Institute of Transport Economics
Norwegian Centre for Transport Research

Circularity and environmental impacts of passenger vehicles

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Summary

This report reviews circular economy principles and life cycle assessment (LCA) application throughout a passenger vehicle's life cycle. Considering these approaches jointly allowed us to map sustainability perspectives and their interconnection. Supporting policy was also reviewed since it provides frameworks that industry must act within - with findings that sustainability requirements will be tightened, in particular for electric vehicle batteries. The review led to identification of key circularity factors and LCA parameters as well as an evaluation of research gaps that can be filled in further LCA work. Seven key gaps were identified, including 1) Vehicle and battery lifetime, 2) Maintenance resource and energy needs, 3) Operation energy use, 4) Vehicle utility, 5) Recyclability, 6) Vehicle and battery End-of-Life options, and 7) Allocation between multipurpose utilities. For these areas, we will collect and compile real-life vehicle and battery information/ statistics from CELECT-partners to compile detailed and comprehensive life cycle inventories.

Kort sammendrag

Denne rapporten gir en gjennomgang av prinsippene for sirkulær økonomi og anvendelsen av livsløps-vurdering (LCA) gjennom livsløpet til en personbil. Ved å betrakte disse tilnærmingene i fellesskap, kart-legges sentrale bærekraftsperspektiver og deres innbyrdes forhold. Støttende politikk ble også gjennomgått, da det gir rammene som industrien må operere innenfor – med funn som indikerer at bærekraftskravene, særlig for elbilbatterier, vil skjerpes betraktelig i årene som kommer. Denne gjennomgangen ledet til identifisering av viktige faktorer for sirkularitet og LCA-parametere, sammen med en evaluering av forskningsbehov som kan adresseres i fremtidig LCA-arbeid. Syv nøkkelområder ble identifisert for videre forskning: 1) kjøretøy- og batterilevetid, 2) ressurs- og energibehov ved vedlike-hold, 3) energibruk i drift, 4) kjøretøyets nytteverdi, 5) resirkulerbarhet, 6) sluttbehandling av kjøretøy og batterier, og 7) allokering ved flerbruksnytte. For disse områdene vil vi samle inn reelle data fra CELECT-partnere for detaljerte livsløpsregistre.

Preface

Transport sector environmental sustainability and reducing transport related environmental impacts have become increasingly important both nationally and internationally in recent years. This drive towards improved sustainability has led to a transition from internal combustion engine vehicles to zero tailpipe emissions vehicles. The transition has resulted in increased electrification of passenger vehicle fleets both in Norway and across Europe. This report forms part of the CELECT project (Circular Economy, Life Cycle Assessment, Electrification and Car Transactions), which has the aim to provide research-based support towards the electrification of the Norwegian passenger vehicle fleet. Led by TØI, CELECT partners include the Norwegian Electric Vehicle Association, the Norwegian Automobile Federation (NAF), the Norwegian Car Industry Association (NBF), the Norwegian Public Roads Administration (SVV), and key players from Norwegian industry including Fremtind Forsikring AS, Bertel O. Steen AS, Autoretur AS, Batteriretur AS, and ECO STOR AS.

The report forms a joint component in the CELECT project for WP2 Environmental impacts (using life cycle assessment, LCA) and WP3 Circular economy. Application of LCA and circular economy principles are two disparate approaches that can be used for assessing and quantifying sustainability. Although the approaches are often applied individually, research shows they should instead be applied together to work towards a holistic sustainability perspective. This is of key importance when considering the overall question of how to achieve the most sustainable car fleet transition. The report provides a review which forms the basis for a circularity-based framework for passenger vehicles, according to which LCA work will be later performed in the CELECT project.

The report was written by Rebecca Thorne and Linda Ager-Wick Ellingsen, with review and input from CELECT project team members from TØI. Final quality assurance was performed on the report according to TØI guidelines, with internal review from Erik Figenbaum, Frants Gundersen, and Bjørne Grimsrud.

Oslo, November 2024
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ENGLISH **Summary**

Circularity and environmental impacts of passenger vehicles

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A significant transformation is underway in Norway's passenger vehicle fleet, driven by ambitious political targets as part of the country's green transition. While this transition will reduce tailpipe emissions and decrease vehicle fossil fuel consumption, it will also shift emissions upstream and increase reliance on various metals and minerals. Some of these materials are defined as critical in the EU with high supply risk. Moreover, the potential environmental benefits of electric vehicles (EVs) are dependent on several factors, including the carbon intensity of the electricity mix used for charging and the vehicles' usage patterns. Collectively, these aspects lead to a need for developing a more circular vehicle system and an increase in overall environmental sustainability.

This report provides an extensive review of circular economy (CE) principles and life cycle assessment (LCA) applications throughout different stages of a passenger vehicle's life cycle, mapping their sustainability perspectives and the interplay between these. The report comprises part of the CELECT project (Circular Economy, Life Cycle Assessment, Electrification and Car Transactions), which has the aim to provide research-based support towards the electrification of the Norwegian passenger vehicle fleet.

CE strategies aim to retain the value of materials within the system by extending product lifespans, reducing waste, and minimizing the need for new raw materials. For passenger vehicles, this involves strategies such as enhancing reuse, repair, refurbishment, and recycling efforts. LCA evaluates emissions, resource consumption, and environmental impacts throughout a vehicle's life cycle, from raw material extraction to end-of-life. The systems perspective that LCA offers is especially relevant for evaluating a transition from conventional internal combustion vehicles to electric vehicles due to the differing environmental profiles. However, applying these principles in isolation may not guarantee a combined reduction in environmental impacts alongside more circular product systems.

Circular Economy and Life Cycle Assessment: A Unified Approach

The review findings underscore the need to combine circularity and LCA approaches to achieve overall environmental sustainability while minimizing burden shifting. Together, the joint use of LCA and CE principles provides a holistic perspective, giving the potential to apply strategies

to increase vehicle circularity whilst ensuring a reduction of environmental impacts across all life cycle stages. This integration is essential as the automotive industry faces mounting regulatory pressures, particularly regarding EV batteries and their overall environmental footprint.

Supporting Policy

This report also reviews supporting policy since it provides a framework that industry must act within. The findings indicate that sustainability requirements will be greatly tightened in coming years, particularly for EV batteries.

Regulatory frameworks are critical in driving the automotive sector toward sustainable practices. The EU's new *Circular Economy Action Plan* identifies automotive and battery sectors as key targets for circularity principles. With the EU's new *Batteries Regulation*, more stringent sustainability requirements are being implemented, focusing on carbon footprint declarations, increased recycling efficiencies, and extended producer responsibility. In Norway, ambitious goals of achieving a zero-emission vehicle fleet by 2025 reflect the necessity for a greener transport sector. In addition, the country has developed its own *National Strategy for a Green Circular Economy* and *Norway's battery strategy*, focusing on recycling and managing EV batteries, as well as aligning with European Union regulations.

Key Findings and Research Gaps

Reviewing the research field allowed us to identify key circularity factors and LCA parameters, as well as evaluating research gaps that should be addressed in further LCA work. Nine research gaps were identified, including seven that are critical. These seven research gaps include:

Vehicle and battery lifetime: These parameters are handled in a generic manner, where typically vehicle lifetime is set to a certain number of years or mileage, while battery lifetime is often considered as equal to vehicle lifetime. Furthermore, differences in vehicle lifetime between powertrain technologies or model year are generally not considered. Some LCA studies address the battery lifetime uncertainty through an uncertainty analysis considering battery replacement, but this is not always the case.

Maintenance resource and energy needs: Maintenance resource and energy needs are often omitted or modelled in a generic manner using data from databases. If included, current LCA studies do not sufficiently distinguish the needs for different powertrain technologies or vehicle sizes/segments.

Operation energy use: Operational energy use is often modelled with an assumed energy use, sometimes using test data for one or several specific vehicle models on the market. Most studies tend to focus on medium sized passenger vehicles, while very few consider energy use of different vehicle sizes/segments.

Vehicle utility: LCA studies considering passenger vehicles do not tend to focus on user numbers (e.g. shared mobility) or capacity but rather on technologies. LCA studies tend to report impacts in terms of vehicle-kilometer (or per vehicle) while passenger-kilometer is primarily considered in studies comparing different transport modes (e.g., buses, planes, ferries). Effects of use characteristic variation (e.g. distance driven) are sometimes studied in LCA sensitivity analyses but are typically not based on real data regarding usage patterns.

Recyclability: Most LCA studies considering end-of-life treatment use generic data from databases or literature with no differentiation between powertrain technologies, size/

segments, or model year. Furthermore, statistics about recycling and recovery rates or reuse of components/materials are not considered specifically.

Vehicles and batteries end-of-life options: A growing body of literature is considering battery reuse, repurposing, and recycling. The access to real-life battery data is limited though and studies often base inventory data and analysis on coarse assumptions about battery life, as well as repurposing needs and second life application and durability.

Allocation between multipurpose utilities: Various allocation alternatives may be used in an attributional LCA to ascribe benefits of multipurpose use of components. Physical (i.e., energy) and economic allocation are particularly relevant, but LCA studies tend to set the allocation keys based on assumptions of use and lifetime, rather than real-life data.

For the seven key research gaps identified, we aim in the CELECT project to collect real-world data from project partners (i.e. key stakeholders in the vehicle industry) to develop more complete, accurate and comprehensive life cycle inventory models.

The Road Ahead

Achieving a fully circular and sustainable automotive sector will require research together with innovation from the industry and continued support from regulatory bodies. By assessing CE strategies together with LCA, and filling the identified research gaps, meaningful steps toward sustainability can be mapped out.

This work will offer valuable scientific insights that can form the foundation for future research, contributing to a more sustainable automotive sector. While it may not directly influence policy in the short term, it aims to provide essential knowledge that can inform future developments in research, policy and industry practices.

Transportøkonomisk institutt
Stiftelsen Norsk senter for samferdselsforskning

NORSK **Sammendrag**

Sirkulærøkonomiske prinsipper og miljømessig bærekraft for personbiler

TØI rapport 2055/2024 • Forfattere: Rebecca Thorne, Linda Ager-Wick Ellingsen • Oslo, 2024 • 49 sider

Norges personbilflåte gjennomgår en betydelig transformasjon, drevet av ambisiøse politiske mål som en del av landets grønne omstilling. Selv om denne overgangen vil redusere eksosgasser og forbruk av fossile drivstoff, vil den også forskyve utslippene oppstrøms og øke avhengigheten av ulike metaller og mineraler. Noen av disse materialene er definert som kritiske i EU, med høy forsyningsrisiko. I tillegg avhenger de potensielle miljøfordelene til elektriske kjøretøy av flere faktorer, inkludert karbonintensiteten i elektrisitetsmiksen som brukes til lading, samt kjøretøyenes bruksmønstre. Samlet sett fører disse aspektene til et behov for å utvikle et mer sirkulært kjøretøysystem og øke den totale miljømessige bærekraften.

Denne rapporten gir en omfattende gjennomgang av prinsippene for sirkulærøkonomi og livsløpsvurdering (LCA) i ulike faser av en personbils livsløp, og kartlegger bærekraftperspektivet deres og samspillet mellom disse. Rapporten er en del av CELECT-prosjektet (Circular Economy, Life Cycle Assessment, Electrification and Car Transactions), som har som mål om å gi forskningsbasert støtte til elektrifiseringen av den norske personbilflåten.

Sirkularitetsstrategier har som mål å bevare verdien av materialer innenfor systemet ved å forlenge produktets levetid, redusere avfall og minimere behovet for nye råmaterialer. For personbiler innebærer dette strategier som å fremme vedlikehold, gjenbruk, reparasjon, ombruk og resirkulering. LCA vurderer utslipp, ressursforbruk og miljøpåvirkning gjennom hele kjøretøyets livsløp, fra råmaterialutvinning til sluttbehandling. Systemperspektivet som brukes i LCA er spesielt relevant for å evaluere en overgang fra konvensjonelle forbrenningskjøretøy til elektriske kjøretøy på grunn av de ulike miljøprofilene. Imidlertid kan ikke anvendelsen av disse prinsippene isolert sett garantere en samlet reduksjon i miljøpåvirkninger samt mer sirkulære produktsystemer.

Sirkulærøkonomi og livsløpsvurdering: en enhetlig tilnærming

Kartleggingen understreker behovet for å kombinere sirkularitet og LCA-tilnærminger for å oppnå overordnet miljømessig bærekraft samtidig som problemforskyvning minimeres. Samlet bruk av LCA- og sirkularitetsprinsipper gir et helhetlig perspektiv, som gjør det mulig å anvende strategier som øker kjøretøyers sirkularitet, samtidig som man oppnår en reduksjon av miljøpåvirkninger gjennom alle livssyklusfaser. Denne integrasjonen er essensiell ettersom bilindustrien står overfor økende regulatoriske krav, spesielt når det gjelder elbilbatterier og deres totale miljøavtrykk.

Politisk tiltak

Denne rapporten gjennomgår støttende politiske tiltak, ettersom tiltakene setter rammene som industrien må handle innenfor. Gjennomgangen indikerer at bærekraftkravene vil bli betydelig skjerpet i de kommende årene, spesielt for elbilbatterier.

Regulatoriske rammeverk er avgjørende for å drive bilindustrien mot bærekraftige praksiser. EUs nye *Circular economy action plan* identifiserer bil- og batterisektoren som hovedmål for sirkularitetsprinsipper. Med EUs nye batteriforordring *(Batteries Regulation)* implementeres strengere bærekraftskrav, med fokus på karbonfotavtrykksdeklarasjoner, økt resirkulerings-effektivitet og utvidet produsentansvar. I Norge gjenspeiler ambisiøse mål om å oppnå en nullutslipps personbilsflåte innen 2025 nødvendigheten av en grønnere transportsektor. I tillegg har landet utviklet sin egen *Nasjonal strategi for ein grøn sirkulær økonomi* og *Norges batteristrategi*, med fokus på resirkulering og håndtering av elbilbatterier, samt samsvar med EUs regelverk.

Nøkkelfaktorer og forskningshull

Ved å gjennomgå forskningsfeltet har vi identifisert nøkkelfaktorer for sirkularitet og LCA-parametere, samt evaluert forskningshull som bør adresseres i videre LCA-arbeid. Ni forsk-ningshull ble identifisert, inkludert syv som er kritiske. Disse syv forskningshullene inkluderer:

Kjøretøy- og batterilevetid: disse parameterne håndteres på en generell måte i LCA, hvor kjøretøyets levetid typisk settes til et visst antall år eller kjørelengde, mens batteriets levetid ofte anses som lik kjøretøyets levetid. Videre tas forskjeller i kjøretøyets levetid mellom drivlinjeteknologier eller årsmodell generelt ikke i betraktning. Noen LCA-studier adresserer usikkerhet knyttet til batterilevetid gjennom en usikkerhetsanalyse som vurderer batteribytte, men dette er ikke alltid tilfelle.

Ressurs- og energibehov for vedlikehold: blir ofte utelatt eller modellert på en generell måte ved bruk av data fra databaser. Hvis det er inkludert, skiller ikke LCA-studier tilstrekkelig mellom behovene til ulike drivlinjeteknologier eller kjøretøystørrelser/-segmenter.

Energibruk under drift: drift modelleres ofte med et antatt energiforbruk, noen ganger basert på testdata for en eller flere spesifikke bilmodeller på markedet. De fleste studier fokuserer på mellomstore personbiler, mens svært få vurderer energiforbruk for ulike kjøretøystørrelser/-segmenter.

Kjøretøyets nytteverdi: LCA-studier som vurderer personbiler har en tendens til å fokusere på teknologi fremfor antall brukere (f.eks. delt mobilitet) eller kapasitet. LCA-studier rapporterer vanligvis påvirkninger i form av kjøretøykilometer (eller per kjøretøy), mens passasjerkilometer primært vurderes i studier som sammenligner ulike transportmidler (f.eks. busser, fly, ferger). Effektene av variasjoner i bruksegenskaper (f.eks. kjørelengde) blir noen ganger studert i LCA-studienes sensitivitetsanalyser, men er vanligvis ikke basert på reelle data om bruksmønstre.

Resirkulerbarhet: de fleste LCA-studier som vurderer sluttbehandling, bruker generiske data fra databaser eller litteratur uten differensiering mellom drivlinjeteknologier, størrelse/segmenter eller årsmodell. Videre blir statistikk om resirkulerings- og gjenvinningsrater eller gjenbruk av komponenter/materialer ikke spesifikt vurdert.

Sluttbehandling av kjøretøy og batterier: stadig mer litteratur vurderer gjenbruk, ombruk og resirkulering av batterier. Tilgangen til reelle batteridata er imidlertid begrenset, og studier baserer ofte inventardata og analyser på grove antakelser om batterilevetid, samt håndterings-behov av brukte batterier og gjenbruk/ombruk og varighet.

Allokering mellom flerbruksnytte: ulike allokeringsalternativer kan brukes i en attribusjonell LCA for å tildele fordeler ved flerbruksnytte av komponenter. Fysisk (f.eks. energi) og økonomisk allokering er spesielt relevante, men LCA-studier har en tendens til å fastsette allokeringsnøklene basert på antakelser om bruk og levetid, i stedet for reelle data fra virkeligheten.

For de syv kritiske forskningshullene vi har identifisert, har vi som mål i CELECT-prosjektet å samle empiriske data fra prosjektpartnere (dvs. sentrale aktører i bilindustrien) for å utvikle mer komplette, nøyaktige og omfattende livsløpsinventar.

Veien videre

Å oppnå en fullt sirkulær og bærekraftig bilindustri vil kreve forskning sammen med innovasjon fra industrien og fortsatt støtte fra regulatoriske organer. Ved å vurdere sirkularitetsstrategier sammen med LCA, og fylle forskningshull, kan meningsfulle grep mot økt bærekraft kartlegges.

