STUDY

Requested by the TRAN Committee

Environmental challenges through the life cycle of battery electric vehicles

ANNEXES

Transport and Tourism

EN

Policy Department for Structural and Cohesion Policies
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RESEARCH FOR TRAN COMMITTEE

Environmental challenges through the life cycle of battery electric vehicles

PART II - ANNEXES

Abstract

This study provides an up-to-date expert assessment and comparison between the life cycle's carbon footprint of battery electric and internal combustion engine passenger cars. It presents evidence from the literature and from life cycle assessment modelling and concludes with policy recommendations. The analysis includes sensitivities, regional variations for six Member States, and also the effects of technical and legislative development on the potential outlook up to 2050.

This document was requested by the European Parliament's Committee on Transport and Tourism.

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LIST OF ABBREVIATIONS

AFID/AFIR	Alternative Fuels Infrastructure Directive/Regulation
ARD_MM	Abiotic Resource Depletion – Minerals & Metals
AQP	Air Quality Pollutants
B7	7%vol biofuel blend in diesel
BAU	Business As Usual
BEV	Battery Electric Vehicle (fully electric)
CBAM	Carbon Border Adjustment Mechanism
CED	Cumulative Energy Demand
CH_4	Methane
Co	Cobalt
CO	Carbon Monoxide
CO_2	Carbon Dioxide
CO_2e	Carbon Dioxide equivalent
CRM	Critical raw material
EC	European Commission
EoL	End-of-Life
ELV	End-of-Life Vehicle
ETS	Emission Trading Scheme
EV	Electric Vehicle
FAME	Fatty Acid Methyl Ester (Biodiesel)
FCEV	Fuel Cell Electric Vehicle (running on hydrogen)
FU	Functional Unit

GHG	Greenhouse Gases
GWP	Global Warming Potential
H_2	Hydrogen
HEV-G	Hybrid Electric Vehicle, with Gasoline ICE
HREE	Heavy Rare Earth Elements
HTP	Human Toxicity Potential
HVO	Hydrotreated Vegetable Oil (Renewable Diesel)
ICE	Internal Combustion Engine
ICEV	Internal Combustion Engine Vehicle
ICEV-D/G	Diesel/Gasoline ICE Vehicle
IEA	International Energy Agency
ISO	International Organisation for Standardisation
kWh	kilo-Watt-Hour
LA	Lifetime Activity
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCV	Light Commercial Vehicle (van)
LDV	Light Duty Vehicle (Car or LCV)
LFP	Lithium-Ion-Phosphate (battery chemistry)
LHV	Lower Heating Value
Li	Lithium
LIB	Lithium ion battery
Li-ion	Lithium Ion
LMO	Lithium-Manganese Oxide (battery chemistry)

MD	Medium Duty
MJ	Mega-Joule
Mn	Manganese
N_2O	Nitrous Oxide
NCA	Lithium-Nickel-Cobalt-Aluminium Oxide (battery chemistry)
NEDC	New European Drive Cycle
Ni	Nickel
NH_3	Ammonia
NMC	Lithium-Nickel-Manganese-Cobalt Oxide (battery chemistry)
NO_X	Nitrogen Oxides (includes nitrogen monoxide and nitrogen dioxide)
OEM	Original Equipment Manufacturer
PCR	Product Category Rules
PEF	Product Environmental Footprints
PHEV	Plug-in Hybrid Electric Vehicle
PMF	Particulate Matter Formation
POCP	Photochemical Ozone Creation Potential
PtX	Power-to-X (where X can be a variety of hydrocarbon liquid fuels or gases)
PV	[Solar] Photo Voltaic
RE	Renewable Energy/Electricity
REE	Rare Earth Elements
RES	Renewable Energy Sources
REEV	Range Extended Electric Vehicle
RW	Real world
SO_2	Sulphur Dioxide

SO_2e	Sulphur Dioxide equivalent
SoC	Available State-of-Charge percentage for battery
SUV	Sports Utility Vehicle
TCO	Total Cost of Ownership
TTW	Tank-to-Wheel
VO	Vehicle Occupancy
VOC	Volatile Organic Compound
V2G	Vehicle to Grid
WLTP	Worldwide harmonised Light vehicle Test Procedure
WTT	Well-to-Tank
WTW	Well-to-Wheel
xEV	Generic term to refer to all electric vehicles (includes BEVs, PHEVs, REEVs and FCEVs)
ZEV	Zero Emission Vehicle (includes BEV and FCEV)

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ANNEXES

ANNEX 1: FREQUENTLY ASKED QUESTIONS (FAQ)

This annex provides short summary responses to common questions, criticisms and misconceptions with regards to the LCA comparisons of ICEV versus BEV. Where relevant, links have been also provided to areas of this report where there is further/more detailed information provided on the topic.

#	FAQ (Frequently Asked Questions)
1	"Is calculating impacts of electricity use by BEV based on grid averages "wrong", or does it significantly over-estimate their benefits relative to ICEVs?"[1]
2	"Will there be enough lithium to enable a global shift to BEVs?"
3	"Do the increased greenhouse gas (GHG) emissions resulting from manufacturing of electric vehicles (EVs) outweigh their reduced emissions (vs ICEVs) in use?"
4	"BEVs have no tailpipe GHG emissions, but have the same or even higher, GHG impacts than for ICEVs often generated upstream, in the supply chain of the electricity used to recharge the batteries?"
5	"Will end-of-life battery packs end up in landfills and cause huge environmental impacts?"
6	"BEVs are heavier than ICEVs; does this lead to higher particulate emissions (due to brake pad and tyre wear)?"
7	"Why the focus on BEVs, if an ICEV running on synthetic/e-fuel produced from renewable electricity can reduce more life cycle GHG impacts than a BEV?"
8	"BEV energy consumption is much higher (/range is lower) in colder climates; does this significantly reduce their potential life cycle benefits versus ICEVs?"
9	"Does the life cycle GHG impacts of BEVs end up being higher than those of ICEVs, because the battery pack needs to be replaced over the course of their service life?"

#1 Is calculating impacts of electricity use by BEV based on grid averages "wrong", or does it significantly over-estimate their benefits relative to ICEVs?

In order to answer this question meaningfully, one has to consider the intended time frame and overall aim of the assessment. Broadly speaking, two alternative cases may be considered:

1) The intention is to calculate the total GHG impact of a BEV over its full service life and assuming that the latter will span a relatively long period of time (e.g. 10+ years).

[1] As has been suggested in an *open letter to the European Parliament.*

In this case, using the "average" grid mix composition to model the electricity input to BEV battery charging may be regarded as the most valid approach, provided that such average composition is estimated by means of a suitable dynamic model[2] which:

(a) captures the expected evolution of the grid mix itself over time; and

(b) is compatible with the gradually increased overall demand for electricity caused by the growing BEV fleet as a whole over the same time period.

It is worth pointing out that, in so doing, such dynamic "average" electricity grid mix model will implicitly entail an element of "consequential" LCA, even if the main BEV LCA itself may still be characterised as a fundamentally "attributional"[3] assessment – see terminology in Annex 2.

2) The intended aim of the assessment is to calculate the initial GHG impact of a BEV over a relatively short period of time (e.g. only a few years), which is characterised by a concomitant rapid increase in BEV penetration in the fleet as a whole.

In this second case, it may be more reasonable/realistic to assume that the resulting surge in electricity demand for battery charging would likely have to be met by ramping up electricity generation by those technologies in the existing grid mix that are readily dispatchable (such as, e.g. natural gas turbines). Hence, the use of a so-called "marginal" grid mix composition model (different from the "average" described above) would be more appropriate.
From a purely methodological perspective, an LCA framed and carried out in this way should be clearly identified as a "consequential"[4] LCA overall, with the additional important caveat/qualifier that the temporal scope of interest is restricted to a shorter time frame (e.g. a few years only). However, such reduced time frame may still encompass the full life cycle of the vehicle if the latter is assumed to be used intensively (e.g. as a taxi cab, or as part of a shared mobility scheme).

Based on all of the above, it can be concluded **the intended time frame and overall aim of the assessment are crucial in co-determining the appropriateness of calculating impacts of electricity use by BEV based on grid averages, with no "one size fits all" clear-cut answers**.

#2 "Will there be enough lithium to enable a global shift to BEVs?"

The extent to which the global availability of lithium may end up representing a bottleneck to the widespread adoption of BEVs worldwide (and to their replacing ICEVs) is not yet fully clear, and it will depend on a number of factors. On the one hand, it has been estimated that, if no improvements were made to end-of-life lithium recovery, nor any new alternative battery concepts were brought onto the market to contribute to the battery technology mix, then indeed the global lithium demand for BEVs may outstrip the available reserve by 2050. However, on-going advancements in hydrometallurgical lithium-ion battery (LIB) recycling are expected to render end-of-life lithium recovery economically viable soon, thereby significantly reducing the pressure on primary lithium supply chains in the future. Also, new sources of lithium have been identified (including geothermal sources, clay deposits, and deep seafloor deposits) which could potentially complement and increase current reserve estimates. Finally, new developments in sodium-ion battery (NIB) technologies are showing promise in terms

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[2] For example, the transport and wider energy system modelling conducted by the European Commission on the impacts future policy scenarios – e.g. (European Commission, 2021a).

[3] Meaning that the LCA is intended to assess the environmental impacts of one unit of product, without explicitly addressing the indirect effects arising from changes to the larger system into which the product is embedded (e.g. the whole fleet, or the grid).

[4] Meaning that the LCA is squarely aimed at providing information about the consequences of a large-scale deployment of the product, including on the larger system into which the product itself is embedded.

of potentially providing a viable alternative to lithium-ion batteries for BEVs, particularly in the more cost-sensitive passenger car segments.

To summarise, when duly accounting for the multiple co-evolution trajectories of lithium mining, **lithium recycling and new battery chemistries, it appears unlikely that a severe lithium shortage will prevent a global shift to BEVs**.

Further information on this can also be found in Section 3.8.1 of this report.

#3 "Do the increased GHG emissions resulting from manufacturing of EVs outweigh their reduced emissions (vs ICEVs) in use?"

It is true that more GHG emissions are produced in the supply of raw materials and manufacturing of BEVs compared to ICEVs. The net life cycle impacts depend in particular on the operation of the vehicle and in particular the electricity mix used to power BEVs. In recent years, substantial improvements have been made to both batteries and in the electricity generation mix used in Europe. Findings from recent robust LCA of BEVs from the literature using up-to-date data and the detailed modelling conducted for this project, show that the additional GHG emissions from BEV manufacturing are far outweighed by the substantial reductions in the use phase in the EU. **Even for countries with the very worst/dirtiest electricity generation mixes in Europe, current life cycle GHG impacts of BEVs are expected to be similar to or better than conventional vehicles.** In the future, with national and European policy driving the uptake of renewables and other low-carbon generation, the advantage of BEVs in terms of life cycle GHG impact is expected to further increase compared to conventional ICEVs.

Further information on this can be found in Section 3.7 and Section 5.2.1 of this report.

#4 "BEVs have no tailpipe GHG emissions; but have the same or even higher GHG impacts than for ICEVs often generated upstream, in the supply chain of the electricity used to recharge the batteries?"

EVs are highly efficient, requiring far less energy to power them than conventional vehicles (typically a third to a quarter of the energy for cars), particularly in urban applications, but also on high-speed roads. As a result, **in the vast majority of cases, the use of BEVs results in very considerably less GHG impacts, when considering emissions from production and use of fuels and electricity on a consistent basis.** Even in the countries with some of the most carbon intensive electricity mixes in Europe, the GHG impact resulting from the operation of BEVs are expected to be below those of equivalent ICEVs in typical operating conditions.

Further information on this can also be found in Section 3.5 and Section 5.2.1 of this report.

#5 "Will end-of-life battery packs end up in landfills and cause huge environmental impacts?"

In Europe, the requirements for end-of-life (EoL) treatment of passenger cars and of batteries used in EVs are controlled by European legislation (i.e. the End of Life Vehicle Directive – (2000/53), and Batteries Directive (2006/66)), and national implementations of this. These **European regulatory instruments already set out the requirements for the collection, recycling and recovery of materials from vehicles and their batteries, to also help minimise environmental impacts.** The recent European Commission proposal for a Battery Regulation (anticipated to be formerly adopted in early 2023), will further strengthen this legislation – setting out more stringent requirements for battery collection, recycling and material recovery rates. Furthermore, there are valuable scarce minerals used in EV batteries, which provides a strong economic incentive to recycle and recover them, and most vehicle manufacturers already have arrangements with 'battery recycling partners' for treatment of their vehicle models in Europe. In many cases, it is expected that there will be potential utility still available in EV batteries after the end of their first life in the vehicle, and there is significant research and exploration into options for repurposing EV batteries for 'second life' applications, prior to their final end-of-life recycling.

Further information on this can also be found in Section 6 of this report.

#6 "BEVs are heavier than ICEVs; therefore, does this lead to higher particulate emissions (due to brake pad and tyre wear)?"

Whilst BEVs do not have exhaust emissions, there are still particulate emissions resulting from tyre and brake wear (as well as road abrasion). **Whilst it is likely for tyre wear emissions from BEVs to be higher, due to their higher weight, brake wear emissions are likely to be lower due to regenerative braking** (Ricardo, 2021a). However, there are currently no robust datasets available on such emissions for BEVs versus conventional vehicles (i.e. ICEVs), as there are no established official test methods to measure such emissions in an accurate and repeatable way (though the topic of ongoing research). From a legislative perspective, the recent Euro 7 proposal from the European Commission also aims to address tyre and brake wear emissions in the future (European Commission, 2022a). Therefore, **it is expected that in the future such emissions will be both measured and regulated to mitigate/reduce the potential for adverse impacts from all vehicle types.**

Further information on this can also be found in Section 5.2.1 of this report.

#7 "Why the focus on BEVs, if an ICEV running on synthetic/e-fuel produced from renewable electricity can reduce life cycle GHG impacts by more than a BEV?"

One of the advantages of electric powertrains used in BEVs is that they are very significantly more efficient than ICEVs, and the losses from transmission and distribution of renewable electricity are relatively small. Whilst the generation of renewable electricity may be considered zero emission, there are still emissions associated with the production of generation equipment, and for e-fuel production and distribution. Some studies (and many 'well-to-wheel' studies that are not full life cycle assessments) exclude some or all of these GHG emissions from the boundaries of their analysis (however they are typically included in full LCA). This can mean that conventional vehicles operating on 100% e-fuels look better than BEVs when manufacturing emissions are included. However, whilst the emissions from generation equipment are relatively small**, because the production of e-fuels and their use in ICEVs consumes many times more energy, they can become more significant,** and can outweigh the additional emissions from battery manufacturing and those resulting from electricity use for BEVs (i.e. when also assuming operation on renewable electricity).

It is generally agreed that e-fuels will have a valuable role in efforts to decarbonise transport and the wider energy-system. However, the availability of renewable electricity is limited (and is anticipated to remain so for the foreseeable future, with the economy-wide demands for this to aid decarbonisation). Therefore, it is important to make the most efficient use of this resource as possible (which is in a BEV for passenger cars, in the majority of situations). There are other sectors/applications (in transport and more broadly) where electrification is more challenging or not feasible as an option to mitigate GHG emissions and impact, therefore **e-fuels are considered by many to be more effectively and efficiently prioritised towards these applications**.

Further information on this can also be found in Section 5.4.2 of this report.

#8 "BEV energy consumption is much higher (/range is lower) in colder climates; does this significantly reduce/remove their potential life cycle benefits versus ICEVs?"

