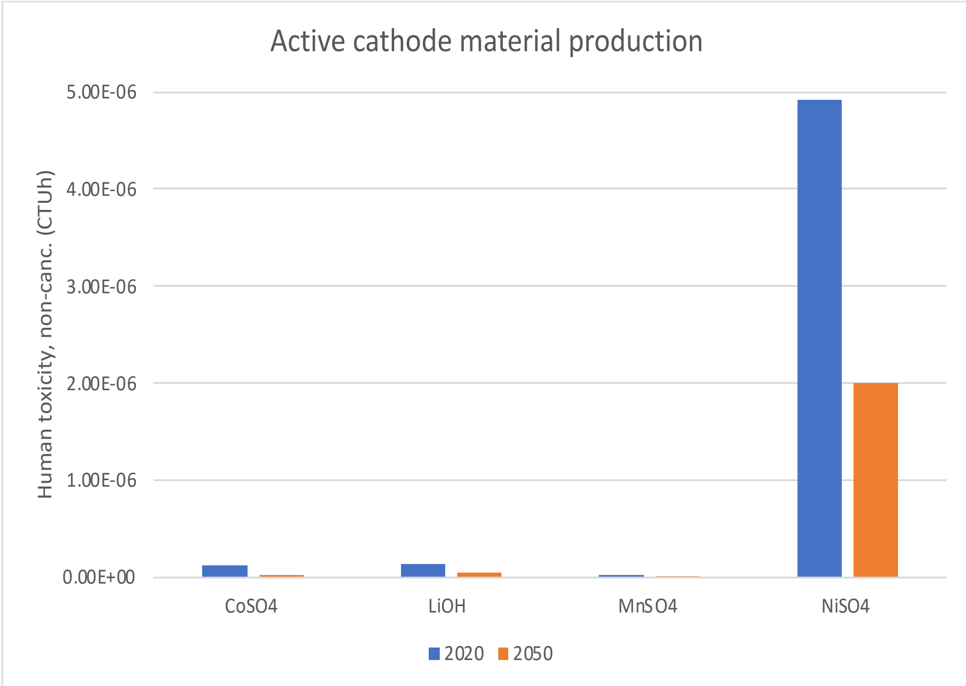
The holistic prospective life cycle assessment of the future of the whole light duty vehicle fleet in the UK has shown positive beneficial effects in terms of reducing the overall demand for non-renewable primary energy sources, curbing greenhouse gas emissions (in large part due to a parallel effort to aggressively decarbonize the grid mix) and reducing overall local air pollution in cities and along busy roads.

However, the consequence of introducing new electric power trains at fleet scale also leads to a sharp rise in demand for metals such as copper, lithium, cobalt, nickel and manganese. Even with large-scale implementation of end-of-life battery take-back and recycling of active cathode material, these risks counterbalance the aforementioned positive effects with significant worsening rates of material resource depletion and human toxicity (the latter primarily occurring overseas along the metal supply chains, which raises issues of environmental justice).

As batteries are expected to move towards higher nickel content (NMC 622 to NMC 811), nickel requirement for the passenger vehicle fleet is responsible for the major contribution of resource depletion and human toxicity, this is more than halved by 2050 (Figure 7.5) despite the increase in number of BEVs and nickel content in the battery pack, when a closed loop recycling pathway is considered for these metals.

Recycling of active material plays an important role in limiting the resource depletion and human toxicity emissions, and in securing the critical active materials required for battery production, especially so when battery closed-loop recycling starts to play a major role beyond year 2035, supplying more than half of the raw material demand for new BEV batteries. However, the rise in resource depletion and human toxicity before significant raw materials become available for recycling cannot be resolved by a battery circularity strategy alone.

Increasing average road vehicle occupancy by introducing TaaS vehicles, has been shown to be a more effective strategy to reverse these challenging environmental trends in resource depletion and human toxicity. If TaaS vehicle were to linearly increase to 45% of all new registered vehicle by 2050, this could lead to significant shift in some of the environmental trends. This is a significant reduction in abiotic depletion potential (ADP) and Human Toxicity Potential (HTP), mainly contributed by the overall reduction in battery and BEV manufacturing and the EoL credits gained from BEVs that have reached their EoL. Figure 7.6 shows the overall abiotic depletion potential for the passenger light duty vehicle fleet up to the year 2050.



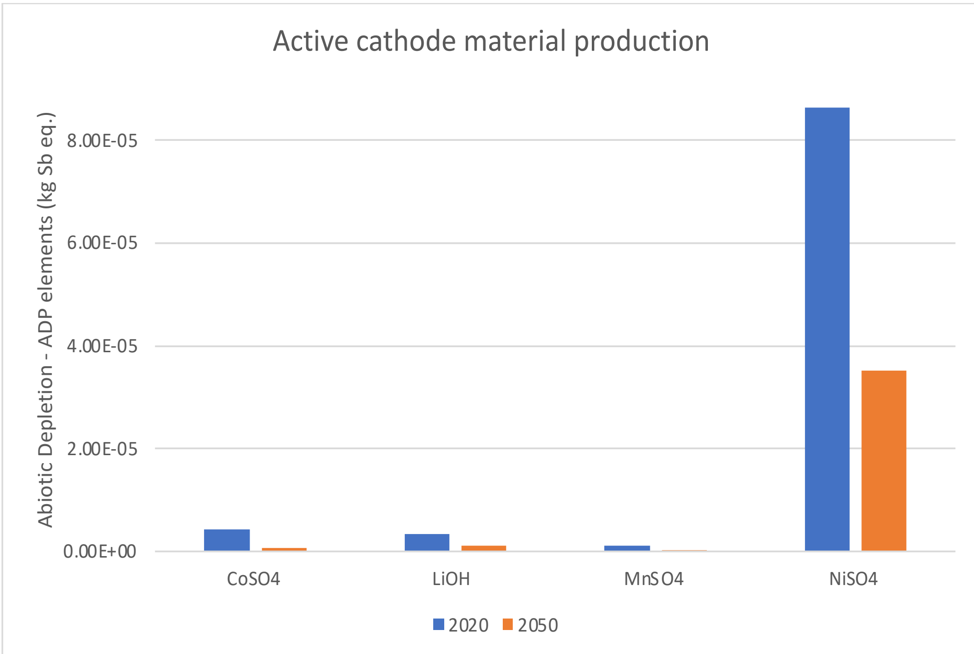


Figure 7.5 Human toxicity (non-cancerous) and abiotic depletion potential of active cathode material production for NMC LIB for the baseline scenario (steadily increasing EoL BEV LIB collection rates and subsequent second life and closed -loop recycling of active cathode material).

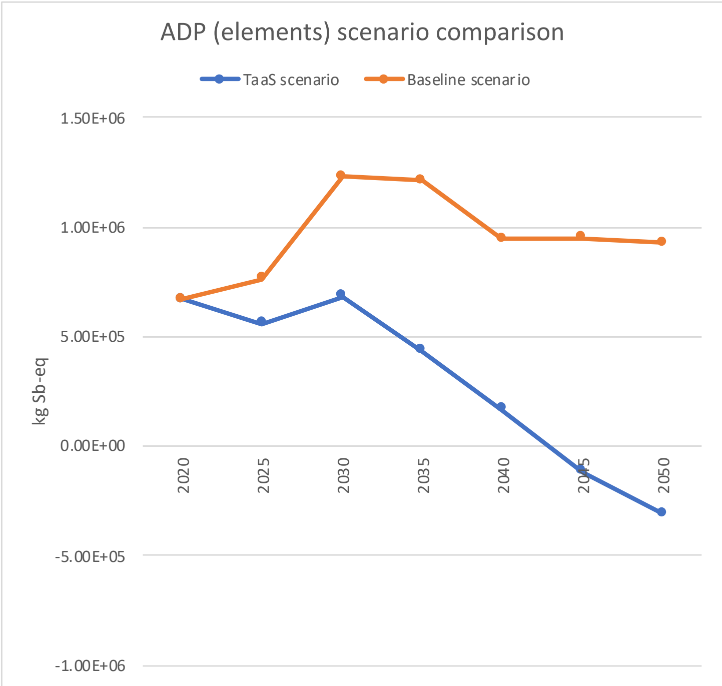


Figure 7.6: Shows the overall abiotic depletion potential for the passenger light duty vehicle fleet up to the year 2050 ("Baseline" Scenario: orange line Vs "TaaS" scenario: blue line).

In the case of ADP, the most significant impact was due to LIB manufacturing, due to the use of significant critical materials. In the case of HTP (Figure 7.7), the impacts are mainly due to the production of both the BEVs themselves, and the LIBs. These also see a significant reduction to the reduction in overall passenger fleet size.

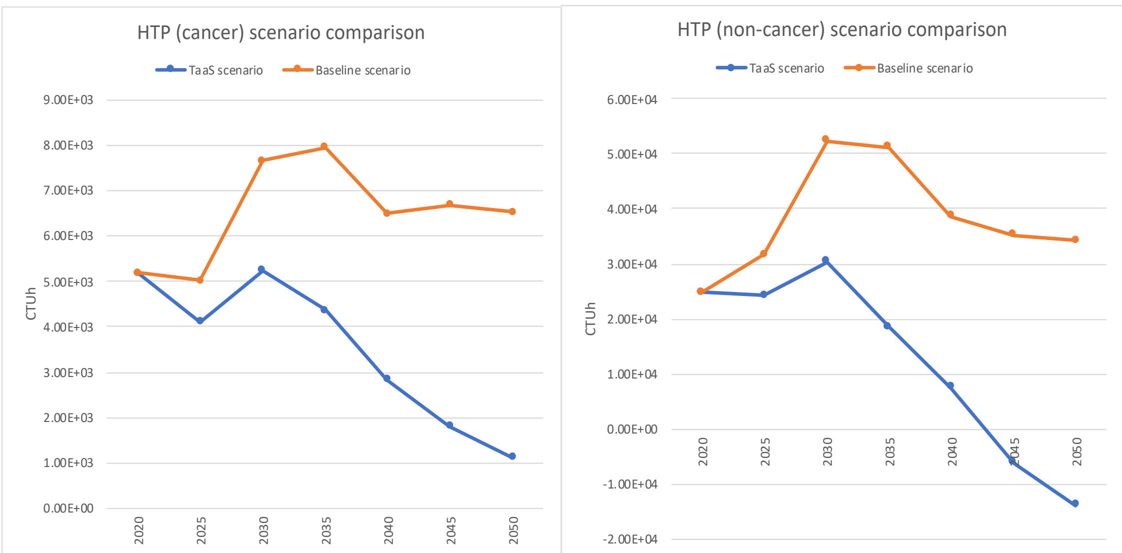


Figure 7.7: Shows the overall human toxicity potential for the passenger light duty vehicle fleet up to the year 2050 ("Baseline" Scenario: orange line Vs "TaaS" scenario: blue line).

In the case of non-renewable energy demand and global warming potential (Figure 7.8), both the scenarios indicate a positive result, as major impacts for both indicators are due to the operation of internal combustion engine vehicles (ICEVs), followed by the manufacturing of LIBs.

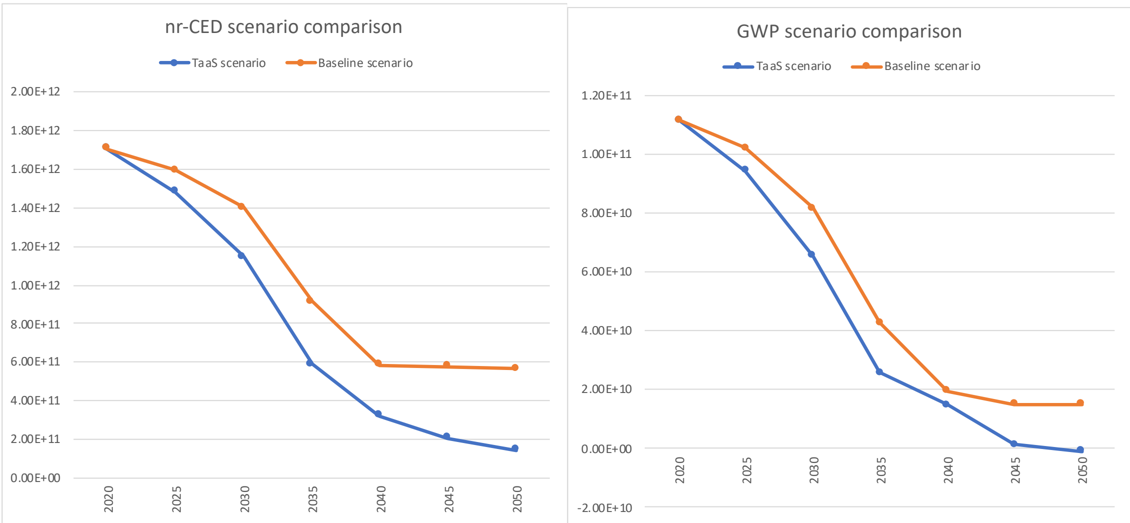


Figure 7.8: Shows the overall non-renewable cumulative energy demand and Global warming potential for the passenger light duty vehicle fleet up to the year 2050.

In the case of Photochemical Ozone Creation Potential (POCP), the results are discussed for the use-phase alone (Figure 7.9), to provide a clearer indication of local air pollution and potential for photo-smog formation in urban centres and along motorways in the UK. The results of POCP are similar for both the scenario as the main benefits are mainly due to the phaseing-out of ICEVs.

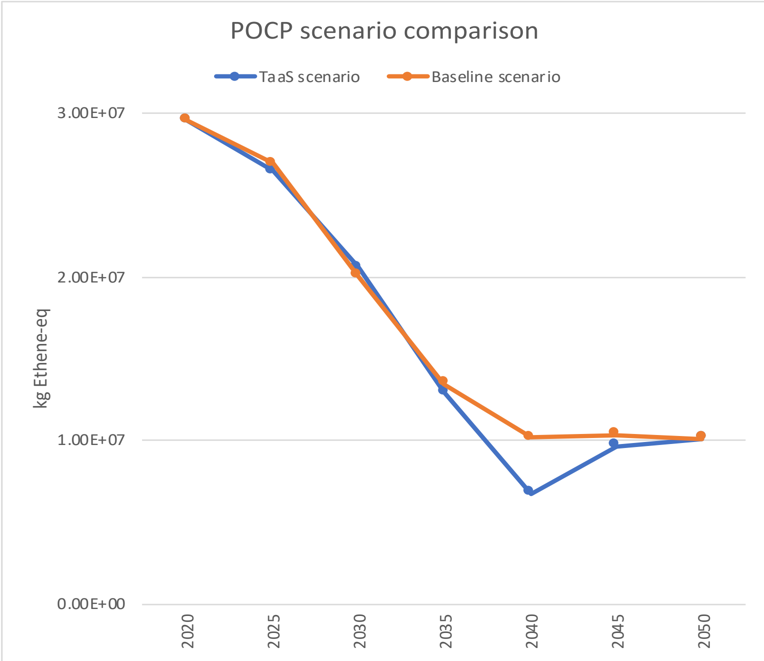


Figure 7.9: Comparison of overall net impact (= total impact – credit) for "Baseline" Scenario: orange line vs "TaaS" scenario: blue line. POCP only represents the use-phase impact.

Overall, the transition to BEVs shows a positive benefit in terms of GWP and nr-CED mainly due to decarbonisation of the grid mix coupled with the transition to BEVs. Battery circularity has seen to be an effective strategy to limit the growing impacts caused by the production of LIBs, however their effect is dependent on when enough raw materials become available from retired BEV batteries. In all cases, integrating TaaS along with battery circularity strategy shows a far greater benefit result in reducing the overall environmental impacts of the transition to BEV fleet.

7.5 Limitation

In this thesis, material flow analysis (MFA) was conducted to understand the battery cathode material demand for future passenger vehicle fleet and the consequences of battery circularity and shared mobility resource strategy. The methodology of integrated MFA and life cycle assessment was used to quantify and compare the environmental burden of the current and future passenger vehicle fleet based on exploratory *"what if"* scenarios analysis on the implementation of these two resource strategies.