Dette arbeidet vil gi verdifulle vitenskapelige innsikter som kan danne grunnlaget for fremtidig forskning, og bidra til en mer bærekraftig bilindustri. Selv om det kanskje ikke direkte påvirker politikk på kort sikt, har arbeidet som mål å bidra med essensiell kunnskap som kan informere fremtidige utviklinger innen forskning, politiske tiltak og industripraksis.

1 Introduction

Within the project CELECT (Circular Economy, Life Cycle Assessment, Electrification and Car Transactions) a subgoal is to unite and optimise circularity and environmental sustainability outcomes for the Norwegian passenger vehicle fleet. A Circular Economy (CE) suggests a system where materials are preserved to close the loops of resource usage in both production and consumption processes. This approach helps extend the life cycle of products and preserves their inherent value, primarily through the practices of reuse, repair, refurbishment, and recycling. The importance of increasing vehicle circularity is compounded when the sector size and importance is considered, and due to the extent of non-circular and non-renewable flows today. Although there is limited consensus in the academic community regarding a universal CE definition, its principles are increasingly being integrated into policies. Notably, in the European Union (EU), the strategic focus on circular practices has been significantly bolstered following the release of the latest Circular Economy Action Plan. This plan underscores the automotive and battery sectors as key areas for implementing CE principles. Nevertheless, as momentum builds, research is increasingly revealing that circularity does not necessarily translate to environmental sustainability. This makes the links between circularity and environmental sustainability increasingly important to understand and means that approaches focusing on the circularity should be complemented with life cycle assessment (LCA) to ensure that a systems perspective is taken.

This report acts as the first step towards developing an environmentally sustainable CE framework for passenger vehicles. It maps CE principles, provides a review of LCA literature, and summarises relevant policy and regulations in the EU and Norway. Therefore, the report represents the outcome of the information gathering phase for the LCA and circularity work packages in CELECT (WP2 and WP3, respectively). The direct output of this work is input for the cradle-to-cradle LCA cases, according to the workflow shown in Figure 1.1.

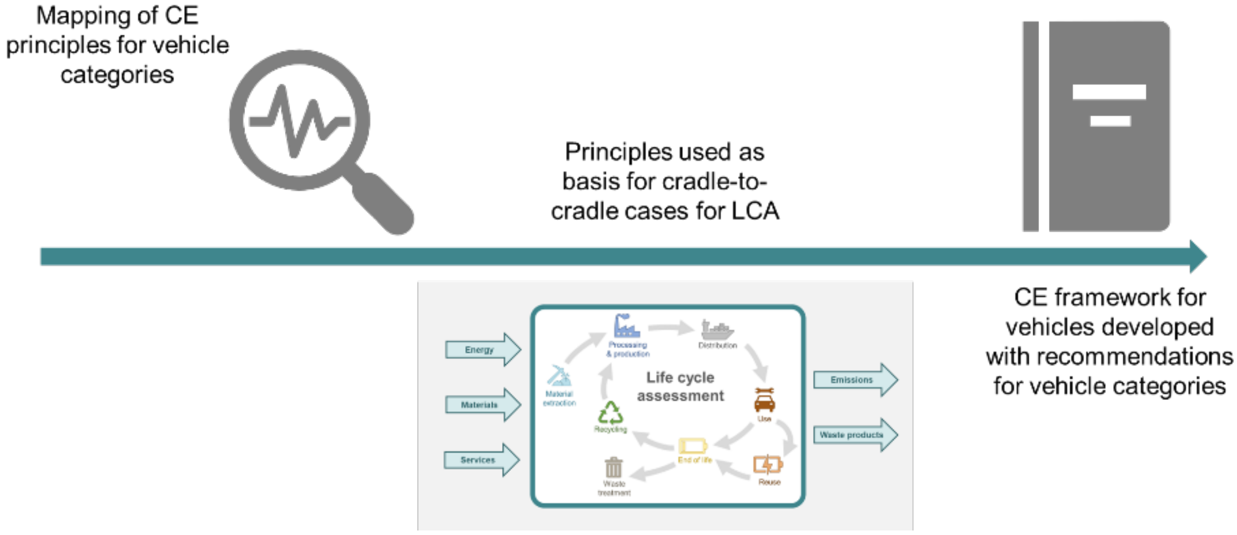
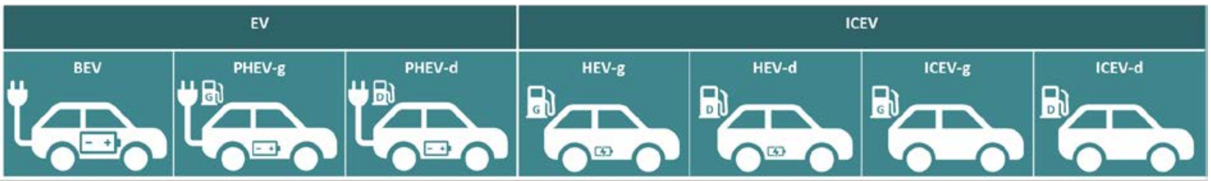


Figure 1.1: Interaction of circularity and LCA work in CELECT project.

The work focuses on passenger vehicles, and only on internal combustion engine vehicles (ICEVs) and electric vehicles (EVs) since these make up 99.99 % of the Norwegian fleet as of 2023 (SSB, 2024a). EVs are further categorized into battery electric vehicles (BEVs) and plug-in hybrid electric vehicles (PHEVs). While the BEV powertrain relies solely on a high-voltage lithium-ion traction battery (LIB) and an electric motor for propulsion, the PHEV powertrain can also utilize gasoline or diesel in an internal combustion engine (ICE) for propulsion. The electric driving range of PHEVs is lower than that of BEVs as the LIB in PHEVs are smaller than those in BEVs. Note that fuel cell electric vehicles (FCEVs) would also fall into the EV category but because fuel cell powertrains have so far not gained traction for passenger vehicles, FCEVs are not considered here.

ICEVs are subdivided into diesel, gasoline, and hybrid electric vehicles (HEVs). In spite of their name, HEVs rely primarily on combustion of fuel in an ICE for propulsion, while the small battery powers an electric motor that provides extra power during starts and acceleration. In contrast to high-voltage LIBs in EVs, the HEV battery is not externally charged by a plug but relies instead on charging through regenerative breaking as well as the ICE (by running the electric motor as a generator). Vehicles relying on combustion of methane gas (from either fossil or bio-based sources) in an ICE would also fall into the ICEV category but because there are very few of these in Norway, gas fueled passenger vehicles are not considered in this work. An overview of vehicle powertrains considered in this report is shown in Table 1.1. Note that “g” and “d” in the table denote gasoline and diesel, respectively. Shared mobility and other CE strategies not directly related to the use of passenger vehicles are outside the scope of this report.

Table 1.1: Overview of vehicle powertrains considered in this report.



The content of the report is as follows: Section 2 broadly describes circularity and LCA principles as well as their general application to passenger vehicles. Although the approaches are often considered in isolation, the benefits of a combination of methods are outlined here. Section 3 expands further to detail strategies to increase circularity at every life cycle stage of a passenger vehicle, along with the relative environmental impacts of these as shown by LCA. Since the composition of a vehicle fleet (and its evolution over time) is ultimately governed by policy, it is of key importance when considering environmental sustainability. Thus, Section 4 considers existing supporting policy infrastructure surrounding the circularity landscape in the EU and in Norway. Section 5 provides an overview of identified key circularity factors and their relation to LCA parameters as well as evaluating the coverage of these in the LCA literature and potential research gaps. Finally, Section 6 offers a brief summary of findings and recommendations for future LCA work within CELECT.

2 Circular economy principles and life cycle assessment as complementary perspectives

Multiple strategies can be taken to minimize environmental impacts of passenger vehicles, depending on the focus and the environmental impact under study. Optimizing material circularity or reducing environmental impacts according to the results of an LCA are two such approaches. In practice, research increasingly shows that these strategies can and should be combined to optimize sustainability from a holistic perspective.

2.1 Circularity principles

Considering the finite resources available, humanity should live within available resources and planetary boundaries[1] to allow consumption and waste generation to balance with resource regeneration and waste absorption (Steffen et al., 2015). However, studies show that globally we live within the equivalent of almost two Earths (Global Footprint Network, 2024) with a mostly linear economy amounting to 'take-make-waste' material and energy flows (Korhonen et al., 2018). Some recycling and reuse feedbacks may be present, emphasizing recovery and recycling of materials from waste streams, but the overall direction of resource flow is from cradle to grave (Figure 2.*1*a). Research indicates that one way to bring human activity within planetary boundaries is to reduce global material extraction and consumption by 33 %, although the extent varies between countries and is related to their development level (Circle Economy Foundation, 2024). In practice, this may involve transitioning to a more CE (Figure 2.*1*b), with objectives to extend useful product life and close system material and energy flows. In this way, materials are circulated and value is maintained, whilst waste generation is minimized via reuse and recycling (Eurostat, 2020). The resilience of supply chains may resultingly be increased due to reduced dependence on primary critical resources (Mansuino et al., 2024). At present, the global economy functions at approximately 7 % circularity, leaving a Circularity Gap where over 90 % of materials are lost, wasted or unavailable for reuse. (Circle Economy Foundation, 2024).

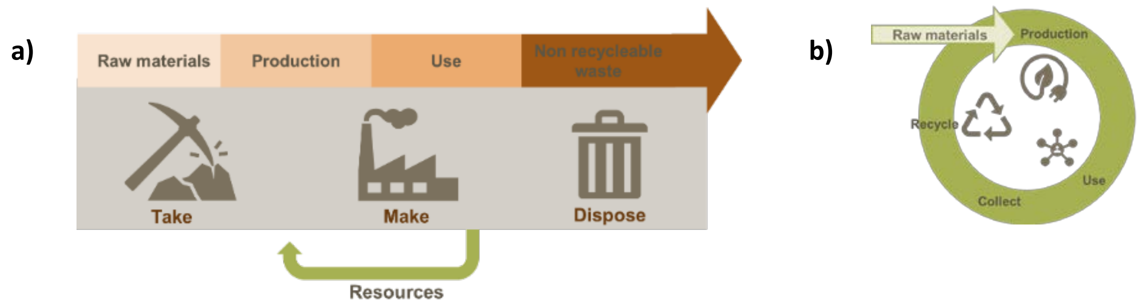


Figure 2.1: Generalised flows of materials and resources through a) Linear economy and b) circular economy.

In the scientific literature, the CE is viewed as an umbrella concept with no hard definition consensus (Jerome et al., 2022). This is due to the multiple schools of thought where CE concepts arose,

[1] Planetary boundaries are limits that, if crossed, could result in irreversible damage to the environment.

including industrial ecology and ecological economics (Wautelet, 2018), as well as the fact that recent approaches shaping concepts into those largely publicly recognized today have been developed mostly by policy and business stakeholder practitioners rather than from scientific research. This has resulted in different definitions of the CE developing from a variety of sources, including business advocacy bodies (e.g. the Ellen McArthur foundation (2013)[2]) and policy makers (e.g. European Parliament (Bourguignon, 2016)[3]), that vary depending on the context or application. As example, Kirchherr, Reike and Hekkert (2017), review 114 definitions of the CE, showing the concept to mean a multitude of different things to different actors. Many definitions stress the significance of optimizing resource use and minimizing waste, primarily by employing strategies such as reduction and recycling, with some others underscoring the necessity for systemic transformation through closed loops, renewable energy, and holistic approaches. However, all point to the importance of retaining resources within a closed-loop system for extended periods through repeated cycles, as exemplified by the ‘9R’ strategy developed by Potting et al. (2017) and adapted in a study by Prochatzki et al. (2023) for the automotive sector (Table 2.1).

Table 2.1: Circular strategies for the automotive sector, adapted from Prochatzki et al. (2023).

CE strategy	Description	Category
R0 refuse	Make a product redundant by abandoning its function or by offering the same function with a radically different product.	Smarter product design, use and manufacture
R1 redesign	Consideration of the CE in the product engineering process, accompanied by modified processes and life cycles (Design for CE concepts).	
R2 rethink	More intensive product use (e.g., by sharing products).	
R3 reduce	Increase efficiency in product manufacture or use by consuming fewer natural resources and materials.	Extend lifespan of product and its parts
R4 reuse	Reuse of a discarded product that is still in condition to fulfil its original function by another consumer.	
R5 repair	Repair and maintenance of defective product so it can be used with its original function during use phase.	
R6 refurbish	Restore an old product and modernize it to fulfil current state of art.	
R7 remanufacture	Use parts of discarded product in a new product with the same function.	
R8 repurpose	Use discarded product or parts of it in a new product with a different function.	
R9 retread	Displace the worn parts of a product to use it for the same function.	
R10 recycle (up)	Recover materials for the qualitatively equivalent (or higher) application purpose.	Useful application of materials
R11 recycle (down)	Recover materials for a qualitatively lower application purpose.	

[2] *‘An industrial system that is restorative or regenerative by intention and design’.*

[3] *‘An economic model based inter alia on sharing, leasing, reuse, repair, refurbishment and recycling, in an (almost) closed loop, which aims to retain the highest utility and value of products, components and materials at all times’.*

Due to the variation in definition and concepts, it has been argued that the CE approach fits the description of an essentially contested concept where there is consensus on concept means and goals but discord on how to define it (Korhonen et al., 2018). However, this picture is changing; various studies in the scientific literature have produced reconciled definitions in the last years, including Nobre and Tavares (2021) and Korhonen et al. (2018), the latter definition of which is below. More significantly, circularity ISO standards have now been published (ISO 59004) containing a review of CE principles along with a definition that highlights an economic system where circular resource flows are maintained (The International Organization for Standardization, 2024a).

"CE is a sustainable development initiative with the objective of reducing the societal production-consumption systems' linear material and energy throughput flows by applying materials cycles, renewable and cascade-type energy flows to the linear system. CE promotes high value material cycles alongside more traditional recycling and develops systems approaches to the cooperation of producers, consumers and other societal actors in sustainable development work." Korhonen et al. (2018)

In practice, cross-over exists within scientific and practitioner bodies, with many scientific studies using the work of the Ellen MacArthur foundation whose principles are illustrated graphically with a butterfly diagram (Figure 2.2). The spine represents the central economy from production to End-of-Life (EoL), with the main part of the figure consisting of two fundamental cycles or 'wings' represent-ting the technical cycle (right) and the biological cycle (left). The technical cycle demonstrates the cycling of non-biodegradable materials whereas the biological cycle demonstrates biodegradable cycles. A key principle is to circulate products at highest value, meaning that the loops are consecu-tive with recycling as last resort. Criticism of this figure has arisen surrounding the challenge of closed-loop production and consumption and biophysical constraints. The latter is since resource flows – particularly relating to anthropogenic use – are often tightly bound organic and inorganic materials, which are challenging to depict in separate cycles (Velenturf et al., 2019; Velenturf & Purnell, 2021). This has led to alternative depictions with integrated resource flows present (Velenturf et al., 2019).

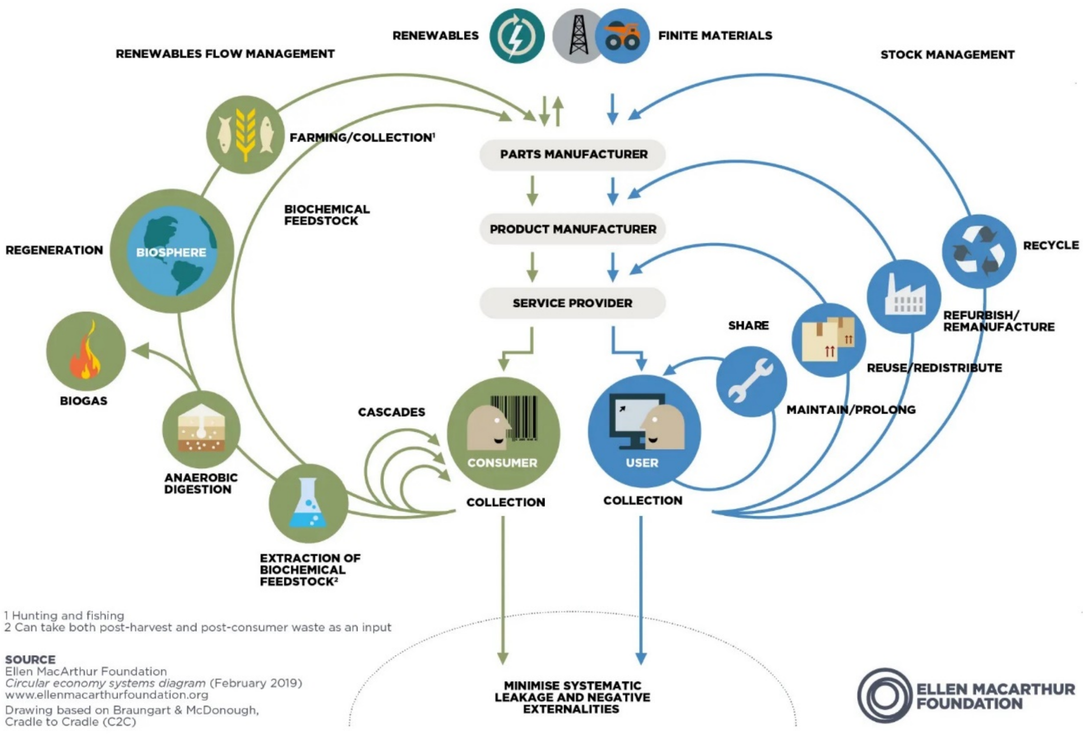


Figure 2.2: Circular economy systems diagram, as taken from Ellen MacArthur Foundation (2019).