EVs are highly efficient, which means that the energy required to provide heating (for passenger comfort) in colder conditions is a much higher share of their energy consumption compared to ICEVs. In addition, battery performance and charging are negatively affected by very cold conditions. This results in significantly higher energy consumption – and consequently also lower electric range, and higher GHG impact when operating in colder real-world conditions. However, the fuel efficiency of ICEVs is also negatively affected by cold conditions, and whilst proportionally it is a smaller effect, the in-use operational emissions for ICEVs are much higher than for BEVs, so it is significant in absolute GHG impact terms. Analysis has been conducted on the overall impacts for BEVs and ICEVs in an unrealistically extreme situation – i.e. operating the entire vehicle lifetime at minus 10 °C. This **analysis has shown that this only has a relatively small effect on the comparison of the life cycle GHG impacts of BEV versus ICEV in otherwise typical European conditions/electricity mix.** Since in the vast majority of operational situations in Europe, passenger cars would be expected to only operate in such conditions for a small proportion of their overall lifetime, it is not expected to materially affect the overall comparison.

The higher energy consumption of BEVs in cold conditions can also be mitigated by the use of more efficient heating systems including heat-pumps, reducing impacts. These are fitted as standard to some BEV models (particularly for vehicles sold into regions with colder climates), or available as an optional extra.

Further information on this can also be found in Section 5.2.1 of this report.

#9 "Does the life cycle GHG impacts of BEVs really end up being higher than those of ICEVs, because the battery pack needs to be replaced over the course of their service life?"

In the vast majority of cases, **it is not expected that the traction battery used in a new BEV passenger sold on the market today in Europe will need to be replaced in the lifetime of the vehicle**. Some earlier BEV models with relatively low battery capacities (compared to new models today) have had issues reported with battery lifetime/durability in extreme conditions (e.g. high temperatures, very frequent use of rapid charging with high mileage, etc.). However, in typical use-cases in moderate European conditions, it is still expected for the batteries of older models to last the life of the vehicle, based also on current experience. Over the last 10 years, there has also been significant technological improvement to batteries and battery-management, resulting in improved lifetime/durability. In addition, new BEV models typically have much higher capacity battery packs, which means they have to do far fewer charge-discharge cycles (a key factor affecting longevity) to drive the same distance, further improving the expected lifetime of these battery packs in normal conditions. It is anticipated, therefore, that these battery packs will have a useful lifetime far beyond the application in the vehicle itself, and there is much research ongoing into so-called 'second-life' applications for these batteries.

Nevertheless, based on recent analysis of the GHG impacts from manufacturing and operation of new BEVs in typical EU conditions, it is expected that the **overall reduction of life cycle impacts would still be significant compared to ICEVs, even with a battery replacement**. However, the latter is unlikely to be needed in the vast majority of cases, except due to some other fault/problem.

Further information on this can also be found in Section 5.2.1 of this report.

ANNEX 2: ADDITIONAL SUPPORTING MATERIAL FOR CHAPTER 3

This Annex provides additional discussion of important LCA methodological details, integrating respectively Sections 3.1 (Introduction to LCA), 3.3 (Literature review and harmonisation) 3.4 (Literature review on vehicle production), 3.5 (Literature review on vehicle use) and 3.8.2 (EV battery technologies and end-of-life recycling and repurposing).

Additional material on life cycle assessment methodology (Section 3.1)

1. Goal and scope definition

The system boundary

A simplified system boundary has been presented in the main body of the report, however Figure A1 provides a more detailed overview of a typical scope and system boundary for vehicle LCA.

Attributional and consequential LCA

A fundamental methodological distinction that must be made when setting the goal of an LCA, and which affects how the entire assessment is then carried out and how the results are to be interpreted, is that between Attributional and Consequential LCA:

Attributional LCA (A-LCA) provides information about the impacts of the processes involved throughout the full life cycle of a product (including raw material and primary energy acquisition, manufacturing, use and end-of-life), but it *does not* consider any indirect effects arising from changes to the larger system into which the product is embedded, such as those which may arise from the large-scale deployment of the product itself. The vast majority of the LCAs in the available literature are A- LCAs.

Consequential LCA (C-LCA) is instead primarily aimed at providing information about the consequences of a change (typically, an increase) in the level of deployment of a product, including the effects on the larger system into which the product itself is embedded (e.g. changes to the electricity grid mix induced by the large-scale deployment of xEVs). The main area of application for C-LCA is to inform policy makers on the broader impacts of policies which are intended to change levels of production and deployment of new products.

Failure to distinguish between C-LCA and A-LCA can lead to the wrong approach being applied for the intended goal of the study or an unwitting combination of the two approaches within a single analysis, which may result in misinterpretation of the results, or an unfair comparison of results derived from the application of different approaches.

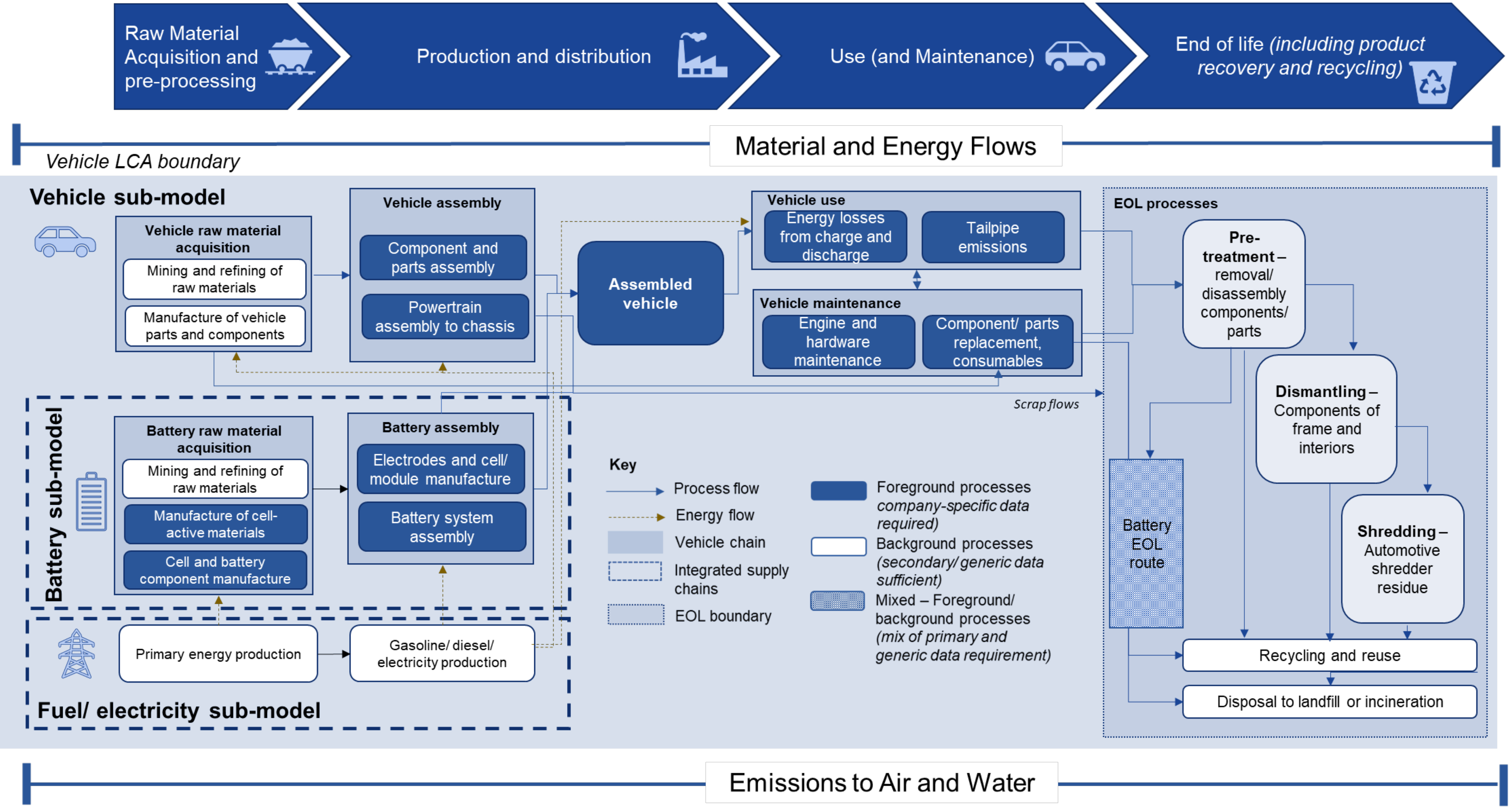
In this report, the term "LCA" is generally assumed to refer to A-LCA, unless otherwise explicitly stated. Specifically, all of the reviewed literature studies were identified as A-LCAs (whether as much was explicitly declared or not). Conversely, some of the research and knowledge gaps discussed in Section 3.8 are best framed within the context of a C-LCA.

End-of-life allocation methods

Another key modelling choice that significantly affects the results of an LCA has to do with end-of-life allocation. In fact, when considering the EoL phase of the product's life cycle, the definition of system boundary is intimately connected and co-determined by the choice of **EoL allocation method**. Three main options are available in this regard:

- **"Cut-off" (a.k.a. "recycled content") method**: This method excludes (i.e. cuts off) any EoL material recycling processes from the system boundary, alongside any environmental benefits or "credits" that may accrue from the ensuing (partial) displacement of demand for virgin materials after the EoL of the vehicle. This first EoL allocation method is therefore well suited to quantifying the benefit of using secondary (i.e. recycled) materials in vehicle manufacturing, but selecting it does not typically produce significant differences in estimated impact when comparing alternative EoL management options (and specifically, of course no difference in the case of alternative recycling processes).
- **"Avoided burden" (a.k.a. "closed loop") method**: This alternative method includes and quantifies the potential future benefit of material recycling at EoL and calculates the associated environmental "credits" by assuming the displacement of equal quantities of their primary (i.e. virgin) counterparts. To ensure internal methodological consistency and avoid any double counting, however, when setting the system boundary in this way, all input materials to vehicle manufacturing must always be modelled as primary (i.e. virgin), regardless of whether any shares thereof may actually be coming from secondary (i.e. recycled) sources. Therefore, in complete contrast to the previous method, this EoL allocation method allows significant differences between alternative EoL management options (and specifically recycling processes) to emerge; however, it does not permit the quantification of any benefit arising from the use of secondary (i.e. recycled) materials in vehicle manufacturing.
- **The Product Environmental Footprints (PEF) "Circular Footprint Formula" (CFF) method:** is by far more sophisticated than either of the previous ones (European Commission, 2021c). It adopts a "balanced" approach, whereby both the benefits of using secondary materials in vehicle manufacturing and the potential environmental credits ensuing from material recycling at EoL are taken into account, while avoiding any methodological inconsistencies that could result in double counting. However, the CFF entails more complex calculations and calls for additional parameters, including two subjective "allocation factors", and none of the current commercially available life cycle inventory databases are yet ready for its fully coherent implementation.

Figure A1: The life cycle stages of a passenger vehicle



Source: Ricardo (own elaboration).

Notes: Excludes energy storage/balancing for the electricity network, and refuelling/charging (or other road) infrastructure for vehicles.

2. Inventory analysis

It is important to underline that, to generate a life cycle inventory (LCI), all material and energy flows that represent direct inputs to the system under assessment during any of its life cycle phases are to be traced back to the points where their supply chains cross the interface between the natural geo-biosphere and the technosphere. For instance, an input of "steel" to manufacturing is to be traced back to all of the natural resources that are extracted/harvested from nature for its production and delivery (e.g. starting from the "iron ore in the ground").

In LCA terminology, a further distinction is also made between "foreground" and "background" processes and data, as follows:

- **Foreground** processes are those "that are under direct control of the producer of the good or operator of the service, or user of the good or where he has decisive influence […]. This covers, firstly, all in-house processes of the producer or service operator of the analysed system. Secondly, […] also all processes and suppliers of purchased made-to-order goods and services, i.e. as far as the producer or service operator of the analysed system can influence them by choice or specification" [Joint Research Centre (JRC), 2010].
- Conversely, **background** processes are those "that are operated as part of the system, but that are not under direct control or decisive influence of the producer of the good (or operator of the service or user of the good). The background processes and systems are hence outside the direct influence or choice of the producer or service operator of the analysed system" [Joint Research Centre (JRC), 2010].

3. Impact assessment

Life cycle impact assessment (LCIA) is comprised of four steps, as outlined below:

1. **Classification.** This first step categorises the inventoried inputs (raw resources from nature) and outputs (emissions to nature) of the full LCI into a number of relevant impact categories. For example, all emissions of GHG (e.g. carbon dioxide, methane, sulphur hexafluoride, etc.) are classified as related to the climate change impact category. The inclusion of multiple impact categories, beyond climate change alone, is mandated by ISO 14040 & 14044 with the aim of preventing the inadvertently missing potential of **"impact shifting"**. The latter is a phrase used to indicate those instances when the reduction of the environmental impact in one category (e.g. climate change) is accompanied by an increase of impact in another category (e.g. human toxicity).
2. **Characterisation.** In this second step, the input and output flows that have been classified within each impact category are scaled according to the magnitude of their contribution to the respective impact categories; this is done on the basis of their comparative impact per unit of mass, relative to the unit impact of a chosen reference chemical compound. For example, in the climate change impact category, all of the inventoried individual GHG emissions are scaled according to their relative potency to contribute to global warming, relative to carbon dioxide (i.e. as CO_2 equivalents). Once all input and output flows have been characterised as appropriate within each impact category, they can be summed to generate an aggregated **"mid-point" impact indicator** for each impact category.
3. **Normalisation.** This step further scales the mid-point indicators obtained at step 2. with regards to the respective overall impact that took place in a pre-defined geographic region over a specific length of time (e.g. in Europe during the year 2010), thereby producing dimensionless ratios (i.e.. "normalised impact indicators").

4. **Weighting.** In this last step, a number of normalised impact indicators are summed together, to generate one or more overall single-score **"end-point impact indicator(s)"**, after multiplication by specific weighting factors. Such end-point indicators attempt to address a point further down the cause-effect chain and attempt to estimate the damage that an increase in emissions (or withdrawal of resources) may have on a pre-defined "area of protection" (the three AoPs are: ecosystem health, human health and resource depletion). For instance, GHG emissions, acidic emissions, toxic emissions, etc. could all have detrimental effects on freshwater species, terrestrial species, etc. and as such they can all be seen as contributing to "end-point" impact on ecosystem health.

ISO 14044 clearly indicates **that steps 3 and 4 (Normalisation and Weighting (N&W) are always optional, and they must not be included in any "comparative assertions intended for public disclosure" (i.e. in comparative LCAs).** The fundamental reason for these ISO rules is that Normalisation and Weighting inevitably always imply subjective value judgements. In particular, the choice of weighting factors implicitly prioritises the importance of different and independent environmental impacts relative to each other (e.g. global warming vs water scarcity vs human toxicity), which is a policy decision that cannot be made on merely physical or scientific grounds.

It is however hereby noted that, in open contrast to ISO 14044, the latest revision of the PEF guidelines [European Commission, 2021] has not only introduced a mandatory requirement for N&W, but actually recommends the repeated use of aggregated single-score impact indicators to (i) establish the relevance of the processes contributing to the system under analysis and set system boundary accordingly and (ii) compare the "overall" environmental performance of the product system under study to that of a pre-defined "benchmark" product.

Additional material from the literature review and harmonisation (Section 3.3)

The harmonisation of the published GHG emission results was carried out using equations 1 and 2 as described below.