The integrated dynamic MFA and LCA is well suited to such type of assessment. The scenarios are built using material flow model to keep track of the transient behaviour of passenger vehicles replacement and additions based on the underlining forecast on growth in mobility service need for passenger fleet dictated by the Department of Transport (2018). However, the impacts of COVID-19 have had an unprecedented impact on transport use and travel patterns, leading to significant drops on travel journey. This study's analysis is also limited by the system boundary focused exclusively on passenger vehicles, specifically four-wheel cars and vans, and does not encompass the growing micro-mobility sector. As shared mobility schemes are increasingly expected to fulfil passenger mobility service needs, this study does not capture the impact and potential growth of alternative forms of transportation, such as electric scooters and bicycles. Consequently, new forecast models are necessary to better understand vehicle stock flow and address the shift in mobility trends, ensuring a comprehensive evaluation of future mobility services.

One of the key aspects of the model was analysing the potential of battery circularity strategy within the UK, the losses in the battery materials are mainly dictated by the battery collection and recovery rate for second life use and closed-loop recycling for the cathode materials. This may be considered a simplistic modelling method of a very complex end of life possibilities of BEV batteries. The model assumes collection rate > 80% based on the available data and estimated figures for BEV batteries and EoL vehicles (European Commission, 2019b) and high recovery of hydrometallurgical recycling process. However, firstly the collection and recovery of battery materials may be hampered by insufficient economic interest and in reality, not all BEV cathode materials would end of up in closed-loop recycling strategy, secondly the model does not consider time delay between battery material recovery and reuse in new BEV battery. Furthermore, currently there isn't any BEV battery hydrometallurgical recycling facility operating in the UK, instead batteries are exported to Europe for processing (Pillot 2019). However, the implication of this would mean that the battery materials may not return back to the UK for a closed loop recycling strategy of UK fleet. Nevertheless, it does not take away on the significant need of battery material recovery and reuse to limit the overall environmental concerns for mass adoption of electric mobility.

Another limitation is that the study does not consider a case where vehicles and their batteries are not meeting proper EoL management, such that vehicles are collected and exported to countries with less developed and unsophisticated EoL management system. This could result in a loss of EoL environmental credits for both ICEVs and BEVs. Currently, the UK has a strict EoL target, which requires 95% recycle and recovery of EoL vehicles (UK Government 2014). However, there is limited information available about the handling of EoL vehicles that are exported. Further research is needed to understand the current streams for EoL vehicles registered in the UK and their EoL implications.

The material flow model results focus solely on the cathode materials of lithium-ion batteries and does not consider other critical materials such as graphite, which is used in the battery anode. Additionally, the study is restricted to lithium-ion batteries with nickel manganese cobalt (NMC) cathode types, which are predominant in current battery electric vehicles (BEVs). One of the primary concerns associated with lithium-ion batteries is their impact on human toxicity and resource depletion. As battery technology rapidly evolves, new types of batteries, such as sodium-ion and solid-state batteries, are expected to enter the market. For instance, CATL battery manufacturer are incorporating a mix of sodium-ion and lithium-ion cells in their battery packs (NY Times 2023), while Nissan is projected to introduce solid-state batteries to the BEV market by 2028 (Tisshaw 2023). These advancements could lead to the selection of cathode materials with lower environmental impacts while maintaining similar energy densities, potentially yielding better life cycle results for future passenger vehicles. Therefore, the study's findings may not fully represent the environmental impacts of emerging battery technologies, necessitating further research into these evolving technologies.

BEVs are currently the main technology for low-emission vehicle strategies. Although the thesis considers that all passenger fleet vehicles will eventually shift to BEVs, this outcome may not fully materialize in practice. In the case where FCVs were to replace passenger ICEVs on a large scale, this could shift pressure on the availability of platinum-group metals (PGMs), which are critical for FCVs. PGM are also use in water electrolysers for the production of "green" hydrogen used in stationary energy storage and is expected to increase in the coming decades. However, a portion of the platinum demand could potentially be met through the recycling of EoL ICEVs which represent relatively small quantities of PGMs compared to FCVs, provided that recycling rates and circularity within the automotive industry are significantly improved (Rasmussen et al., 2019; Tong et al., 2022). This aspect was not explored, as it falls outside the thesis's scope. However, given BEV has a 2.5 times higher powertrain efficiency in comparison to FCV (Salahuddin et al. 2018), it is unlikely FCV will dominate the passenger fleet market. FCV are expected to be suitable for long range and heavy-duty vehicles (National Composites Centre 2021).

One significant limitation of this study is the modelling approach used for the electricity grid mix in the life cycle assessment (LCA). While the grid mix for the UK was modelled in a prospective manner, accounting for future changes and improvements, the battery manufacturing process was assumed to occur in Europe and grid mix for the European countries was not modelled with a prospective approach. This discrepancy may lead to an underestimating the benefits of carbon emissions associated with battery manufacturing. Future studies should aim to incorporate a more detailed and prospective grid mix model for battery manufacturing to provide a more accurate assessment of the environmental impacts. Furthermore, not all manufacturing of BEV LIB will occur in Europe. Europe is considered due to their planned ramp-up of battery production capacity.

Lastly, the thesis assumes all manufacturing of vehicles occur in the UK, however, in 2022 vehicles were the second most imported goods in the UK mainly from Germany, followed by a relativity a small proportion from China, Spain, Czechia and South Korea (OEC n.d.). Likewise, prior to Bexit, EU27 represented 81% of UK vehicle imports in 2019 (ACEA 2020). Due to the uncertainty surrounding Brexit at the time of the study, particularly regarding future trade agreements, it was assumed that all vehicles registered in the UK were manufactured domestically. While this introduces an uncertainty on vehicle manufacturing emissions, both the UK and the EU are progressing toward low-carbon grid mixes and remain committed to achieving Net Zero emissions by 2050 (European Commission, 2019a). As both policy landscape continues to evolve towards stricter emissions standards, the findings on the environmental impact of vehicle manufacturing is not expected to make a significant change to the overall results.

8 Conclusion and Further Work

8.1 Conclusions

Transport plays a critical role in decarbonisation; therefore, it is important to deliver a transport system which is truly low carbon, not only during its operation but over its entire life cycle, whilst preventing environmental consequences from this extensive systemic shift. This thesis has laid the methodological groundwork to support and direct future decision-making in maximising decarbonisation and limiting the overall environmental impact of the transport system in the UK, in particularly focusing on the passenger light duty vehicle fleet (LDV).

This thesis aimed to capture and assess expected changes in the transport system to evaluate the consequences of transition to BEVs with regards to different resource strategies, i.e., battery circularity and uptake of TaaS through to 2050. The scenarios were developed based on the expected changes influencing the passenger vehicle fleet, including:

- Uptake of Electric Mobility
- Evolution of the Electricity Grid Mix
- Evolution of the Battery Chemistry
- Battery Reuse and Recycling Opportunity
- Transition from Vehicle Ownership to Transportation as a Service

Given the complexity of various on-going changes influencing the transition to BEV and their associated resource and waste flows over-time, a multi-pronged approach was used to provide better and more comprehensive understanding on the role of technological improvement in batteries, the shift to EV and implementing battery circularity and shared mobility strategies.

A prospective and dynamic material flow analysis (MFA) and life cycle assessment (LCA) tool were used as methods to assess the resource and environmental benefits and consequences based on these different pathways. Where the dynamic MFA captures the flow of resources in real-time, laying the inventory for the LCA. To assess the consequences of shared mobility and battery circularity, consequential approach to LCA was taken for the uptake of shared mobility, battery elements, the improvement in battery technology and battery reuse has a direct consequence on the resource for battery materials. This allowed the evaluation of the resource and environmental trade-off occurring in real time as the passenger fleet transitions to BEVs through to 2050.

The thesis follows a whole system approach, that considers the impact of a service, instead of focusing on individual product or processes (Garcia & Freire 2017). The thesis expends the methodology of previous work by not solely focusing on the impact of transitions (Garcia et al. 2015; Rietmann et al. 2020; Xiong et al. 2021; Shui et al. 2024), but also focusing on different resource strategies. Although there have been several fleet-based studies, they are mainly limited to grid mix and battery evolution. Few studies considered the effect of battery circularity (Bobba et al 2020; Liu et al. 2020; Kamath et al. 2023) and shared mobility (Liao et al 2021; Vilaça et al. 2022; Roca-Puigròs 2023) on the fleet transition, where the focus was again limited to carbon footprint, limited to the partial mass adoption of EVs excluding the current fleet mix of fossil-fuel based vehicles. The either exclusion limits the comprehensive understanding of the shift in environmental burden and overall trade-off of the fleet transition. Therefore, the full impacts and benefits of resource strategy may be under looked by not considering other environmental indicators or the impact of service as a whole.

The novelty of this study was brought about by simultaneously and consistently modelling all aspects that are expected to change dynamically as a consequence of their complex interlinkages and its influence on the passenger vehicle fleet as a whole. The methodological framework developed provides a holistically in-depth understanding of the trends of energy and mass flows and environmental impacts into the future to help in future decision making to assess the temporal trade-off of different possible pathways, which becomes increasing important with the increase shift to technologies heavily replying on critical materials to meet the expected demand. To the best of author's knowledge, a comprehensive life cycle assessment of the passenger vehicle fleet analysing the environmental consequences of battery circularity and shared mobility has not been done before.

When carrying out the MFA and LCA, it was found that the direct shift from a fleet based on ICEVs to BEVs coupled with low carbon electricity, contributes to a decrease in GHG emissions. However, in addition to the increased penetration of renewables such as wind energy and solar PV, the future electricity grid mix scenarios also rely somewhat on the successful implementation of biomass energy carbon capture and storage (BECCS) technology to achieve grid decarbonization, which in turn plays a key role in reducing the emissions from the operation of LDV sector. Also, with the on-going energy transition, a significant rise in demand for various critical raw materials is expected. In particularly, battery materials play a dual role in both the energy and transport sectors, respectively to meet the rising demand of energy storage in the grid and BEVs. The role of both reuse and recycling plays a critical role in securing these raw materials. In terms of environmental impact, batteries have been shown here to be the main contributor in most impact categories, followed by the other components of BEVs. More specifically, a large share of the impact is associated with the nickel used for the battery cathodes. However, less attention is often dedicated to nickel as a raw material, due to its relative abundance, while more emphasis is given to lithium and cobalt in most discussions of critical raw materials. Yet, according to this study, at a fleet level, nickel use indicates significant concerns in terms quantity and environmental contribution.

Overall, circular battery strategies and a transition to lower-carbon electricity lead to a significant reduction in the carbon emissions of the transport system. However, the performance in terms of other environmental indicators (especially abiotic depletion and toxicity) is not as good, which is mainly due to the production of both the BEV powertrains and batteries. In the goal of reducing carbon emissions and to avoid shifting environmental burdens, it is necessary to not only aim to recycle and reuse materials, but also to enable a shift towards a successful implementation of transport as a service (TaaS), which has been shown to have great potential for a significant environmental benefit at the overall fleet level.

Overall, achieving sustainability in the transport sector goes beyond shifting from ICEVs to BEVs. This investigation of the transition to BEVs points out the importance of resource strategies, to look beyond mere technological improvements, which may result in only marginal overall reduction in environmental impact over time, resulting instead in a shift of pollution and resources. Rather, emphasis should be put on strategies based on elements of behavioural change, resulting in conservation of resources and more significant reductions in environmental burdens.

8.2 Recommendations for Further Work

Whilst the main focus of this thesis was on exploring the impact of resource utilization strategies on the landscape of interlinkages between the co-evolving energy and transport sectors; however, the scope was limited to set of transition pathways at the passenger fleet level.

Several areas require further investigation to gain a more comprehensive understanding of the broader implications of these strategies. Mobility demands and patterns are subject to significant uncertainty due to change in travel pattern post-Covid due to stoical due. Moreover, the potential for some recovered materials to fall outside of a circular strategy has not been explored in this thesis. Understanding the impact of this limitation is crucial to comprehending the real-world challenges and constraints of implementing a fully circular economy.

Further work can expand the scope to also include mobility service as a whole (including other means of transport e.g. public modes), allowing to better understand the wider implications of resource strategies which were not captured within the current fleet-level scope.

Current research globally is focused on new battery technologies, beyond lithium-ion, to achieve higher theoretical limits for battery packs coupled with use of less critical elements. These were not included in the current study due to the level of uncertainties involved on the performance and commercialization in the future. Given that battery materials play a significant impact on the environmental performance of the vehicle fleet, further work could expand on exploring several battery technology options and what implications this may have on the overall vehicle fleet.

8.3 Policy Recommendation

The finding highlights the impact of a circular economy approach, which involves UK-based recycling and reuse of BEV batteries, as well as the move from private vehicle ownership towards TaaS for passenger vehicles in the context of passenger vehicle fleet transition to BEV coupled with decarbonising electricity grid mix.

The finding highlights a circular battery approach can recover valuable materials like lithium, cobalt, and nickel, thereby alleviating supply concerns regarding the supply of these metals investigated in the thesis. This study suggests that starting from 2040, recycling could potentially meet more than half of the cathode material requirements for new BEV LIBs by utilising EoL batteries.Although the study was limited to NMC cathode type of lithium-ion batteries but highlights the sheer benefit of domestic recycling infrastructure to reduce the long-term reliance of critical materials and the potential to support global battery supply chain. Therefore, this offers an opportunity to mitigate some concerns about supply chain disruptions of cathode materials while also creating the potential for integration with battery manufacturing markets. The finding also highlights UK grid battery energy storage will account for only a small portion of the overall battery demand for BEVs. Therefore, second-life applications for BEV batteries offer a valuable opportunity to extend their lifecycle and contribute to energy storage needs without causing significant disruption in the availability of recycled battery materials. Policy should encourage reuse of battery, domestic recycling, incentivise high recovery rates and prioritise a closed-loop system to enhance resource security. Ensuring policy for vehicle and battery EoL traceability is crucial for achieving high collection rates of BEV batteries and minimising resource loss.

The study indicates that the circular approach to battery materials can significantly reduce the UK transport carbon footprint[11]. However, while battery circular strategy also helps mitigate the risks of resource depletion and health impacts, it is not sufficient to fully address these significant environmental concerns. The thesis shows to meet the 2030 ban on ICEV and replacement for ICEV passenger vehicles by BEVs, an estimate of around 900,000 tonnes of BEV battery would be needed in 2035 as well in 2050[12]. A massive shift to shared mobility vehicles, has a potential to decrease this by 1.8 and 9 times in 2035 and 2050 respectively[13]. A successful transition to Transport as a Service (TaaS) can substantially reduce the demand for new vehicles and the associated need for critical battery materials, leading to improved overall environmental performance. Similar recommendations have been by Vilaça, M et al. (2022) on the importance of promoting an increased use of shared mobility to improve the overall environmental performance of the passenger fleet. The policy should encourage the adoption of alternative

[11] This study shows a decrease from 1.12E+11 to 1.47E+10 kg CO_2-eq for battery circularity strategy.

[12] Assuming the battery chemistry stays lithium-ion with the current major share of NMC cathode type.

[13] With a 1.5% increase each year in new vehicles being part of TaaS and an average high mileage of 64,000 km and a low lifetime of 3 years.

environmentally friendly transport options to shift away from the need of private vehicle ownership.

Decarbonising the passenger vehicle fleet by transitioning to BEVs, the UK will help manage a significant rise in battery demand, address potential human toxicity issues, and tackle concerns about resource depletion. The policy should develop long-term strategies that integrate circular economy principles, focusing on recycling, shared mobility, and the adoption of sustainable battery technologies.