As part of informed decision making, quantitative measurement of circularity is needed in the form of indicators to develop goals, measure progress and simplify information (de Oliveira & Oliveira, 2023). Circularity can be measured in various ways (both with or without life cycle thinking) as discussed by de Oliveira and Oliveira (2023), Moraga et al. (2019) and Jerome et al. (2022). A review by Jerome et al. (2022) found that circularity indicators can measure production losses (energy intensity), material composition (recycled content rate), technical lifetime (potential reuse index), recycling activity (recyclability rate), energy recovery (recoverability rate) or have a multi-focus. Another review found that the two most used circularity strategies that indicators cover are material recycling and reduction of production losses (de Oliveira & Oliveira, 2023). Less commonly captured principles include repurposing. It is therefore recommended to use a set of indicators to ensure wide coverage of circularity principles, but complementarity between indicators is not well understood (Jerome et al., 2022).

In addition, circularity indicators can focus on micro (typically product or company), meso (supply-chain), and macro (sector) levels (Mansuino et al., 2024), complicating the picture. Many indicators also exist, creating a wealth of perspectives that are challenging to compare; Typical indicators at a micro-level are described in Figure 2.3[4], which have different focuses, strengths, weaknesses, and ideal application areas. High indicator relevance in the figure is emphasised by underlining, whilst low relevance is emphasised by grey colour.

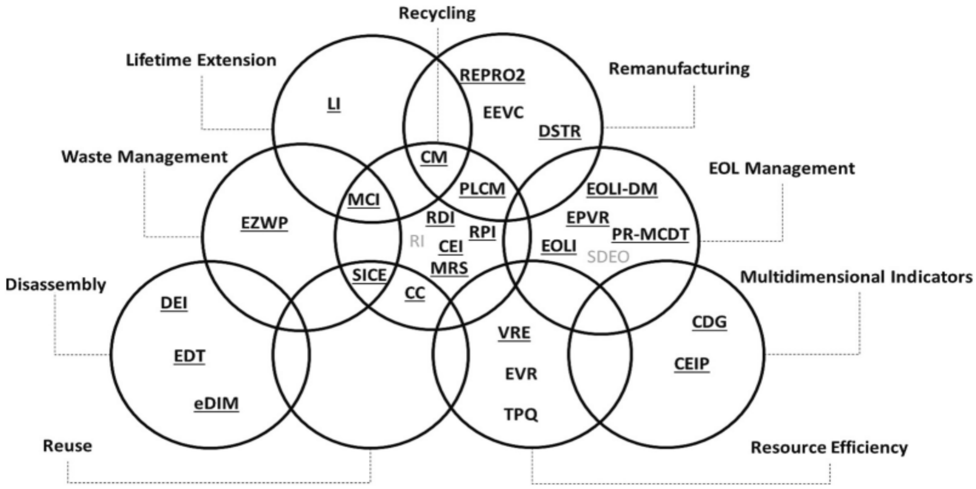


Figure 2.3: Example micro-level indicators for quantifying automotive plastic circularity, as taken from Matos et al. (2023).

[4] *Acronym explanations as listed and described in Matos et al. (2023) include: Circularity Calculator (CC), Circularity Design Guidelines (CDG), Circular Economy Index (CEI), Circular Economy Indicator Prototype (CEIP), Circular Economy Performance Indicator (CEPI), Combination Matrix (CM), Disassembly Effort Index (DEI), Decision Support Tool for Remanufacturing (DSTR), Ease of Disassembly Metric (eDIM), Effective Disassembly Time (EDT), Eco-efficient Value Creation (EEVC), End-of-life Index (EOLI), End-of-life Indices - Design Methodology (EOLI-DM), End-of-use Product Value Recovery (EPVR), Eco-cost/Value Creation (EVR), Model of Expanded Zero Waste Practice (EZWP), Longevity Indicator (LI), Material Circularity Indicator (MCI), Material Reutilization Score (MRS), Product-level Circularity Metric (PLCM), Product Recovery Multi-criteria Decision Tool (PR-MCDT), Recycling Desirability Index (RDI), Remanufacturing Product Profiles (REPRO2), Recycling Indices (RI), Reuse Potential Indicator (RPI), Sustainable Circular Index (SCI), Sustainable Design and End-of-life Options (SDEO), Sustainability Indicators in Circular Economy (SICE), Typology for Quality Properties (TQP), and Value-Based Resource Efficiency (VRE).*

As an example for one indicator, the Material Circularity Indicator (MCI) evaluates the proportion of recyclable or biodegradable materials in a product, alongside recycling efficiency and unrecoverable waste, and connects these factors to the product's lifespan and functionality (Soo et al., 2021, Matos et al. 2023). Indicators are also closely related, with for example the Product Circularity Indicator (PCI) extending the MCI model to incorporate material losses during the feedstock and component production (Soo et al., 2021). Other more novel indicators are proposed in the literature; Mansuino, Thakur and Lakshmi (2024) propose an approach measuring industrial waste, energy use, and electronic material waste to evaluate circularity at a company level in the Swedish automobile industry. The sensitivity of resulting circularity indicator values to data availability underscores the importance of understanding accuracy and uncertainty.

Although there has previously not been any real consensus or standardized guidance as to how and where to apply indicators, recent publication of the ISO 59020 standard regarding circularity measurement now provides a framework (The International Organization for Standardization, 2024b). The new guidance supplies 13 quantitative core indicators which are categorized according to five categories (see The International Organization for Standardization (2024b) for more details. These encompass: 1) resource inflows, including the average reused, recycled, and renewable content of inflows; 2) resource outflows, measured by the average product or material lifespan compared to the industry standard, the percentage of reused products and components from outflows, the percentage of recycled materials from outflows, and the percentage of biological cycle recirculation; 3) energy, focusing on the average percentage of renewable energy consumption; 4) water, covering the percentage of water withdrawal from circular sources, the percentage of discharged water meeting quality standards, and the on-site or internal water reuse and recirculation ratio; and 5) economic indicators, such as material productivity and the resource intensity index.

The standard additionally outlines the minimum (mandatory) set that should be used for circularity measurement and assessment, which surround the resource flow indicators (The International Organization for Standardization, 2024b). Core indicators can be supplemented by additional indicators if desired.

2.1.1 Circularity of passenger vehicles

Production of current mass-produced passenger vehicles involves extensive material use (including critical rare earth elements), requiring large degrees of mining and high energy needs. Research shows that increasing circularity is crucial, since currently mostly non-circular (linear) and non-renewable flows dominate over a vehicle life cycle (Mansuino, Thakur and Lakshmi, 2024, Aguilar Esteva et al., 2020), as exemplified by Figure 2.4 for a conventional ICE (gasoline) vehicle in the United States. In the figure arrow widths are roughly indicative of flow magnitudes, and all materials and energy sources use upstream fossil energy. Coupled to this, the linear consumption model encourages vehicles to have a shorter lifespan, and there are currently limited recycling options at EoL (Mansuino et al., 2024). For example, analysis by Bruno and Fiore (2023) shows that existing European recycling capacities may be only capable of processing < 20 % of the EoL LIBs generated from the EV fleet by 2030. The study assumes a 10-year lifetime for LIBs and projects the number of EoL LIBs in 2030 based on the EV sales in 2020. The findings from Bruno and Fiore are not unexpected, however, since it makes sense from an economic perspective for companies to time when they establish needed capacity.

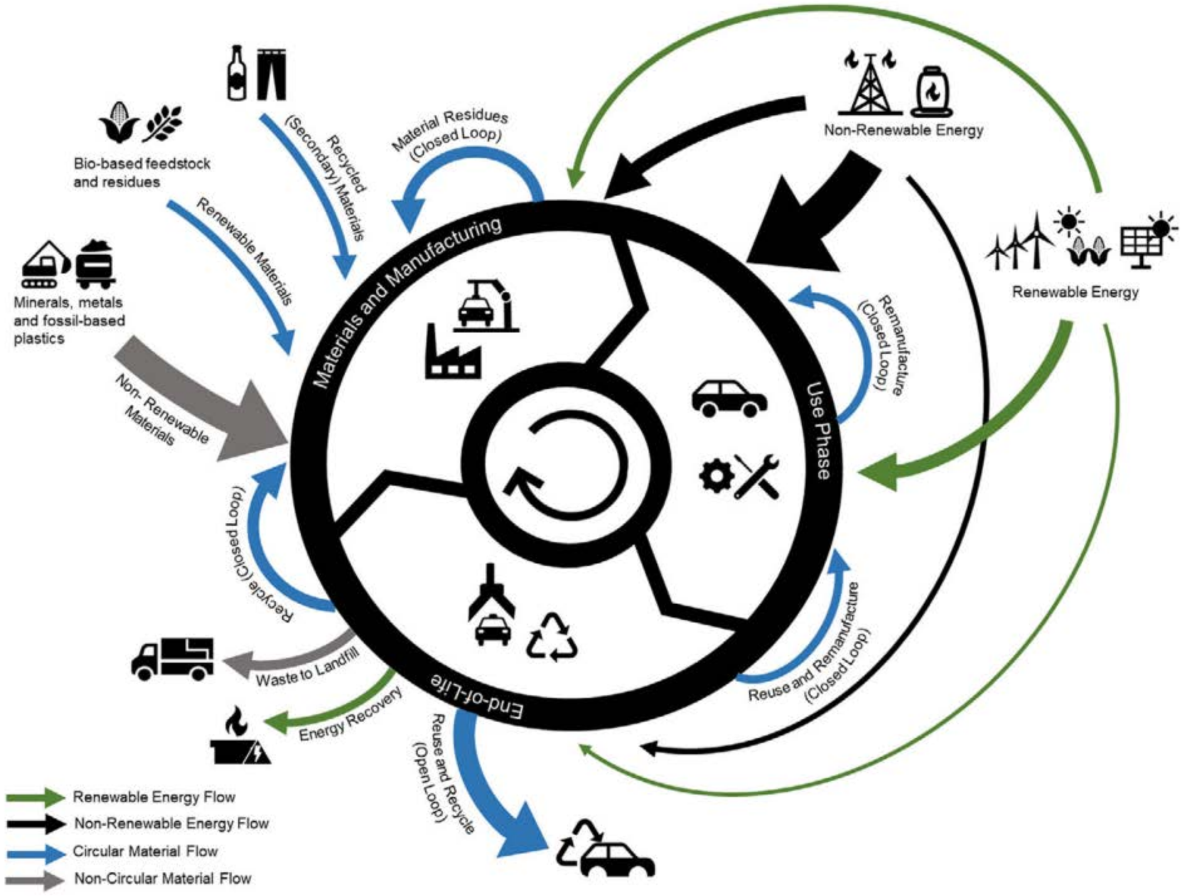


Figure 2.4: Current circular economy framework for vehicles, as taken from Aguilar Esteva et al. (2020).

The importance of increasing vehicle circularity is compounded when the sector size and importance is considered. Globally, light duty vehicles account for approximately 15 % of global energy consumption, with vehicle manufacturing responsible for ~12 % and 27 % of global steel and aluminum (Al) use (Aguilar Esteva et al., 2020). High use of materials and energy are tied together; results from Sato and Nakata (2020) indicate that of the total 41.8 MJ energy input required per kg of vehicle production, mining and material production processes are responsible for 68 %. In addition, natural gas represents the most consumed energy resource.

Due to the size of the flows, increasing emphasis is being placed on increasing vehicle circularity at a policy level (Bonsu, 2020; Soo et al., 2021), with circularity strategies increasingly explored by automotive manufacturers and the scientific literature to enhance vehicle sustainability (e.g. Martins et al., 2021; Schulz-Mönninghoff, Neidhardt and Niero, 2023). Circular systems approaches are also increasingly important for vehicle components, with LIBs particularly under current consideration.

Section 3 goes into detail about circularity strategies that can be utilized, but in brief vehicle circularity can be increased by optimizing production (e.g. using increased renewable energy shares or increased shares of recycled material), use (e.g. optimizing efficiency) or EoL stages (e.g. implementing reuse and recycling pathways). LIB circularity can be increased in similar ways (Heath et al., 2022), by using more renewables in electricity production (used as input), optimizing battery material composition (e.g. for recycling and reuse potential), reducing input of water and energy during production of LIB, or decreasing numbers of LIB required via reuse, remanufacture, or repurpose pathways (Ahmed et al., 2023).

The current state of circularity has been quantified in the literature for both an overall vehicle and its components. At a vehicle level, circularity has been estimated at between ~30-37 %, dependent on powertrain type, based on U.S conditions (Ahmed et al., 2023). At a component level, research focuses on EV battery circularity, due to their heavy reliance on rare earth elements (Silvestri et al., 2021). Using Circular Transition Indicators (CTIs) – based on circular company material inflows and outflows – Schulz-Mönninghoff, Neidhardt and Niero (2023) found that European EV battery circularity for critical battery materials is currently only around 5 % but has potential for 23 % by 2030. Other components such as a plastic automotive component have been estimated to have circularity of around 10 % when calculated with a Circularity Calculator, corresponding to an MCI value of 0.189 (Matos et al., 2023). Results are challenging to compare due to different indicators used, study scopes and focuses and case study geographic regions chosen.

2.1.2 Needs for life cycle assessment

Although circularity is a necessary condition for sustainability, as circularity indicators generally focus on resource management they do not fully encompass the three pillars of sustainability along social, environmental, and economic axes. Thus, a circular system may not directly or indirectly equal a holistically sustainable one, or even an environmentally sustainable one. This may run counter-intuitive to circularity principles but is supported by a large body of research demonstrating the non-causal link between circularity and sustainability (Jerome et al., 2022; Velenturf & Purnell, 2021).

To ensure that circular solutions are environmentally sustainable, systems approaches must be taken. This requires the application of holistic methods to assess resulting environmental impacts relating to energy and resource use using e.g. LCA. At present, LCA-based CE strategies are not widely explored in the literature, leading to inconsistencies in methodological approaches and research results (Picatoste et al., 2022). In addition, reducing resource consumption and recycling are the most commonly explored CE strategies in the LCA literature, with fewer studies investigating strategies such as repurposing (Picatoste et al., 2022).

More research is also needed to understand the relationship between CE indicators and these existing frameworks. In a comparison of indicator results and LCA results, Jerome et al. (2022) found that conclusions from LCA results and CE indicators are not well aligned; LCA accounts for and differentiates by resource flows and translates to environmental impacts, whilst CE indicators only account for resource flows. As trade-off between circularity and environmental impacts may occur Lonca et al. (2018) and Picatoste, Justel and Mendoza (2022) recommend integrated CE/LCA studies and provide a range of recommendations, and Pedneault et al. (2021) emphasize that:

“Typical circular economy indicators focusing on the circularity […] should be complemented with life cycle assessment (LCA) indicators to ensure that improvements are made at a systems perspective.” Pedneault et al. (2021)

This complementarity is also reflected in the new ISO 59020 standards for circularity measurement, which highlight the usefulness of other approaches in addition to circularity indicator quantification. This includes use of LCA to provide additional data needed to assess e.g. environmental impacts related to resource extraction occurring inside and outside the system (The International Organization for Standardization, 2024b).

2.2 Life cycle assessment principles

The LCA method is widely recognized as the *“best framework for assessing environmental potential impact of products”* (*European Platform on LCA | EPLCA*, 2003). The method is standardized in the ISO 14040/14044 series and offers a systematic approach and framework to quantify the environmental performance of products and services throughout their life cycle – from raw material extraction, manufacturing, distribution, use, and EoL treatment. For each of the life cycle stages, the inputs (materials, energy, and services) as well as outputs (emissions and waste products) are considered to quantify the environmental impact potentials. The life cycle stages as well as their inputs and outputs for a vehicle are depicted in Figure 2.5.

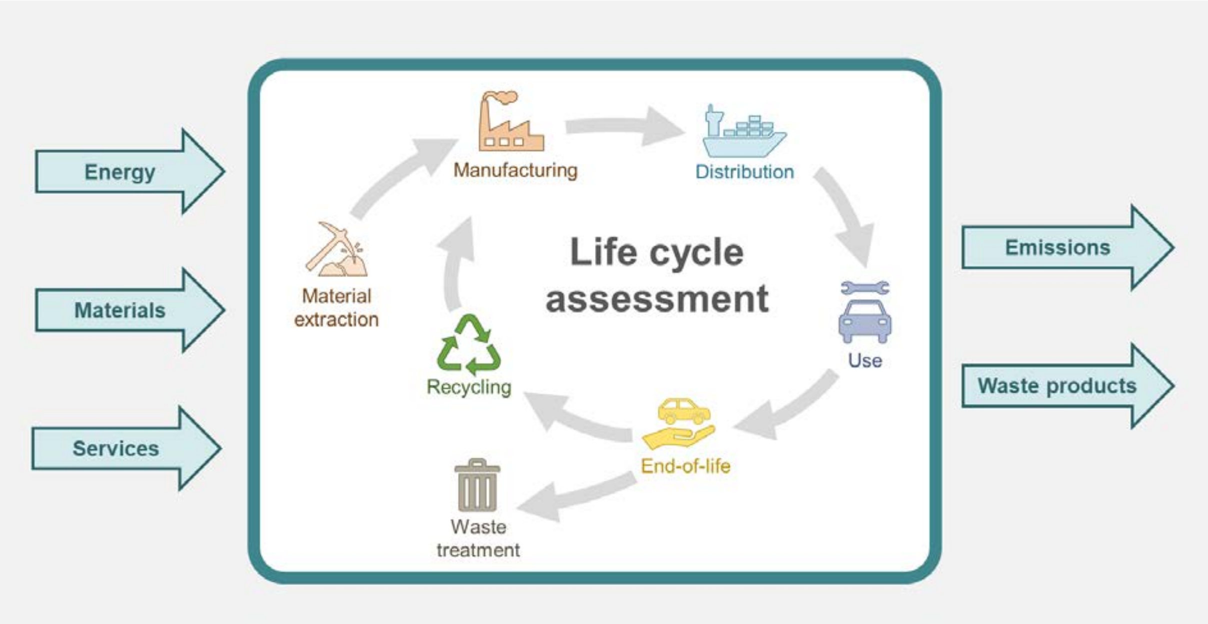


Figure 2.5: Overview of various life cycle phases from extraction of materials, processing, production and distribution via use to waste products as well as inputs and outputs that affect the life cycle.