Harmonisation of GHG results for vehicle production phase:

$$\text{(Eq.1)} \begin{cases} IF\ (UoA = vehicle)\ THEN\ GWP_{P,H} = \frac{GWP_P}{LA_H} \\ IF\ (UoA = vehicle \cdot km)\ THEN\ GWP_{P,H} = \frac{GWP_P \cdot LA}{LA_H} \\ IF\ (UoA = passenger \cdot km)\ THEN\ GWP_{P,H} = \frac{GWP_P \cdot VO \cdot LA}{LA_H} \end{cases}$$

Where:
$GWP_{P,H}$ = Harmonised Global Warming Potential of vehicle production phase
GWP_P = Global Warming Potential of vehicle production phase, as originally published
LA_H = harmonised vehicle lifetime activity
LA = vehicle lifetime activity, as assumed in original study
VO = vehicle occupancy, as assumed in original study

Harmonisation of GHG results for vehicle use phase:

$$\text{(Eq.2)} \begin{cases} IF\ (UoA = vehicle)\ THEN\ GWP_{U,H} = \frac{GWP_U}{LA} \\ IF\ (UoA = vehicle \cdot km)\ THEN\ GWP_{U,H} = GWP_U \\ IF\ (UoA = passenger \cdot km)\ THEN\ GWP_{U,H} = GWP_U \cdot VO \end{cases}$$

Where:
$GWP_{U,H}$ = Harmonised Global Warming Potential of vehicle use phase
GWP_U = Global Warming Potential of vehicle use phase, as originally published
LA = vehicle lifetime activity, as assumed in original study
VO = vehicle occupancy, as assumed in original study

The total harmonised life cycle GHG impact ($GWP_{LC,H}$, excluding EoL phase) were then simply calculated as the sum of the two previous terms:

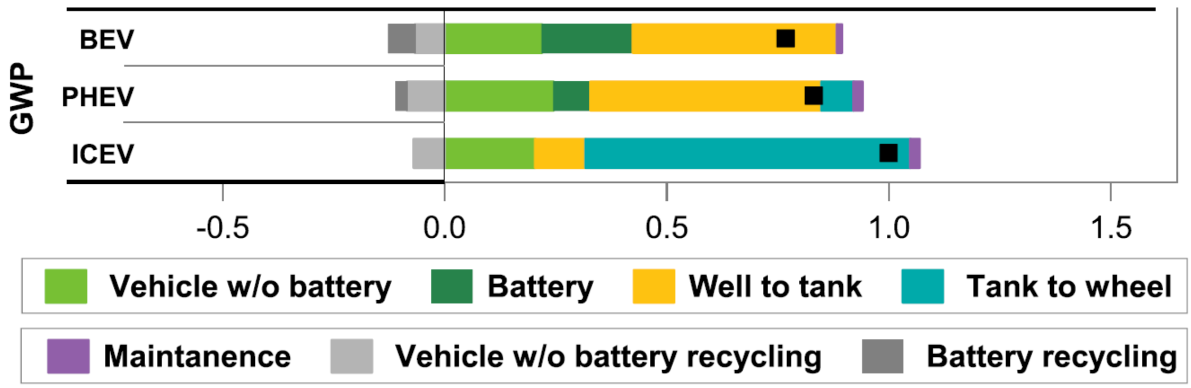
(Eq.3) $GWP_{LC,H} = GWP_{P,H} + GWP_{U,H}$

Additional material on literature findings

1. Production (Section 3.4)

In general terms, and within the same vehicle size class, the GHG impact associated to the first two phases of the vehicle's life cycle (i.e. raw material sourcing and vehicle production) tend to be higher for BEVs than for ICEVs. This is due for the most part to the comparatively heavy and resource-intensive battery packs, which can be responsible for up to 50% of the total GHG BEV production emissions – see for instance the light and dark green bars (respectively referring to production of the vehicle excluding the battery and to production of the battery) in Figure A2 below (adapted from (Zeng, et al., 2021)).

Figure A2: Life cycle GHG impact of passenger vehicle, per km travelled



Source: adapted from (Zeng, et al., 2021).
Notes: Different colours refer to different phases of the vehicle's life cycle. The intended focus here is on vehicle and battery production impacts (in light and dark green, respectively). Well-to-tank (in yellow) = fuel and electricity production; Tank-to-wheel (in teal) = direct emissions/impacts from the vehicle itself (e.g. due to fuel combustion). Impacts of recycling (in grey) are negative due to the credits associated with the displaced virgin materials. Black squares indicate total life cycle results, which are normalised to the highest total impact among the three considered power train options.

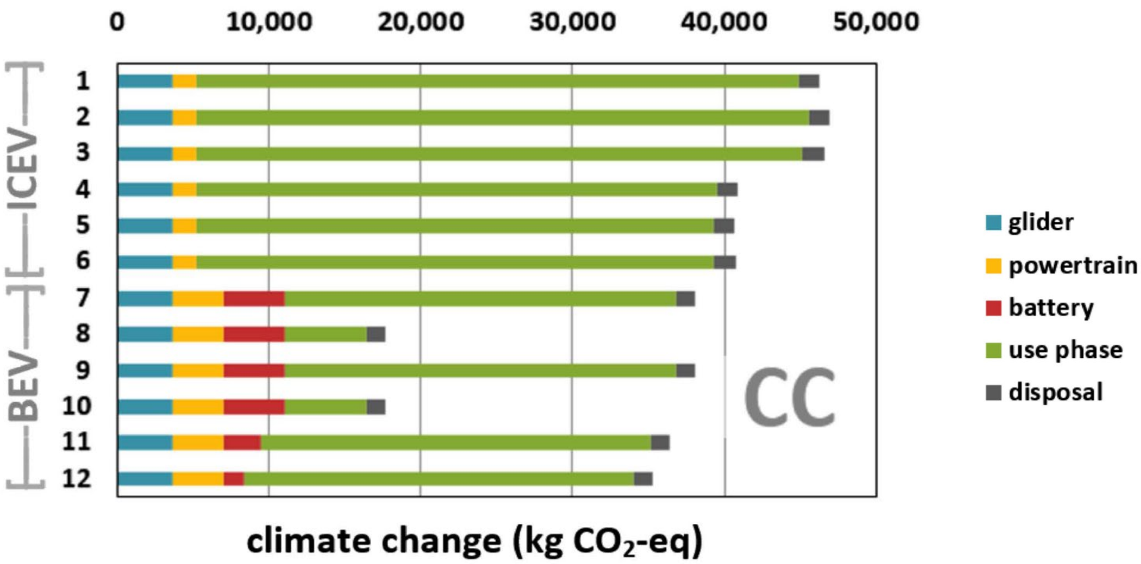
2. Use (Section 3.5)

The carbon intensity per unit of energy delivered to the power train of BEVs is highly variable, depending on the specific electricity grid mix that is assumed to be used to recharge the BEV batteries during the vehicle use phase. This point is well illustrated by the LCA calculations reported in two studies focusing on light duty vehicles in Germany, which are discussed briefly below. Incidentally, the literature review indicated a dearth of studies focusing specifically on some of the other countries of special interest here.

In the first of these two case studies (Helmers et al., 2020), the authors compared the life cycle GHG impact of a VW mini-van alternatively powered by an ICE (scenarios 1-6) or an electric power train (scenarios 7-12). The first six scenarios differ in terms of the fuel used (gasoline for 1-3 and diesel for 4-

6) and of the specific assumed emission profiles per km travelled, while the latter six scenarios span a range of different assumptions on the location of the battery production (which affects the red bars in Figure A 3), and the electricity mix used in the use phase (which affects the green bars in the same figure, and which is by far the factor that produces the largest variations in the overall results). Specifically, while scenarios 7, 9, 11 and 12 assume the average German grid mix for 2013, scenarios 9 and 10 assume a future, heavily decarbonised German grid mix for 2050. The results point to very large margins for reductions (>60%) in the overall life cycle GHG impact of BEVs vs ICEVs when the former are powered by electricity generated by a low-carbon grid mix (despite the initially almost double GHG impact in the vehicle production phase).

Figure A3: Life cycle GHG impact from mini-van vehicles in Germany



Source: (Helmers et al., 2020).
Notes: The figure in the original study reports climate change impacts for all vehicle life cycle phases. Specifically: (1+2) raw material sourcing and production, disaggregated in glider (blue), powertrain (yellow), battery (red); (3) use (green); and (4) end-of-life (grey). The intended focus here is on the significant differences during the use-phase, across the different vehicle scenarios (see text).

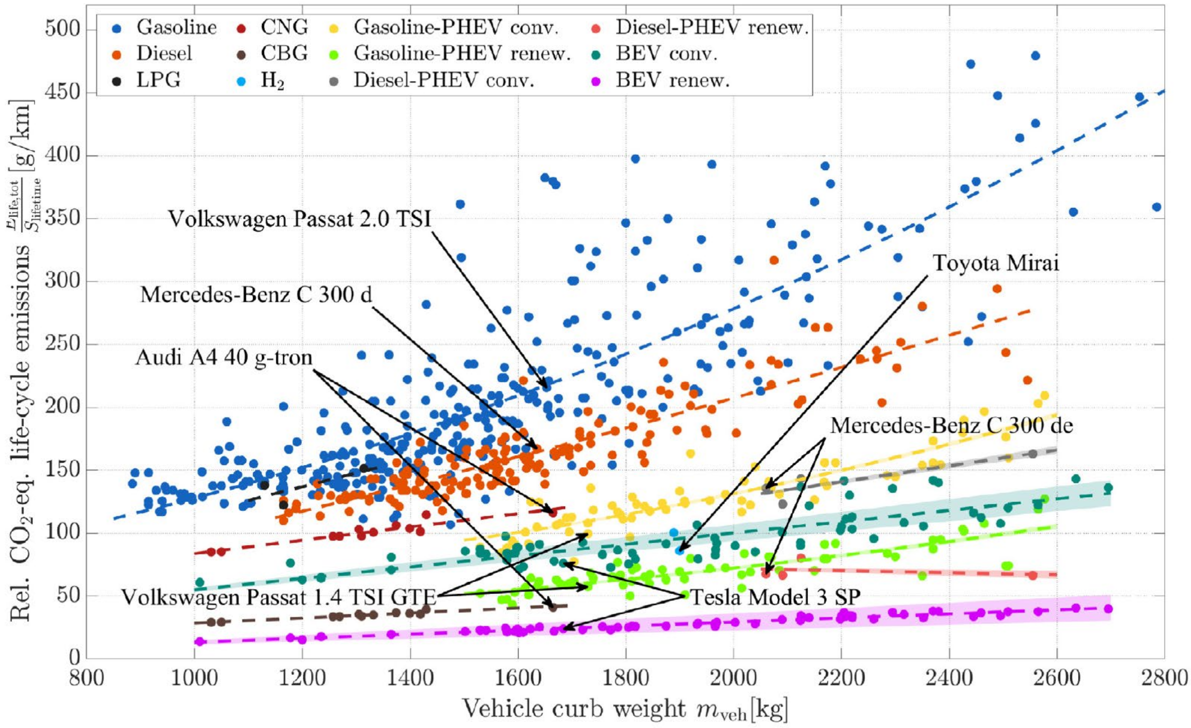
In the second study (Buberger, et al., 2022), the authors assessed the life cycle GHG impact of 790 vehicle types in Germany, encompassing a range of power train and energy carrier options: ICEVs powered by gasoline, diesel, LPG, CNG, CBG, and H_2; PHEVs powered by gasoline or diesel + current German grid mix or 100% renewable electricity; and BEVs powered by current German grid mix or renewable electricity. For all vehicles, a Lifetime Activity (LA) = 230,000 km was assumed. The full set of the resulting 790 data points is reported in Figure A4, where the vertical axis spans the range of calculated life cycle GHG impact per [vehicle×km], while the horizontal axis refers to the vehicle curb weight [kg]. The superimposed trend lines, individually plotted for each power train and energy carrier combination, clearly point to:

(i) already significantly lower GHG impact for BEVs (even when the electricity is generated by the current German grid mix: dark green data points and trend line) vs ICEVs (blue = gasoline and red = diesel data points and trend lines);

(ii) even lower GHG impact for BEVs when the electricity is generated using renewable technologies (purple data points and trend line);

(iii) a clear correlation between vehicle curb weight and life cycle GHG impact, across all power train and energy carrier options. This is obviously due to the increased energy consumption in the use phase, even if the data themselves are not disaggregated per life cycle phase; it is also noteworthy that the trend lines are progressively "steeper" for the less efficient power train options (meaning that curb weight is a much bigger factor in determining life cycle GHG impact for ICEVs than it is for BEVs).

Figure A4: Life cycle GHG impact from passenger vehicles in Germany vs vehicle curb weight



Source: (Buberger, et al., 2022).

Notes: The figure in the original study reports total climate change impacts over the full life cycle of the vehicles (excluding end-of-life). However, the intended focus here is to discuss the observed dependency of the life cycle GHG impact on the curb weight of the vehicles, which – to an overwhelming extent – is due to the associated energy consumption during the use phase.

Additional material on methodological implications of electricity grid mix

The phrase "electricity grid mix" refers to the proportion of primary energy sources used, ranging from fossil fuels to renewables, to generate the electricity that is supplied to the grid. Calculating the electricity mix of a country or region allows to accurately account for the emissions generated by its power generation sector, and thus also those associated to the use phase of BEVs.

The EU electricity mix was 40% renewables, 26% nuclear and 35% fossil fuels in 2020. According to the Reference Scenario 2020 (which includes policy in place at this point) and the scenarios assessing the impacts of the Fit for 55 policy package (European Commission, 2021a), the proportion of renewables is expected to increase over time, as one of the aims of the European Green Deal is to reduce power generation sector emissions to reach climate neutrality by 2050. Currently, each Member State has a different carbon intensity for the power generation sector. For example, the Czech Republic's electricity mix was only 8% renewables, 40% nuclear and 52% fossil in 2020, compared to Romania's with 42% renewables, 19% nuclear and 39% fossil in the same year. Therefore, using each individual nation's electricity grid mix is important for conducting accurate EVs life cycle analysis across Member States.

The carbon intensity of electricity generation is expected to gradually decrease in the future due to EU policy interventions and climate advocacy, with EU Member States taking different approaches to decarbonise their electricity grid. Correctly estimating future grid electricity mix compositions is crucial in LCA to obtain accurate carbon intensity figures, and to this end, using dynamic modelling can ensure more realistic projections of GHG impact for EVs reflecting the decreasing trend of carbon intensity of electricity over time, vs the simplistic (and typically worst-case) use of current grid mix compositions.

Several modelling techniques have emerged in LCA studies attempting to capture the changes in electricity mix over time, with each method using a different algorithm to calculate the total emissions during a product life cycle.

The first and the most common method is the average grid mix assumption, i.e. using one set of electricity mix data during a fixed time period to calculate the impacts over the whole life cycle. The chosen time period will typically be the median point of the use phase, thereby assuming that the carbon intensity reduction of the electricity mix is linear over time. For example, an LCA of an EV with a use phase spanning 15 years may use the same set of electricity mix data to estimate its emissions per a certain distance travelled or unit of energy consumed, using the electricity mix data estimated for the 7th year of operation. The problem with this method is that the linear carbon intensity decrease assumption might not be realistic, and it could lead to inaccurate results.

A more sophisticated, and potentially more accurate, family of methods variously break down the use phase into more intervals and adopt a corresponding number of different grid mix composition. It is estimated for each individual interval, thus enabling a better approximation of a non-linear evolution pathway for the grid mix over time. The trade-off in doing so is, however, an increase in modelling complexity; also, reliable grid mix composition projections may not be available with the required time granularity.