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Appendix A: Lithium-ion Battery Chemistry

A LIB consists of a cathode (positive electrode), anode (negative electrode), separator, and an electrolyte. Cathode and the anode are made of intercalation compound that allows the flow of lithium-ion from the cathode to anode and vice versa. The cathode of the battery represents the transition metal oxide or phosphate that have been lithiated during construction of the battery to provide the supply of lithium ions. The anode consists of porous carbon mainly graphite (sometimes graphene) (Battery University (2022), the structure of graphite and its high conductivity are favourable for ensuring high efficiency intercalation of lithium-ions in the anode (Heß & Novak 2013). The electrolyte is an ionic conductive insulating material that transports lithium ions between electrodes during discharge and charge cycling. The battery reaction is mainly the movement of the lithium-ions and the electrons in the battery from cathode to the anode. When the battery is being charged, an oxidation reaction takes place in the cathode and a reduction reaction takes place at the anode, where the lithium-ion combines with graphite (Liu et al. 2016). During discharge this process is reserved, and the lithium-ion is released from the graphite anode and combines with the cathode material (Satyavani et al. 2016).

There are currently four main cathode chemistries of lithium-ion batteries widely available in commercial BEVs: (1) NMC-LMO blend - spinel (2) NMC - layered (3) NCA - layered and (4) LFP - olivine. Each combination has distinct advantages and disadvantages in terms of performance, cost, safety, and other parameters.

Olivine LFP ($LiFePO_4$) is inexpensive and less toxic compared to cobalt-based materials. However, LFP have a lower conductivity and low diffusion coefficient of lithium-ions, which results loss in the capacity during high-rate discharge and therefore, reducing its operating efficiency (Julien et al. 2014, Ramasubramanian et al. 2022). However, carbon coated LFP enhances conductivity which increases the overall capacity of the LFP reaching the theoretical value (~170 Wh/kg) (Julien et al 2014; Ramasubramanian et al. 2022). LFP LIBs is one of the main choices for the electric buses, power tools and grid energy storage (Belharouak et al. 2020). Recently Tesla EV model 3, Ford Mustang Mach-E have moved towards LFP cathodes due to their long cycle life and thermal stability, up to 270 ◦C (Tesla 2023, CNN 2022, Ford Media Center n.d.).

LMO ($LiMn_2O_4$) has a stable spinel structure, lower cost (manganese is five times cheaper than cobalt and is found in abundance in nature (Julien et al 2014), it is non-toxic, and it has a three-dimensional lithium-ion diffusion pathway which improves lithium-ion flow on the electrode and the structure gives it a high thermal stability (Tran et al 2014; Battery University 2023). However, they lack in terms of long-term cycle life (Nitta et al. 2015). There is a move towards NMC-LMO blended lithium-ion as the system can be built economically with less cobalt and it achieves good overall performance in terms of improved specific energy and the lifespan (Battery University 2023). The LMO part of the battery provides high current for acceleration and the NMC part provides long driving range. Blended LMO(NMC) is used in most BEVs such as Nissan Leaf, Chevy Volt and BMW i3 and in stationary grid energy storage systems that need frequent cycling (Battery University 2023).

The layered NMC ($LiNi_xMn_yCo_zO_2$) compounds has a better stability during cycling at a higher temperature (50° C), higher reversible capacity and has a high voltage potential (Julien et al. 2014, Nitta et al. 2015). NMC is known for higher energy/power density. However, NMC suffer from the mixing of nickel-ion and lithium-ions (due to the similar ionic radius size), this impacts the transport performance of lithium-ion and causes low reversible capacity which leads to poor stability (Julien et al. 2014, Belharouak et al. 2020). Small amount of cobalt can stabilize the electrode, leading to a significant increase in terms of capacity, structural stability, and cycle life (Belharouak et al. 2020). Since the development of NMC 111, (i.e., 111 represents the equal composition ratio between nickel, manganese and cobalt respectively), there has been on-going research to achieve high nickel content in NMC batteries, this includes NMC 532, NMC 622 and NMC 811. NMC 532 and NMC 611 and NMC811 are the current adopted cathode composition for BEVs (Mckinsey, 2021). Ni-rich NMC cathode materials such as NMC-811 are likely to dominated in the automotive industry owing to their higher specific energy and low cobalt content (Houache et al. 2022). However, although Ni-rich NMC can efficiently enhance the specific energy, it is very hard to exceed its theoretical limitation (350 Wh/kg) at a cell level (Ding et al. 2019b).

NCA cathode share similarities with NMC as they both are layered structure (Tran et al. 2021). NCA is prepared by dual doping of cobalt and aluminium, the use of aluminium instead of manganese improves the specific energy and lifespan of the battery when compared to its NMC.

Aluminium-ions can keep the crystal structure stable and its presence also increases the operating voltage, NCA batteries also have a high specific energy of 200 Wh/kg (Tran et al. 2021). The main disadvantage of NCA batteries is that they are not as safe as other battery types and require special monitoring (Tran et al 2021). By 2025, the specific energy for NCA is expected to reach 300 Wh/kg at a cell level (Myung et al. 2017). To date, NCA technology has been successfully employed in Tesla electric cars.

Cathode (+ve) active material	**$LiMn_2O_4$**	**$LiFePO_4$**	**$LiAl_{0.05}Co_{0.15}Ni_{0.8}O_2$**	**$LiNi_{0.33}Mn_{0.33}Co_{0.33}O_2$**
Formula	$Li_xMn_yO_4$	$Li_xFe_yPO_4$	$LiNi_xCo_yAl_zO_2$	$LiNi_xMn_yCo_zO_2$
Crystal structure	Spinel	Olivine	layered	layered
Average voltage	3.86 - 4.1	3.22 - 3.45	3.65 - 3.7	3.7
Specific Capacity (Wh/kg)	148	170	200	280

Table 1.1: Comparison of different commercial LIB cathodes, adopted from (Nitta et al. 2015 and Satyavani et al. 2016).

Appendix B: Critical Elements for a Successful Energy Transition

B.1: Systematic Review Flow Diagram

The systematic review process was structured following the Preferred Reporting Items for Systematic reviews and Meta-Analyses (PRISMA) methodology (Page et al. 2020) represented in Figure A1.1, which allows for the transparent and unbiased collection of studies related to a set of research questions. The information sources were selected to benefit from a range of repositories for academic journals specific to the areas of research of interest, to guarantee the quality of the returned articles. Specifically, two main search engines were selected and used to retrieve peer-reviewed journal papers, reviews, and editorial materials: Google Scholar and Web of Science. In addition, three publisher-specific search engines relevant to the field were also identified and used in parallel: Science Direct, Nature Publishing Group, and MDPI.

The search terms were combined using Boolean operators. The asterisk (wildcard) symbol was used in combination with some of the keywords where possible and appropriate to find papers using the same root keywords but with different suffixes. Science Direct and MDPI do not support the use of wildcards, however. Searches were done using the "topic" field where possible, which includes title, abstract and keywords. However, the Google Scholar and Nature Publishing Group search engines are limited to searches in the "title" or "article" fields only, and hence, for better comprehensiveness, the latter field was used in these cases. The MDPI search engine is instead limited to the "keyword" and "title" fields, both of which were employed. Papers from fields such as physics, biology and chemistry were deemed too specific and out of scope, and hence they were ruled out or excluded in the search engine process where possible. Likewise, certain words were explicitly excluded from the Google Scholar search (e.g., "biology" and "physics") to limit the number of irrelevant papers returned and reduce the burden of the subsequent manual screening stages. A full list of these 161 papers with brief accompanying notes is provided in Table A1.1.

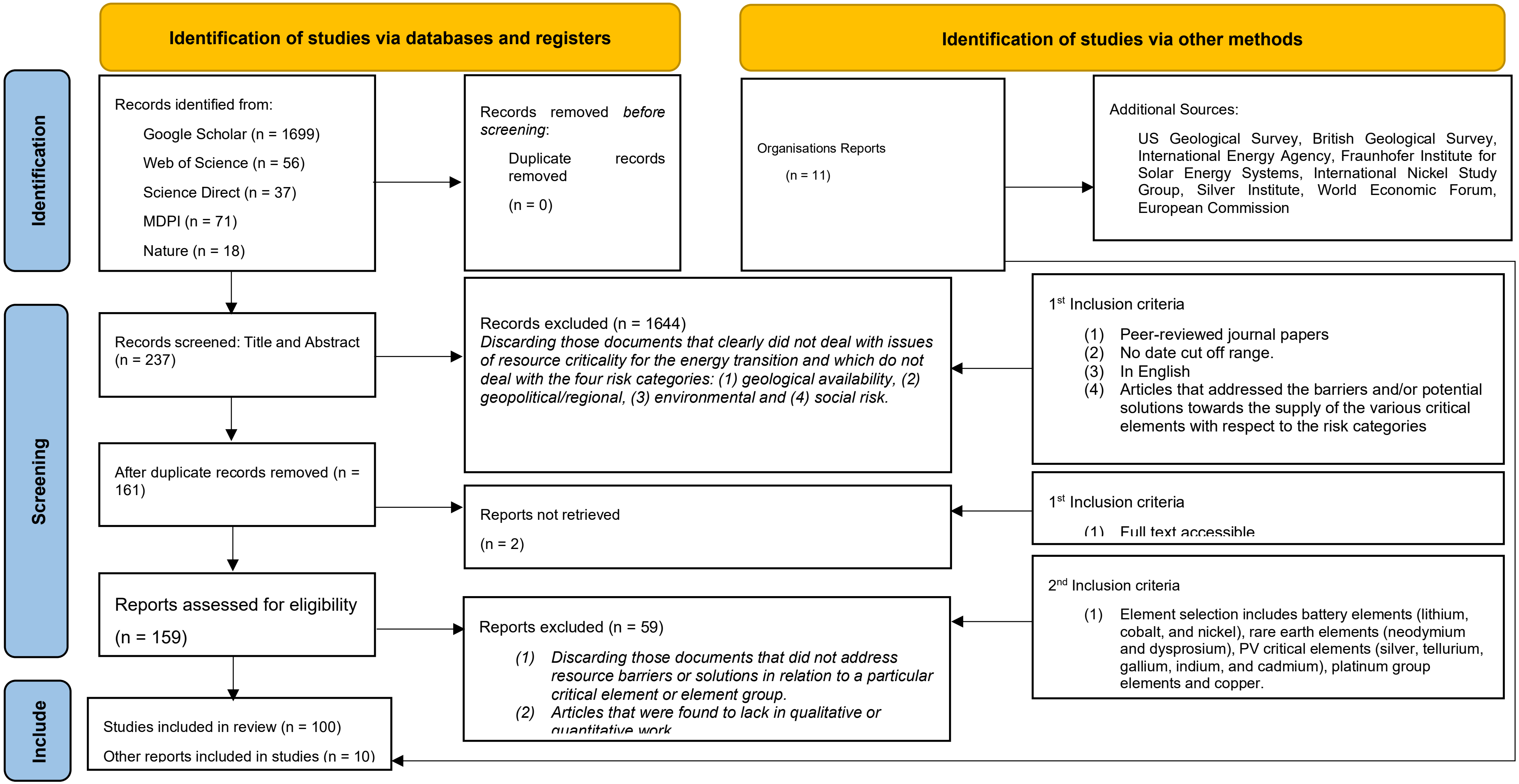

Figure A1.1 PRISMA FLOW DIAGRAM. From: Page MJ, McKenzie JE, Bossuyt PM, Boutron I, Hoffmann TC, Mulrow CD, et al. The PRISMA 2020 statement: an updated guideline for reporting systematic reviews. BMJ 2021;372:n71. doi: 10.1136/bmj.n71. For more information, visit: http://www.prisma-statement.org/

B.2: List of Papers from Systematic Search

Authors	Year	Paper Title	Digital Object Identifier	Journal
A. Elshkaki; T. Graedel	2013	Dynamic analysis of the global metals flows and stocks in electricity generation technologies	https://doi.org/10.1016/j.jclepro.2013.07.003	Journal of Cleaner Production
O. Vidal; B. Goffe; N. Arndt	2013	Metals for a low-carbon society	https://doi.org/10.1038/ngeo1993	Nature Geoscience
R. L. Moss; E. Tzimas; H. Kara; P. Willis; J. Kooroshy	2013	The potential risks from metals bottlenecks to the deployment of Strategic Energy Technologies	https://doi.org/10.1016/j.enpol.2012.12.053	Energy Policy
A. Elshkaki; T. Graedel	2014	Dysprosium, the balance problem, and wind power technology	https://doi.org/10.1016/j.apenergy.2014.09.064	Applied Energy
A. Stamp; P. A. W√§ger; S. Hellweg	2014	Linking energy scenarios with metal demand modeling‚ÄìThe case of indium in CIGS solar cells	https://doi.org/10.1016/j.resconrec.2014.10.012	Resources, Conservation and Recycling

W.-Q. Zhuang; J. P. Fitts; C. M. Ajo-Franklin; S. Maes; L. Alvarez-Cohen; T. Hennebel	2015	Recovery of critical metals using biometallurgy	https://doi.org/10.1016/j.copbio.2015.03.019	Current opinion in biotechnology
L. Grandell; M. H√∂√∂k	2015	Assessing rare metal availability challenges for solar energy technologies	https://doi.org/10.3390/su70911818	Sustainability
P. Viebahn; O. Soukup; S. Samadi; J. Teubler; K. Wiesen; M. Ritthoff	2015	Assessing the need for critical minerals to shift the German energy system towards a high proportion of renewables	https://doi.org/10.1016/j.rser.2015.04.070	Renewable & Sustainable Energy Reviews
K. S. Stegen	2015	Heavy rare earths, permanent magnets, and renewable energies: An imminent crisis	https://doi.org/10.1016/j.enpol.2014.12.015	Energy Policy
N. T. Nassar; D. R. Wilburn; T. G. Goonan	2016	Byproduct metal requirements for US wind and solar photovoltaic electricity generation up to the year 2040 under various Clean Power Plan scenarios	https://doi.org/10.1016/j.apenergy.2016.08.062	Mineral Economics

B. C. McLellan; E. Yamasue; T. Tezuka; G. Corder; A. Golev; D. Giurco	2016	Critical minerals and energy‚Äìimpacts and limitations of moving to unconventional resources	https://doi.org/10.3390/resources5020019	Resources
A. Rollat; D. Guyonnet; M. Planchon; J. Tuduri	2016	Prospective analysis of the flows of certain rare earths in Europe at the 2020 horizon	http://dx.doi.org/10.1016/j.wasman.2016.01.011	Waste Management
K. Habib; H. Wenzel	2016	Reviewing resource criticality assessment from a dynamic and technology specific perspective - using the case of direct-drive wind turbines	https://doi.org/10.1016/j.jclepro.2015.07.064	Journal of Cleaner Production
L. Grandell; A. Lehtil√§; M. Kivinen; T. Koljonen; S. Kihlman; L. S. Lauri	2016	Role of critical metals in the future markets of clean energy technologies	https://doi.org/10.1016/j.renene.2016.03.102	Renewable Energy
C. Helbig; A. M. Bradshaw; C. Kolotzek; A. Thorenz; A. Tuma	2016	Supply risks associated with CdTe and CIGS thin-film photovoltaics	https://doi.org/10.1016/j.apenergy.2016.06.102	Applied Energy

K. Tokimatsu; H. Wachtmeister; B. McLellan; S. Davidsson; S. Murakami; M. H√∂√∂k; R. Yasuoka; M. Nishio	2017	Energy modeling approach to the global energy-mineral nexus: A first look at metal requirements and the 2 C target	https://doi.org/10.1016/j.apenergy.2017.05.151	Applied energy
B. Zhou; Z. Li; C. Chen	2017	Global potential of rare earth resources and rare earth demand from clean technologies	https://doi.org/10.3390/min7110203	Minerals
O. Vidal; F. Rostom; C. Fran√ßois; G. Giraud	2017	Global Trends in Metal Consumption and Supply: The Raw Material–Energy Nexus	https://doi.org/10.2138/gselements.13.5.319	Elements
S. Davidsson; M. H√∂√∂k	2017	Material requirements and availability for multi-terawatt deployment of photovoltaics	https://doi.org/10.1016/j.enpol.2017.06.028	Energy policy
M. Frenzel; J. Kullik; M. A. Reuter; J. Gutzmer	2017	Raw material 'criticality'-sense or nonsense?	https://doi.org/10.1088/1361-6463/aa5b64	Journal of Physics D-Applied Physics
T. Watari; B. C. McLellan; S. Ogata; T. Tezuka	2018	Analysis of potential for critical metal resource constraints in the international energy agency,Äôs long-term low-carbon energy scenarios	https://doi.org/10.3390/min8040156	Minerals