LCA is a tool that can be used for multiple purposes and can provide comprehensive environmental insight as to where in the supply chain different emissions arise, as well as identifying emissions sources and improvement opportunities. If applied comparatively, LCA can be used to compare and rank different technology alternatives.

An LCA will often consider multiple environmental impact potentials (e.g. climate change, acidification, eutrophication, depletion of the ozone layer, etc.) to obtain a holistic understanding of the product's environmental performance. The holistic mapping can help highlight problem shifting between different types of environmental impact as well as providing insights about the importance of different life cycle stages.

2.2.1 Life cycle assessment of passenger vehicles

LCA is widely used in industry, including the automotive industry where LCA has been used for nearly three decades (Mikosch et al., 2022; Neef et al., 2023; Tarne et al., 2017). European automakers consider LCA as an important tool for managing environmental improvements over the life cycle of a vehicle (ACEA, 2021) and a review found that 13 of the top 15 passenger vehicle manufacturers (accounting for 85 % of the volume of vehicles produced globally) employ LCA (Tarne et al., 2017). While many of the passenger vehicle manufacturers use LCA as an instrument of environmentally oriented production and process development, some also publish the LCA results to document environmental performance (ACEA, 2021; Tarne et al., 2017). Outside of the automotive industry, numerous research studies have assessed and reviewed the environmental sustainability of passenger vehicles (Dolganova et al., 2020; Ellingsen et al., 2022; Hawkins, Singh, et al., 2012; Nordelöf et al., 2014; Sacchi et al., 2022; Xia & Li, 2022).

A vehicle-LCA typically considers two main aspects: the equipment life cycle and the energy life cycle. The equipment life cycle relates to the vehicle and its components, while the energy life cycle relates to the energy carriers. The energy life cycle is often referred to as Well-to-Wheel (WTW), which consists of the Well-to-Tank (WTT) and Tank-to-Wheel (TTW). Figure 2.6 depicts the complete vehicle life cycle.

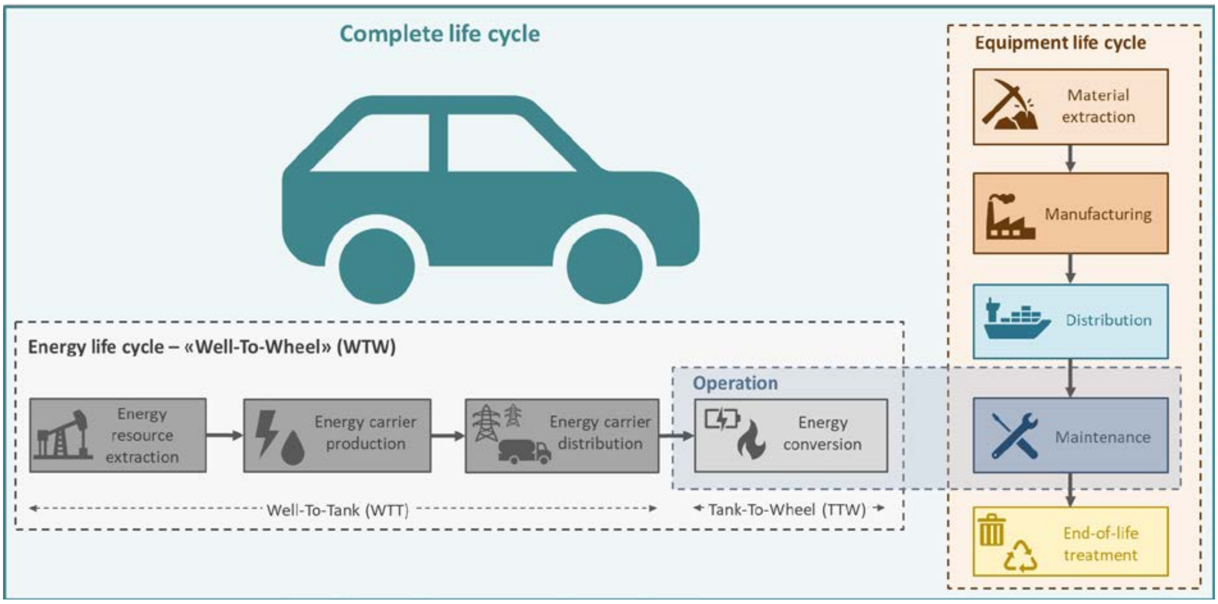


Figure 2.6: The complete life cycle of passenger vehicles consists of the equipment life cycle and the energy life cycle.

LCA studies commonly report environmental impacts in terms of three main life cycle stages: production, use, and EoL. '*Production'* typically includes both material extraction and equipment manufacturing, but in some cases distribution of the finished vehicle is also considered in this life cycle stage. Material extraction and equipment manufacturing is commonly referred to as Cradle-to-(Factory) Gate (CtG). '*Use'* refers to upstream activities associated with production and delivery of the energy carriers (i.e., WTT) and the operation of the vehicle with onboard energy conversion (i.e., TTW) as well as maintenance, which includes service needs, repairs, and replacements. '*EoL*' involves treatment of the vehicle at its EoL, and may include treatment processes like disassembly, crushing, sorting, and recycling. While EoL could also include repurposing and reuse, these aspects are less commonly considered in vehicle LCA studies.

LCA studies assessing passenger vehicles find that vehicles with different powertrain technologies have different environmental profiles. Figure 2.7 provides an example of carbon footprint profiles from a comparative LCA considering a BEV (charged with average Norwegian, European, and Global electricity mixes) and ICEVs (diesel and gasoline). The carbon footprint profile of BEVs is characterized by a relatively high contribution from production while the use phase emissions depend on the electricity mix used for charging. As can be seen in the figure, the climate footprint of BEVs varies significantly depending on the electricity mix used for charging. The carbon intensity of an electricity mix is not static and generally expected to decrease moving forward due to increasing use of renewables (and in some cases, nuclear power) following the global push to reduce greenhouse gas (GHG) emissions and the increasing cost of GHG emission permits in Europe; note that a change in carbon intensity for electricity production will impact the use phase of BEVs (and PHEVs), as well as production, maintenance, and EoL treatment of all vehicle types. The carbon footprint profile of ICEVs is characterized by their relatively high use phase emissions while production emissions play a relatively smaller role. Note that there are some small differences in the carbon footprint profile between diesel and gasoline vehicles; the diesel vehicle has slightly higher production emissions, but lower use phase emissions compared to the gasoline vehicle. This is due to the diesel-based powertrain generally being heavier but more energy efficient than the gasoline-based powertrain. For both BEVs and ICEVs, contributions from EoL treatment are relatively small but the emissions (both relative and absolute) are higher for BEVs due to recycling of the traction batteries.

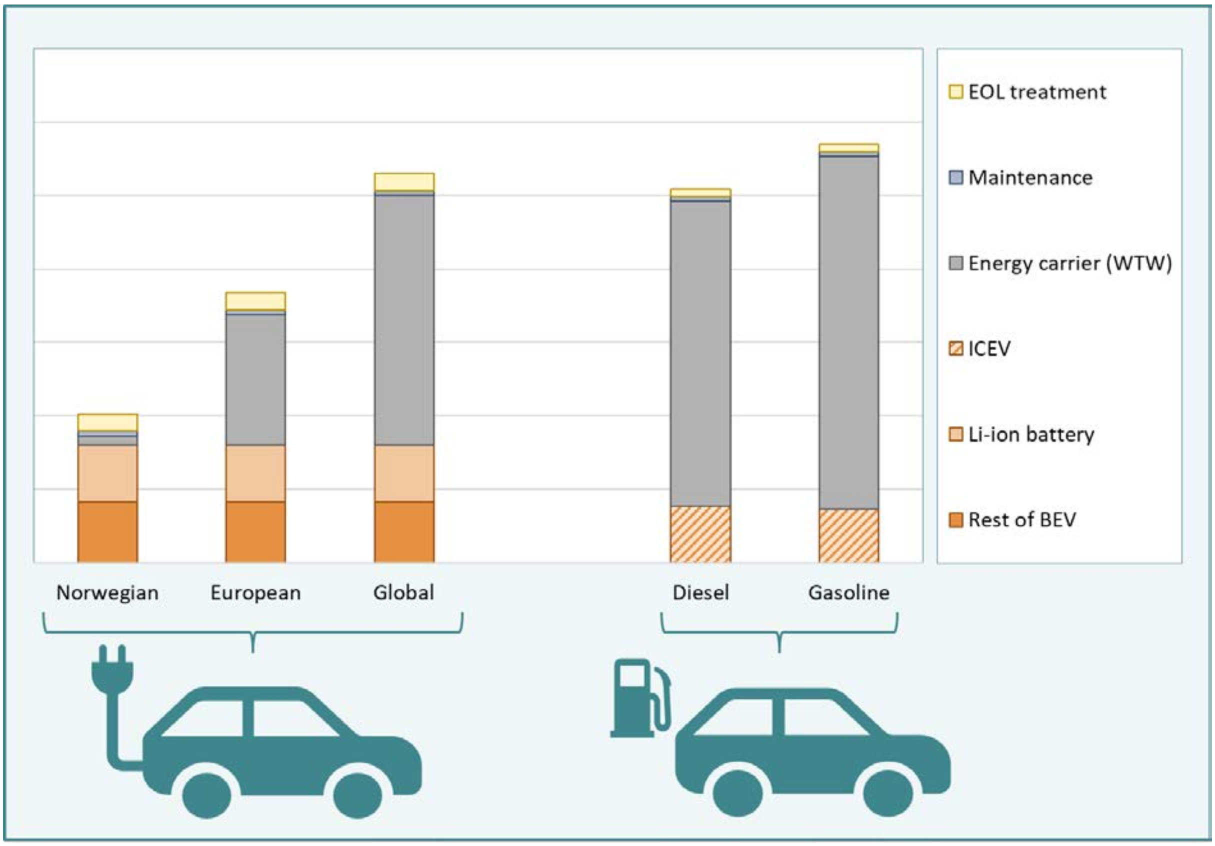


Figure 2.7: Example of relative contributions to carbon footprint profiles for BEV (left) and ICEVs (right), adapted from Ellingsen, Thorne and Figenbaum, 2022.

3 Application of circular economy principles and life cycle assessment to passenger vehicle life cycle stages

Different circularity principles can be applied to each of the life cycle stages of a vehicle (and its components) (Kifor & Grigore, 2023), which can be considered broadly as production (raw material extraction to assembly), use (energy carrier use and maintenance), and EoL (including reuse and recycling). An outline of the relevant CE principles over the three life cycle stages (production, use, and EoL) and supporting infrastructure considered in this report is depicted in Figure 3.1. Life cycle stages depicted in the figure include raw material extraction to vehicle assembly (orange), vehicle use including energy carriers (WTW) and maintenance (blue), and EoL (yellow). Circular economy principles that can be included in supporting infrastructure are shown in grey Since these principles are interconnected, these should in practice be viewed collectively with regards to outcomes. Following the recommendations of Pedneault et al. (2021), we evaluate the CE principles in parallel with LCA results to determine the significance of various strategies to the overall environmental sustainability of passenger vehicles in this section.

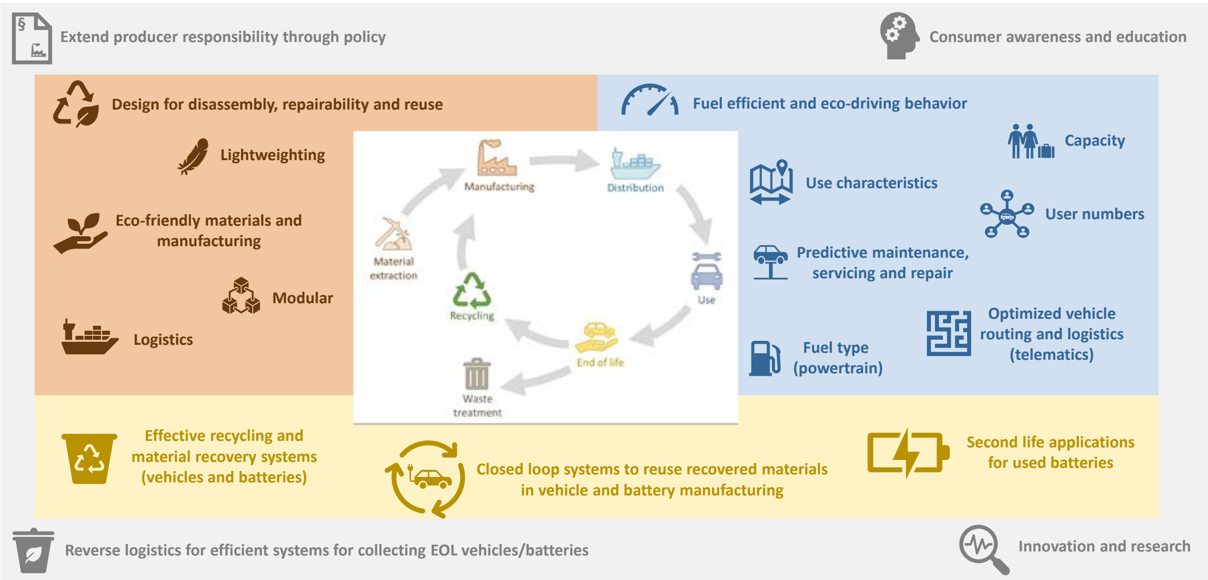


Figure 3.1: Relevant circular economy principles for different stages of a vehicle life cycle.

3.1 Production: raw material extraction to assembly

Circular economy processes initiate at the start of a product's life cycle with efficient and innovative production methods, where it is often quoted that 80 % of circularity is determined (European Commission, 2024a). This is also the stage that Original Equipment Manufacturers (OEMs) have most direct control over due to their role in product design (including logistics) and materials manufacturing (Aguilar Esteva et al., 2020). At the highest production level, vehicle powertrain selection affects resulting circularity, with e.g. analysis by Ahmed et al. (2023) estimating circularity scores of 36.8 % for BEVs and 32.9 % for ICEVs. Here, the scores reflect the overall circular economy performance of these vehicle types based on sustainability assessments across multiple dimensions, where a score of 100 % would represent an ideal circular system. Circularity differences for the BEVs were due to reduced overall energy consumption for fuel and materials production (and associated water use) and high recyclability rate and efficiency. BEV circularity can be further improved by utilising a more renewable energy mix.

To enhance circularity of all vehicle types, effective strategies include dematerialization to maintain product functionality with reduced material use, decreasing material intensity by thinning parts while ensuring vehicle durability, and optimizing material efficiency and selection. Circularity can also be increased by designing vehicles and parts with longer lifetimes, through use of durable materials that can be easily serviced, remanufactured, or reused. A variety of tools and methods are available related to eco-design and life extension for this purpose (Royo et al., 2023).

Vehicle lightweighting is a form of both dematerialization and reduction of material intensity, referring specifically to the process of reducing vehicle weight to improve performance and fuel economy. It is commonly achieved by substituting heavier metals (e.g., iron and steel) with lightweight alloys (e.g. aluminum and magnesium), which can theoretically achieve weight reductions up to 25 % (H. Kim et al., 2010). Both cast and wrought Al metal types are used for different applications[5], where cast Al typically has a higher recycled material content (85 %) than wrought Al (11 %) (Aguilar Esteva et al., 2020). Use of plastics, alternative glazing, seat, and engine materials and designs can also contribute to lightweighting. However, use phase fuel economy benefits need to be weighed up against its energy and (often) GHG intensive production, due to higher manufacturing emissions. Figure 3.2 a) shows a schematic comparison between "maximum lightweight design," "optimum lightweight design," and "conventional design.", while Figure 3.2 b) illustrates an example of lightweighting a component, demonstrating its effect on GHG emissions during both the production and use stages (Daimler AG, 2011).

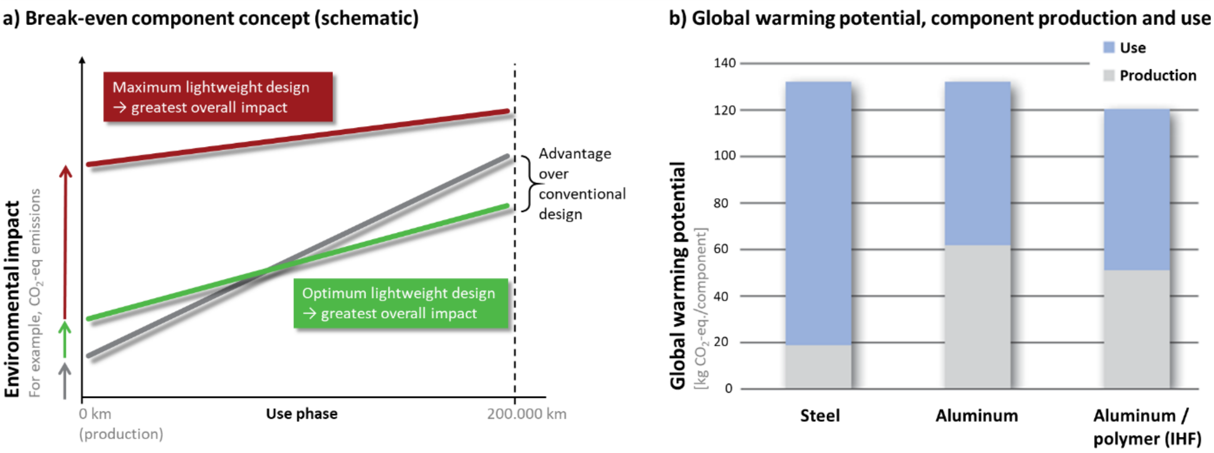


Figure 3.2: Examples of problem shifting due to lightweighting, with figures reproduced from an LCA report for the Mercedes-Benz B-class (Daimler AG, 2011).