Additional material on EV batteries: future technology roadmaps and end-of-life recycling (Section 3.8.2)

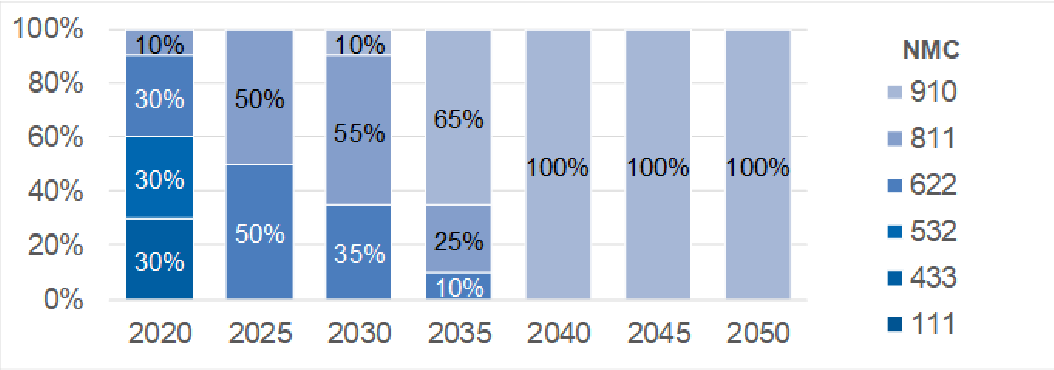
1. Evolutionary trends in the EV battery sector

i. Battery technology mix trends

As already mentioned in Section 2, the two LIB types that are most widely employed in passenger BEVs are Lithium-Nickel-Cobalt-Aluminium Oxide (battery chemistry)(NCA) and Lithium-Nickel-Manganese-Cobalt Oxide (NMC). Over time, the trend has been to aim for a reduction in cobalt content, with a corresponding increase in nickel. NMC formulations have evolved from NMC-111, to NMC-622 and NMC-811, while batteries containing cathodes with aluminium have also transitioned to variants with more nickel (NCA+).

In line with this general shift of the industry towards batteries containing less and less cobalt, there have been efforts to draft potential roadmaps in terms of the composition of future LIB chemistries. There are speculations that NMC 111, 433 and 532 cathode chemistries will be phased out by as early as 2025, while the NMC mix will be dominated by NMC 910 in 2050 (Figure A5).

Figure A5: Battery technology mix speculations/expectations for NMC cathode chemistries



Source: (Hill , et al., 2020).

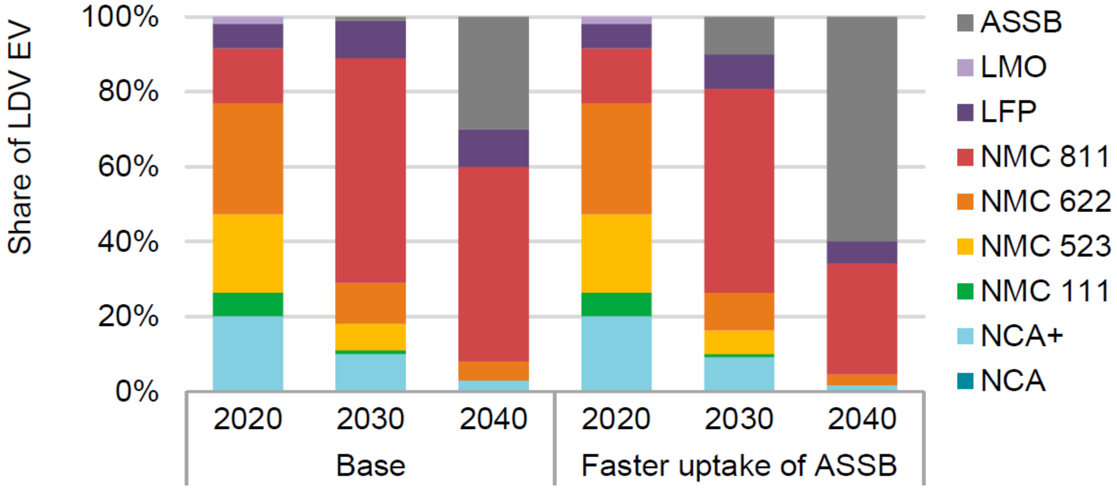
In terms of the broader market mix of different battery chemistries, beyond the currently employed LIB options, there have been estimations that by 2030 the most commonly used cathode mix will be the NCA, but that the latter will be phased out in the future, in favour of new technologies such as all-solid-state batteries (ASSBs) and sodium-ion batteries (NIBs) in 2050.

ASSBs, including those with a metallic lithium anode, have the key advantage of doing away with the liquid (and flammable) electrolyte, and they are often considered a key player for the future of energy storage. They are composed of non-flammable solid electrodes and a solid electrolyte, which makes them a type of battery with ultimate safety (Kotobuki, Munakata, & Kanamura, 2013). This inherently safe nature makes ASSBs a very versatile and efficient solution for the electric transportation sector. The fact that the associated hardware can be manufactured with nanotechnology could further improve this aspect, as electrochemically active nanomaterials with a large surface area can assist in shortening the diffusion distances for the movements of both Li-ions and electrons (Karuthedath Parameswaran, et al., 2023). Furthermore, ASSBs that are equipped with lithium metal anodes are able to achieve volumetric energy densities that are 70% higher than today's LIBs with graphite anodes. This makes them ideal and more sustainable batteries for future EVs (IEA, 2021).

Sodium-Ion Batteries (NIBs) are another promising electrochemical energy storage system which has raised attention during the last few years. They are so far lower in gravimetric energy density, compared to the current crop of LIBs, but they are characterised by a good life cycle (Sarkar, Rashid, & Hasanuzzaman, 2022). Their key advantage is that they completely remove dependence on a whole range of critical raw materials (such as lithium, nickel and cobalt), while the natural abundance of sodium resources make them a low cost solution that is very competitive with respect to LIBs (Yasin, Muhammad, Nguyen, & Nguyen-Tri, 2021). They have also been characterised as more "green" and sustainable than other chemistries since they host environmentally friendly electrodes and are cobalt-free (Tarascon, 2020). These batteries further offer the significant advantage of being able to be stored or transported at an empty energy state (0 V) (Sarkar, Rashid, & Hasanuzzaman, 2022), which essentially means that there would be no transportation limitations and that the safety risks would be minimised.

In the recent report on critical raw materials for the energy transition (IEA, 2021), the IEA drafted two scenarios for the progressive shift to lower-Co NMCs and NCAs to ASSBs over the next two decades (Figure A6).

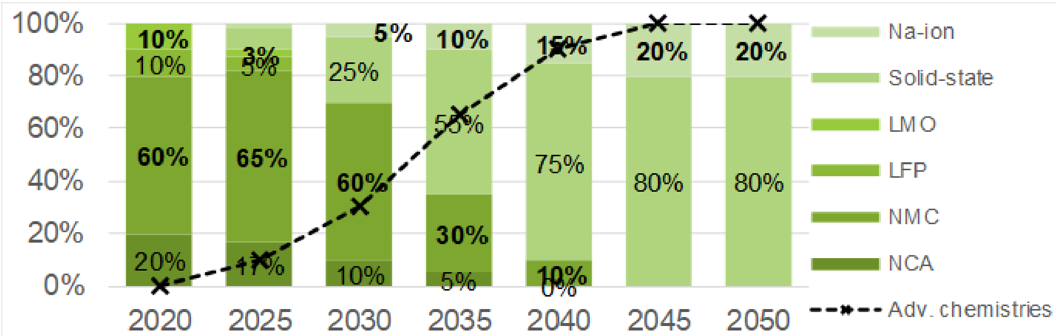
Figure A6: IEA scenarios for future EV battery technology mix



Source: (IEA, 2021).

In Ricardo's own report to the European Commission (Hill , et al., 2020), another battery technology mix scenario was considered, in broad alignment with IEA's "Faster uptake of ASSB" one, but also considering the penetration of NIBs in the mix (Figure A7).

Figure A7: Ricardo scenarios for future EV battery technology mix



Source: (Hill , et al., 2020).

However, the above data on the evolution of battery chemistries in the medium/long run are to be complemented by a study focusing on the market trends of future LIBs (Bajolle, Lagadic, & Louvet, 2022), which highlighted conflicting opinions among different stakeholder groups. Specifically, one large group was reported to expect NMC chemistries to gain a lot of ground in the forthcoming future of the automotive industry, and lithium-ion technologies in general to remain dominant even in the long-run, regardless of the new technologies that will be introduced, such as ASSBs and SIBs. Some other experts suggested an even more business-as-usual scenario for the future, where they do not foresee any LFP development, nor the appearance of any disruptive new technologies. Conversely, a third group of stakeholders was confident about the disruption that new technologies will bring to the industry by 2030.

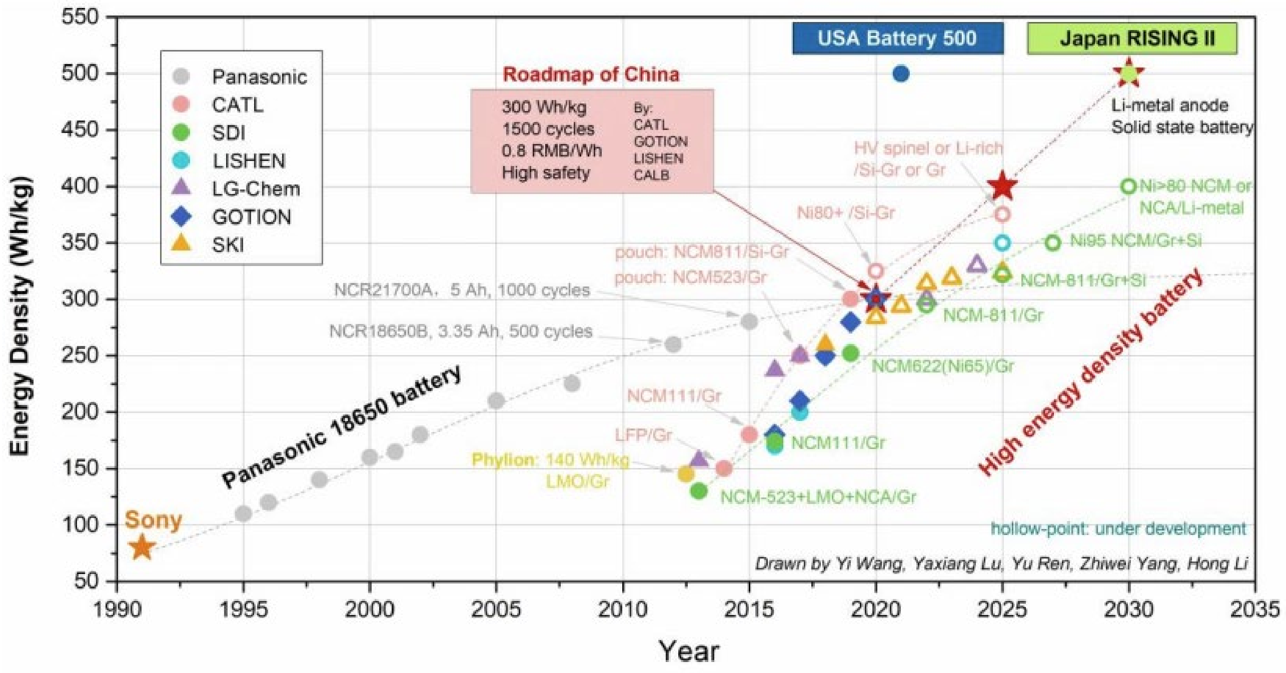
ii. Battery energy density trends

One particular aspect that has not been sufficiently addressed by the existing life cycle assessment literature, is that of projected battery energy density improvements. As of 2020, the highest reported gravimetric energy density for a Li-Ion cell at near-commercial level was 304 Wh/kg, which is based on a silicon/graphite anode, paired with NMC811 cathode (The Faraday Institution , 2020). In terms of other LIB technologies, a US company, Sion Power, has developed a 400 Wh/kg cell, the Licerion Electric

Vehicle (Sion Power, 2022), comprising a conventional nickel-rich cathode and an 'ultrathin' lithium anode with protective coating. Other researchers have developed a battery with a $LiCoMnO_4$ cathode and graphite and lithium metal anodes, which, by using a specially designed electrolyte, can provide 720 Wh/kg for 1000 cycles at 5.3 V (The Faraday Institution , 2020). The ASSBs technologies are showing high energy densities in relation to other technologies, as a US company named Solid Power (spinoff from the University of Colorado at Boulder) has manufactured laboratory scale cells at 400 to 500 Wh/kg, reaching up to 500 cycles (Solid Power, 2022). As the industry is witnessing a small deceleration in conventional LIBs advancements, it will not be an exaggeration to speculate that other technologies and mixes will start overtaking them in the next few years in terms of energy density (The Faraday Institution , 2020). For instance, since existing lithium metal prototypes have already reached 500 Wh/kg specific energy, they could gain significant shares of the market, if their cycle life is improved.

The BATTERY 2030+ research initiative (Amici, et al., 2022) aims to invent and develop safe and sustainable high-performance batteries in order to help Europe reach the European Green Deal targets. In this context, the R&D hub has gathered various international roadmaps internationally and by the EU Strategic Energy Technology (SET) Plan, and has compared them in Figure A8 below.

Figure A8: Comparison of the gravimetric performance of different batteries for automotive applications

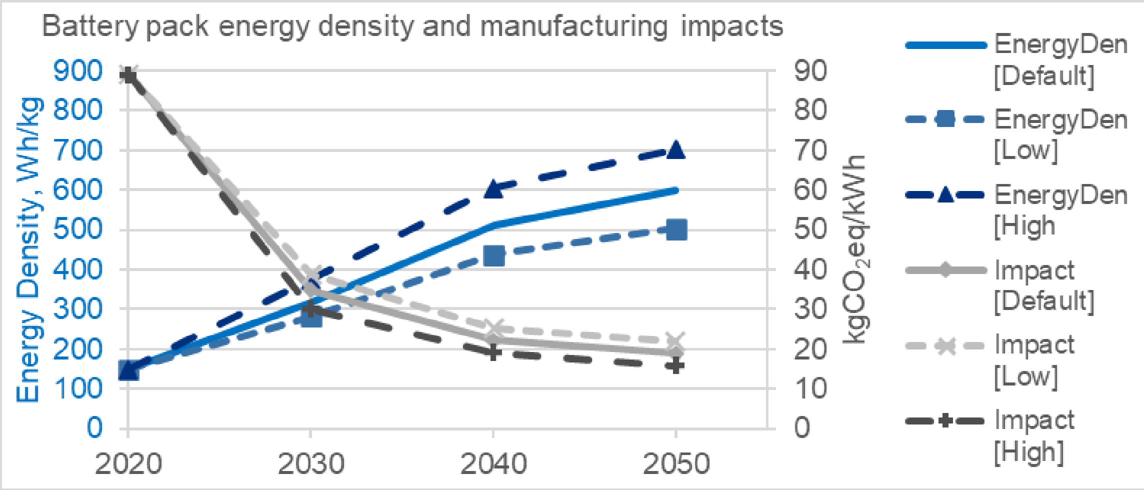


Source: (Amici, et al., 2022).

It can be seen from this figure that 2030 targets have been set from the SET Plan (green line), Japan (light green point) and China (red stars). More precisely, the EU aims for NCM or NCA/Li-Metal batteries, with more than 80% Nickel, to reach 400 Wh/kg by 2030, while Japan and China are more ambitious, targeting Li-metal anode solid state batteries to reach 500 Wh/kg.

Figure A9 below illustrates a series of possible future battery energy density (and associated GHG emission) trends developed by Ricardo (Hill , et al., 2020), in line with the battery technology mix scenarios previously reported in Figure A7.