J. H. Hodgkinson; M. H. Smith	2018	Climate change and sustainability as drivers for the next mining and metals boom: The need for climate-smart mining and recycling	https://doi.org/10.1016/j.resourpol.2018.05.016	Resources Policy
D. Son; S. Kim; H. Park; B. Jeong	2018	Closed-loop supply chain planning model of rare metals	https://doi.org/10.3390/su10041061	Sustainability
B. J. Smith; R. G. Eggert	2018	Costs, substitution, and material use: the case of rare earth magnets	https://doi.org/10.1021/acs.est.7b05495	Environmental science & technology
P. Buchholz; T. Brandenburg	2018	Demand, supply, and price trends for mineral raw materials relevant to the renewable energy transition wind energy, solar photovoltaic energy, and energy storage	https://doi.org/10.1002/cite.201700098	Chemie Ingenieur Technik
K. Tokimatsu; M. H√∂√∂k; B. McLellan; H. Wachtmeister; S. Murakami; R. Yasuoka; M. Nishio	2018	Energy modeling approach to the global energy-mineral nexus: Exploring metal requirements and the well-below 2 C target with 100 percent renewable energy	https://doi.org/10.1016/j.apenergy.2018.05.047	Applied Energy

A. M√•nberger; B. Stenqvist	2018	Global metal flows in the renewable energy transition: exploring the effects of substitutes, technological mix and development	https://doi.org/10.1016/j.enpol.2018.04.056	Energy Policy
A. De Koning; R. Kleijn; G. Huppes; B. Sprecher; G. Van Engelen; A. Tukker	2018	Metal supply constraints for a low-carbon economy?	https://doi.org/10.1016/j.resconrec.2017.10.040	Resources, Conservation and Recycling
M. D. Bazilian	2018	The mineral foundation of the energy transition	https://doi.org/10.1016/j.exis.2017.12.002	The Extractive Industries and Society
B. C. McLellan	2019	A Dark Materials for a Brighter Energy Future	https://doi.org/10.1016/j.oneear.2019.11.009	One Earth
E. Hache; G. S. Seck; M. Simoen; C. Bonnet; S. Carcanague	2019	Critical raw materials and transportation sector electrification: A detailed bottom-up analysis in world transport	https://doi.org/10.1016/j.apenergy.2019.02.057	Applied Energy
A. Elshkaki; L. Shen	2019	Energy-material nexus: The impacts of national and international energy scenarios on critical metals use in China up to 2050 and their global implications	https://doi.org/10.1016/j.energy.2019.05.156	Energy

V. Moreau; P. C. Dos Reis; F. Vuille	2019	Enough Metals? Resource Constraints to Supply a Fully Renewable Energy System	https://doi.org/10.3390/resources8010029	Resources
P. Wang; L. Y. Chen; J. P. Ge; W. J. Cai; W. Q. Chen	2019	Incorporating critical material cycles into metal-energy nexus of China's 2050 renewable transition	https://doi.org/10.1016/j.apenergy.2019.113612	Applied Energy
A. Elshkaki	2019	Materials, energy, water, and emissions nexus impacts on the future contribution of PV solar technologies to global energy scenarios	https://doi.org/10.1038/s41598-019-55853-w	Scientific Reports
K. D. Rasmussen; H. Wenzel; C. Bangs; E. Petavratzi; G. Liu	2019	Platinum demand and potential bottlenecks in the global green transition: a dynamic material flow analysis	https://doi.org/10.1021/acs.est.9b01912	Environmental science & technology
H. Hao; Y. Geng; J. E. Tate; F. Liu; X. Sun; Z. Mu; D. Xun; Z. Liu; F. Zhao	2019	Securing platinum-group metals for transport low-carbon transition	https://doi.org/10.1016/j.oneear.2019.08.012	One Earth

X. Sun; H. Hao; P. Hartmann; Z. Liu; F. Zhao	2019	Supply risks of lithium-ion battery materials: An entire supply chain estimation	https://doi.org/10.1016/j.mtener.2019.100347	Materials Today Energy
A. M√•nberger; B. Johansson	2019	The geopolitics of metals and metalloids used for the renewable energy transition	https://doi.org/10.1016/j.esr.2019.100394	Energy Strategy Reviews
B. K. Sovacool	2019	The precarious political economy of cobalt: Balancing prosperity, poverty, and brutality in artisanal and industrial mining in the Democratic Republic of the Congo	https://doi.org/10.1016/j.exis.2019.05.018	The Extractive Industries and Society
E. Van der Voet; L. Van Oers; M. Verboon; K. Kuipers	2019	Environmental implications of future demand scenarios for metals: methodology and application to the case of seven major metals	https://doi.org/10.1111/jiec.12722	Journal of Industrial Ecology
T. Watari; B. C. McLellan; D. Giurco; E. Dominish; E. Yamasue; K. Nansai	2019	Total material requirement for the global energy transition to 2050: A focus on transport and electricity	https://doi.org/10.1016/j.resconrec.2019.05.015	Resources, Conservation and Recycling

P. Greim; A. A. Solomon; C. Breyer	2020	Assessment of lithium criticality in the global energy transition and addressing policy gaps in transportation	https://doi.org/10.1038/s41467-020-18402-y	Nature Communications
A. Ortego; G. Calvo; A. Valero; M. Iglesias-√âmbil; A. Valero; M. Villacampa	2020	Assessment of strategic raw materials in the automobile sector	https://doi.org/10.1016/j.resconrec.2020.104968	Resources, Conservation and Recycling
S. Bobba; I. Bianco; U. Eynard; S. Carrara; F. Mathieux; G. A. Blengini	2020	Bridging Tools to Better Understand Environmental Performances and Raw Materials Supply of Traction Batteries in the Future EU Fleet	https://doi.org/10.3390/en13102513	Energy
L. A. Levin; D. J. Amon; H. Lily	2020	Challenges to the sustainability of deep-seabed mining	https://doi.org/10.1038/s41893-020-0558-x	Nature Sustainability
G. S. Seck; E. Hache; C. Bonnet; M. Simo√´n; S. Carcanague	2020	Copper at the crossroads: Assessment of the interactions between low-carbon energy transition and supply limitations	https://doi.org/10.1016/j.resconrec.2020.105072	Resources, Conservation and Recycling

J. Haas; S. Moreno-Leiva; T. Junne; P.-J. Chen; G. Pamparana; W. Nowak; W. Kracht; J. M. Ortiz	2020	Copper mining: 100% solar electricity by 2030?	https://doi.org/10.1016/j.apenergy.2020.114506	Applied Energy
C.-L. Huang; M. Xu; S. Cui; Z. Li; H. Fang; P. Wang	2020	Copper-induced ripple effects by the expanding electric vehicle fleet: A crisis or an opportunity	https://doi.org/10.1016/j.resconrec.2020.104861	Resources, Conservation and Recycling
T. Junne; N. Wulff; C. Breyer; T. Naegler	2020	Critical materials in global low-carbon energy scenarios: The case for neodymium, dysprosium, lithium, and cobalt	https://doi.org/10.1016/j.energy.2020.118532	Energy
J. Li; K. Peng; P. Wang; N. Zhang; K. Feng; D. Guan; J. Meng; W. Wei; Q. Yang	2020	Critical rare-earth elements mismatch global wind-power ambitions	https://doi.org/10.1016/j.oneear.2020.06.009	One Earth
Y. Zhou; J. Li; H. Rechberger; G. Wang; S. Chen; W. Xing; P. Li	2020	Dynamic criticality of by-products used in thin-film photovoltaic technologies by 2050	https://doi.org/10.1016/j.jclepro.2020.121599	Journal of Cleaner Production

S. M. Jowitt; G. M. Mudd; J. F. H. Thompson	2020	Future availability of non-renewable metal resources and the influence of environmental, social, and governance conflicts on metal production	https://doi.org/10.1038/s43247-020-0011-0	Communications Earth & Environment
S. N. Kamenopoulos; Z. Agioutantis	2020	Geopolitical Risk Assessment of Countries with Rare Earth Element Deposits	https://doi.org/10.1007/s42461-019-00158-9	Mining, Metallurgy & Exploration
D. Paulikas; S. Katona; E. Ilves; S. H. Ali	2020	Life cycle climate change impacts of producing battery metals from land ores versus deep-sea polymetallic nodules	https://doi.org/10.1016/j.jclepro.2020.123822	Journal of Cleaner Production
J. Cristobal; M. Jubayed; N. Wulff; L. Schebek	2020	Life cycle losses of critical raw materials from solar and wind energy technologies and their role in the future material availability	https://doi.org/10.1016/j.resconrec.2020.104916	Resources, Conservation and Recycling
A. Elshkaki	2020	Long-term analysis of critical materials in future vehicles electrification in China and their national and global implications	https://doi.org/10.1016/j.energy.2020.117697	Energy
J. Wang; M. Guo; M. Liu; X. Wei	2020	Long-term outlook for global rare earth production	https://doi.org/10.1016/j.resourpol.2019.101569	Resources Policy

B. Liu; J. Chen; H. Wang; Q. Wang	2020	Renewable Energy and Material Supply Risks: a Predictive Analysis Based on An LSTM Model	https://doi.org/10.3389/fenrg.2020.00163	Frontiers in Energy Research
J. Lee; M. Bazilian; B. Sovacool; S. Greene	2020	Responsible or reckless? A critical review of the environmental and climate assessments of mineral supply chains	https://doi.org/10.1088/1748-9326/ab9f8c	Environmental Research Letters
T. Watari; K. Nansai; K. Nakajima	2020	Review of critical metal dynamics to 2050 for 48 elements	https://doi.org/10.1016/j.resconrec.2019.104669	Resources, Conservation and Recycling
M. Henckens; E. Worrell	2020	Reviewing the availability of copper and nickel for future generations. The balance between production growth, sustainability and recycling rates	https://doi.org/10.1016/j.jclepro.2020.121460	Journal of Cleaner Production
J. Lee; M. Bazilian; B. Sovacool; K. Hund; S. M. Jowitt; T. Nguyen; A. M√•nberger; M. Kah; S. Greene; C. Galeazzi	2020	Reviewing the material and metal security of low-carbon energy transitions	https://doi.org/10.1016/j.rser.2020.109789	Renewable and Sustainable Energy Reviews

B. K. Sovacool; S. H. Ali; M. Bazilian; B. Radley; B. Nemery; J. Okatz; D. Mulvaney	2020	Sustainable minerals and metals for a low-carbon future	https://doi.org/10.1126/science.aaz6003	Science
J. Wang; L. Yang; J. Lin; Y. Bentley	2020	The Availability of Critical Minerals for China's Renewable Energy Development: An Analysis of Physical Supply	https://doi.org/10.1007/s11053-020-09615-5	Natural Resources Research
B. Jones; R. J. Elliott; V. Nguyen-Tien	2020	The EV revolution: The road ahead for critical raw materials demand	https://doi.org/10.1016/j.apenergy.2020.115072	Applied energy
F. Heredia; A. L. Martinez; V. S. Urtubey	2020	The importance of lithium for achieving a low-carbon future: overview of the lithium extraction in the 'Lithium Triangle'	https://doi.org/10.1080/02646811.2020.1784565	Journal of Energy & Natural Resources Law
I. de Blas; M. Mediavilla; I. Capell√°n-P√©rez; C. Duce	2020	The limits of transport decarbonization under the current growth paradigm	https://doi.org/10.1016/j.esr.2020.100543	Energy Strategy Reviews

√â. L√®bre; M. Stringer; K. Svobodova; J. R. Owen; D. Kemp; C. C√¥te; A. Arratia-Solar; R. K. Valenta	2020	The social and environmental complexities of extracting energy transition metals	https://doi.org/10.1038/s41467-020-18661-9	Nature Communications
B. Ballinger; D. Schmeda-Lopez; B. Kefford; B. Parkinson; M. Stringer; C. Greig; S. Smart	2020	The vulnerability of electric-vehicle and wind-turbine supply chains to the supply of rare-earth elements in a 2-degree scenario	https://doi.org/10.1016/j.spc.2020.02.005	Sustainable Production and Consumption
N. R. Rachidi; G. T. Nwaila; S. E. Zhang; J. E. Bourdeau; Y. Ghorbani	2021	Assessing cobalt supply sustainability through production forecasting and implications for green energy policies	https://doi.org/10.1016/j.resourpol.2021.102423	Resources Policy
K. Ren; X. Tang; P. Wang; J. Willerstr√∂m; M. H√∂√∂k	2021	Bridging energy and metal sustainability: Insights from China's wind power development up to 2050	https://doi.org/10.1016/j.energy.2021.120524	Energy Research & Social Science

F. Calv√£o; C. E. A. Mcdonald; M. Bolay	2021	Cobalt mining and the corporate outsourcing of responsibility in the Democratic Republic of Congo	https://doi.org/10.1016/j.exis.2021.02.004	The Extractive Industries and Society
R. Vakulchuk; I. Overland	2021	Central Asia is a missing link in analyses of critical materials for the global clean energy transition	https://doi.org/10.1016/j.oneear.2021.11.012	One Earth
V. Klimenko; S. Ratner; A. Tereshin	2021	Constraints imposed by key-material resources on renewable energy development	https://doi.org/10.1016/j.rser.2021.111011	Renewable and Sustainable Energy Reviews
S. Kiemel; T. Smolinka; F. Lehner; J. Full; A. Sauer; R. Miehe	2021	Critical materials for water electrolysers at the example of the energy transition in Germany	https://doi.org/10.1002/er.6487	yes
Y. Guohua; A. Elshkaki; X. Xiao	2021	Dynamic analysis of future nickel demand, supply, and associated materials, energy, water, and carbon emissions in China	https://doi.org/10.1016/j.resourpol.2021.102432	Resources Policy
T. L. Yao; Y. Geng; J. Sarkis; S. J. Xiao; Z. Y. Gao	2021	Dynamic neodymium stocks and flows analysis in China	https://doi.org/10.1016/j.resconrec.2021.105752	Resources Conservation and Recycling

W. Liu; D. B. Agusdinata	2021	Dynamics of local impacts in low-carbon transition: Agent-based modeling of lithium mining-community-aquifer interactions in Salar de Atacama, Chile	https://doi.org/10.1016/j.exis.2021.100927	The Extractive Industries and Society
C. Harpprecht; L. van Oers; S. A. Northey; Y. Yang; B. Steubing	2021	Environmental impacts of key metals' supply and low, carbon technologies are likely to decrease in the future	https://doi.org/10.1111/jiec.13181	Journal of Industrial Ecology
K. Ren; X. Tang; M. H√∂√∂k	2021	Evaluating metal constraints for photovoltaics: Perspectives from China's PV development	https://doi.org/10.1016/j.apenergy.2020.116148	Applied Energy
X. Tian; Y. Geng; J. Sarkis; C. Gao; X. Sun; T. Micic; H. Hao; X. Wang	2021	Features of critical resource trade networks of lithium-ion batteries	https://doi.org/10.1016/j.resourpol.2021.102177	Resources Policy
√â. L√®bre; J. R. Owen; M. Stringer; D. Kemp; R. K. Valenta	2021	Global Scan of Disruptions to the Mine Life Cycle: Price, Ownership, and Local Impact	https://doi.org/10.1021/acs.est.0c08546	Environmental Science & Technology