Materials manufacturing includes multiple processes from extraction to parts manufacturing, also including material sourcing and process selection, meaning there are many opportunities to optimize material efficiency and selection. According to Aguilar Esteva et al. (2020) who based their analysis on the situation in the U.S., at the time of the analysis, ICEVs are made using around 27.5 % recycled materials, whilst BEVs are made using ~21 % (see Figure 3.3). Batteries and EoL management are

[5] Cast Al is typically used for powertrain applications whilst wrought Al is typically used for body-frame fabrication.

included in the analysis. The authors find that conventional light duty vehicle material composition has remained similar over past decades despite increasing shares of plastics and Al, and is dominated by 40-60 % steel (of which only around 25 % is secondary metal). Even though steel is (by mass) the most recycled material globally, it is often open loop recycled into alternate applications meaning that OEMs can improve vehicle circularity by incorporating increased recycled steel in their production processes (Aguilar Esteva et al., 2020). Use of critical raw materials in batteries and catalytic converters is problematic from a circularity point of view, making reducing consumption through material-use efficiency, substitution, or closed-loop recycling of importance (Aguilar Esteva et al., 2020).

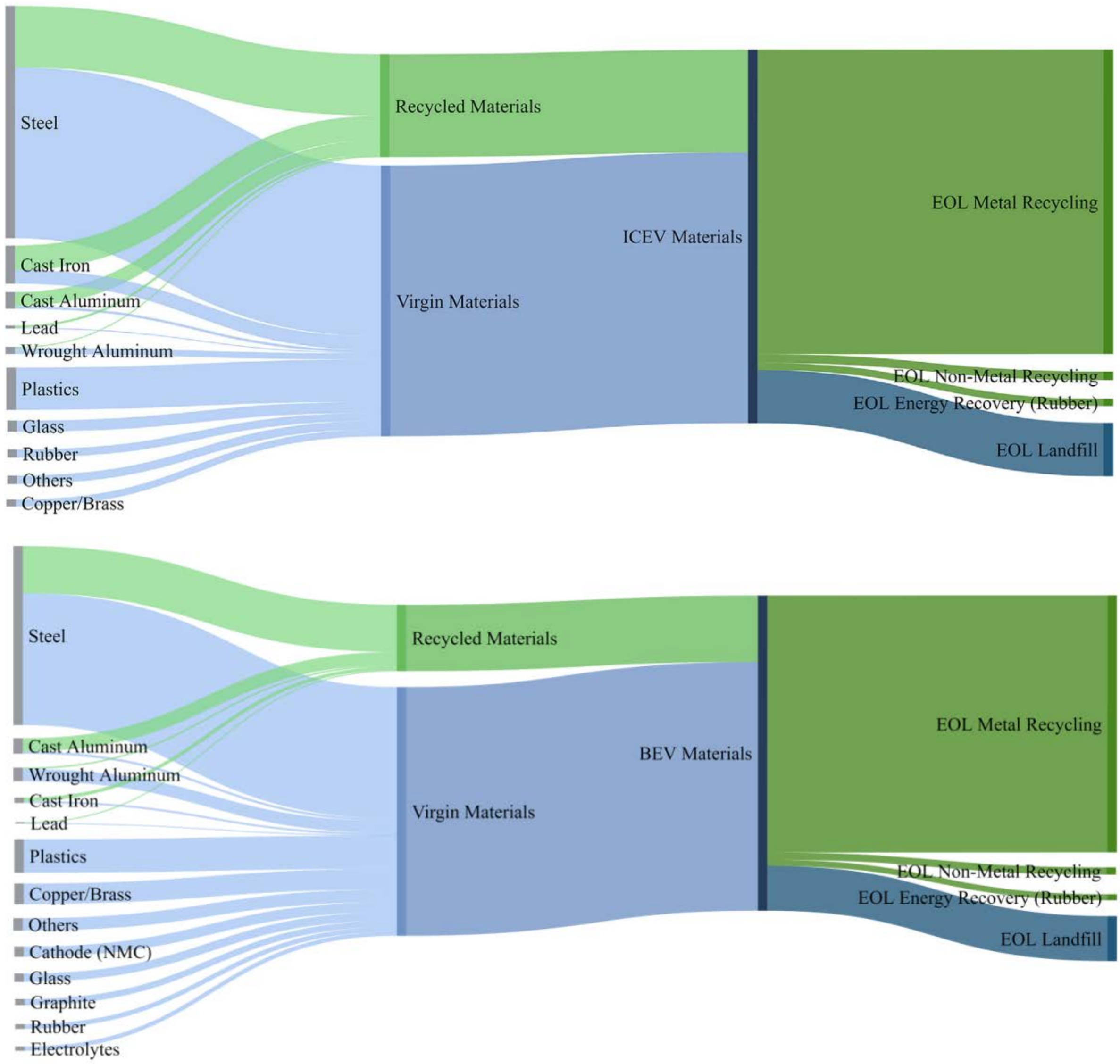


Figure 3.3: Typical U.S. ICEV (top) and BEV (bottom) composition, by share of total mass, as taken from Aguilar Esteva et al. (2020).

Design for EoL management has also been of increasing importance in recent years, via approaches focused on ease of disassembly and modularity. However, a trade-off is that longevity may delay deployment of newer and more efficient vehicles, meaning that there is an optimal point of vehicle replacement (H. C. Kim et al., 2003). This is where the environmental costs of keeping an older vehicle may outweigh the benefits of upgrading to a more efficient newer model, although in practise this could be somewhat mitigated since older vehicles are often used less than newer ones. Fuel economy is also largely determined at the design phase, and can be improved through aerodynamic vehicle design, development of more efficient powertrains, and via lightweighting (Aguilar Esteva et al., 2020).

3.1.1 Life cycle assessment results for production

LCA studies report that EVs typically have higher production GHG emissions compared to ICEVs. In terms of relative GHG emissions, studies typically report that production contributes to ~50 %, ~35 %, and ~20 % for BEVs, PHEVs, and ICEVs, respectively (ACEA, 2021; Daimler AG, 2019; Ellingsen et al., 2022; Kelly et al., 2024). Note that the emission shares for BEVs and PHEVs depend on various factors, such as the electricity mix used for BEV charging, battery size, and assumed mileage (the latter applies to all powertrains). In terms of absolute GHG emissions, the higher production-related GHG emissions of BEVs and PHEVs compared to ICEVs stem primarily from production of the high-voltage LIB (Daimler AG, 2019, 2022; Ellingsen et al., 2022; Kelly et al., 2024; Volvo, 2021). For example, Daimler reports that the LIB contributes to approximately 50 % for their BEVs Mercedes-Benz EQC and EQE SUV and 25 % for their Mercedes-Benz C 300 e PHEV, respectively (Daimler AG, 2019, 2022, 2023). The GHG production emission of HEVs is similar in size to that of ICEVs as the small HEV battery adds no significant production emissions compared to ICEVs, but it does improve fuel economy and reduce use phase emissions. The lower use phase emissions results in a higher production emission share (~35 %) for HEVs compared to ICEVs (Kelly et al., 2024; Toyota Europe, n.d.-c). When comparing same size ICEVs, diesel vehicles tend to have somewhat higher production emissions than gasoline vehicles (Ellingsen et al., 2022; Volkswagen AG, 2014). And perhaps somewhat expectedly, when it comes to vehicles, size matters; larger vehicles tend to have higher production emissions (Ellingsen et al., 2016; Kelly et al., 2024). For EVs, battery size also matters as vehicle models with larger battery packs have higher production emissions than the same vehicle model with smaller battery packs (Ellingsen et al., 2022; Toyota Europe, n.d.-a).

When considering a wider range of environmental impacts, we also find that BEVs tend to have higher production impacts compared to ICEVs (Dolganova et al., 2020; Ellingsen et al., 2022; Pero et al., 2018). For example, in terms of acidification, human toxicity, and particulate matter formation, the higher demand for metals in the electric powertrain is the source of higher production impacts for BEVs compared to ICEVs. In particular, activities related to mining and processing of metals used in the high-voltage LIB, such as aluminum (Al), copper (Cu), and nickel (Ni), cause high impacts (Bauer et al., 2015; Ellingsen et al., 2022; Pero et al., 2018). Studies also report that in terms of resource depletion, BEV production cause higher burdens due to the electric powertrain relying on rare earth elements, such as neodymium (Nd) and dysprosium (Dy) in the permanent magnets for electric motors (Dolganova et al., 2020; Pero et al., 2018). For PHEVs and HEVs, few studies assess their impacts beyond GHG emissions; one comparative study finds that the HEV has slightly higher production impact than the ICEV but not as high as the BEV (Bauer et al., 2015). None of the reviewed studies assess the wider production impacts of PHEVs specifically but when considering their powertrain components and reported importance of these components for BEVs, we expect that the size of the PHEV production impacts fall between that of the HEVs and BEVs primarily because their battery packs are larger than those used in HEVs but smaller than those used in BEVs (following the same trend as the production-related GHG emissions).

3.2 Use: energy use and maintenance

The vehicle use phase relates to the use of energy in the form of fuels and electricity and maintenance in form of service needs, repairs, and replacements. Since most life cycle primary energy use of a vehicle derives from the use phase (estimated by Aguilar Esteva et al. (2020) to be 92 %), fuel economy, fuel and powertrain type are key principles determining circularity. Switching to EVs (both

BEVs and PHEVs) has the potential to offer more energy efficient powertrains, with the potential for incorporation of renewables upstream during the manufacturing of the energy carrier (with lower GHG life cycle impacts). Although fuel/powertrain type and (to a large extent) fuel economy are largely determined at the design stage, vehicle operation patterns also influence circularity, most notably through effects on lifetime.

Average lifetime for a passenger vehicle in the Norwegian fleet, as of 2023, was 18.2 years (SSB, 2024b). Since current generation EVs were only introduced to the mass-market early in the 2010s, scrapping of BEVs are currently due to accidents and takeback campaigns while true vehicle lifetimes (as well as the lifetime of the batteries they contain) are yet uncertain. Whilst lead-acid batteries are used in ICEVs, high-voltage LIB are used in EVs for traction. Table 3.2 shows battery warranties from vehicle importers, which imply large economic liabilities for the vehicle producers. The real battery life is expected to be considerably longer than the battery warranties. Battery lifetime is highly affected by vehicle usage patterns (Pagliaro & Meneguzzo, 2019) since degradation accelerates with charging current, cycling, climate, and calendaring (age) (and combinations thereof).

Vehicle usage patterns are complex; data shows that EVs are driven more than gasoline/diesel vehicles, but this is mainly because they are, on average, newer. Figures for vehicles of the same age show that EVs are only driven slightly longer distances (2.4 % more when adjusted insured mileage when swapping from an ICEV to a BEV is used as an indicator) (Figenbaum & Nordbakke, 2019). Differences between segments are also expected.

Table 3.1: Comments provided by BEV manufacturers regarding battery lifetime, based on information from Hatrem (2021) and adapted from Lone (2022).

Manufacturer	Battery warranty	Comments
Tesla	8 y or 240,000 km (70 % capacity)	
Mercedes	EQA, EQB, EQC: 8 y or 160,000 km EQS: 10 y or 250,000 km	
Volvo	8 y or 160,000 km (78 % capacity)	Replacement is a guarantee case since battery is supposed to last the vehicle lifetime. Cells may be replaced
Volkswagen, Audi, Skoda, Seat, Cupra	8 y or 240,000 km (70 % capacity)	Assumed to last the vehicle lifetime (lifetime not given)
Nissan	Leaf 24 kWh: 5 y or 100,000 km Leaf 30-62 kWh: 8 y or 160,000 km	According to Nissan, many of the vehicles have been driven 260,000 km and still have 70-80 % battery capacity left. Few battery packs are replaced (cells are replaced if possible).
Ford	8 y or 160,000 km (70 % capacity)	Expect it to be longer than the guarantee. Modules and other components may be replaced
Kia	7 y or 160,000 km (70 % capacity)	
Hyundai	8 y or 160,000 km	

Maintenance including service needs such as refueling, cleaning, repair[6], and reconditioning[7] are key to extending lifetime and can include use of remanufactured parts. Where vehicle high-voltage LIBs are involved, a remanufacturing definition is suggested to differentiate remanufacturing from reuse, the aim of which is to restore battery capacity to >90 % original rated capacity (European Parliament, 2023b). Most BEV batteries are modular by design which means that modules can be replaced to restore battery capacity. Newer battery pack concepts such as cell-to-pack will have to take remanufacture capability into consideration to be able to preserve battery life over the life of the vehicle. Certain parts such as bumpers and headlights do not need to be remanufactured before reuse and can be taken from undamaged EoL vehicles (Aguilar Esteva et al., 2020).

The text below considers use-related environmental impacts (separated into energy use and maintenance categories), first in terms of GHG emissions and then in terms of other environmental impacts.

3.2.1 Life cycle assessment results for energy use (well-to-wheel)

In terms of GHG emissions, studies report that WTW typically contributes to ~50 %, ~65 %, and ~80 % of the complete life cycle GHG emissions for BEVs, PHEVs, and ICEVs, respectively (ACEA, 2021; Daimler AG, 2019, 2022; Ellingsen et al., 2022; Volvo, 2021). Note that the emission shares depend on assumed mileage for all powertrains as well as the charging electricity for BEVs and PHEVs; the abovementioned shares relied on the average European electricity mix[8] for battery charging. The significance of the charging electricity mixes on GHG emissions of BEVs and PHEVs has been highlighted in numerous LCAs from both research and the automotive industry (Daimler AG, 2023; Ellingsen et al., 2022; Kelly et al., 2024; Sacchi et al., 2022; Volvo, 2021). Typically, WTW emissions across the different powertrains rank BEV, PHEV, HEV, and ICEV when ranked lowest to highest (Kelly et al., 2024). An example of how the charging electricity mix influences the GHG emissions of a BEV is illustrated in Figure 3.4. The figure displays the total cumulated GHG emissions of Volvo XC40 ICE and XC40 Recharge relying on various electricity mixes (Volvo, 2021). In the figure, the XC40 Recharge is depicted with different electricity mixes in the use phase (EU28 electricity mix – light blue, global –

[6] Returning a faulty part back to a usable state.

[7] Returning a part back to working condition by rebuilding or repairing major components.

[8] The effects of the EU Emissions Trading Scheme (ETS) putting an aggregate cap on emissions from e.g., electricity generation and industry are not included in this assessment.

green, and wind – dark blue) and the XC40 ICE (dashed grey line). Break-even between the two vehicles occurs where the lines cross.

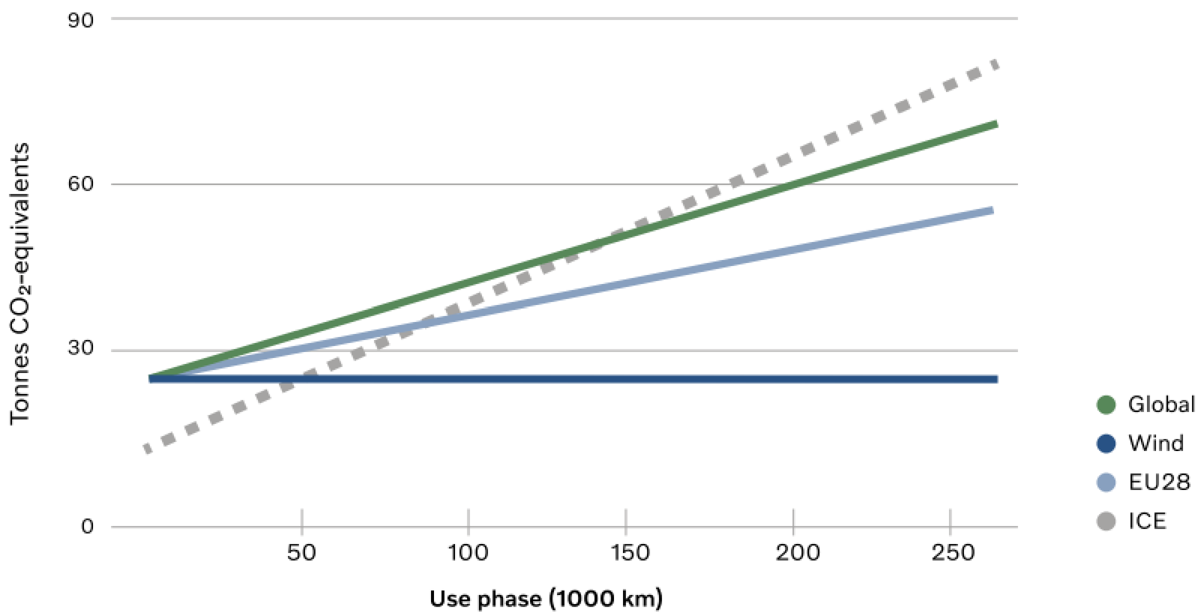


Figure 3.4: Total cumulated amount of GHG emissions for Volvo XC40 BEV and ICEV, as taken from Volvo (2021).

Since BEVs have higher GHG emissions associated with production compared to ICEVs, they must compensate with lower use phase emissions to achieve lower total life cycle emissions. In Figure 3.4, the steepness of the curve indicates the carbon intensity of the energy carriers; the ICE curve is steeper than all of the BEV curves, meaning that per kilometer the ICE has higher WTW emissions compared to the BEV – regardless of electricity mix. Another point is that the carbon intensity of the electricity mix affects how long it takes to reach the break-even point where the BEV offers a GHG emission benefit. As such, stricter demands are placed on sufficient BEV lifetimes in areas relying on more carbon intensive electricity mixes. When comparing gasoline and diesel ICEVs, studies report that diesel vehicles have lower WTW emissions than gasoline vehicles (Bauer et al., 2015; Ellingsen et al., 2022; Volkswagen AG, 2014). Similar to production, size matters when it comes to WTW GHG emissions; studies find that larger vehicles tend to have higher emissions (Ellingsen et al., 2016; Kelly et al., 2024). For BEVs, longer range offered by larger batteries comes at a cost of higher WTW emissions as heavier battery packs require more energy (Ellingsen et al., 2022). For PHEVs, however, a larger battery pack reduces the overall WTW emissions as it extends the electric driving range and thereby reduces the combustion of fuels (Toyota Europe, n.d.-b).