Figure A9: Ricardo scenarios for future EV battery energy density and GHG impact



Source: (Hill , et al., 2020).

Along with the trends toward higher cell energy densities, there are also advancements being made in battery packaging. Such developments are the cell-to-pack (CTP) designs, a design placing the cells directly into the battery pack and presented by CATL and BYD (Battery Design, 2022), as well as alternative structural battery designs being developed by others such as Tesla (Teslarati, 2022).

iii. Price and materials trends

It has been noticed that in the last 10 years the average cost of LIBs has declined by 90% in total, approaching $130 per kWh in 2021 (Frith, 2021). This phenomenon occurred as EV sales grew exponentially, and the trend is expected to continue, as new designs and falling manufacturing costs have the potential to result in more cost reductions. It has been forecasted that by the mid-2020s the price will drop below $100 per kWh, but this is highly dependent on maintaining steady supply with the same drop-in costs, and on mineral scarcity. The vast majority of mineral elements used in manufacturing batteries are geographically concentrated (see Section 3.8.1), and thus it cannot be certain that their supply will still be uninterrupted in the years to come.

Significant reductions in costs, and thus prices, may also be achieved through a shift to new, disrupting technologies, both in the anode and in cathode compartments of the battery (IEA, 2021).

Since ASSBs offer high safety levels and do not use liquid electrolytes, they do not require expensive systems for cooling, and it is expected that when a robust and cost-effective scale-up process is developed, electric cars powered by ASSBs would only take three to five years to be designed and manufactured. This has created significant incentives for researchers and manufacturers, who, in the last few years, have invested in R&D for this technology. The progress that is expected to be made will result in further reducing costs and scaling up production, to make this solution commercially viable (IEA, 2021).

Cathode materials have been the primary aspect of research in the NIBs industry so far. Polyanionic compounds and layered transition metal oxides have been the protagonists in R&D processes, while minimum emphasis was given to anode materials. However, recent research is directed towards NIB anode materials that will increase the cell energy density and also help contain cost (Karuppasamy, et al., 2020). The main alternatives for NIB anodes are carbon-based materials since they are chemically and thermally stable while also having a low voltage against sodium. Alloy-forming, as well as conversion-alloying materials are additionally used as NIB anodes, amongst others, where the most suitable technology of the former is cobalt-based sulphide compounds (Mohan, et al., 2022). Although these compounds show exceptional sodium storage capacity, they contain cobalt, which goes against current R&D trends in trying to mitigate its use.

In this context, the fact that manufacturers are making efforts to reduce cobalt content in battery mixes while ensuring equally high, and even higher, energy densities, points to mineral prices playing a huge part in determining the costs of batteries. Indicative of this is the fact that, as of 2021, 50-70% of total battery costs were due to the raw materials used in the battery manufacturing (Argonne National Laboratory, 2020). Researchers suggest that if an event of high mineral prices took place, it could mitigate the "anticipated learning effects associated with a doubling of capacity" (IEA, 2021). Therefore, manufacturing stakeholders (governments and industry) should aim to prevent price spikes by ensuring a reliable supply of battery metals. Nonetheless, it is of paramount importance that technologies making use of less scarce materials be further developed, as the trend of cost decline that was observed during the last decade is not certain to continue without further innovative technology acceleration.

iv. Battery manufacturing location trends and renewable electricity

Currently the biggest manufacturers of EV batteries are located in Asia, as China, Japan and South Korea represent the largest shares of the manufacturing industry (Hung, Völler, Agez, Majeau-Bettez, & Strømman, 2021). Conversely, as of 2018, cell manufacturing for traction batteries in Europe amounted to only 1% of the global total (Eddy, Pfeiffer, & Staaij, 2019). The European Commission has identified this gap between the two regions and has set clear goals, aiming for a 7-25% manufacturing share to be located in Europe by 2028 (European Commission, 2019). In this context, the planned EU battery manufacturing capacity would reach 200-290 GWh/year by 2025 (Eddy, Pfeiffer, & Staaij, 2019).

The largest European facilities, in terms of annual manufacturing capacity, are located in Germany, France, Italy, Poland, Sweden and Norway. In Berlin, the Tesla Gigafactory is the company's first manufacturing location in Europe, which, in addition to vehicles, also produces cells in-house, and has a capacity of 100 GWh/year approximately (Tesla, 2022). In Germany, France and Italy, the Automotive Cells Co. (ACC) are creating high performance LIBs in their three 40 GWh/year factories (Automotive Cells Co., 2022), and they are planning to start producing EV battery cells by 2025 in their new plant in Lauter, Germany. LG Chem's Polish cell manufacturing plant, which has the potential to annually supply over than 295,000 EVs with batteries, has been granted a 95 million Euros aid from the European Commission, in order to grow its production capacity in the European Economic Area (EEA) (Newsroom, 2022). Finally, in the Scandinavian countries, the Swedish company Northvolt, which produces Li-Ion cells, has set a target of 150 GWh/year in annual cell output by 2030 (Northvolt, 2022), while the Norwegian industrial cell manufacturer Fryer, intends to install 50 GWh of battery cell capacity by 2025 and 100 GWh annual capacity by 2028 and 200 GWh of annual capacity by 2030 (FREYR, 2022).

Although the EU makes active efforts to achieve the transition to EVs as quickly as possible, and at the same time to gain shares in the industry of the battery cell manufacturing, there needs to be a solid plan to achieve this in a sustainable and environmentally friendly way. In China, Japan and South Korea, where the majority of EV batteries are currently made, fossil fuels still dominate the electricity mix (Hung, Völler, Agez, Majeau-Bettez, & Strømman, 2021), and previous studies (Ellingsen, Singh, & Strømman, 2016; Kim, et al., 2016) have shown that battery manufacturing contributes by 31-46% to the total production emissions for BEVs. However, research (Bryntesen, et al., 2021) has shown that the environmental impact of electrode manufacturing can be reduced by up to 85%, if fossil derived energy sources are replaced with technologies harnessing renewable energy.

2. End-of-life battery recycling and second life

i. Recycling processes of EoL batteries

When a battery reaches the end of its life (EoL) manufacturers and operators have three options. The first one would be to dispose of it; however, this is not the most sustainable solution, as not only could the battery potentially release toxic chemicals to the environment, but also none of the valuable materials in the battery would be made available for re-use. For these reasons, current regulations exist that prohibit mass disposal of batteries. A second option is to recycle the battery and extract critical raw materials (CRMs) that can be re-used for new batteries; doing so helps close a potentially significant gap between future supply and demand for these CRMs, which is crucial, as there is research suggesting that the supply of Ni and Co will become tight by the late 2020's (Engel, Hertzke, & Siccardo, 2019). There are also concerns that a future exponential demand for lithium in Li-Ion batteries might not be met by the current annual lithium production (Kamran, Raugei, & Hutchinson, 2021). The third option for an EoL battery would be to repurpose it for stationary energy storage applications, where it will be used less intensively than in an EV. This last third option is discussed more at length in a later section iii.

There are currently three major recycling procedures for waste Li-ion batteries: pyrometallurgical, hydrometallurgical, and direct physical recycling. The recovery of CRMs from LIBs involves both physical and chemical processes, while spent LIBs have to be discharged before they proceed to any recycling procedures in order to avoid risks of explosion, combustion and poisonous gas creation. Literature (Zhou, Yang, Du, Gong, & Luo, 2020) has identified the advantages, disadvantages and challenges that occur for different recovery methods, and these are presented in Table A1 below.

Table A1: Advantages, disadvantages and challenges for the three types of battery recycling processes (Zhou, Yang, Du, Gong, & Luo, 2020)

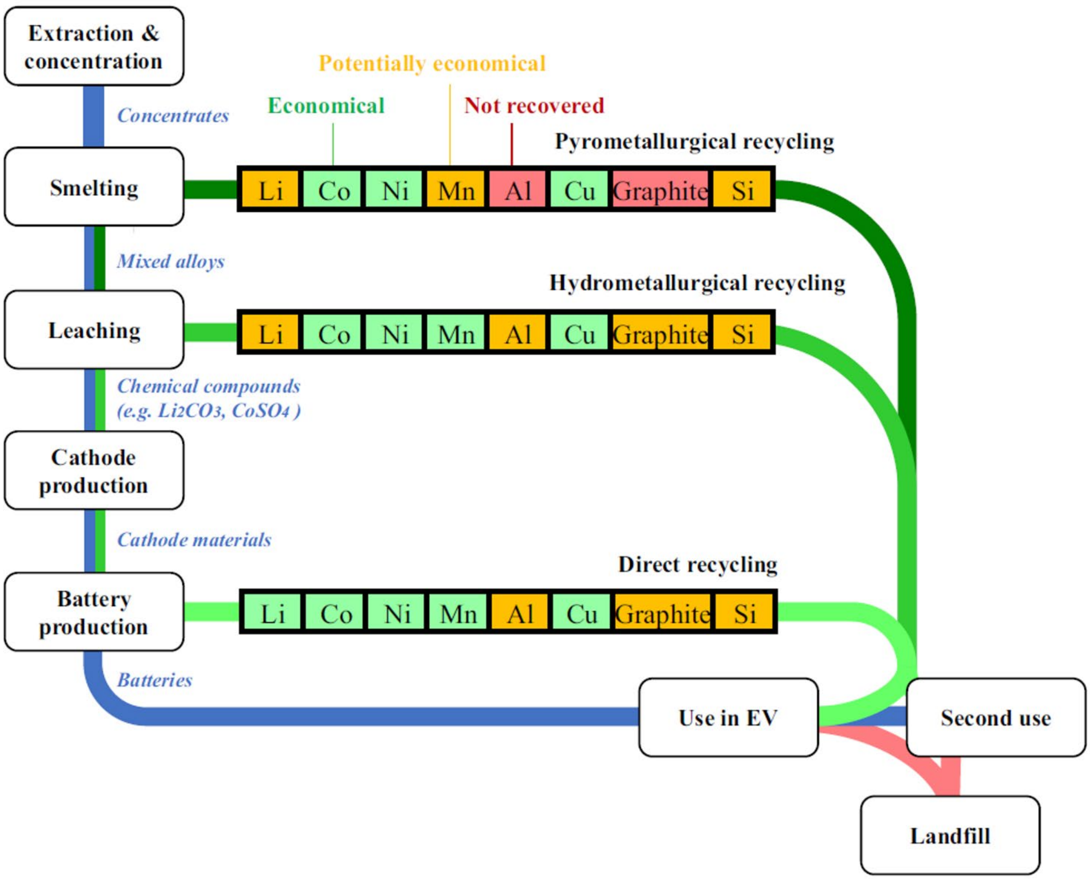
Process	Advantages	Disadvantages	Challenges
Hydrometallurgical Process	+ High recovery rate + Product is of high purity + Low energy consumption + Less waste gas + High selectivity	– More wastewater – Long process compared to the others	→ How to treat the produced wastewater → How this process can be optimized
Pyrometallurgical Process	+ Simple operation + Short flow + No requirement for categories and the size of inputs	– Product is of low purity – Lithium and Manganese are not recovered – High energy consumption – Low recovery efficiency – More waste gases and thus increased costs on waste gas treatment	→ How to reduce energy consumption and pollutant emissions → How to reduce environmental hazards → How to combine with hydrometallurgy for further purification of recovered metals

Process	Advantages	Disadvantages	Challenges
Direct Physical Recycling Process	+ Short recovery route + Low energy consumption + Environmentally friendly + High recovery rate	– High operational and equipment requirements – Incomplete recovery – Only applicable if cathode formulation remains the same	➔ How to reduce recovery costs ➔ How to lower category requirements ➔ How to further optimize product performance

The pyrometallurgical process is based on smelting entire batteries or, after pre-treatment, battery components, while the hydrometallurgical recovery procedure is based on acid leaching[5] and subsequent recovery of battery materials (Xu, et al., 2020). Finally, direct recycling fetches and recovers active materials from LIBs while keeping their original compound structure (Chen, et al., 2019).

Xu et al. (2020) have produced a conceptual schematic showing how different recycling scenarios and procedures can close battery material loops, as well as which materials are recovered through every method.

Figure A10: Schematic presenting recycling methods and scenarios



Source: (Xu, et al., 2020).

5 Leaching dissolves the metals existing in EoL LIBs, and the product is subject to further treatment to chemically separate the solubilized metal ions (Chen et al., 2019).

Out of the processes listed, the most commonly used industrially are pyrometallurgy and hydrometallurgy, or a combination of both.

1. Pyrometallurgy

The most widely used industrial methods globally for battery recycling involve the use of pyrometallurgical processes. Pyrometallurgy, also known as smelting, exposes the process input to high temperatures (>1100 °C), to cause physical and chemical transformations to the input material (Makwarimba, et al., 2022). The combined input components are treated like an ore, whereby high temperature exposure causes them to volatilise, combust or melt. The output product is a mixed alloy containing cobalt, nickel and copper. However, these elements only account for approximately 30wt% in LIBs for electronics (Chen, et al., 2019). The metal alloy must undergo hydrometallurgical treatment to be separated out into its component elements (Gaines, Dai, Vaughey, & Gillard, 2021), leading to so-called "hybrid" recycling. Lithium is difficult to separate using pyrometallurgy due to its high melting and boiling point. The lithium and aluminium bound in the slag can be recovered when a (hydrometallurgical) leaching step is applied after the pyrometallurgy. Overall material recovery rates for pyrometallurgical recycling are typically limited to up to 60% of the battery cell mass (Green Car Reports, 2021), due to lost battery cell materials.

A key reason why pyrometallurgy has significant prevalence is that it adopts well-established existing technology. Smelting is already used to processes virgin ores. Therefore, in some cases, incorporating spent batteries into the input stream requires only a limited number of technical changes to be implemented. As a result, pyrometallurgy can be seen as an immediate solution for battery recycling. However, based on the associated energy demand it may not be widely implemented for future, purpose-built recycling facilities.

2. Hydrometallurgy

This process consists of two main steps, an initial mechanical separation, followed by chemical one, (leaching). In the mechanical step, cells are shredded, with the product stream sieved to remove copper and aluminium foil current collectors. The chemical separation involves dissolving the remaining material in strong acid to break up the crystal structure of the cathode. This is followed by a series of solvent extraction and precipitation steps, to separate out the dissolved metals. A significant advantage of hydrometallurgy is that can generate higher purity materials compared to pyrometallurgy, however it can be a challenge to separate some of the components due to their similar properties (Trower, Raugei, & Hill, 2022).

Different hydrometallurgical techniques achieve varying degrees of leaching efficiencies, however, the significantly greater recovery levels achieved when compared to pyrometallurgical processes are seen across most leaching processes. As an example, inorganic acid leaching is capable of 99.7% efficiency for Ni, Co, Mn, and Li (Trower, Raugei, & Hill, 2022). The high recoverability of lithium may prove to be significant in the future. If lithium becomes more valuable, which seems likely, then hydrometallurgy could be key. The additional investment required to incorporate lithium leaching for hydrometallurgy is likely to be minimal when compared with pyrometallurgy which requires leaching of lithium slag.