L. T. Peiro; N. Martin; G. V. M√©ndez; C. Madrid-L√≥pez	2021	Integration of raw materials indicators of energy technologies into energy system models	https://doi.org/10.1016/j.apenergy.2021.118150	Applied Energy
C. Minke; M. Suermann; B. Bensmann; R. Hanke-Rauschenbach	2021	Is iridium demand a potential bottleneck in the realization of large-scale PEM water electrolysis?	https://doi.org/10.1016/j.ijhydene.2021.04.174	International Journal of Hydrogen Energy
J. D. Graham; J. A. Rupp; E. Brungard	2021	Lithium in the Green Energy Transition: The Quest for Both Sustainability and Security	https://doi.org/10.3390/su132011274	Sustainability
A. Pi√ßarra; I. R. Annesley; A. Otsuki; R. de Waard	2021	Market assessment of cobalt: Identification and evaluation of supply risk patterns	https://doi.org/10.1016/j.resourpol.2021.102206	Resources Policy
S. Nate; Y. Bilan; M. Kurylo; O. Lyashenko; P. Napieralski; G. Kharlamova	2021	Mineral Policy within the Framework of Limited Critical Resources and a Green Energy Transition	https://doi.org/10.3390/en14092688	Energies

E. Lewicka; K. Guzik; K. Galos	2021	On the Possibilities of Critical Raw Materials Production from the EU,Äôs Primary Sources	https://doi.org/10.3390/resources10050050	Resources
Y. Zhou; H. Rechberger; J. Li; Q. Li; G. Wang; S. Chen	2021	Dynamic analysis of indium flows and stocks in China: 2000–2018	https://doi.org/10.1016/j.resconrec.2021.105394	Resources, Conservation and Recycling
M. C. Bonfante; J. P. Raspini; I. B. Fernandes; S. Fernandes; L. M. Campos; O. E. Alarcon	2021	Achieving Sustainable Development Goals in rare earth magnets production: A review on state of the art and SWOT analysis	https://doi.org/10.1016/j.rser.2020.110616	Renewable and Sustainable Energy Reviews
C. Jovanovic	2021	Precious and Few: Solving Renewable Energy's Critical Minerals Problem	https://digitalcommons.law.lsu.edu/jelr/vol9/iss1/6	LSU J. Energy L. & Resources
D. Mulvaney; R. M. Richards; M. D. Bazilian; E. Hensley; G. Clough; S. Sridhar	2021	Progress towards a circular economy in materials to decarbonize electricity and mobility	https://doi.org/10.1016/j.rser.2020.110604	Renewable and Sustainable Energy Reviews
G. Calvo; A. Valero	2021	Strategic mineral resources: Availability and future estimations for the renewable energy sector	https://doi.org/10.1016/j.envdev.2021.100640	Environmental Development

A. Elshkaki	2021	Sustainability of emerging energy and transportation technologies is impacted by the coexistence of minerals in nature	https://doi.org/10.1038/s43247-021-00262-z	Communications Earth & Environment
C. W. Babbitt; S. Althaf; F. C. Rios; M. M. Bilec; T. Graedel	2021	The role of design in circular economy solutions for critical materials	https://doi.org/10.1016/j.oneear.2021.02.014	One Earth
C. A. Gallego	2021	The role of Rare Earth Elements in the deployment of wind energy in Colombia	https://doi.org/10.32685/0120-1425/bol.geol.48.2.2021.552	Geological Society of America Bulletin (peer reviewed)
T. Watari; S. Northey; D. Giurco; S. Hata; R. Yokoi; K. Nansai; K. Nakajima	2022	Global copper cycles and greenhouse gas emissions in a 1.5C world	https://doi.org/10.1016/j.resconrec.2021.106118	Resources, Conservation and Recycling
G. S. Seck; E. Hache; C. Barnet	2022	Potential bottleneck in the energy transition: The case of cobalt in an accelerating electro-mobility world	https://doi.org/10.1016/j.resourpol.2021.102516	Resources Policy

J. van Oorschot; B. Sprecher; B. Roelofs; J. van der Horst; E. van der Voet	2022	Towards a low-carbon and circular economy: Scenarios for metal stocks and flows in the Dutch electricity system	https://doi.org/10.1016/j.resconrec.2021.106105	Resources, Conservation and Recycling
X. Tong; H. Dai; P. Lu; A. Zhang; T. Ma	2022	Saving global platinum demand while achieving carbon neutrality in the passenger transport sector: linking material flow analysis with integrated assessment model	https://doi.org/10.1016/j.resconrec.2021.106110	Resources, Conservation and Recycling

B.3: Statistical analysis of the screened literature

The statistical analysis highlighted a comparative larger emphasis on supply risk vs. the associated environmental and social concerns. This may in part be due to the sheer difficulty of collecting reliable quantitative information on the latter, but it also points to potentially significant knowledge gaps in terms of these important issues. The third and final step of the statistical analysis involved categorizing the papers according to the elements taken into consideration, grouped according to their key roles in specific applications, as discussed in Section 1, namely: elements for battery storage (Li, Co, Ni, Mn), elements for permanent magnets used in wind turbines and electric motors (REE), elements for photovoltaics (Ag, Te, Ga, In, Se, Ge), elements for catalysts used in "green" hydrogen production (PGM), copper (used in all electrical applications). The two groups of elements that appear to have attracted the most attention thus far are the battery elements and the REE; it is noteworthy that both are key to enable the transition to electrical mobility.

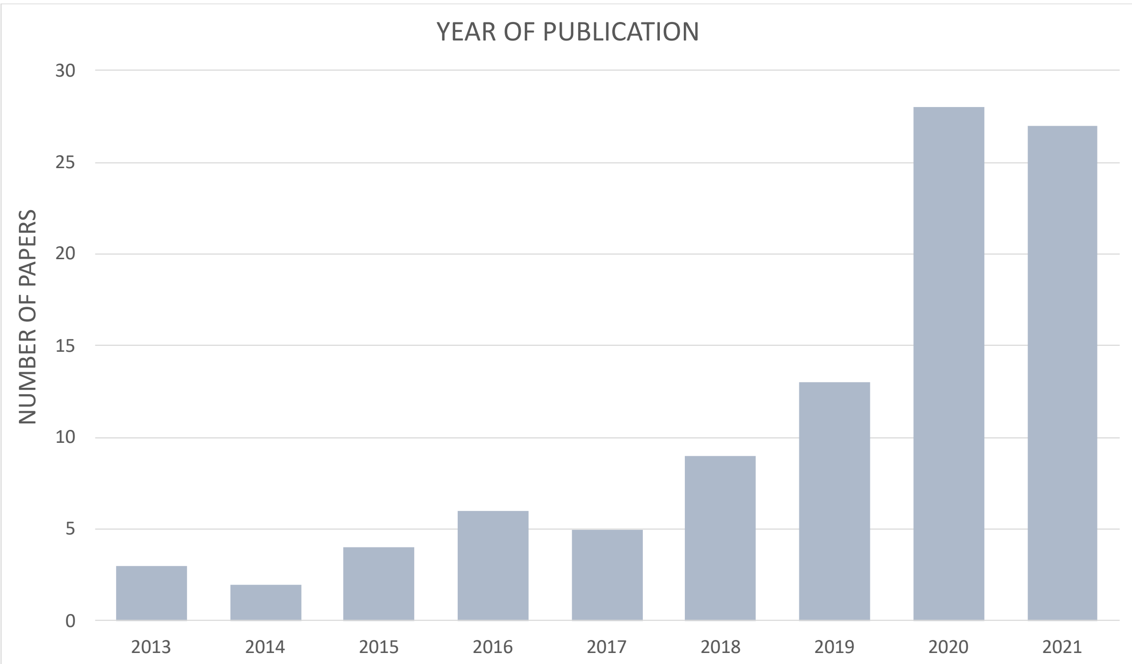

Figure B.1: Number of papers per year of publication.

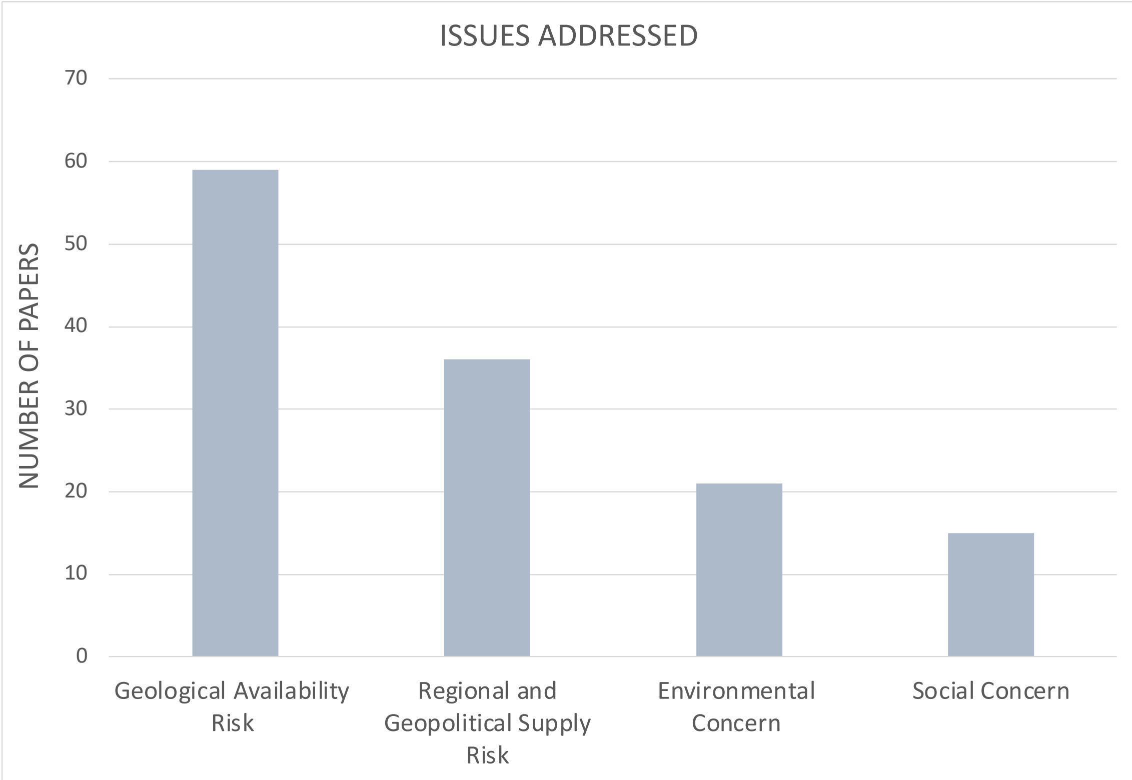

Figure B.2: Number of papers addressing each of the four types of issues/barriers identified in Section 1.

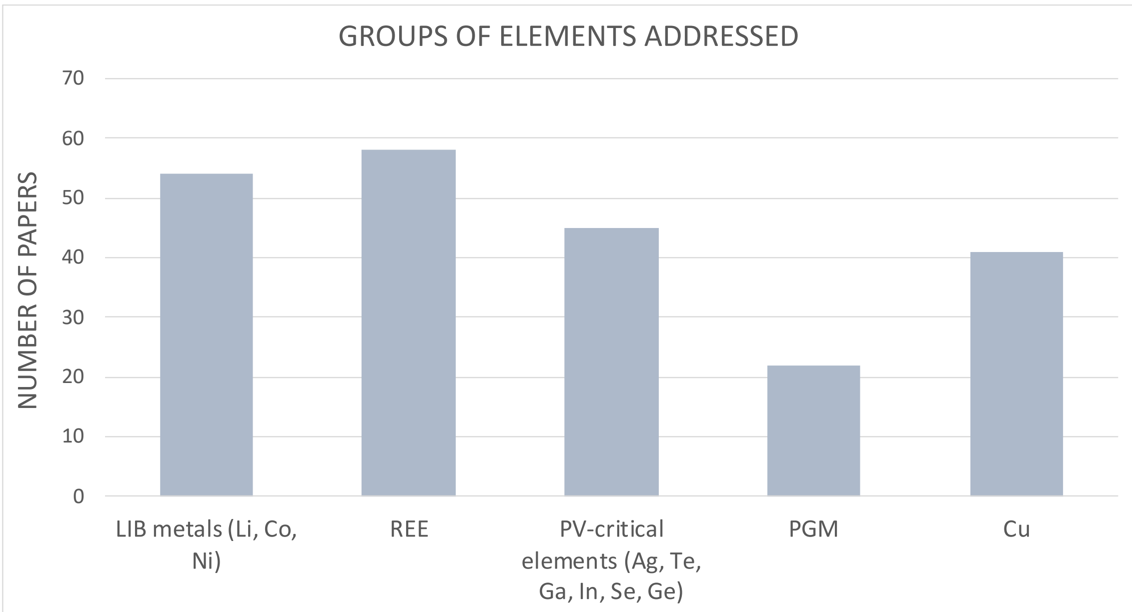

Figure B.3: Number of papers addressing each of the six groups of elements considered. REE = Rare Earth Elements; PGM = Platinum Group Metals.

B.4: Summary Review of Critical Elements

This section represents the summary review of other critical elements except for battery elements. A complete systematic review for other elements can be found at (Kamran et al 2023).

B4.1 Rare Earth Elements

Rare earth elements, especially neodymium and dysprosium used in neodymium iron boron (NdFeB) permanent magnets (PMs), are critical for mainly offshore wind turbine generators and electric mobility motors. Their ability to provide high energy density and performance makes them suitable for the use in vehicle applications which call for lightweight and compact magnets, as well as for wind turbines, by allowing for a lighter turbine design, requiring less structural materials, and consequently fewer efficiency losses specially at low wind speeds (Grandell et al. 2016). The shares of REEs in NdFeB are mainly dominated by neodymium, which represents 29 – 31% of the magnet mass, followed by dysprosium for higher temperature stability and sometimes in small quantities also praseodymium and terbium (Grandell et al. 2016). Figure B.4 shows the demand projection of REEs up to 2050 vs the current reserve estimates. The shortage of dysprosium, as well as the current high concertation of REE mining in China, are likely to be the main factors constraining the use of REE in PMs both in the short and long terms (Rollat et al. 2016; Grandell et al. 2016; Junne et al. 2020). Furthermore, environmental impacts associated with REE are major concerns on their production (Li et al. 2020). These elements are linked with radioactive waste and contamination of ecosystems through intensive chemical use for their refinement, which calls for stricter environmental measures to be put in place. Furthermore, efforts must also be made to improve efficiency of mining as well as increasing investment in REE exploration and extraction outside of China, improve recycling and possible substitutes specially for dysprosium to prevent long-term availability constraints and geopolitical risk (Wang et al. 2020; Li et al. 2020; Gallego 2021). Table B.2 summaries key literature findings.