When considering WTW impact in a wider environmental scope, we find that there are significant differences among BEVs and ICEVs. Studies report that for most impact categories, BEVs can have both lower and higher WTW impact compared to ICEVs depending on the charging electricity (Bauer et al., 2015; Ellingsen et al., 2022). However, in terms of toxicity impact categories BEVs have been found to have higher WTW impact regardless of electricity source due to metals use for power transmission and distribution grid (Bauer et al., 2015; Ellingsen et al., 2022). While there are few studies comparing BEVs and ICEVs across multiple impact categories, there are even fewer comparing gasoline and diesel vehicles. Because the few studies comparing different ICEVs consider different impact categories and use different characterization methods, a definite conclusion regarding their ranking may not be drawn although environmental trade-offs between the two ICEVs appear to exist (Bauer et al., 2015; Ellingsen et al., 2022; Volkswagen AG, 2014).

3.2.2 Life cycle assessment results for maintenance

Relatively few LCA studies assess maintenance and those that do often provide very limited details or use generic data from databases (Bein et al., 2023; Danilecki et al., 2021; Ellingsen & Hung, 2018; Ricardo et al., 2020). The *ecoinvent* dataset for maintenance includes electricity use at the workshop, the replacement of tyres, motor oil, coolant, battery and sundry items for a lifetime set to 150 000 km, which is based on a life cycle inventory analysis of the Golf A4 with a vehicle mass of 1240 kg. The few studies that assess maintenance report that it contributes little to the overall environmental impacts of the vehicle life (Bein et al., 2023; Hawkins, Singh, et al., 2012; Ricardo et al., 2020; Schweimer & Levin, 2000; Toyota Europe, n.d.-a).

Even though maintenance GHG emissions contribute little to the overall life cycle emissions of vehicles, studies report that there are interesting differences between ICEVs and EVs (Hawkins, Gausen, et al., 2012; Ricardo et al., 2020). One report finds that GHG emissions stemming from maintenance in form of replacement parts and consumables are highest for PHEVs, followed by HEVs, ICEVs, and BEVs in decreasing size (Ricardo et al., 2020). The PHEVs were found to have the highest maintenance emissions as the authors assume that these will require one LIB replacement; for the BEVs, the authors assume that the LIBs will last the lifetime of the vehicle. For this reason, and because BEVs require fewer replacements and consumables compared to PHEVs, HEVs, and ICEVs (that require exhaust replacement and engine oil), BEVs were found to have the lowest maintenance emissions. Furthermore, the authors find that diesel vehicles have slightly higher maintenance emissions than gasoline vehicles due to the assumed AdBlue consumption in the aftertreatment systems controlling NOx emissions.

Studies have also considered wider environmental impacts; one LCA study focusing exclusively on modelling maintenance (regular and irregular) and repair of diesel and gasoline ICEVs reports environmental impact in terms of three end-point indicators: human health, ecosystem quality, and resources (Danilecki et al., 2021). In the study, the maintenance and repair were modelled individually for the diesel and gasoline vehicles and was modelled using data from 40 Ford Focus II vehicles. Across the three impact categories, the authors find that the diesel vehicle has 4 %–66 % higher impact than the gasoline vehicle. The authors report that impacts stemming from irregular maintenance caused by exchanging components due to their normal wear and tear (e.g., brake pads and discs, wiper blades, tires, clutch, etc.) is the most significant contributor to the maintenance impacts for both vehicles; tires, suspension, exhaust system, drivetrain, and also oil changes have the biggest impact. The higher overall maintenance impacts of the diesel compared to the gasoline vehicle is primarily due to impacts associated with the diesel exhaust system. In addition to the three end-point indicators, the authors also report maintenance impact in terms of the ecoindicator single score and compare the result with that from the *ecoinvent* database. They find that their single score results for maintenance of diesel and gasoline vehicles are 11-24 % higher than the results obtained using the generic *ecoinvent* inventory. While the study focuses on comparing maintenance and repair impact between only diesel and gasoline vehicles, it highlights there are important differences between powertrain technologies. Thus, when we consider the life cycle environmental impact of different powertrain technologies, it is important to consider all life cycle phases to obtain a complete mapping of potential benefits and disadvantages between powertrain alternatives.

3.3 End-of-life

The End-of-life (EoL) phase is central to ensuring circularity through remanufacture, repurposing, reuse, and recycling processes. Here, remanufacturing is a process to allow older products to perform as new or sufficiently to allow continued use[9]. Repurposing is product utilization in a role it was not designed to perform. Reuse is usage in the same purpose. Recycling entails retrieving or extracting a product's raw materials for use in creating new products.

EoL treatment varies by country. In Norway (from personal communication with Autoretur AS), EoL practises begin when a vehicle is delivered by its owner to authorized treatment (dismantling) facilities. There is no requirement that the vehicle should be delivered complete to dismantlers. The dismantler takes ownership of the vehicle and writes a wreck report, before dismantling and collecting parts that can be reused or sold. These may include the fuel, antifreeze, coolant and washer fluid, engine, gearbox, rear axle and body parts, LIB and lead batteries as well as tires and rims. Fractions are then removed that are hazardous waste or require special treatment, such as those fractions that can be recycled. Fractions for material recycling may include the tires and rims, LIB and lead batteries, oil filters, catalysts, gearbox, balancing lot, airbags and glass. Fractions for energy recovery include fuel, antifreeze, brake, coolant and washer fluid, waste oil and oil waste from oil separators, tires and a small share from lead-acid batteries and LIB. Remaining fractions for disposal include oil waste from oil separators and mercury-containing components. Largely, EoL practices have remained consistent for the last years, although the amounts of materials recovered have varied.

Scrapped vehicles that have been environmentally remediated are then transported to shredding facilities, most often after being pressed. Shredders then separate the vehicle into fractions. According to Aguilar Esteva et al. (2020), mechanical and magnetic separation processes are utilized to separate ferrous (e.g. steel and iron) and nonferrous (e.g. Al and Cu) metals, and results in a shredder residue byproduct consisting of light nonmetallic fractions usually landfilled[10], and heavy nonferrous metallic fractions. An overview of vehicle EoL treatment is given in Figure 3.5. Note that for EVs the battery and other specific components (e.g. electric motor) can be additionally recycled. Further information on treatment processes can be obtained from consulting Aguilar Esteva et al. (2020) or Pero, Delogu and Pierini (2018).

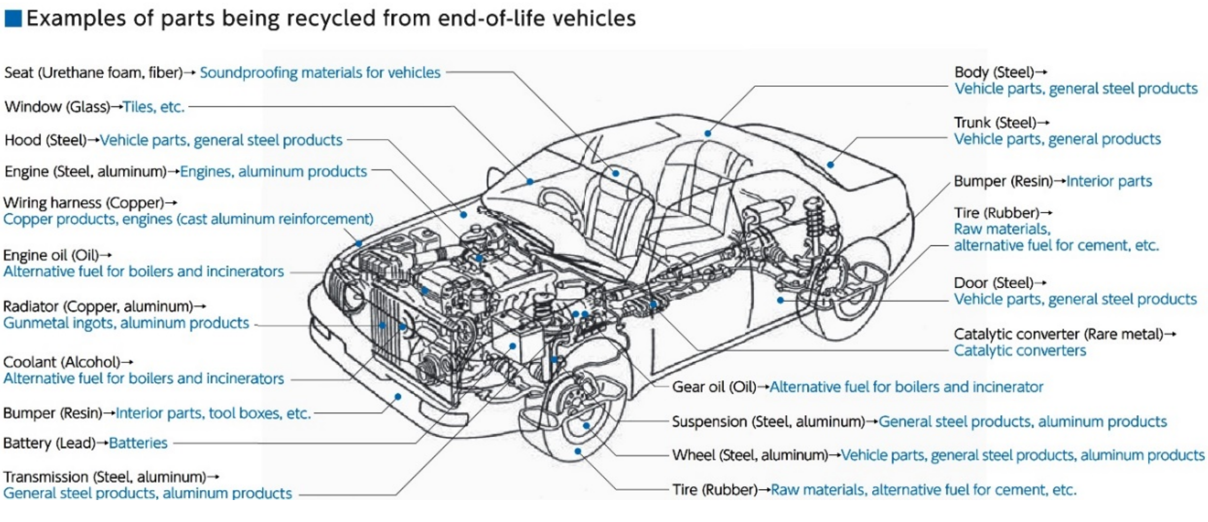


Figure 3.5: Parts of the vehicle that can be recycled and reused, using the example of an ICEV (Toyota, 2013).

[9] Note that replacing a defective battery module will not result in a vehicle being as new as the battery modules will have degraded somewhat with long-term use.

[10] Fines can also be potentially used as a composite filler.

Reuse, recovery, and recycling targets currently in place in Europe for vehicles are set in the End-of-Life Vehicles Directive (described further in Section 4). On average, 85 % of the vehicle weight should be recovered and utilized in some form through reuse or recycling processes, and 95 % should be recovered through reuse or other forms of recovery including energy recovery. Much of the vehicle can be reused and recycled, as shown in Figure 3.5, with the remains used for energy recovery. Reuse may either be direct, or parts may be remanufactured, reprocessed and upgraded before reuse. Although some remanufacturing occurs at EoL, most occurs during the vehicle use phase (H.-J. Kim et al., 2008). Regarding recycling, the part that is most reused and recycled by weight is the metal body or frame of the vehicle. The body or frame typically consists of steel or aluminum, which are highly recyclable materials. The percentage of weight that a metal frame represents in a vehicle can vary depending on the type and size of the vehicle.

In recent years, a special focus has arisen regarding reuse and recycling potential for the LIBs contained in EVs (Nurdiawati & Agrawal, 2022). Although there are no specific guidelines for what should constitute the EoL of the battery, below a threshold vehicle performance is affected due to e.g. reduced range, and the battery is not fit for purpose (Pagliaro & Meneguzzo, 2019) or violates the battery warranty. At this point the batteries may retain around 80-85 % of their initial capacity or power (Nurdiawati & Agrawal, 2022), but can also vary more widely. Thereafter, the complete battery packs are removed and passed for testing and evaluation for potential reuse or second life.

While recycling of lead-acid batteries is the accepted standard, recycling of LIBs has only become profitable recently due to increased demand for materials as reflected by increasing prices of lithium (Li), cobalt (Co), manganese (Mn) and nickel (Ni) (Pagliaro & Meneguzzo, 2019), and because the volumes of batteries available for recycling has increased. The recovery focus is on valuable metals/ materials as recycling of all materials is not necessarily economical (Keoleian & Sullivan, 2012). New (first) targets for recycling efficiency and material recovery have been set in the new Batteries Regulation (European Parliament & Council of the European Union, 2023c). The target for recycling efficiency is 65 % by 2025, extending to 70 % by 2030. Material recovery targets are 90 % for Co, Ni, and lead (Pb) and 30 % for Li in 2025, rising to 95 % for Co, Ni, and Pb and 70 % for Li in 2030.

Various battery recycling methods have been developed, which can involve direct, pyrometallurgical and hydrometallurgical processes, or a combination of methods (Figure 3.6). Regardless of method, the first step is generally pretreatment through discharge or inactivation, disassembly, and separation. Pyrometallurgical processing involves addition of reductant (e.g., charcoal) and/or slag forming agents followed by reductive smelting to convert metal oxides into metals or metal compounds, and has typically been used to extract Co commercially (Baum et al., 2022; Jiang et al., 2022; Zhou et al., 2020). An Al/Li containing slag is also produced, and metals are recovered as mixed transition metal alloys. Hydrometallurgical recycling involves pretreatment, acid or biological leaching, precipitation using pH variation or extracted using organic solvents, with recovery of transition metals and/or Li salts (Jiang et al., 2022). High material recovery rates can be obtained, particularly through hydrometallurgical treatment. In many cases, combinations of hydrometallurgical and pyrometallurgical processing methods are used for battery recycling; partly because pyrometallurgical methods allow feedstock flexibility and due to fixed investment in existing facilities (Baum et al., 2022). Direct recycling is a relatively early-stage technology process to recover useful components without using chemical methods and involves reuse or regeneration of active material for a new cathode by addition of Li source and calcination (Baum et al., 2022; Jiang et al., 2022). Kallitsis, Korre and Kelsall (2022) give more detail regarding the processing steps of a spent automotive traction battery pack from collection to generation of secondary materials.

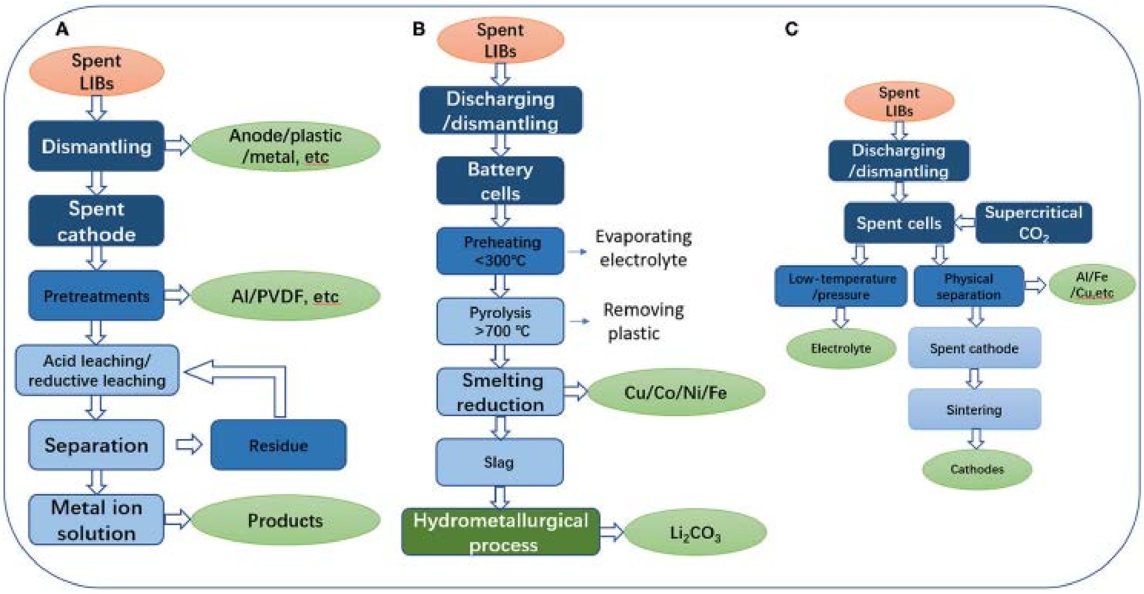


Figure 3.6: Recycling process for LIBs showing a) Hydrometallurgical process, b) Pyrometallurgical process, and c) Direct physical recycling process, as taken from Zhou et al. (2020).

Extending LIB lifetime is of increasing interest, which primarily involves remanufacturing and repurposing for use in different applications than originally designed for (see Figure 3.7). This is typically for second life in high-energy or high-power stationary applications (Pellow et al., 2020), or hybrid solutions encompassing both (Faessler, 2021), which can be categorized as grid-side (in-front-of-the-meter) applications, customer-side (behind-the-meter) applications, and off-grid applications. Typical applications are for stationary energy storage in utility-scale grid or buildings, or for peak shaving (Pagliaro & Meneguzzo, 2019). In Norway, Wrålsen and Faessler (2022) find that reuse of batteries in this way is economically viable, with (for the system they analyse) a payback period of minimum seven years before the investment costs of the battery system are recovered[11]. However, the expected lifespan, determined by charge/discharge cycles, ranged from 3 to 10 years for a second-life battery and 5 to 15 years for a new battery (Wrålsen & Faessler, 2022). Casals, Amante García and Canal (2019) modelled second life lifespan in detail accounting for several ageing mechanisms (e.g. calendar ageing and C-rate), and found it varied significantly dependent on application. This ranged between 30 years for fast charging applications, to six years for grid services. Giving the battery a second life is of additional benefit since recycling technologies are currently underdeveloped, allowing more time for LIB recycling technologies to improve before true EoL is reached (Kotak et al., 2021).

[11] The system in question with minimum payback time involved solar PV generation with self-consumption and electricity arbitrage.

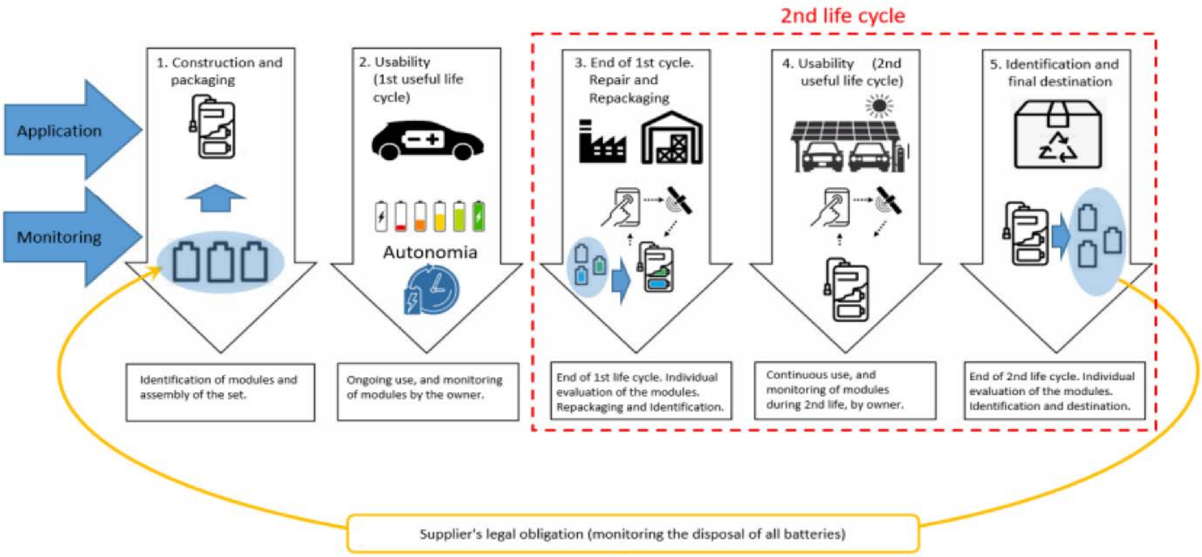


Figure 3.7: Processes involved in battery second life, as taken from Illa Font et al. (2023).