An interesting discussion point on the hydrometallurgical EoL recycling of EV batteries, is the effect that this process will have in their cradle-to-grave assessment, if it becomes more efficient and widely used. As literature suggests (Zhou, Yang, Du, Gong, & Luo, 2020), the hydrometallurgical procedure produces more wastewater than other forms of recycling. This has direct implications on the environmental impacts of the battery recycling, and thus it is expected that if the wastewater is minimised, or efficient treatment ways are developed the overall footprint will be significantly reduced. Moreover, there are studies suggesting that the full potential of this recycling method has not yet been realised completely (Thompson, et al., 2021). What is highlighted, is that more sophisticated methods and approaches in battery pre-treatment and disassembly of batteries can offer increased metal recovery rates, making this a very important step to the path for further battery price reductions. This essentially means that improved hydrometallurgical processes will improve the sustainability of LIB manufacturing by conserving materials and the energy and resources invested in their production.

ii. LCAs of EoL battery recycling

A recent literature review of EV battery LCAs found that only a small proportion of studies have included battery recycling in the scope of their research, even by sensitivity analysis (Aichberger & Jungmeier, 2020). For those that have, most of the data sources relied on secondary data such as the open-source EverBatt and GREET2 models. These open-source models provide a means of modelling recycling processes and supply chains, allowing comparisons to be made between the impacts of virgin materials used in battery manufacturing vs their recycled counterparts; however, they also rely on a range of in-built assumptions that may limit the accuracy of the results obtained.

Current EoL battery recycling LCA studies emphasise studying the recycling of cathode materials like cobalt, nickel and aluminium, as these materials account for the largest impacts in battery production (Aichberger & Jungmeier, 2020). The same literature review found that the median recycling benefit in terms of reduced GHG impact thanks to recycling is about 20kg CO_2/kWh, as the recycled materials substitute the primary materials, including the additional energy required to recycle the batteries. Only one study showed that pyrometallurgical recycling might lead to a net increase in GHG impact, whereas hydrometallurgical recycling performs (Romare & Dahllöf, 2017). If the recycled materials do not require further refining process to meet battery-grade requirements or are used in other purposes requiring less processing, battery recycling can deliver even greater environmental benefits with less energy used (Dai, Kelly, Gaines, & Wang, 2019).

The focus on cathode materials in existing LCA literature has painted a partial image of the overall impact of battery recycling, and has tended to favour processes, such as pyrometallurgical recycling, that does not recover as many metal types as hydrometallurgical recycling does. Also, battery types without cobalt or nickel are reported as not being recycled, due to the economic value of battery recycling being currently driven by the relative high prices of these two cathode metals (Romare & Dahllöf, 2017).

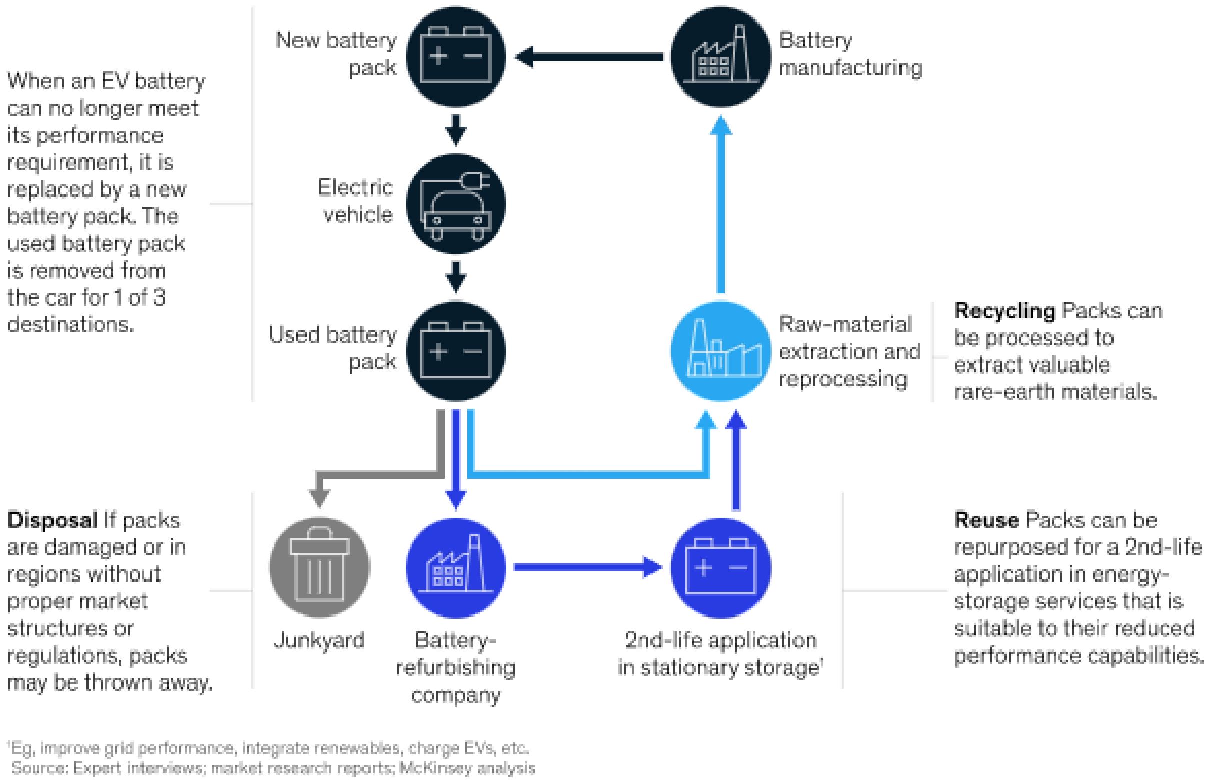
There are several areas requiring further investigation by LCAs of EoL battery recycling. For instance, recycling methods that are still under development like the direct recycling approach are lacking in terms of LCA coverage, despite these methods having sometimes shown technical advantages and increased economic value for recycled batteries compared to other more established methods (Aichberger & Jungmeier, 2020). Additionally, an earlier report for the European Parliament TRAN committee suggested that future studies should also account for other environmental effects, such as eutrophication impacts from solvents involved in hydrometallurgical processes, to draw a better picture of the overall environmental impacts and economic analysis of EoL battery recycling (Thomas, Ellingsen, & Hung, 2018).

iii. Second-life batteries as a storage solution

As mentioned in the previous chapters, the growing demand for Li-Ion batteries is expected to result in more challenges in the future of this technology and its environmental impacts. The end of life battery life cycle can follow different paths, with one of them being the second-life application in stationary energy storage as seen in Figure A11: Alternative options for EoL EV batteries. It is estimated that more than 50 GWh of grid storage capacity may become available by 2050 from second life EV batteries in the UK alone (Kamran, Raugei, & Hutchinson, 2021), a phenomenon also enhanced by an expected widespread adoption of EVs and Transport-as-a-service (TaaS) schemes. Repurposing end-of-life batteries for grid storage can significantly bring down the levels of manufacturing new grid-storage batteries and abate additional emissions in more intense extraction efforts for raw metals from lower-quality metal ores (Manjong, Usai, Burheim, & Strømman, 2021). The large supply of decommissioned batteries also enhances the economic viability of adopting LIB recycling schemes as the material extraction activities can be done efficiently.

Certain technical requirements are required for EoL EVs battery to be usable in second-life applications. Specifically for NMC batteries, one of the requirements is that the batteries have more than 70-80% remaining charge capacity, which means that they must not have reached their "ageing knee", roughly corresponding with 800-1900 full equivalent charge cycles (Martinez-Laserna, et al., 2018).

Figure A11: Alternative options for EoL EV batteries



Source: (McKinsey, 2020).

In terms of second-life batteries applications, three choices are deemed practicable. The first one is to maintain utilities' power reliability at lower cost by offering reverse energy capacity. Economic analysis has identified profitable opportunities for deploying these second-life batteries in several fields such as smart grid load dispatch in residential settings and to store variable renewable energy temporarily to support grid surges (Shahjalal, et al., 2022). In the latter application, EoL batteries defer transmission and distribution investments, while they also serve as renewable energy storage, which can be deployed in times of scarcity. A use case in China saw EoL EV batteries storing renewable energy over a short-term when demand is low and energy is readily available, then supplying it back to the grid to meet additional demand (Song, et al., 2019). This flexibility has led to expectations that second life batteries may be 30% to 70% less expensive than newly manufactured packs in 2025 (Engel, Hertzke, & Siccardo, 2019). Additionally, the increased residual value of new EV batteries by accounting their second-life usage could be transferred to new EVs and make substantial savings, as the EV battery cost makes up most of the EV cost, especially if the second-life use of the battery spans a prolonged period. Neubauer et al.'s LCA study modelled that EV batteries with 70% of the initial capacity after 15 years of first-life service can be extended for a further 10 years of second-life service to shave peak load demand and replace peak load power plants, as the fewer charge cycles per day and longer discharge duration in second-life application extends the lifespan of the batteries (Neubauer, Smith, Wood, & Pesaran, 2015).

Although the idea of second-life batteries as grid storage seems very promising, some challenges could surface in the near future. One of the main challenges is that power supply and demand need to always be balanced and this becomes more difficult as the rates of transport electrification grow. As second-life batteries become more established as energy storage solutions in the future, the flexibility of more conventional dispatchable power plants will become less readily available, and thus the grid may become less stable (Kamran, Raugei, & Hutchinson, 2021). Moreover, cycle and calendar ageing, with the second being the most significant in electric vehicles (Redondo-Iglesias, Venet, & Pelissier, 2018), have a significant effect on the capacity fading of EV LIBs at the end of their first life, resulting in some of the EoL batteries only being suitable for operation in less demanding applications. In terms of second life batteries price, the expected 30-70% cost advantage against newly manufactured batteries could drop to 25% by 2040, as the rate of decline in remanufacturing costs is expected to be more modest than the rate of decline in manufacturing cost for new batteries (Engel, Hertzke, & Siccardo, 2019). Finally, there are some studies (Xu, et al., 2020) suggesting that using batteries in their second life could cause further delays in critical raw material recycling, as the rates of recovery may not be able to catch up with future demand if batteries are repurposed and their service life is extended. In summary, because the economic viability and use cases for second-life batteries vary across geographical regions, further research is needed to carefully assess their potential economic and environmental benefits.

ANNEX 3: ADDITIONAL SUPPORTING MATERIAL FOR CHAPTER 5

This Annex provides additional material on important details on the LCA modelling conducted for this project by Ricardo on the current and future outlook for ICEVs and BEVs.

Additional information on the LCA impact categories (Section 5)

The following Table A2 provides a summary of the LCA environmental impact categories used in this study, and the abbreviations for these used in this report.

Table A2: List of impact categories for the study.

Impact Category	Abbreviation in Report	Indicator and Unit	Original Source
Climate change	GWP	Greenhouse gas emissions GWP100 in CO_2 eq	IPCC 2013
Energy consumption	CED	Cumulative energy demand in MJ (fossil and renewable)	ecoinvent 3.5 (Bourgalt 2017)
Photochemical ozone formation	POCP	Photochemical Ozone Creation Potential POCP in NMVOC eq	ReCiPe 2008
Particulate matter	PMF	Particulate matter formation in PM2.5 eq	De Leeuw 2002
Human toxicity, cancer and non-cancer	HTP	Comparative Toxic Unit for Human Health in CTUh	USEtox (Rosenbaum et al 2008)
Resource depletion - minerals and metals	ADP_MM	ADP ultimate reserves in Sb eq	Van Oers et al. 2002
Water scarcity	WaterS	Scarcity-adjusted water use in m^3	AWARE 2016

Source: Ricardo, based on a sub-set of the impact categories analysed in (Ricardo et al., 2020).

Additional material on the LCA modelling analysis (Section 5)

Modelling updates - overview

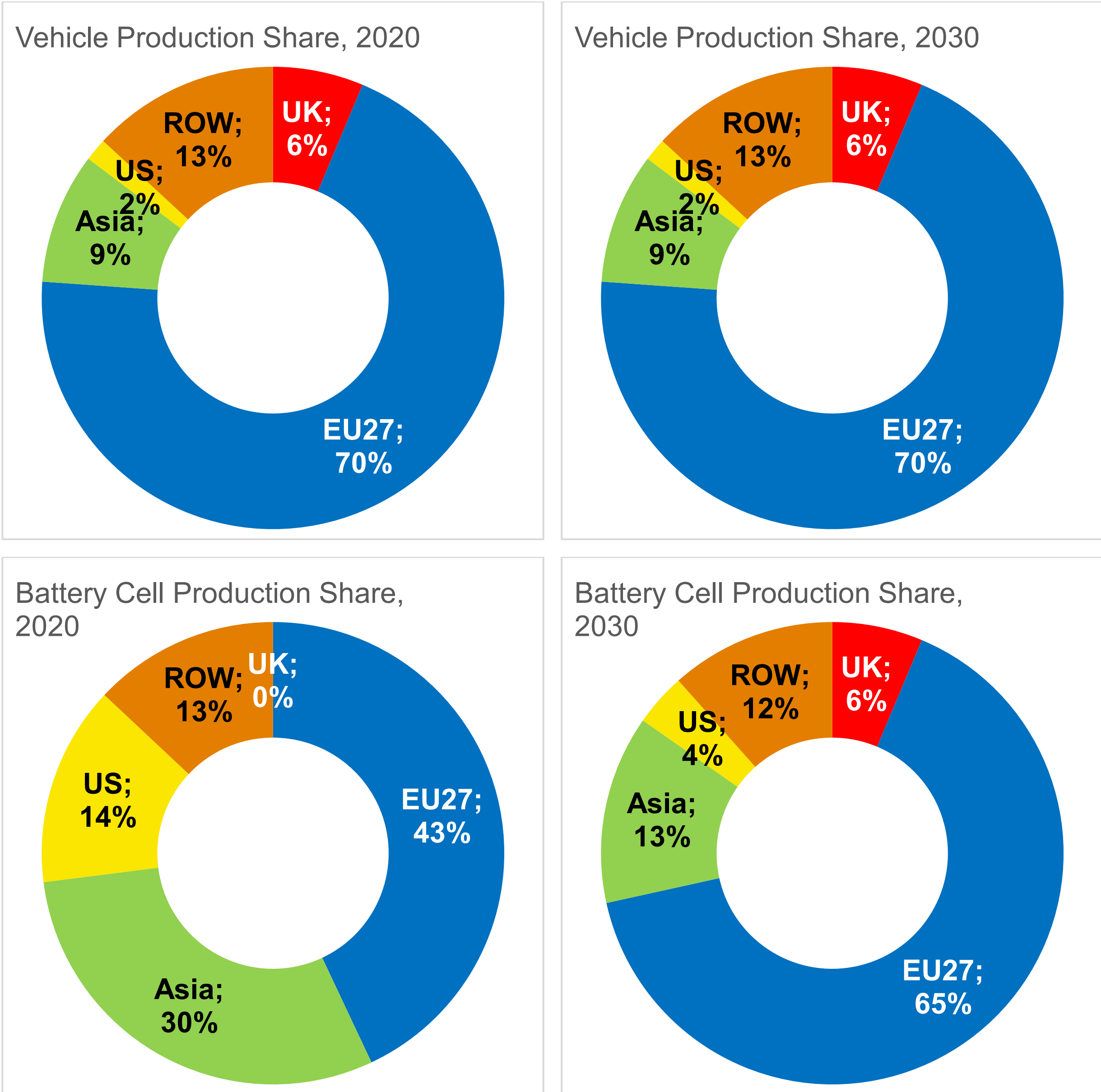
Table A3: Outline of updates/improvements to 2020 EC Vehicle LCA modelling

Area	Summary of Proposed Updates Versus 2020 CLIMA Project	Status
Electricity mix projections	Updates to EU27 and individual country mix based on Fit For 55 (to 2030 based on public data; with projections to 2050 based on previous analysis). Improvements have also been made to the assumptions on the future reduction in embedded emissions (i.e. 'capital goods') for solar electricity and in the overall modelling. Updates were also made to the future projected electricity mix for non-EU regions (which affects the projected future decarbonisation of key materials), based on (IEA, 2022).	Partial (EU27, MS)
Fuel mix projections	Updates to the liquid and gaseous fuel mix based on Fit for 55 as far as feasible (limited by the availability of sufficiently detailed data on this).	Limited update
Raw materials	Additional options and improvements for future decarbonisation of steel, aluminium and plastics (e.g. also as potentially influenced by CBAM).	Updated
Vehicle specification and performance (i.e. energy efficiency)	Significant updates/improvements to BEV (and ICEV) characterisation and performance (particularly energy consumption per km) based on Ricardo analysis for the EC supporting impact assessments for proposals for revisions to the car/van CO_2 regulations (i.e. Fit for 55).	Updated
Battery specification and manufacturing	Review and update of the battery energy density/battery chemistry projections and share of manufacturing by geography/location.	Updated*
End-of-life modelling	Updates to account for future proposals on battery recycling/material recycled content as part of the Sustainable Battery Regulation.	Partially updated**

Source: Ricardo.
Notes: Updated = implemented in Ricardo's latest vehicle LCA models; Partial or Limited = partially updated in Ricardo's models, additional work to be conducted in this project limited by data availability/detail. * Ricardo updated the current and projected battery production mix (by region/country) as part of recent work on vehicle LCA for UK Department for Transport (Dft); further updates were also made to the assumptions for future battery chemistry mix and improvement in battery energy density for this study (i.e. these were more conservative than for the previous analysis in (Ricardo et al., 2020)). ** Ricardo conducted an analysis of the effects on recycled content and end-of-life recycling and recovery targets resulting from the proposed Battery Regulation for a recent conference (ACI, 2022); no further updates have been possible.