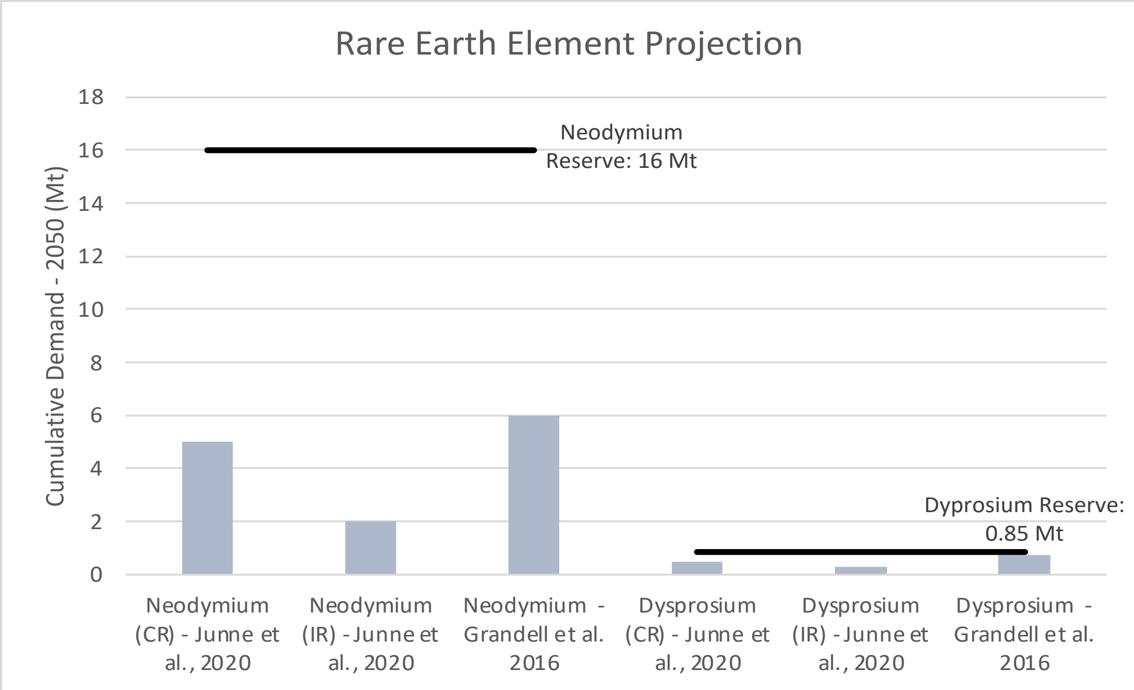

Figure B.4: Cumulative demand projection for Neodymium and Dysprosium in all sectors, adopted from Junne et al., 2020 and Grandell et al., 2016. CR: Current recycling rate; IR: Improved recycling rare. Resource estimates are not available in the literature for REE.

Table B.2: Summary of key barriers/challenges and suggested solutions for rare earth elements.

Category	Issues	Elements	Potential Solutions	References
Geological Availability Risk	Insufficient reserves	Dysprosium	Increase recycling of magnets by imposing legal responsibility and improving recycling efficiency. Replace production process by dual alloying or gain boundary diffusion to reduce dysprosium content; replace dysprosium by terbium. Improve efficiency of mining and chemical processing to maximize metal output from ores Possible recovery of dysprosium from dilute ores or industrial and other waste streams Exploration and development of REE mining.	Gallego 2021; Wang et al. 2020; Li et al. 2020 ; Smith and Eggert 2018; Grandell et al. 2016
Geopolitical and Regional Risk	Heavy rare earth only mined in China.	Dysprosium, Neodymium	Increase investment in REE mining outside of China. Increase recycling and substitution.	Rollat et al. 2016; Ballinger et al. 2020; Smith and Eggert 2018; Li et al. 2020; Wang et al. 2020

	Light rare earth mainly concentrated in China. Lack of rare earth reserve availability in Europe.			
Environmental Risk	Presence of radioactive element; contamination of water and land. Lack of environmental investment in China rare earth mining.	Dysprosium, Neodymium	Implement and monitor proper waste and safety regulations to ensure appropriate prevention of exposure to workers and communities. Extraction of REE and harmful substances from waste stream using bioleaching techniques to reduce harmful accumulation	Zhuang 2015 ; Li et al., 2020
Social Risk	Illegal production of rare earth in China	Dysprosium, Neodymium	Enforce strict regulation and stable pricing of REEs. Protect labour rights and provide a safe working environment.	Elshkaki et al. 2014; Bonfante et al 2021; Lee et al 2020b.

B4.2 PV Critical Elements

PV technologies elements discussed are silver used in crystalline silicon (c-Si) solar cells; tellurium and cadmium used in CdTe solar cells; and indium, gallium and selenium used in CIGS solar cells. Currently, c-Si makes the majority of the market share, whereas CdTe and CIGS thin films comprise less than 6% of the total PV market (Fraunhofer ISE 2022). Other, third generation PV technologies such as perovskites are not discussed, due to the lack of information represented in the literature paper reviewed. For PV elements, the extraction of several PV critical elements is constrained by the production and refinery of the host metals, zinc and lead for the cases of indium and cadmium, copper for the cases of tellurium and selenium, and aluminium for the case of gallium. Although Silver is also extracted on its own, 70% of it is extracted together with zinc, lead and copper (Elshkaki 2013; McLellan 2016). Tellurium, gallium, and indium production are geographically limited to few countries: 60% of tellurium in China, 98% of gallium in China and 80% Indium in East Asia. However, since copper, zinc and aluminium are mined throughout the globe, further refineries could be built to overcome the possibility of geopolitical constraints. Almost all these elements would require production ramp up except for cadmium, whose historical production is greater than the expected growth (and additionally, recycling spent cadmium batteries could provide a significant supply of cadmium). Figure B.5 represents the demand projects for PV elements up to 2050 vs the current reserve estimates. Indium and selenium used in the production of CIGS solar cells are the most concerning in terms of global availability, whereas Tellurium is of concern for CdTe. Their supply for this PV technology alone could exceed the global reserve regardless of the efforts taken to their specific demand per unit of product. Silver and cadmium are instead unlikely to hinder the future growth of PV technologies. In the case of tellurium, competing demand, low recovery rates and declining copper grade ore could impact its availability and consequently CdTe PV growth. This would require improving and expanding copper refinery and extracting from other ore deposits such as gold ores. In terms of recovery, most of the elements, except for silver and cadmium, suffer from highly dispersive uses, not enough recycling facilities or lack of economic incentive to recycle small concentration in end products. In terms of social and environmental implications, there was a dearth of information in the literature papers reviewed. Table B.3 provides the literature finding's key issues and suggestion for PV elements.

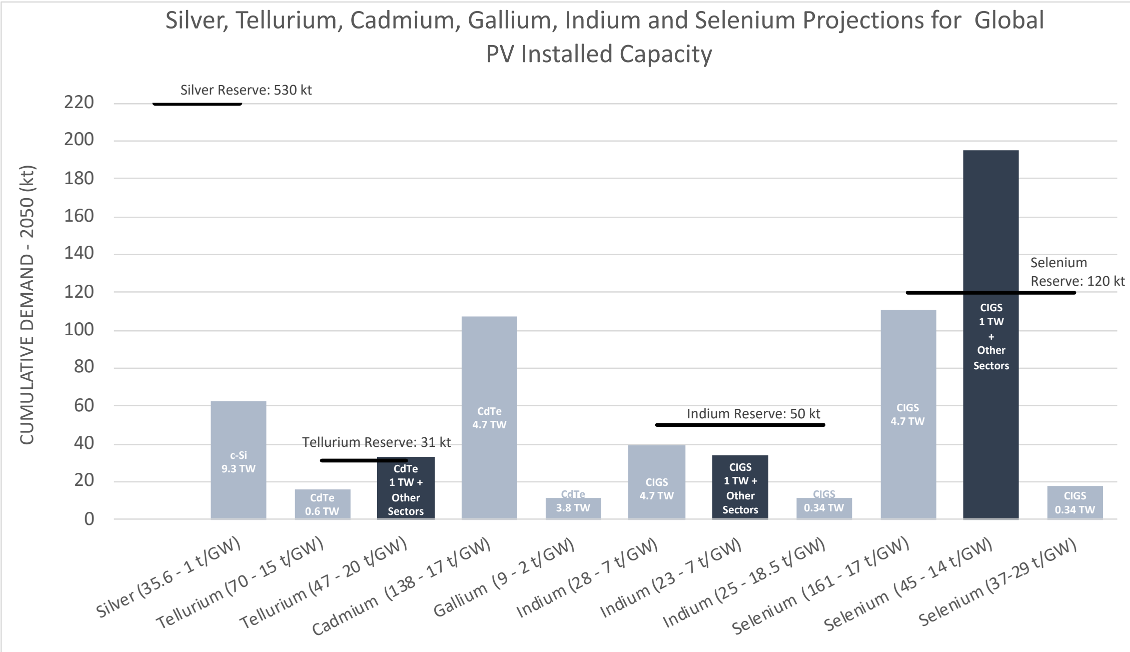

Figure B.5: Reserve estimates and cumulative demand projections for silver, tellurium, cadmium, gallium, indium, and selenium based on decreasing intensities in PV manufacturing and expected installed capacities to 2050, adapted from Elshkaki 2013; Månberger & Stenqvist 2018; Davidsson & Höök, 2017 and Zhou et al. 2020. Reserve estimates for cadmium and gallium are unavailable. Resource estimates are not available in the literature for any of these elements.

Table B.3: Summary key barriers/challenges and suggested solution through literature finding of PV critical elements.

Category	Issues	Elements	Potential Solution	Reference
Geological Availability Risk	Insufficient reserves	Tellurium Selenium Indium	Increase recovery from electrolytic copper and zinc refineries. Increase recycling and scrap supply from mine waste.	Davidsson and Höök 2017; Tokimatsu et al. 2018
	Low recovery during extraction:	Tellurium, Selenium, Gallium Indium	Improve recovery rates and refine mine waste (significant quantities available in tailings, slags, smelting, and refining processes for recovery of host metals). Have smelters with indium recovery capabilities to reduce losses.	Stamp 2014; USGS 2022; Watari et al., 2022; Zhou et al 2021

	Recycling Barriers: -High dispersion losses for tellurium gallium and selenium -Low concentration uses in end products for indium	Tellurium Gallium Selenium Indium	Improve recycling and collection of EoL products, such as LCDs for Indium	USGS 2022
Geopolitical and Regional Risk	Mining and/or refinery concentrated in a single region	Tellurium, Indium Gallium	Diversify supply by increasing refining and treatment at host element extraction.	USGS 2022; Helbig et al. 2016
Environmental Risk	Highly toxic	Cadmium	Increased use of waste cadmium to prevent harmful accumulation of cadmium on ecosystem.	Grandell & Höök 2015; Ren et al. 2021
	Mildly toxic, exposure hazardous to human health	Indium	N/A	Watari et al. 2019

B4.3 Platinum Group Metals

Platinum group metals (PGMs), i.e, ruthenium, rhodium, palladium, osmium, iridium, and platinum, are widely used in catalytic converters of internal combustion engine vehicles (ICEV) operating on fossil fuels, to remove harmful combustion chemicals from their tailpipe emissions: around 39% of platinum, 50% of palladium, and 83% of rhodium are used by the automotive industry for catalytic converters (Hao et al. 2019). With the move towards low carbon transport systems and EVs, the demand for PGMs for automobile catalytic converters is expected to drop; however, at the same time PGM use in water electrolysers for the production of "green" hydrogen to be used in fuel cell electric vehicles (FCEVs) and for stationary storage is expected to increase significantly in the coming decades (Rasmussen et al. 2019; Tong et al. 2022). Figure B.6 presents PGM group demand projection up to 2050 vs the current reserve estimates. The adoption of hydrogen storage and fuel cell technologies is expected to be hindered by geological availability constraints of PGMs, declining ore grades and continuous price fluctuations and production in politically unstable countries, which may also prevent further exploration in those same regions (Hao et al. 2020; Minke et al. 2021; Rasmussen et al. 2019). A significant proportion of platinum demand can be met by end-of-life ICEVs, if the recycling rates and circularity flows are significantly improved globally within the automotive industry (Mulvaney et al. 2021; Tong et al, 2022). Improvement is also required in terms of reducing the iridium and platinum content in electrolyzers, and platinum content in fuel cells to mitigate availability concerns. It is expected that, due to the future phase out of ICEVs, there will also be an increase in palladium and rhodium supply from recycling which can be used to substitute iridium and platinum in certain applications (Rasmussen et al. 2019; Tong et al, 2022). Deep sea deposits could also be a potential source of PGMs; however, the environmental implications are unclear (McLellan et al. 2016; Mulvaney et al. 2021). Table B.4 summaries the key literature findings.

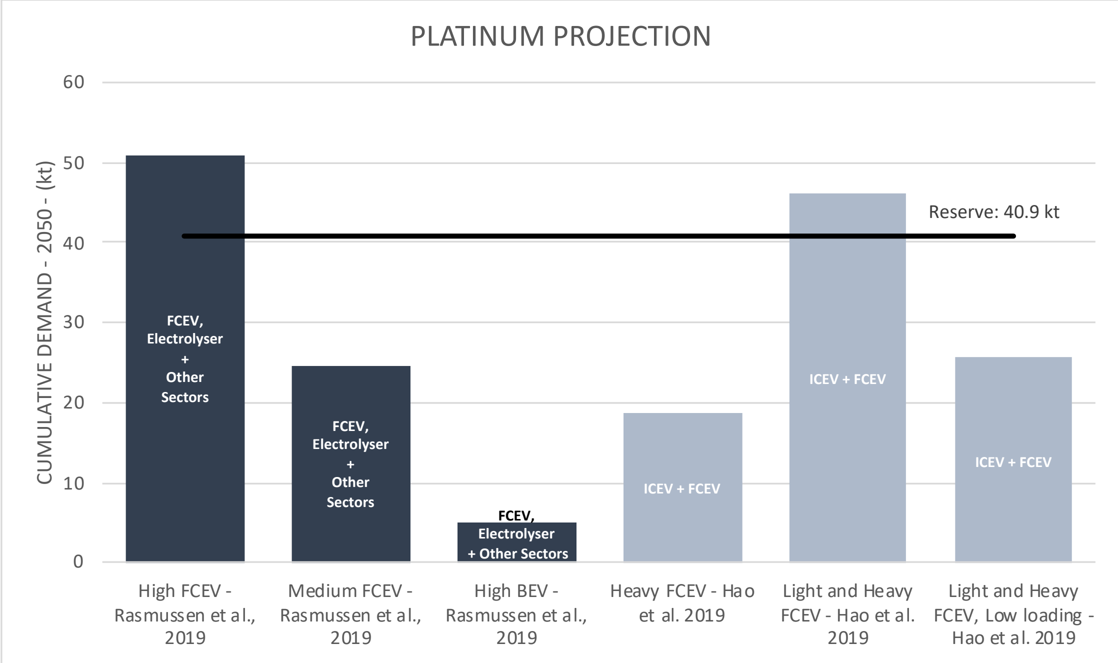

Figure B.6: Cumulative demand of platinum up to 2050 and estimated current reserve base, adopted from Rasmussem et al., 2019 and Hao et al., 2019. ICEV = Internal combustion engine vehicles, FCEV = Fuel cell electric vehicles and BEV = Battery electric vehicle. High FCEV = 30% of the vehicle share is FCEVs. High BEV = 80% of the vehicle share is BEVs and the rest is ICEVs. Resource estimates are not available in the literature for PGMs.

Table B.4: Summary of key barriers/challenges and suggested solution through literature finding of platinum group elements used in fuel cell and electrolysers for energy storage applications.