3.3.1 Life cycle assessment results for end-of-life

The vast body of LCA literature considering the life cycle environmental impacts of vehicles is performed in an attributional manner where relevant impacts are assigned to the vehicle product system. In line with the attributional approach, LCA studies considering vehicle EoL treatment tend to focus on dismantling and shredding followed by incineration of combustibles and landfilling of non-combustibles, while material recycling is typically considered as outside of the system boundary (as depicted in Figure 3.8).

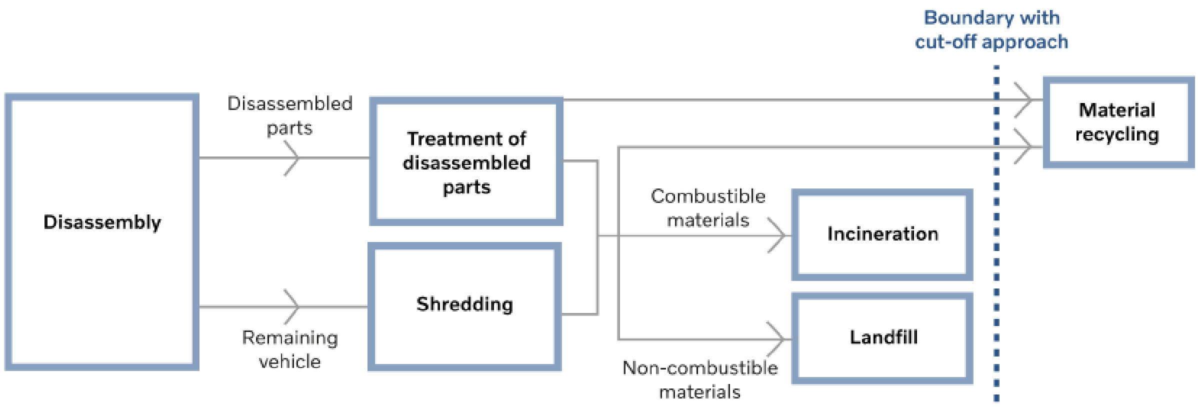


Figure 3.8: EoL system boundaries for Volvo XC40 Recharge and XC40 ICE, as taken from Volvo (2021).

Following this attributional approach, neither impacts nor credits associated with material recycling are ascribed to the vehicle as these are considered outside the system boundary. Any potential credit associated with recycling is given in production of the vehicle where assumed shares of material input stem from recycled (secondary) material (as recycled materials tend to have lower environmental impacts than virgin (primary) materials). This way of handing recycling impact and potential credits is commonly referred to as the cut-off (or recycled content) approach.

Studies following the cut-off approach report that EoL treatment contribute little to the total life cycle impacts. In terms of GHG emissions, the contribution from EoL treatment to total impacts is reported in a range around ~1-3 % for all powertrains (Daimler AG, 2019, 22022, 2023; ACEA, 2021; Ellingsen, Thorne and Figenbaum, 2022; Toyota Europe, no date a, no date c; Volvo, 2021). GHG emissions associated with EoL treatment generally increase slightly with size and higher degree of electrification (Ellingsen, Singh and Strømman, 2016). Fewer studies consider a wider range of impacts associated with EoL treatment; generally, the share of impacts associated with EoL treatment are in similar magnitude as GHG emission (Daimler AG, 2019; Hawkins et al., 2012; Ellingsen, Singh and Strømman, 2016; Ellingsen, Thorne and Figenbaum, 2022). However, in terms of freshwater ecotoxicity, impacts stemming from EoL treatment are reported to be more significant (Hawkins *et al.*, 2012; Ellingsen, Thorne and Figenbaum, 2022).

Environmental impacts relating to battery reuse and recycling have been widely investigated using LCA. In an LCA considering hydrometallurgical recycling of lithium nickel manganese cobalt oxide ($LiNiMnCoO_2$) and lithium iron phosphate ($LiFePO_4$) batteries in China, Jiang et al. (2022) found that recycling can give net environmental benefits when used to manufacture new batteries (-1 and -1.7 kg CO_2-eq for batteries made with hydrometallurgical and direct recycled material vs. virgin LIB, respectively). Higher benefits were found with direct recycling, due to reduced chemical and energy requirements, as also noted by Baum et al. (2022). Xiong, Ji and Ma (2020) also show that lithium nickel manganese cobalt oxides with 33 % nickel, 33 % manganese, and 33 % cobalt (NMC 111) remanufacturing through hydrometallurgy routes is of environmental benefit, with almost 7 % reduction of GHG compared to virgin production. Kallitsis, Korre and Kelsall (2022) performed an attributional LCA comparing pyrometallurgical and hydrometallurgical processes, finding the latter to be beneficial due to the less energy intensive processing and additional recovery of Li as hydroxide. Yu et al. (2021) also found environmental benefits of remanufacturing NMC 111 compared to virgin production, particularly with direct recycling. These results harmonize with results from the sensitivity analysis from Jiang et al. (2022) that environmental benefits associated with recycling are highly dependent on recovery efficiency and electricity use. As recycling technologies are still developing it is expected that 'first-generation' recycling technologies will evolve in coming years to greener chemistry processes (Pagliaro & Meneguzzo, 2019).

Regarding second life, LCA studies show that there are potential environmental benefits related to battery second life use. Bobba et al. (2018) performed an adapted LCA based on the comparison of different scenarios concerning use of repurposed lithium manganese oxide (LMO) or NMC batteries to increase domestic (building) photovoltaic self-consumption. The study showed benefits when a repurposed BEV battery was used in place of a virgin storage battery, but if a repurposed battery was used in a grid-connected configuration without replacing a battery, environmental drawbacks were revealed. Life cycle GHG emissions results from a consequential LCA performed by Wrålsen and O'Born (2023) demonstrated that remanufacturing used batteries for a second life generates only 16 % of the greenhouse gas emissions associated with producing a new battery. Thus, there is potential for net environmental benefits of circular business models with second life batteries, although market price and remaining battery capacity are important factors determining overall benefits, as well as the avoided production emissions related to the substitution coefficient[12]. Uncertainties in these types of analyses relates to primary data availability associated with battery repurposing (Wrålsen & O'Born, 2023), with ongoing discussion as to the extent to which the battery's first life should be included in the system boundaries (Bobba et al., 2018).

3.4 Relative sustainability of circularity principles

Although not well mapped out in literature, relative environmental sustainability of circularity principles differs due to variation in vehicle characteristics, which can relate to powertrain efficiency,

[12] In consequential LCA, the substitution coefficient represents the quantity of production avoided by utilizing a second-life product or material.

type and – where applicable – battery chemistry. Studies have used both circularity indicators and LCA to map this.

Jerome et al. (2022) and Jerome, Ljunggren and Janssen (2023) performed cradle-to-grave attributional LCAs to investigate whether extending lifetime via repair is environmentally beneficial for long-lived and energy intensive 'energy-using-products' such as motors before recycling. Results for two types of motor showed that if energy efficiency is reduced significantly then repair may not be beneficial in terms of environmental impacts, regardless of how long the motors are used subsequently. Figure 3.9 shows the resulting extra lifetime that is required with repairs that cause energy efficiency reductions, in order to obtain lower impact (global warming potential, GWP, and resource depletion) per year of use than a new motor. Both a) induction motor and b) synchronous motor are shown. For the motors (with an initial 20 years usage and use of Swedish electricity mix), the maximum allowable efficiency reduction after repair should be 0.04–0.05 % to ensure that the repair results in lower GWP compared to using a new motor, and 0.3-0.4% for resource depletion. Jerome, Ljunggren and Janssen (2023) also note that potential benefits from reuse are low when low carbon electricity mixes are used, which can be relevant for countries like Norway that use a high percentage of renewables in the electricity mix. This means that powertrain efficiency (also related to vehicle lifetime) is a key parameter in determining optimum circularity strategies, due to its significance in determining overall system energy needs.

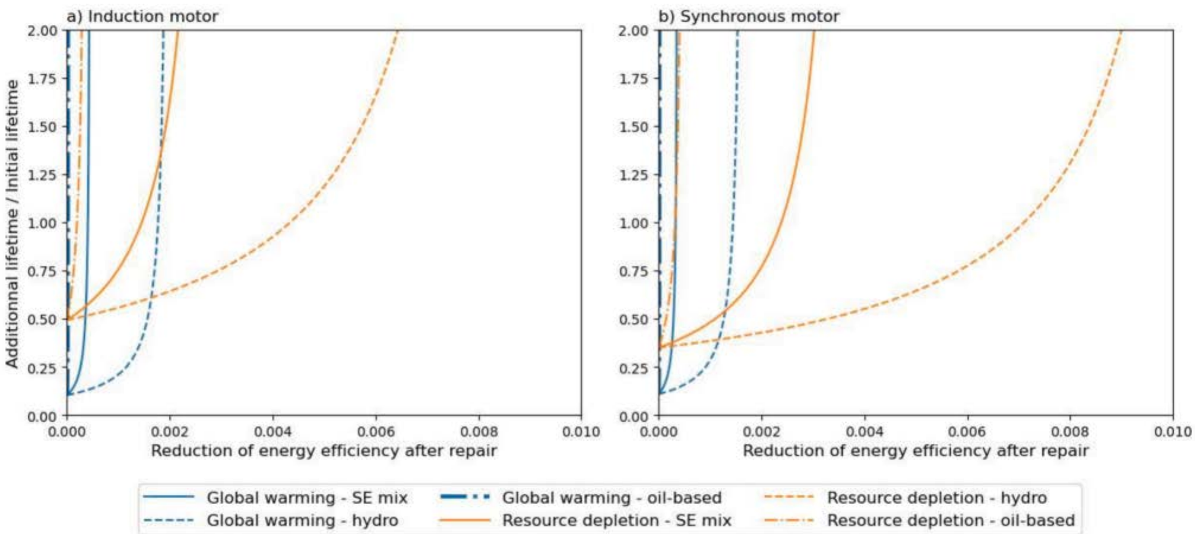
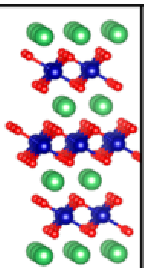
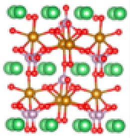
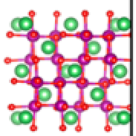
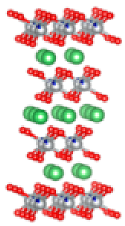
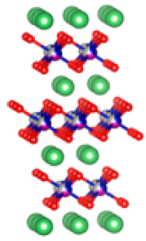


Figure 3.9: Minimum additional lifetime for a repair to result in lower impact per year of use than the motor without repair with changing repair efficiency reduction, as taken from Jerome, Ljunggren and Janssen (2023).

Powertrain *type* is also of key importance. Life cycle impacts of replacing a gasoline vehicle with a BEV were explored in a consequential LCA by Schaubroeck et al. (2020), showing the optimum replacement time to vary with different timeframe scenarios. Vehicle lifetime was set at 10 years with usage of 15,000 km/y, with the number of vehicles required in the product system accounted for. Additional secondhand usage of the vehicles was considered in most scenarios assuming a demand-constrained market, with one scenario assuming all products are disposed of as waste after initial usage. Results suggest that with second-hand usage, greatest reductions of life cycle GHG emissions are achieved when a gasoline vehicle is replaced early with a BEV.

Applicability of recycling and reuse pathways also vary with battery state of health, battery chemistry, and scale. Popien, Thies and Spengler (2022) find that it is beneficial to build large recycling facilities instead of recycling at smaller scales locally, especially with respect to social and economic impacts. Differences in suitability of recycling process type for different battery types were explored in a review by Baum et al. (2022), based on economic profitability. As shown in Table 3.2, the authors found that while hydrometallurgical (H), pyrometallurgical (P), and direct (D) recycling processes are suitable for lithium cobalt oxide (LCO), lithium nickel cobalt aluminum oxide (NCA) and NMC (denoted as NCM in the table) types, it is currently only profitable for direct recycling with lithium iron phosphate (LFP) and LMO. Baum et al. (2022) can be consulted for further details.

Table 3.2: Relevance of different battery recycling pathways to various battery chemistries, as taken from Baum et al. (2022).

Structure	 **LCO**	 **LFP**	 **LMO**	 **NCA**	 **NCM**
Composition	$LiCoO_2$	$LiFePO_4$	$LiMn_2O_4$	$LiAl_xCo_yNi_{1-x-y}O_2$	$LiCo_xMn_yNi_{1-x-y}O_2$
Energy Density/ Wh kg^{-1}	624	544	410	740	592-740
Material price/ \$ $(kWh)^{-1}$	88	32	26	39	40-50
Battery price/ \$ $(kWh)^{-1}$	357	222	251	199	145-230
Publications on Direct Recovery	110	98	43	9	56
Economical Recycling Pathways	**H, P, D**	**D**	**D**	**H, P, D**	**H, P, D**

Regarding repurposing, Wu et al. (2020) estimate that it is profitable where the difference between the 'value for energy storage applications' minus the 'willing to sell price' is positive. Effectively the authors find this is when remaining battery capacity in retirement falls below 85 %, and assume abandonment at 50 % (Figure 3.10). The grey and blue shaded regions in the figure depict the profitability range, depending on whether the pricing is based on the market evaluation or the willing-to-sell price. Point A marks the point where these two pricing strategies intersect, indicating a critical juncture for assessing the profitability of battery reuse. The segment B-C represents the profit potential when the market evaluation price is applied, with the highest profit observed at Point C, where the battery's remaining capacity is most favourable for energy storage profitability. Conversely, the segment D-E outlines the profit potential when the pricing is determined by the willing-to-sell price, with maximum profit at Point E, reflecting the optimal balance of economic and practical considerations for battery repurpose.

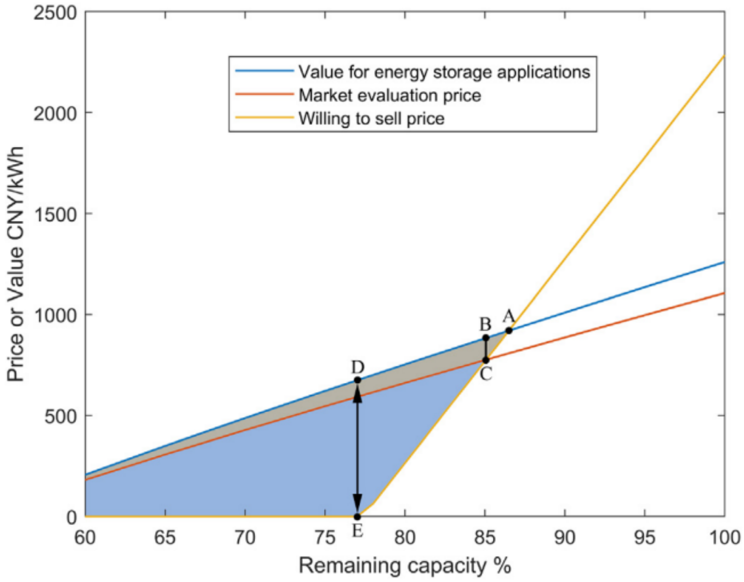


Figure 3.10: Potential profit of second life batteries for energy storage, as taken from Wu et al. (2020).

Application of Circular Transition Indicator (CTI) indicators by Schulz-Mönninghoff, Neidhardt and Niero (2023) show that automotive manufacturers can boost material circularity for battery materials from 5 % (current status) to 23 % by 2030, by integrating a closed-loop production system with various EoL strategies (remanufacturing, repurposing, and recycling) (Schulz-Mönninghoff et al., 2023). Nevertheless, from a circularity perspective, focusing on EoL strategies alone is not effective. Soo et al. (2021) used the Material Circularity Indicator (MCI) and Product Circularity Indicator (PCI) indicators to measure the effectiveness of EV legislation; The authors found that the greatest potential for circularity improvement is through higher scrap utilization, along with greater vehicle usage intensity and improved designs for reuse, remanufacturing, and repurposing, and that focusing solely on material recycling rates in the EoL phase affects material circularity by up to 17.3 %. Regarding the battery, Picatoste, Justel and Mendoza (2022) find that repurposing NMC batteries and improving their recycling process can cut environmental impacts by up to 50 %. Furthermore, the authors find that implementing an optimized recycling approach can greatly enhance material circularity, with potential for recovery of over 80 % of metals.

4 Supporting policy infrastructure for passenger vehicles and batteries

4.1 Selected policy at European level

Transitioning to a CE involves transforming the entire economic system to achieve resource efficiency and waste reduction, with potential indirect benefits relating to job creation and resource scarcity/ volatile price protection. This has led to CE concepts gaining prominence in EU policy making, with the CE being described in the 7th Environment Action Programme 2014-2020 and forming part of the key objectives of the 8th Environment Action Programme to 2030 (European Parliament & Council of the European Union, 2022)[13]. In addition, EU focus has shifted in recent years beyond a focus on waste to a more systems orientated approach including reuse (Watkins & Meysner, 2022).

Amongst the various initiatives, the launch of the first EU Circular Economy Action Plan in 2015 'Closing the loop – An EU action plan for the Circular Economy', and its adoption in March 2020 as the new 'Circular Economy Action Plan for a Cleaner and More Competitive Europe' (European Commission, 2020) was significant (Rask, 2022). These are described in more detail by Völker, Kovacic and Strand (2020) and Watkins and Meysner (2022). The CE plan has synergies with EU objectives on climate and energy and the EC package on 'Clean Energy for all Europeans' and is regarded as a key foundational element of the European Green Deal. (European Council & Council of the European Union, 2024)[14,15]. The Action Plan contains a set of interrelated initiatives to establish a product policy framework and new business models for sustainable growth, with the idea that a product policy framework will be progressively rolled out. Concepts of circularity are described along the entire product life cycle, from the design phase to waste collection and treatment (with focus on reuse).