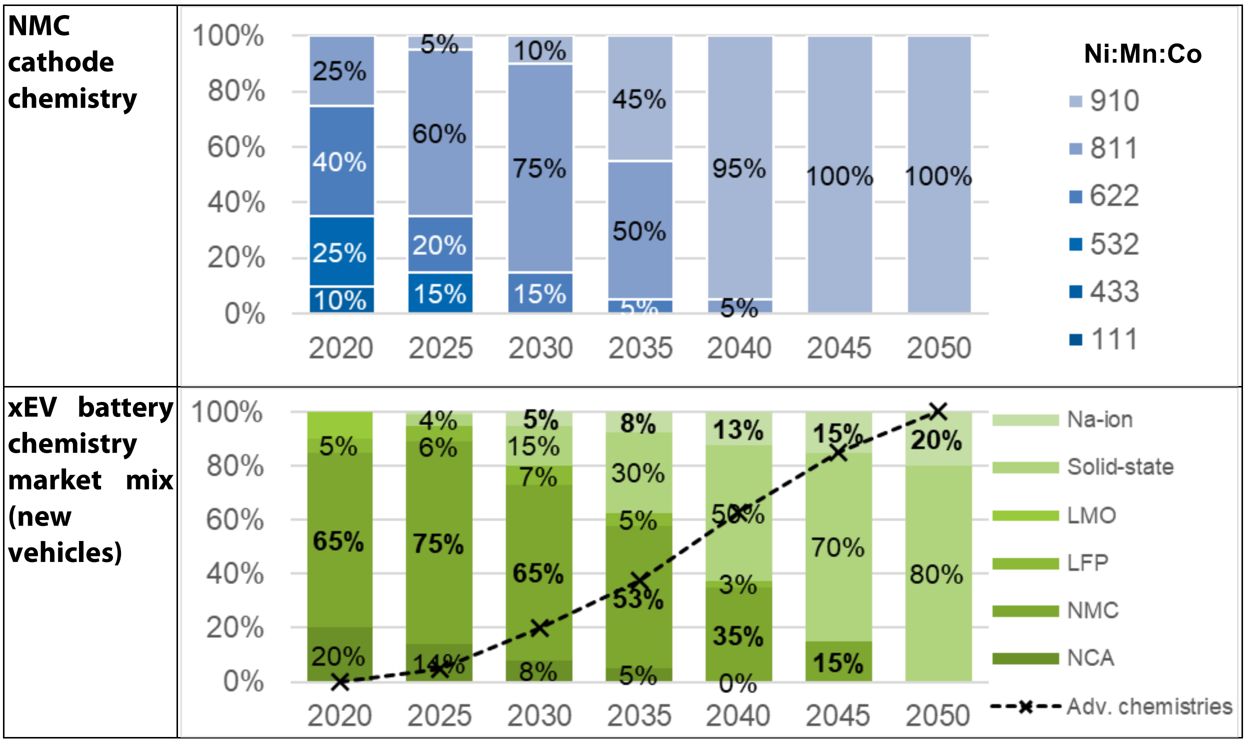
Manufacturing

Figure A12: Shares of vehicle and battery production as a % of total vehicles registered in the EU27



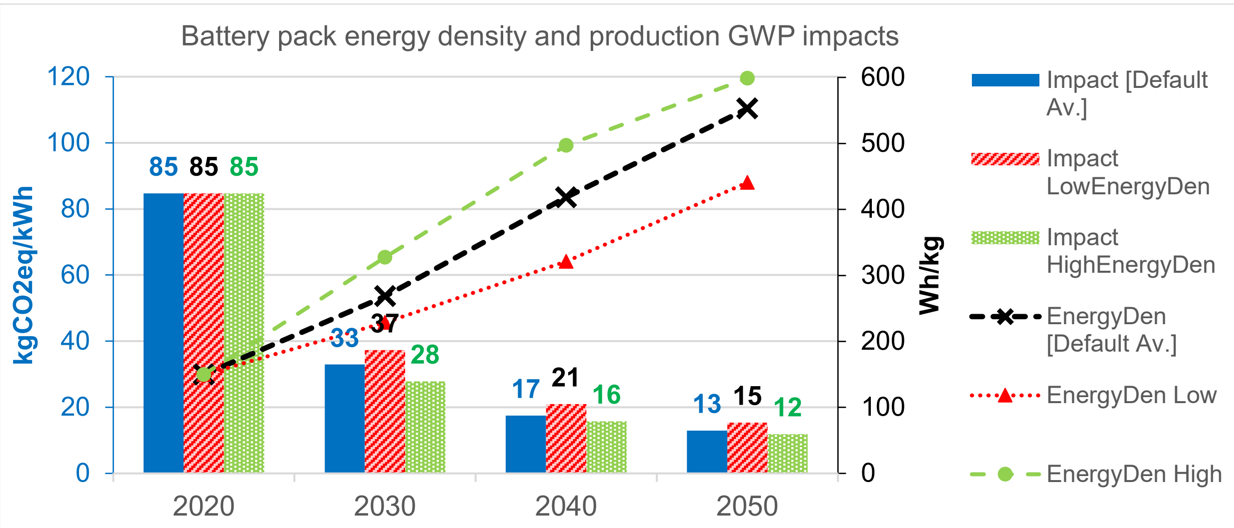
Source: Based on (Ricardo, 2021a).
Notes: ROW = Rest of the World. Manufacturing shares for Asia and for the EU27 are broken down into further detail in the LCA modelling based on confidential data provided by UK DfT, which cannot be presented here. Based on analysis of the planned/projected European battery production capacity, it is expected that Europe will be self-sufficient in battery manufacturing for automotive applications before 2030, e.g. (EUROBAT, 2021).

Figure A13: Summary of updated assumptions for default battery technology mix assumptions used in the LCA modelling for this project



Sources: Ricardo modelling, January 2023.
Notes: Na-ion = sodium ion battery chemistries, Solid-state = a range of solid-state battery chemistries, Lithium-ion battery chemistries: LMO = lithium manganese oxide cathode, LFP = lithium iron phosphate cathode, NMC = nickel, manganese, cobalt cathode chemistries, NCA = nickel cobalt aluminium cathode chemistries. 'Adv. Chemistries' = Advanced chemistries, including Na-ion and Solid-state battery chemistries.

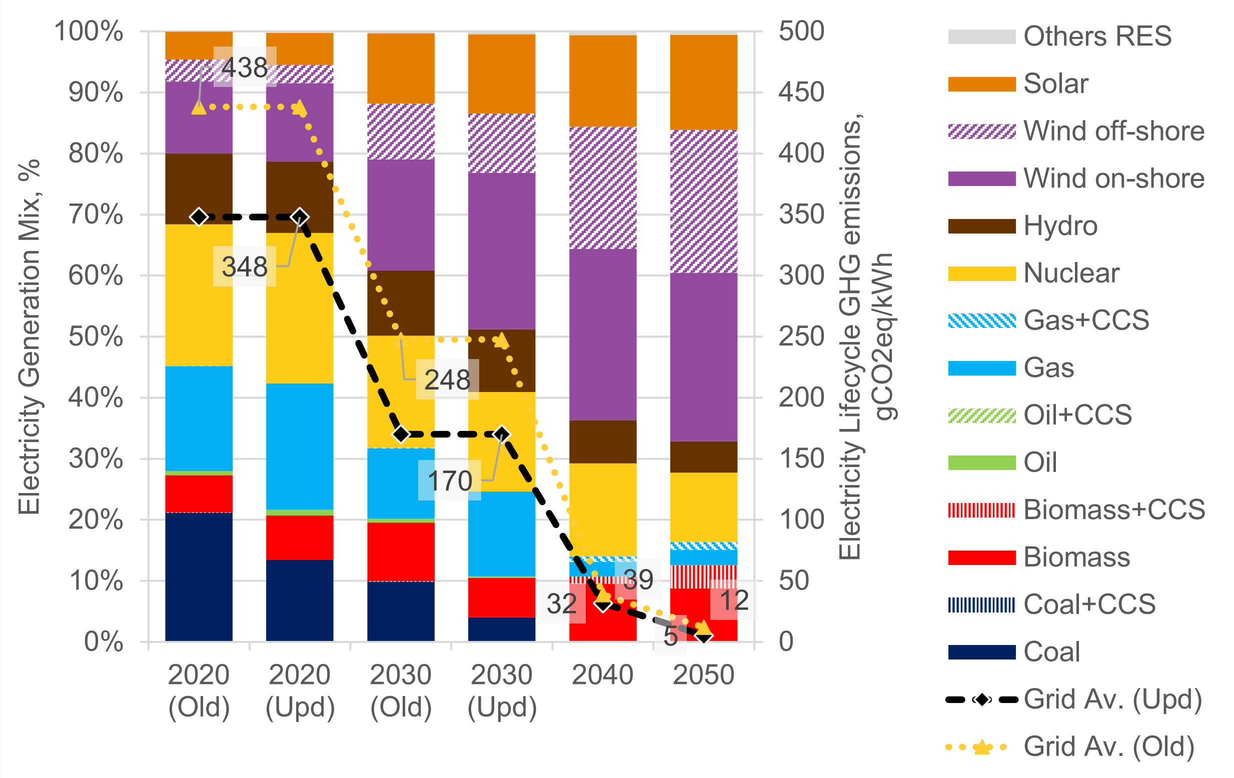
Figure A14: Updated battery pack energy density assumptions based on revised input data, and the calculated battery manufacturing GHG impact



Sources: Ricardo modelling, January 2023.

Fuels and electricity

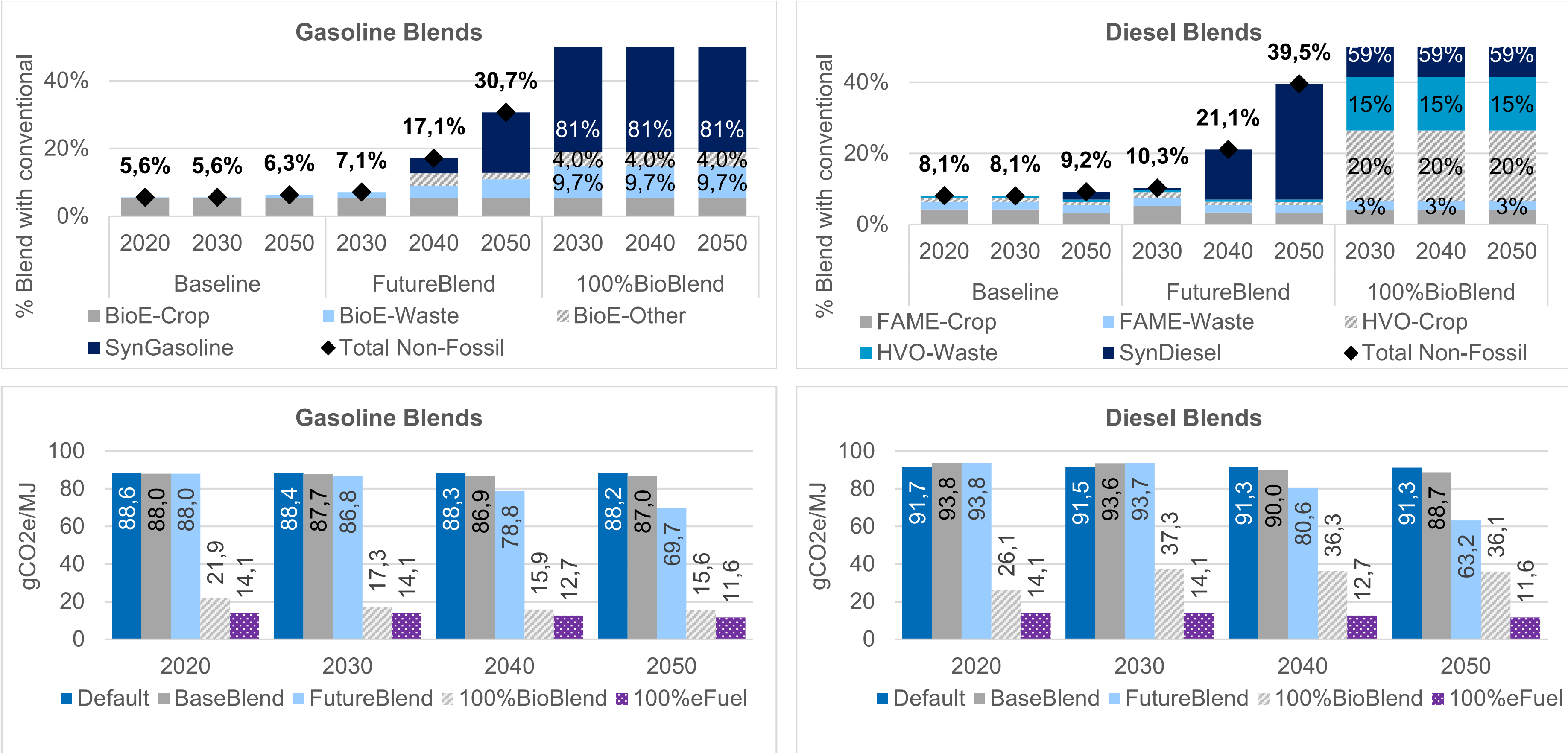
Figure A15: Overview of updates to the EU27+UK electricity mix projections for this project, compared to previous analysis



Sources: Ricardo modelling, January 2023.

Notes: Results are for the updated 'Tech1.5' scenario, presented for EU27+UK for comparability with the EU28 analysis performed in Ricardo's previous work for DG CLIMA (Ricardo et al., 2020); however, an EU27 electricity mix was also characterised and used as part of the current project (as well as updated electricity mixes for all the EU member states) based on statistical data for 2020 from (Eurostat, 2022) and modelled data future periods from (European Commission, 2021a).

Figure A16: Fuel blend/production mix assumptions and resulting GHG impacts used in the overall Vehicle LCA modelling



Source: Ricardo.