Category	Issues	Elements	Potential Solution	Reference
Geological Availability Risk	Insufficient reserves and resource constraints Significant losses during extraction process Declining ore grade in South Africa	Platinum Iridium	Deep sea mining (but the environmental implications are unclear). Improve extraction rates and increase secondary production from mine waste. Significantly increase closed-loop recycling and end-of-life collection rates. Strategic mix of BEVs and FCEVs Reduce PGM content in fuel cells and electrolysers.	Mulvaney et al. 2021; Minke et al. 2021; Hao et al. 2020; Rasmussen et al. 2019; McLellan et al. 2016; Tong et al., 2022

			Substitute with other PGMs.	
Geopolitical and Regional Risk	High proportion of mining in unstable regions	Platinum Iridium	Increase exploration and secondary supply to reduce dependency	Minke et al. 2021; Rasmussen et al. 2019
	Price fluctuation	Platinum Iridium	Substitute between other PGMs	Rasmussen et al. 2019
Environmental Risk	High environmental impact associated with PGM mining and processing	Platinum Iridium	Increase secondary supply of PGMs from end-of-life products, scraps and wastewater streams	Zhuang et al 2015; McLellan et al. 2016; Tong et al., 2022
Social Risk	Labour dispute and safety concern	Platinum	N/A	USGS, 2022

B4.4 Copper

Copper plays a vital role in the development of the on-going energy transition, from building new cables for expanding grid infrastructure to supporting the growth of energy transition technologies. Copper demand is expected to grow due to the demand for new power distribution lines and its intensive use for wiring in most low carbon energy generation and transport technologies, in addition to the expected increase for copper in developing countries for various applications (Henckens & Worrell 2020; Calvo & Valero 2021; Lee et al. 2020; Vidal et al. 2017). Figure B.7 represents copper demand projection up to 2050 2050 vs the current reserve and resource estimates. Copper demand is expected to grow between 30 to 102 Mt in 2050; an increase of 102 Mt in 2050 would mean that most of the known resource would be depleted (Vidal et al. 2017; Bonnet et al. 2019; Watari et al. 2022). Copper is internationally traded in many different forms across the supply chain. The literature identified no major geopolitical concerns related to copper supply, although there has been mention that increasing copper demand growth rates may make it expensive for future generations, especially in less wealthy nations, to obtain (Henckens & Worrell 2020). It is expected that most of the demand will be driven by the building and construction sector, followed by increases in global EV market and renewable generation (Bonnet et al. 2019; Watari et al. 2022). Improvements in recycling and recovery of copper alongside of substitution could mitigate some of the availability concerns (Watari et al. 2022). Further concerns are related to the deteriorating copper ore, which would mean more emission and energy requirement per unit of commodity (Mulvaney et al. 2021). Solar copper mines in sunny regions such as Chile (which represents a major copper production region), improvements in energy efficiency of mining, and electrification of heating processes could help reduce energy investment and emission in the copper sector (Haas et al. 2020; Watari et al. 2022). Table B.5 summaries the key literature findings.

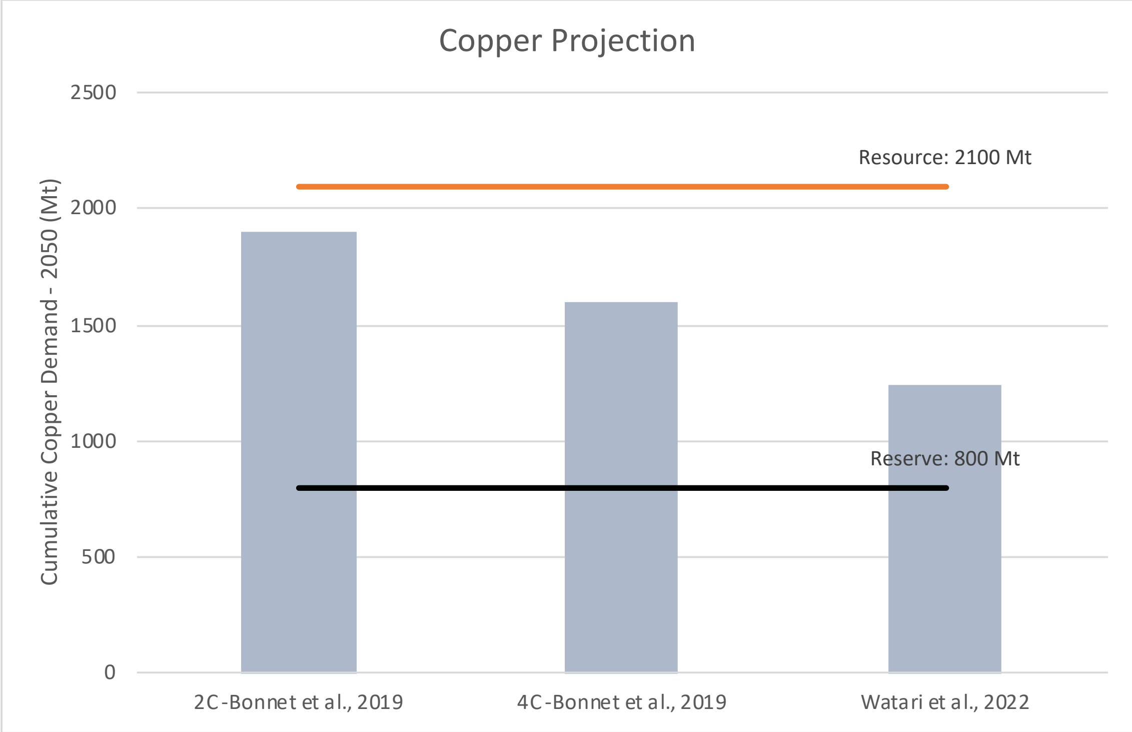

Figure B.7: The cumulative demand of copper up to 2050 and estimated current resource and reserve base, adopted from Bonnet et al., 2019 (current recycling rate - 45%) and Watari et al., 2022. 2C: 2-degree climate scenario, 4C: 4-degree climate scenario.

Table B.5: Summary of Copper used in energy technologies, wiring and other.

Category	Issues	Elements	Potential Solutions	References
Geological Availability Risk	Insufficient reserves and resource	Copper	Improve copper production efficiency (copper smelting and refining). Significantly increase recycling and end-of-life collection rates of copper products and scraps. Improve material efficiency and substitution. Encourage shared practices of certain copper end products. Deep sea mining (but the environmental implications are unclear).	Watari et al, 2022; Månberger and Johansson 2019; Henckens and Worrell 2020; Bonnet et al. 2019

Environmental Risk	Increase in energy demand and emissions due to decline in copper ore grade	Copper	Electrify mining processes, improve energy efficiency, and use renewable generation for mining. Increase recycling and end-of-life collection of copper end products and scraps.	Lee et al. 2020; Haas et al. 2020; Watari et al. 2022
Social Risk	Social unrest due to increase in pollution, lack of environmental compensation and inconsistent displacement of local communities	Copper	N/A	Vakulchuk 2021; Ren et al. 2021

Appendix C: Average Lithium-ion Battery Pack Mass

Share of Registered UK Light Duty Vehicles in 2017	
Mini	2.8%
Supermini	32.4%
Lower Med	27.3%
Upper Med	9.5%
Exec	4.7%
Luxury	0.4%
Sports	1.8%
Dual Purpose	16.3%
Multi-purpose	4.7%
Total	**100%**

Table C.1: Represents the market share of LDV in the UK in 2017

BEV Make	Current EV Market Share[14]	Type of Light Duty Vehicle	Projected BEV market Share (based on	Battery Capacity (KWh)	Battery weight (kg)

[14] Represents the market share of BEV in the UK in 2019 alongside their current battery capacity and driving weight.

			Registered Vehicles)[15]		
NISSAN LEAF	17%	Small family - Lower Med	20%	40	272
NISSAN LEAF	17%	Small family - Upper Med	10%	60	303
BMW I3	9%	Supermini	15%	42	272
VOLKSWAGEN GOLF	6%	Small family - Lower Med	7%	32	303
TESLA MODEL 3	12%	Executive	4%	73	478
RENAULT ZOE	12%	Supermini	20%	52	326
TESLA MODEL S	11%	executive	3%	95	625
NISSAN E-NV200	6%	Small Van	16%	40	272
JAGUAR I-PACE	6%	SUV	2%	85	603
TESLA MODEL X	6%	SUV	2%	95	625
Weighted Average[16]				**49.4**	**323**

Table C.2: Average battery weight and capacity based on vehicle market share.

[15] Market share of BEV recalculated based on market share of registered LDV in 2017.

[16] Sum product of battery capacity/weight and projected market share of BEV.

Appendix D: Material Flow Analysis Structure

D.1 Vehicle Fleet

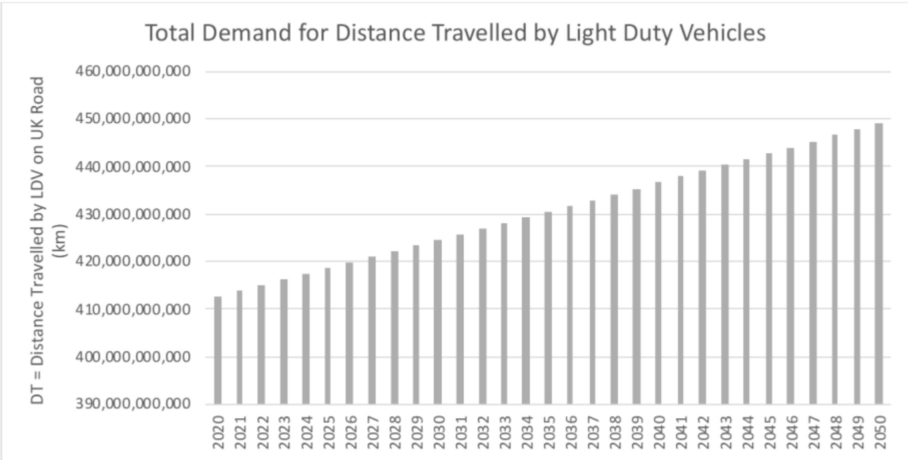

Figure D.1: The total demand for distance travelled by light duty based on the DfT Road traffic forecasts 2018 “reference” scenario given for combined Wales and England. It is assumed all light duty vehicles in the UK will follow the same trend for simplification.

D.2 Grid Battery Storage

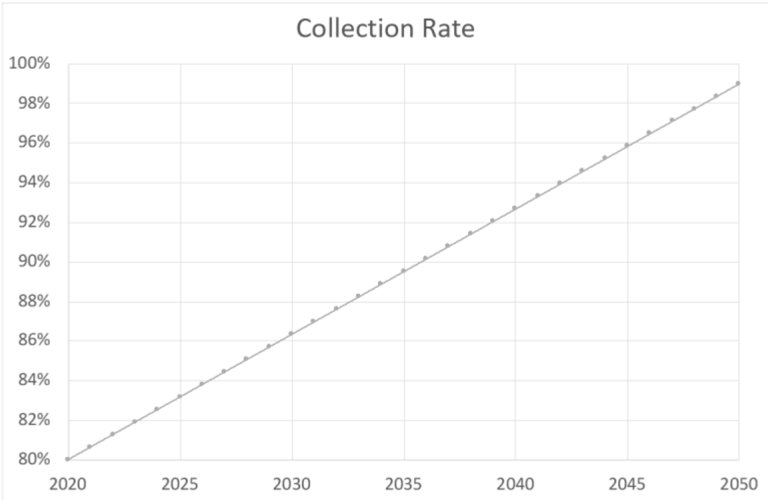

Figure D.2: The collection Rate is assumed to grow linearly from 80% in 2020 to 99% by 2050. The “Worst-Case” assumes zero collection rate of end-of-life EV batteries.

Appendix E: Electricity Grid Mix

E.1: Life Cycle Assessment Results

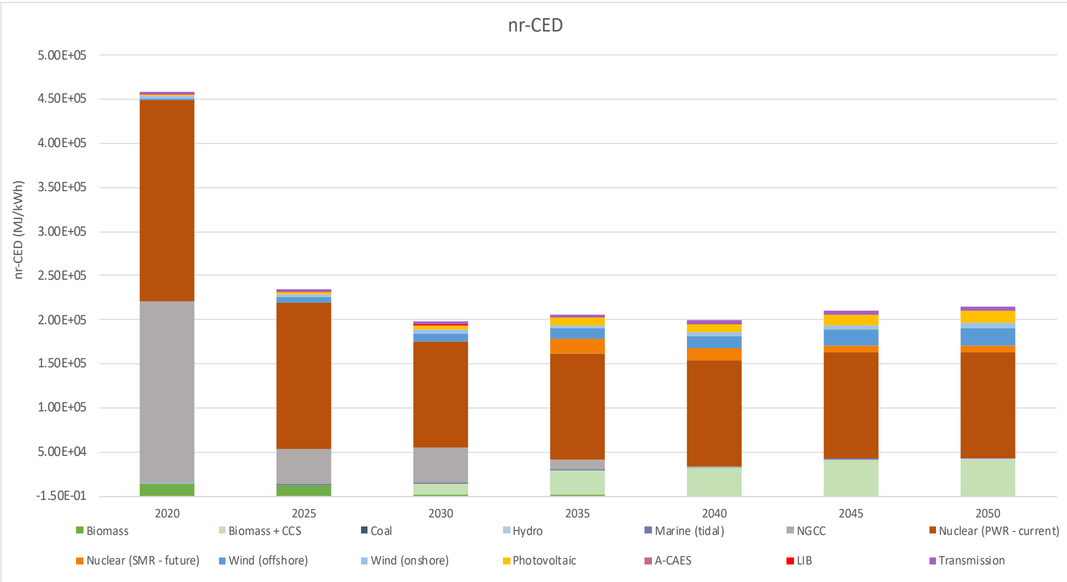

Figure E.1: Represents the UK grid mix non-renewable cumulative energy demand per kWh of delivered electricity from 2020 to 2050 for National Grid 2020 "Leading the way

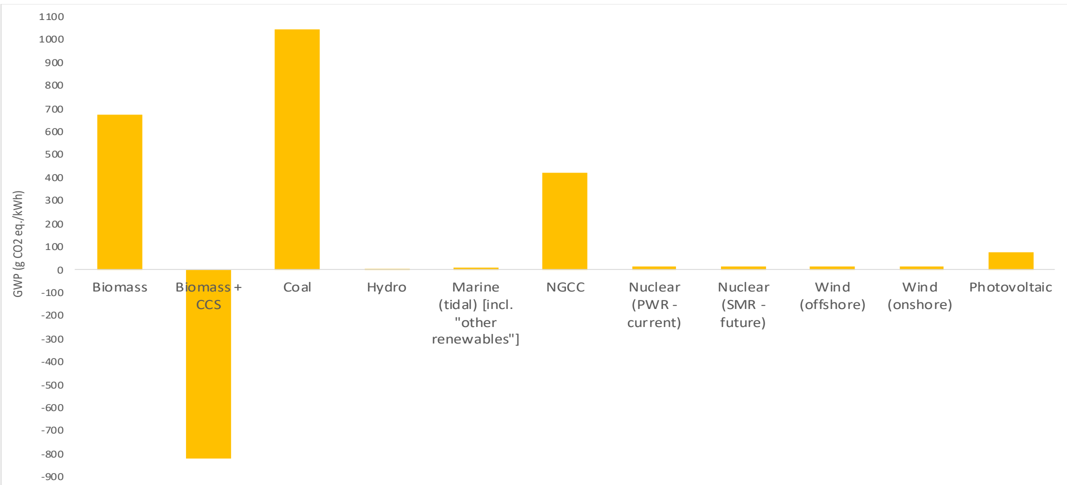

Figure E.2: Represents the global warming potential impact of each energy source technology per kWh of delivered electricity.

Appendix F: Lithium-ion Battery Life Cycle Inventory Information

F.1: LiOH Production

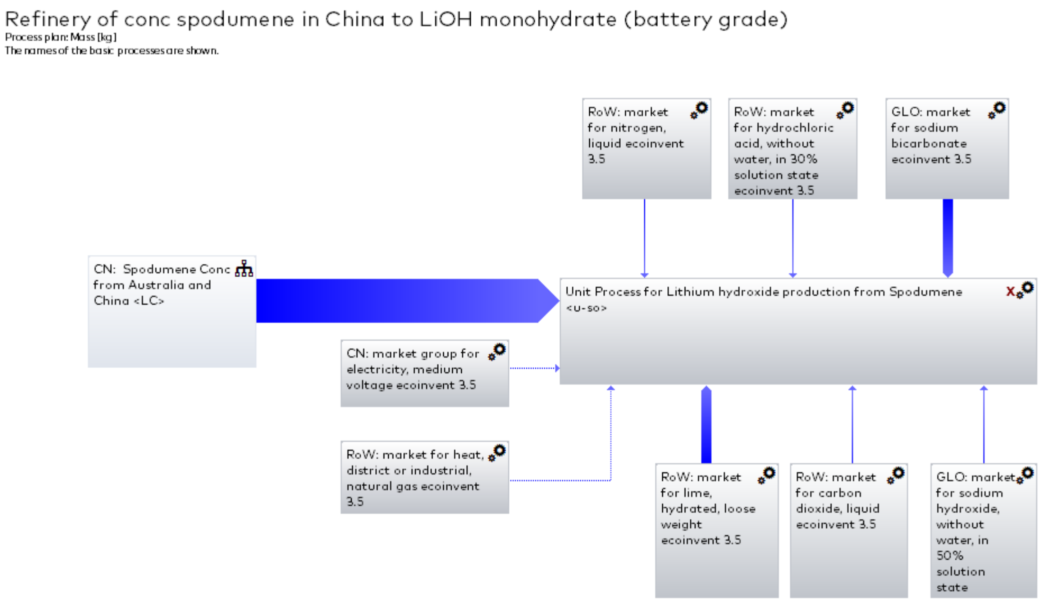

Figure F.1: Material Inventory diagram for refinement of concentrated spodumene rock in China

Flows	Quantities	Amount	Units
Calcium hydroxide	Mass	1.33	kg
CN: electricity, medium voltage, at grid	Energy (net calorific value)	43.92	MJ
GLO: sodium hydroxide, without water, in 50% solution state	Mass	0.112	kg
GLO: spodumene	Mass	5.81	kg

RER: carbon dioxide liquid, at plant	Mass	0.084	kg
RER: nitrogen, liquid, at plant	Mass	0.4	kg
RoW: heat, district or industrial, natural gas	Energy (net calorific value)	42.5	MJ
RoW: hydrochloric acid, without water, in 30% solution state	Mass	0.072	kg
Soda (sodium carbonate)	Mass	1.58	kg
CH: disposal, gypsum, 19.4% water, to inert material landfill	Mass	1.15	kg
CH: disposal, aluminium, 0% water, to municipal incineration	Mass	5.73	kg

Table F.1: Material Inventory for refinement of concentrated spodumene rock in China

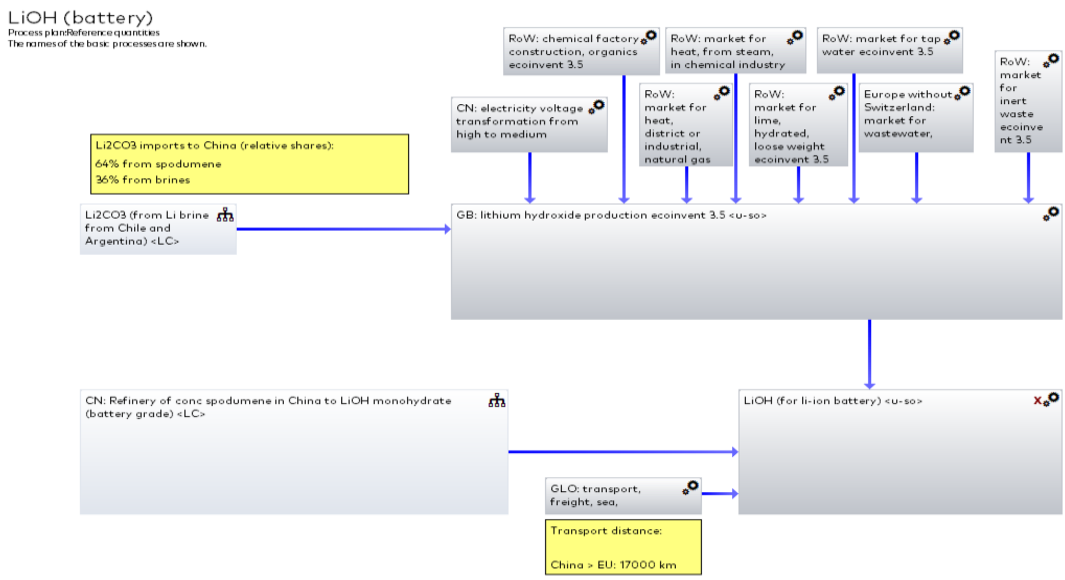

Figure F.2: Material Inventory diagram for the production of lithium hydroxide (LiOH)

Flows	Quantities	Amount	Units

Input			
Calcium hydroxide	Mass	1.33	kg
CN: electricity, medium voltage, at grid	Energy (net calorific value)	43.92	MJ
GLO: sodium hydroxide, without water, in 50% solution state	Mass	0.112	kg
GLO: spodumene	Mass	5.81	kg
RER: carbon dioxide liquid, at plant	Mass	0.084	kg
RER: nitrogen, liquid, at plant	Mass	0.4	kg
RoW: heat, district or industrial, natural gas	Energy (net calorific value)	42.5	MJ
RoW: hydrochloric acid, without water, in 30% solution state	Mass	0.072	kg
Soda (sodium carbonate)	Mass	1.58	kg
CH: disposal, gypsum, 19.4% water, to inert material landfill	Mass	1.15	kg
CH: disposal, aluminium, 0% water, to municipal incineration	Mass	5.73	kg
Output			
lithium hydroxide (from brine)	Mass	1	kg

Table F.2: Material Inventory for the production of lithium hydroxide (LiOH).

Flows	Quantities	Amount	Units
Input			
GLO: transport, freight, sea, transoceanic ship with reefer, cooling	Ecoinvent quantity ton kilometer (tkm)	17	tkm
lithium hydroxide (from brine)	Mass	0.36	kg
lithium hydroxide (from hard rock)	Mass	0.64	kg
Output			
lithium hydroxide (battery grade)	Mass	1	kg

Table F.3: Material Inventory diagram for the production of lithium hydroxide (LiOH).

F.2: NMC LIB Manufacturing

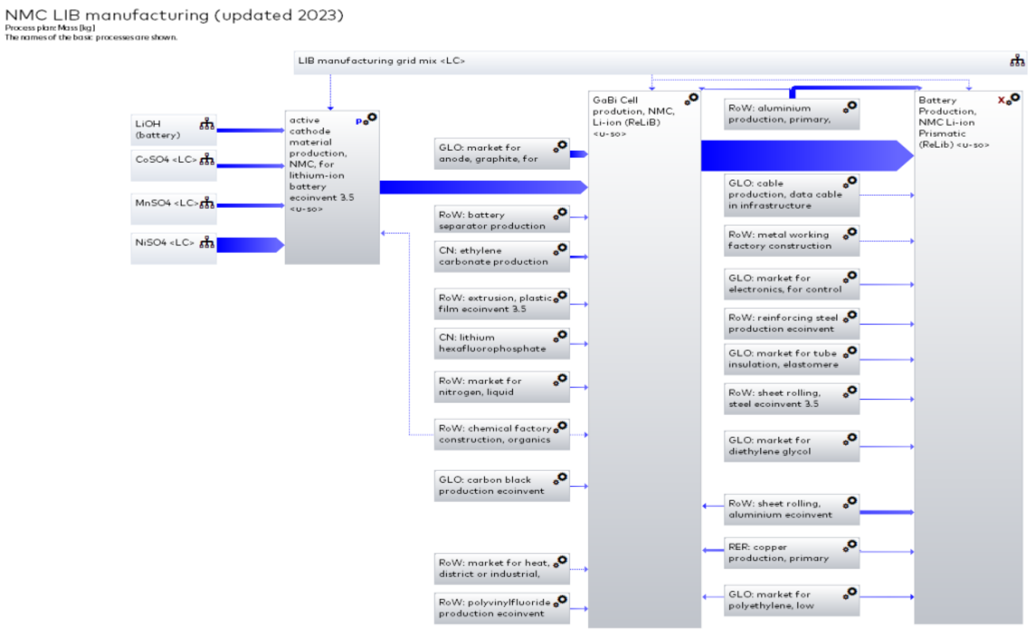

Figure F.3: Material inventory for electric vehicle lithium-ion battery manufacturing

Flows	Quantities	Amount	Units
Input			
CoSO4	Mass	0.366924133	kg
LiOH	Mass	0.283464478	kg
MnSO4	Mass	0.358351403	kg
NiSO4	Mass	1.101842659	kg
CN: electricity, medium voltage	Energy (net calorific value)	25.2	MJ
GLO: chemical factory, organics	Number of pieces	4.00E-10	pcs.
Output			
cathode, NMC [ReLIB]	Mass	1	kg

Table F.4: Material Inventory for electric vehicle lithium-ion battery active cathode manufacturing. The inventory is provided for the year 2020. the amount of CoSO4, LiOH, MnSO4 and NiSO4 is dictated by the change in active cathode chemistry (NMC 611 to NMC 811) and the share of virgin material met through closed-loop recycling.

Flows	Quantities	Amount	Units
Input			
CN: electricity, medium voltage	Energy (net calorific value)	89.7	MJ
GLO: aluminium, wrought alloy	Mass	11.5	kg
GLO: anode, graphite, for lithium-ion battery	Mass	62.52	kg
GLO: battery separator	Mass	11.75	kg

GLO: Carbon black, at plant	Mass	2.21	kg
GLO: cathode, $LiMn_2O_4$, for lithium-ion battery	Mass	105.87	kg
GLO: chemical factory, organics	Number of pieces	4.00E-08	pcs.
GLO: ethylene carbonate	Mass	22.38	kg
GLO: extrusion, plastic film	Mass	5	kg
GLO: lithium hexafluorophosphate	Mass	4.01	kg
GLO: polyethylene, low density, granulate	Mass	3.22	kg
GLO: sheet rolling, aluminium	Mass	11.5	kg
RER: copper, primary, at refinery	Mass	20.25	kg
RoW: heat, district or industrial, natural gas	Energy (net calorific value)	15.23	MJ
RoW: nitrogen, liquid	Mass	2.35	kg
US: polyvinylfluoride, at plant	Mass	3.48	kg
Output			
battery cell, Li-ion	Mass	235.02	kg

Table F.5: Material inventory for electric vehicle lithium-ion battery cell manufacturing

Flows	Quantities	Amount	Units
Input			
DE: tube insulation, elastomere	Mass	1.05	kg
GLO: battery cell, Li-ion	Mass	235.02	kg

GLO: cable, data cable in infrastructure	Length	30	m
GLO: electricity, low voltage	Energy (net calorific value)	81	MJ
GLO: metal working factory	Number of pieces	4.58E-08	pcs.
GLO: reinforcing steel	Mass	1.76	kg
GLO: sheet rolling, steel	Mass	1.76	kg
RER: aluminium, cast alloy	Mass	40.84	kg
RER: copper, primary, at refinery	Mass	0.52	kg
RER: diethylene glycol, at plant	Mass	8.65	kg
RER: electronics, for control units	Mass	6	kg
RER: polyethylene, LDPE, granulate, at plant	Mass	0.13	kg
sheet rolling, aluminium	Mass	40.84	kg
Output			
battery, Li-ion, rechargeable, prismatic	Mass	293.3	kg

Table F.6: Material inventory for electric vehicle lithium-ion battery pack manufacturing

Parameter	Flows	Quantities	Amount	Units
Input				
Hungary	HU: electricity mix	Energy (net calorific value)	50	MJ
Poland	PL: electricity mix	Energy (net calorific value)	70	MJ

Sweden	SE: electricity mix	Energy (net calorific value)	32	MJ
Output				
Mix	RER: electricity, medium voltage	Energy (net calorific value)	152	MJ

Table F.7: Material inventory for electric vehicle lithium-ion battery manufacturing grid mix

F.3: LIB Recycling

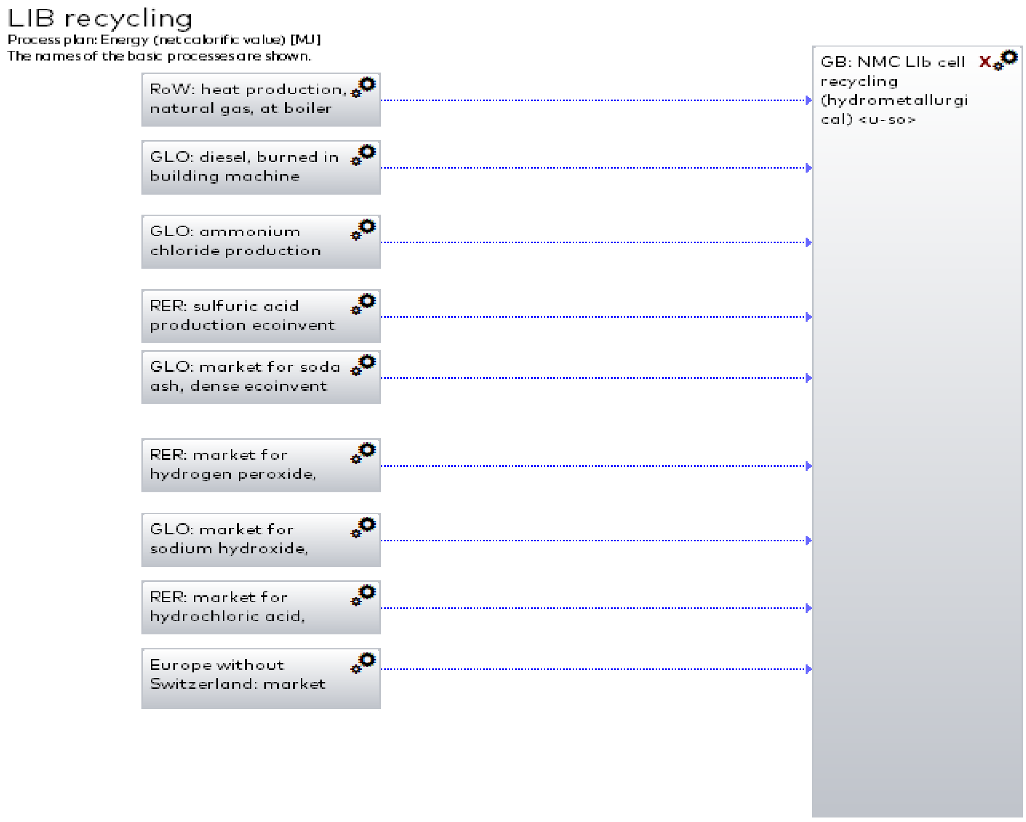

Figure F.4: Material inventory diagram for the operation of lithium battery cell recycling.

Flows	**Quantities**	**Amount**	**Units**
ammonium chloride	Mass	0.022	kg

GB: electricity, medium voltage, at grid	Energy (net calorific value)	2.088	MJ
GLO: diesel, burned in building machine	Energy (net calorific value)	0	MJ
hydrochloric acid, without water, in 30% solution state	Mass	0.0085	kg
RER: hydrogen peroxide, without water, in 50% solution state	Mass	0.26	kg
RER: natural gas, burned in industrial furnace low-NOx >100kW	Energy (net calorific value)	0	MJ
RER: sodium hydroxide, without water, in 50% solution state	Mass	0.4	kg
RER: sulfuric acid	Mass	0.77	kg
RER: tap water, at user	Mass	2.69	kg
soda ash, dense	Mass	0.015	kg

Figure F.8: Material inventory for the operation of lithium battery cell recycling.

Appendix G: Latest Update After Paper Publication

Chapter 4: At the time of publication, the paper was based on National Grid Electricity Future Energy Scenario (FES) 2019 (National Grid, 2019). This has been updated to FES 2020 (National Grid, 2020) in this chapter with the addition of baseline and TaaS scenarios, to take account of reduced demand for purpose-built grid storage battery due to second life EV batteries and uptake of TaaS. This form the part of the full hybrid LCA for the evolution of the transition to BEV. The main changes from FES 2019 to FES 2020, is that biogas from anaerobic digestion of organic matter such as waste food is no longer used to meet electricity demand. Instead, biogas is refined to produce biomethane to achieve decarbonization of gas network.

Chapter 6: This chapter represents the full hybrid LCA for the evolution of the transition to BEV. The paper based on this chapter was updated to reflect recent changes. Battery supply chain was updated to represent the current global production. The spodumene production and refining route was also updated based on the current processing method (see Chapter 6, Section 3). The manufacturing of electric vehicle lithium-ion battery was updated from China to EU to also reflect the current progress on battery manufacturing in Europe (see Chapter 2, Section 3). The lithium-ion battery production was updated to be based on the GREET model 2020 to reflect the latest material inventory for battery production.