Notes: BioE = bioethanol, SynGasoline / SynDiesel includes biomass-to-liquid (BtL) chains. The blend/mix of fuel production chains assumed are only indicative as these were limited by the subset of the currently available fuels that have been modelled as part of this project. The diesel and gasoline blends for 2030 from (Ricardo et al., 2020) have been updated by Ricardo, for a FutureBlend scenario to better reflect the increased targets in the Fit for 55 legislative package for renewable energy, and the EC modelling of this (European Commission, 2021a). The blend/mix of fuel production chains assumed are only indicative as these were limited by the high-level information available from the EC's published scenarios. The 100% BioBlend scenario represents a sensitivity whereby an illustrative 100% biofuel substitution is achieved for all liquid and gaseous fuel types from 2030 onwards, as defined by Ricardo. No e-fuel / PtX chains are included in these fuel mix scenarios, however a separate sensitivity on 100% e-fuels was also conducted.

Vehicle operation

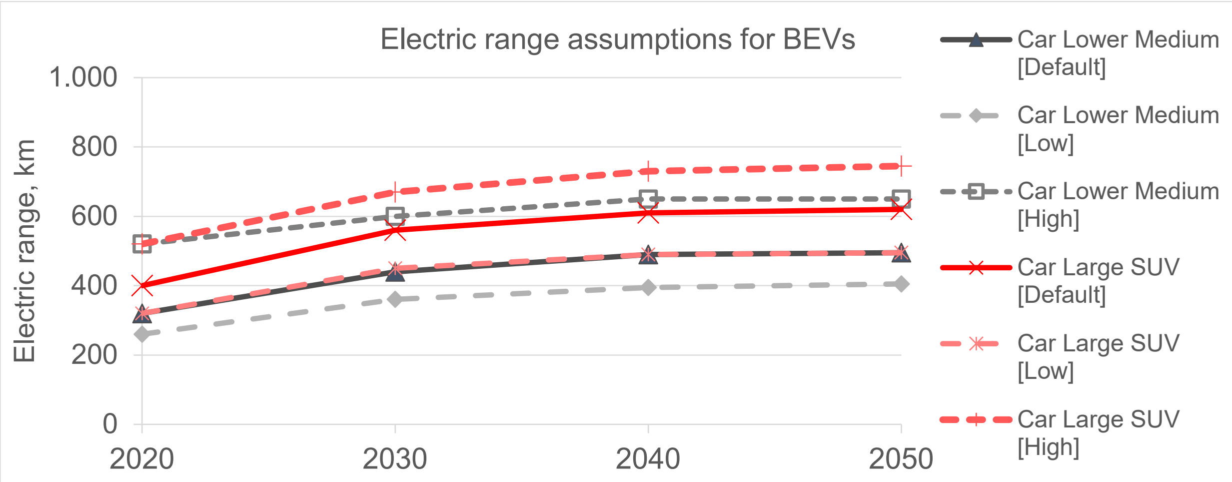
Table A4: Reference vehicle and powertrain characteristics used in the analysis

Vehicle Type	Powertrain Reference	Cycle	Energy, MJ/km	Mass, kg	GVW, kg	Power, kW	Lifetime km	Lifetime (years)
Car Lower Medium	ICEV-P	WLTP	2.15	1,325	2,500	96	225,000	15
Car Large SUV	ICEV-D	WLTP	3.07	2,149	3,500	182	270,000	15

Source: Ricardo, based on market average data for passenger cars.

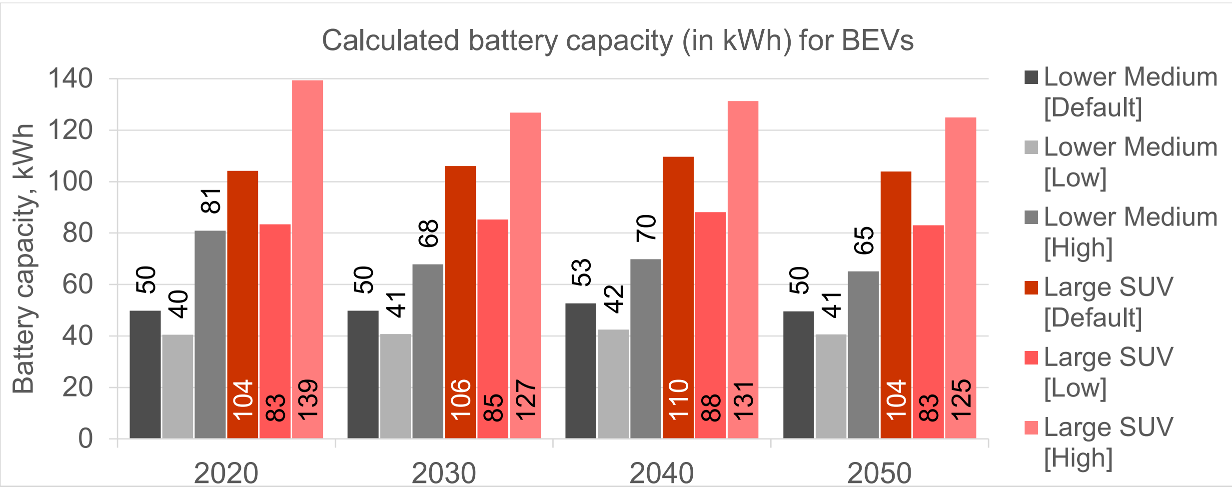
Notes: Energy consumption and unladen mass are calculated within the LCA model for the other different powertrain types. Mass is calculated based on the scaling parameters for different system components and other factors, such as the electric range (which affects the size/mass of the required battery).

Figure A17: BEV electric range default assumptions for passenger cars



Source: Ricardo.
Notes: Electric range defined based on standard test cycle, which is WLTP for LDVs. Study assumptions for electric range based on a market analysis by Ricardo for available and proposed models, and future expectations based on mass deployment and battery technology improvements and cost reduction.

Figure A18: BEV battery capacities for passenger cars, calculated based on the study methodology using electric range and vehicle energy consumption projections



Source: Ricardo.
Notes: Future battery capacities include accounting for assumed future improvements in overall vehicle efficiency, therefore requiring smaller batteries to achieve the same overall electric range.

End-of-life

The end-of-life recycling/material recovery rates used in the analysis are mostly the same as those utilised in Ricardo's previous analysis for EC DG CLIMA, as reported in Appendix A4.3 of that report (Ricardo et al., 2020). Updates have been made to the modelling to align the assumed future battery collection rates in the proposed Battery Regulation (i.e. 70% by 2030).

Further detail on vehicle LCA results - Sensitivity scenario settings

Table A5: Key parameters varied in the different sensitivities explored for the current situation, Lower Medium Car, 2020, EU27

Sensitivity	Lifetime km	Average Temp, °C	EoL Method	Electric Range, km	Battery Size, kWh
Default	225,000 km	+10 °C	PEF CFF	320 km	50 kWh
Activity low	**150,000 km**	+10 °C	PEF CFF	320 km	50 kWh
Activity high	**300,000 km**	+10 °C	PEF CFF	320 km	50 kWh
Activity SharedM	**450,000 km**	+10 °C	PEF CFF	320 km	50 kWh
Temp low	225,000 km	**-10 °C**	PEF CFF	320 km	50 kWh
Temp low+HP	225,000 km	**-10 °C + HP**	PEF CFF	320 km	50 kWh
Temp high	225,000 km	**+35 °C**	PEF CFF	320 km	50 kWh
EoL RecCon	225,000 km	+10 °C	**Recycled Content**	320 km	50 kWh
EoL AvoidB	225,000 km	+10 °C	**Avoided Burden**	320 km	50 kWh
Range low	225,000 km	+10 °C	PEF CFF	**260 km**	**40 kWh**
Range high	225,000 km	+10 °C	PEF CFF	**520 km**	**81 kWh**

Sources: Ricardo modelling input assumptions.

Notes: HP = Heat Pump, used to provide more efficient electric cabin heating and battery conditioning. Electric range assumptions based on regulatory (WLTP) conditions. PEF CFF = Product Environmental Footprint Circular Footprint Formula.

Table A6: Key parameters varied in the different sensitivities explored for the future projections, Lower Medium Car, 2030, EU27

Sensitivity	Lifetime km	Electric Range, km	Battery Size, kWh	Battery Energy Density, Wh/kg
Default	225,000	440 km	50 kWh	268 Wh/kg
Range/Battery low	225,000	**360 km**	**41 kWh**	**327 Wh/kg**
Range/Battery high	225,000	**600 km**	**68 kWh**	**228 Wh/kg**

Sources: Ricardo modelling input assumptions.
Notes: Range/Battery sensitivities explore the best and worst case options – i.e. (i) low range/small battery and high battery energy density improvement, plus (ii) high range/large battery and low battery energy density improvement.

ANNEX 4: PEER REVIEW SUMMARY

Commissioner:	Policy Department for Structural and Cohesion Policies, Directorate-General for Internal Policies
Report Developer	Ricardo
Reviewers:	1.Prof. Marzia Traverso (PhD), senior expert in LCA, Marbach am Neckar Germany, marziatraverso@gmail.com
References:	1. ISO 14040 (2006): Environmental Management - Life Cycle Assessment 2. Principles and Framework, 3. ISO 14044 (2006): Environmental Management - Life Cycle Assessment 4. Requirements and Guidelines, 5. ISO/TS 14071(2014): Environment Management-Life Cycle Assessment- Critical review processes and reviewer competencies: Additional requirements and guidelines to ISO 14044:2006

Scope of the critical review

The objective of the project is "to conduct critical review according to the ISO/TS 14071:2014, ISO 14040 (2006, 2018) and ISO 14044 (2006, 2018) for the report "Environmental challenges through the life cycle of battery electric vehicles. The report is intended to be published externally. The review was performed according to paragraph 6.3 of ISO 14040, because the study is intended to be disclosed to the public.

The critical review had the task to assess whether:

1. The methods used to carry out the LCA are consistent and in accordance with international standards (ISO 14040 and ISO 14044) particularly.
2. The methods used to carry out the LCA are scientifically and technically valid.
3. The information and data used are appropriate and reasonable in relation to the goal of the study.
4. The interpretations reflect the limitations identified and the goal of the study.
5. The report of the study is transparent and consistent.

This review is valid for the report issued in February 2023.Excluded from the scope of the audit were:

Verification of the assumptions and data on the examined facilities and disposal, the verification of the LCA models created, and the verification of the data sets and databases used.

The review process

The review process was coordinated between Ricardo team, the Policy Department for Structural and Cohesion Policies of the European Parliament, and the reviewer. The goal and scope, timeline of the project, level of detail, and target of the critical review have been discussed and established during the inception meeting.

The first round of review to the interim report contained 16 comments: 11 general and technical and 5 editorial. A clarification meeting online on the interim report has been organised to discuss the comments received and identifying improvements.

Eight comments to the final report were delivered.

Besides few issues in the first round and necessity to align the terminology, all comments were adequately addressed, and the related modifications in the report were completed. The critical reviewer checked the implementation of the comments in the first draft report while closing down all comments with the final report.

The reviewer acknowledges unrestricted access to all requested information as well as open and constructive dialogue during the critical review process with the Policy Department for Structural and Cohesion Policies of the European Parliament and with Ricardo, the study developer.

General evaluation

The study aimed to give provision of policy recommendations based on an extensive overview of BEVs and battery technology development, a comparison of BEV vs ICEV vehicle environmental impacts throughout the life cycle assessment technique according to the ISO 14010(2006) and ISO 14044(2006, 2018), and by understanding the current and future outlook based on Fit For 55 and technological development. It means that the study is definitely broader of the usual standard LCA study where a state of the current literature is a summary and no political recommendations are drawn.

The scope has been properly and detailed described by giving: vehicle and powertrains included, geographical and temporal conditions as well impact categories considered. The main focus has been given to the GHG/ global warming potential (GWP) and resource efficiency impact categories. However according to ISO 14040/44 other impacts have been assessed and reported in chapter 4.2.2.

The study consists of three main parts: an overview of the battery technology development followed by a comprehensive literature overview on the comparison of LCA on ICEV and BEV vehicles along their life cycle; an outlook of the comparison of life cycle impacts of BEVs vs ICEVs; and summary of policy recommendations.

The overview of BEV and battery technology (Chapter 2) is very relevant considering that the performance and environmental impact of this component strongly affect the environmental impact of a BEV vehicle. Further details on the LCA methodology, such as the definition of consequential and attributional LCA, allocation procedures, inventory data typologies are reported in the Annex 2.

The focus of the LCA literature review (Chapter 3) has been given to the GHG Impacts in Global Warming Potential according to the goal and scope of the study. After a first introduction on the LCA technique according to the ISO 14040 (chapter 3.2) to introduce the topic to and drive the readers who are not LCA experts, a detailed comparison between ICEV and BEV GHG impacts has been given for each life cycle stage. It helps communication but also gives the framework of the study according to the ISO 14040 and ISO 14044. The literature review and the report on the current BV and battery technology is very good, well documented and transparently reported.

Chapter 3 gives a very good overview of the diverse modelling assumptions taken in the studies in this field and the consequent different results obtained. Particularly relevant is also chapter 3.8 which highlights the limits of the literature studies and gaps which are still remaining in the field. This part is also well structured and well documented.

Chapter 4 is the key one where the main findings are reported. It confirmed the better performance of the battery electric cars in terms of life cycle GHG impacts compared with conventional gasoline and diesel vehicles operating in average EU conditions. However, battery technology together with the electricity grid represents the main component whose environmental impacts have the highest potential to be reduced in the future to achieve a reduction of 86% of CO2e in the transport sector by 2050. The chapter is well structured and clusters the results in a consistent way. Most of the results are derived by the LCA modelling of passenger vehicles, previously developed by Ricardo for the European Commission and which the framework is described in figure 4-1 of the report. Details and background data are reported in the main text or in the annex, for helping the reader to not be lost in too many details.

The last chapter 5 after an overview of the most important EU norms, directives and initiative, and having confirmed compatibility between the LCA findings and the current policy framework targets, reports a set of possible policy recommendations. The recommendations are consistently derived by the results.

Conclusion

The LCA studies reported in literature are based on the ISO 14040 (2006) and ISO 14044 (2006). The comparison of LCA results BEV vs ICEV is also based on the ISO 14040/44. The methods used in the LCAs and the modelling of the product system correspond to the state of the art. They are suitable for meeting the objectives formulated for the study. The report is very comprehensive and describes not only the scope, assumptions and limitations of the study in a transparent manner, but give also a good overview of the battery technology, and political context. All of these support the authors in developing consistently a set of political recommendations.

The reviewer found the overall quality of the methodology and its execution to be adequate for the purposes of the study. All data necessary are reported or the literature references are reported. The study is reported in a very comprehensive manner including a transparent documentation of its scope. The used secondary data sources, the used software and background data, the transparent documentation, the adequate combination with scenarios and sensitivity checks, as well as the discreet and careful interpretation and consequently recommendations make this report and its results very consistent, applicable and valuable.

Self-declaration of independence

I, the signatory, hereby declare that:

- I am not a full-time or part-time employee of the commissioner or practitioner of the LCA study;
- I have not been involved in defining the scope or carrying out any of the work to conduct the report, i.e. I have not been part of the commissioner's or practitioner's project team(s);
- I do not have vested financial, political or other interests in the outcome of the study.

I declare that the above statements are truthful and complete.

Marbach am Neckar, Germany, 8th Feb. 2023

Marzia Traverso

Prof. Marzia Traverso

This study provides an up-to-date expert assessment and comparison between the life cycle's carbon footprint of BEV and ICEV passenger cars. It presents evidence from the literature and from LCA modelling and concludes with policy recommendations. The analysis includes sensitivities, regional variations for six Member States, and also the effects of technical and legislative development on the potential outlook up to 2050.

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