Under the Circular Economy Action plan, the vehicle sector and (associated) batteries industry are priority sectors for implementation of CE principles (European Commission, 2020). This is outlined in a roadmap that emphasizes sustainable product design, extended producer responsibility, resource efficiency, and the utilization of recycled materials. These objectives tie in well within the general policy landscape for vehicles and batteries within the EU, where significant progress has already been made in the last decades towards fostering sustainability and resource efficiency. This progress, as well as initiatives developed under the Circular Economy Action Plan, is detailed in the sections below.

4.1.1 Selected policies for passenger vehicles

Central to current regulation that addresses the environmental impact of vehicles across their life cycle in line with circularity principles, the End-of-Life Vehicles Directive (ELV Directive 2000/53/EC), has been fundamental in establishing a framework for the proper treatment and disposal of EoL vehicles and promoting recycling and material recovery (European Parliament & Council of the European Union, 2000). As such, it sets requirements for the collection, reuse, recycling, and

[13] These multi-year plans provide a framework for addressing environmental challenges and guiding EU action.

[14] Launched in 2019, the European Green Deal provides a roadmap for climate neutrality by 2050 and a 55 % reduction of GHG by 2030 compared to 1990 levels.

[15] Additional initiatives under the Green Deal include the 'Fit for 55' package, a collection of legislative proposals aimed at updating climate, energy, and transport legislation to align EU laws with climate objectives.

recovery of vehicles and their components, as well as for Member States to ensure that systems are in place for the collection of all EoL vehicles. Reuse and recycling targets denote the percentage by weight of the vehicle that should be recovered and utilized in some form through reuse, recycling or recovery[16] processes. The Commission has now performed a joint review of the Directive 2000/53/EC on EoL vehicles and its corresponding Directive 2005/64/EC (European Parliament & Council of the European Union, 2005) on the type-approval of motor vehicles regarding their reusability, recyclability and recoverability, and has proposed new regulation to repeal and replace them (Ragonnaud, 2023). The new proposed directive covers the entire life cycle from design to EoL and aims to achieve more circular design processes along with requirements for recycled material content use in new vehicles and strengthened extended producer responsibility. A full comparative overview, with relevance to circularity, is given in Table 4.1.

Table 4.1: Key features of the End-of-Life Vehicles Directive and Type-Approval Directive as well as the proposed new directive.

Sections from the Proposal	Existing Directives (ELV 2000/53/EC and Type-Approval 2005/64/EC)	End-of-life vehicle regulation, proposed 2023
Reusability, recyclability, and recoverability of vehicles	Vehicles must be designed to allow for at least 85 % of their mass to be reused or recycled and at least 95 % of their mass to be reused or recovered	Must meet minimum of 85 % recycling and 95 % recovery targets
Requirements for substances in vehicles	Restriction on hazardous substances like Pb, Hg, Cd, and hexavalent Cr	Continuation of hazardous substances restrictions, with potential additional restrictions under other regulations (e.g., REACH and batteries and waste batteries)
Minimum recycled content in vehicles	Not specified	Minimum 25 % recycled plastics in new vehicles from post-consumer plastic waste. At least 25 % to derive from plastics recycled from EoL vehicles in the vehicle type concerned. Future targets for recycled steel, Al, Mg, and other (mostly critical) raw materials (Nd, Dy, Pr, Tb, Sm or B) under consideration
Design for circularity	Not explicitly detailed. Vehicles must be designed considering future dismantling and recovery	Vehicles should be designed to facilitate the easy removal and replacement of specific parts, such as batteries and motors, by authorized treatment facilities or maintenance/repair operators
Obligations of manufacturers and importers	Manufacturers and importers must provide information on component materials for proper waste treatment and demonstrate that new vehicles meet the required reusability, recyclability, and recoverability targets	Manufacturers and importers required to demonstrate compliance with circularity requirements and submit a circularity strategy for each vehicle type
General management of EoL of vehicles	Extended producer responsibility for EoL vehicles, although implementation varies	More standardised and strengthened extended producer responsibility for vehicle manufacturers and importers, including financial contributions for the management of EoL vehicles

[16] Recovery encompasses various processes such as energy recovery, which involves using the non-recyclable parts of the vehicle for energy generation.

Sections from the Proposal	Existing Directives (ELV 2000/53/EC and Type-Approval 2005/64/EC)	End-of-life vehicle regulation, proposed 2023
Collection of EoL vehicles	Free take-back systems required from manufacturers and importers for EoL vehicles	Manufacturers and importers are required to establish and participate in collection systems to ensure EoL vehicles are delivered to authorized treatment facilities
Treatment of EoL vehicles	Vehicles must be depolluted; targets for reuse, recycling, and recovery. Treatment operations to promote recycling are also given	A targeted annual recycling rate of 30 % is set for plastics recovered from end-of-life vehicles. All extracted parts and components must undergo evaluation to determine their suitability for reuse, remanufacturing, refurbishment, recycling, or other processing methods. Additionally, there is a prohibition on landfilling non-inert waste untreated with post-shredder technology
Export of used vehicles	Not specified	Used vehicles must be roadworthy and not classified as EoL vehicle for export; criteria and roadworthiness certification required for export

Extensive legislation is also in place that indirectly promotes low and zero-emission tailpipe vehicle technologies, aligning with circularity principles (European Commission, 2024b). Setting emission limits is one such example, with Regulation (EU) 2019/631 entering into force in 2020 setting new CO_2 emission performance standards for new passenger cars and vans[17]. Amended by Regulation (EU) 2023/851, the Regulations together set fleet-wide CO_2 emission targets (Table 4.2) and include a mechanism to incentivize the uptake of low and zero tailpipe emission vehicles. Other standards have been set for other emissions, with new Euro 7 regulations now agreed upon for carbon monoxide (CO), total hydrocarbons, non-methane hydrocarbons, nitrogen oxides (NO_x) and particulate matter (PM), including evaporative emissions and those from tires and brakes (Dornoff & Rodríguez, 2024; European Parliament & Council of the European Union, 2024). Vehicles are expected to comply with these limits for defined vehicle lifetimes, which are strengthened in the new standards; for passenger vehicles a main vehicle lifetime of 160,000 km or 8 years[18] is defined, with an extended lifetime to 200,000 km or 10 years. During the extended lifetime, emission limits are modified by applying a durability multiplier.

Table 4.2: CO_2 emissions targets according to Regulation (EU) 2019/631 and its amendment.

Years	CO_2 target		
	Cars	Vans	Unit
2020-2024	95	147	g/km
2025-2029	15	15	% reduction vs 2021 baseline
2030-2034	55	50	% reduction vs 2021 baseline
>2035	0	0	g/km

[17] Financial penalties (for each vehicle newly registered) apply to manufacturers whose emissions exceed a target in a given year.

[18] Whichever is reached first.

This is complemented by other EU legislation promoting a switch in powertrain technologies or increase in energy efficiency. The Energy Efficiency Directive (European Parliament & Council of the European Union, 2023a) aims to reduce energy consumption by an additional 9 % compared to baseline projections from 2020, which aligns with the Climate Target Plan's energy efficiency targets of a 39 % improvement in primary energy consumption and a 36 % improvement in final energy consumption, respectively. Since this target was made, the REPowerEU Plan presented in 2022 (European Commission, 2024c), proposes to raise the energy efficiency target from 9 % to 13 %. In parallel, the Renewable Energy Directive and revisions (European Parliament & Council of the European Union, 2023b) set a binding target for a 32 % share of renewable sources in energy consumption by 2030. A provisional agreement was reached March 2023 to increase this target to 42.5 % by 2030, whilst aiming for 45 %. Direct forms of support for use of zero-emission vehicles include agreements under the Fit for 55 deal for the deployment of charging and fueling stations for alternative fuels. This includes installing electric vehicle charging points with a minimum 400 kW output every 60 km along the core TEN-T network by 2026, increasing to 600 kW by 2028, and establishing hydrogen refueling stations at least every 200 km along the core TEN-T network by 2031 (European Parliament, 2023a).

4.1.2 Selected policies for batteries

The first initiative to be delivered under the Circular Economy Action Plan was the new Batteries Regulation 2023/1542, adopted after review in 2023 (European Parliament, 2023b, European Parliament and Council of the European Union, 2023c). This builds on the EU Strategic Action Plan for Batteries (European Parliament & Council of the European Union, 2019), amends Regulation (EU) 2019/1020 and Directive 2008/98/EC, and repeals Directive 2006/66/EU on batteries and accumulators (the Batteries Directive, last amended in 2018). With relevance to transport, it covers light means of transport batteries, starting lighting and ignition batteries, and specifically traction batteries for BEVs and PHEVs.

Key features of the new Batteries Regulation, with key relevance to circularity, are described in Table 4.3. Note that in the table, [a] denotes that this applies to industrial batteries two years later, [b] denotes that portable batteries of general use are excepted, and [c] denotes that the Commission may bring further materials into scope. Some summary information in the table was sourced from Flash Battery (2024). The new requirements cover and improve traceability along the entire battery life cycle, including stages from raw material extraction through manufacturing, product design, use, collection, and finally processes for recycling and reuse. Furthermore, EV batteries require an accompanying LCA-based carbon footprint declaration that must be expressed in terms of kg CO_2-eq/kWh as well as differentiation between life cycle stages. Second-life batteries are exempt from recycled content and carbon footprint requirements, as operators have no control over the criteria set for the original battery. In addition, an annex has been included outlining minimum criteria to distinguish used batteries from waste batteries for export purposes, aiming to prevent waste batteries from being misclassified as second-hand products for export.

These features go further than the previous Batteries Directive (European Parliament & Council of the European Union, 2006), where EV LIBs were included in the category ‘industrial batteries’. Nonetheless, this laid the groundwork for Member state responsibility for battery collection schemes; under these schemes battery and battery product producers are responsible for waste management and financing of collection of recycling schemes under the extended producer responsibility principle (Choi & Rhee, 2020), as well as setting recycling efficiencies.

Table 4.3: Key features of the new Batteries Regulation 2023/1542.

<table>
<tr><th>Feature</th><th>Year</th><th>Specifics</th></tr>
<tr><td rowspan="3">Carbon footprint requirements for BEVs, light transport and rechargeable industrial batteries[a]</td><td>2025</td><td>A carbon footprint declaration is required to stay with the product until it is accessible via a QR code, detailing both direct and indirect GHG emissions calculated by LCA across the full battery value chain. This must include:
- Manufacturer information (including location of manufacturing facility), battery model information (including ID number) and web link to public document
- The carbon footprint of the battery, measured in kg CO_2-eq per kWh of total energy delivered over its expected lifespan
- A breakdown of the battery's carbon footprint by life cycle stages, covering everything from raw material extraction to production and recycling</td></tr>
<tr><td>2026</td><td>Classification into a carbon footprint performance category with corresponding labeling</td></tr>
<tr><td>2027</td><td>Requirement to comply with maximum life cycle carbon footprint threshold</td></tr>
<tr><td>Due diligence policy to economic operators except SMEs</td><td>2025</td><td>Actors offering batteries on the market must establish and follow a “due diligence policy”. This must include:
- Definition of measures to manage social and environmental risks associated with the sourcing, trade, and handling of primary and secondary materials used in battery production and throughout the battery value chain.
- Implementation of control processes for suppliers to e.g. ensure they are not contribution to cobalt mine exploitation</td></tr>
<tr><td rowspan="2">European Battery passport, QR code and CE label requirements</td><td>2027</td><td>Development of an electronic information exchange system, including a battery passport that accompanies the battery throughout its entire life cycle. The passport will be required to have the following information:
- Manufacturer information, and date of manufacture/placement on market
- Type of battery and batch number/model identifier
- Chemical composition including harmful substances and recycled material content
- Details on procedures and actions for repair, reuse, and dismantling
- Approaches for treatment, recycling, and recovery of EoL batteries</td></tr>
<tr><td>2027</td><td>Batteries >2 kWh placed on the Union market will need to be electronically registered with QR code/CE label, for safety, traceability and key parameter communication</td></tr>
<tr><td rowspan="2">Collection targets for EoL</td><td>2028</td><td>For light means of transport: 51 %</td></tr>
<tr><td>2031</td><td>For light means of transport: 61 %</td></tr>
<tr><td rowspan="2">Recycling efficiencies from EoL batteries</td><td>2025</td><td>Targets (by average weight):
- Pb acid batteries: 75 %
- Li-based: 65 %
- Ni-Cd: 80 %</td></tr>
<tr><td>2030</td><td>Targets (by average weight):
- Pb acid batteries: 80 %
- Ni-based: 70 %</td></tr>
<tr><td rowspan="2">Material recovery targets from EoL batteries</td><td>2027</td><td>Targets for recycling (minimum quantities for materials recovered from spent batteries):
- Li: 50 %
- Co, Cu, Pb and Ni: 90 %</td></tr>
<tr><td>2031</td><td>Targets for recycling (minimum quantities for materials recovered from spent batteries):
- Li: 80 %
- Co, Cu, Pb and Ni: 95 %</td></tr>
<tr><td></td><td>2028</td><td>Recycled content declaration</td></tr>
</table>

Feature	Year	Specifics
Recycled material content requirements for all batteries[b,c]	2031	A certain proportion of recovered materials is to be made mandatory for use in new batteries: - Co: 16 % - Pb: 85 % - Li and Ni: 6 %
	2036	A certain proportion of recovered materials is to be made mandatory for use in new batteries: - Co: 26 % - Pb: 85 % - Li 12 % - Ni:15 %

Requirements for EV battery lifetime (both BEV and PHEV) have now been set in the new Euro 7 standards; the first time that durability standards have been set in regulation (Dornoff & Rodríguez, 2024; European Parliament & Council of the European Union, 2024). According to this, passenger vehicle batteries must retain an energy storage capacity of 80 % after 5 years or 100,000 km[19]. After 8 years or 160,000, battery capacity must be at least 72 %. A revision is expected to the durability requirements in future, with placeholders existing for achievable range requirements in addition (Dornoff & Rodríguez, 2024).

4.2 Selected policies at Norwegian national level

As part of the European Economic Area (EEA), Norway aligns with EU regulations and policies surrounding circularity related to vehicles (including their emissions) and batteries, but has also set its own domestic policies and initiatives to promote circularity within the transport sector. Following up to the EU Circular Economy Action Plan, Norway developed its own strategy for a CE in 2021, the National Strategy for a Green Circular Economy ('Nasjonal strategi for ein grønn, sirkulær økonomi', (Regjeringen, 2021). Regarding the vehicle system, the government notes that a significant challenge is to establish effective recycling solutions for the large number of EV batteries from the transportation sector as they become obsolete, whilst properly handling hazardous substances and components. The strategy aims to help implement comprehensive battery regulations in Norway and Europe that cover the entire value chain, promote high environmental standards, and contribute to increased circularity and sustainable growth. Publication of Norway's Battery Strategy in 2022 continued the focus on LIB, outlining key initiatives and presented sustainable industrialization actions (Norwegian Ministry of Trade Industry and Fisheries, 2022). The strategy discusses the importance of recycling/reuse at EoL and included measures that the Norwegian Government will work to ensure that Norway contributes to development of sustainable value chains and follow up on EU CE strategy. The report also notes that the European Battery Alliance (EBA) anticipates the Nordic Region will become one of three key hubs for the European battery industry, alongside Germany and Hungary, due to its involvement across all segments of the battery value chain (Norwegian Ministry of Trade Industry and Fisheries, 2022).

Emissions reductions from transport are also targeted. Norway has set an overall goal to reduce territorial (direct) CO_2 by 55 % by 2030, and 90-95 % by 2050, compared to the 1990 baseline (Climate Action Tracker, 2024). In the transport sector specifically, the goal by 2030 is to reduce

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[19] Whichever is reached first.

emissions by 50 % compared with 2005, as described in the Climate Action Plan for 2021-2030 (Norwegian Ministry of Climate and Environment, 2021). Among the primary policy instruments to achieve this are carbon taxes, biofuel quota obligations, investment support schemes, and promotion of the switch from ICE powertrains to so-called zero-emission. Regarding the latter, ambitions for zero-emission (tailpipe) vehicles (see Table 4.4) are set (Samferdselsdepartementet, 2024), which have been stimulated through various measures including substantial incentives. These have included a high tax on CO_2 emissions per km, exemption from purchase/import tax on EVs between 1990 to 2022, exemption from 25 % VAT on purchase between 2001 and 2022, and exception from annual road tax between 1996 and 2021[20] (Norsk Elbilforening, 2024). As of Sept 2022, 89 % of first-time registered passenger vehicles in Norway were electric (BEVs and PHEVs combined), which amounts to the highest percentage for any European country (EEA, 2023). Almost 80 % of these were BEVs alone. As of Sept 2024, 96.4 % of new passenger vehicles were zero-emission, with the most sold vehicle being a Model Y Tesla (OFV, 2024).

Table 4.4: Zero-emission vehicle targets set in the Norwegian National Transport Plan (Samferdselsdepartementet, 2024).

Year	Target
2025	All new passenger vehicles and small vans will be zero-emission
	All new city buses will be zero emission or use biogas
2030	All heavy-duty vehicles >3500 kg will be zero-emission or use biogas (includes lorries, long distance buses), and all heavy vans will be zero-emission.

[20] Starting in 2023, purchase tax applies to all new EVs, with 25 % VAT on the purchase price exceeding 500,000 NOK. A reduced tax rate was introduced in 2021, followed by full taxation in 2022.

5 Circularity principles in CELECT

CELECT will continue building upon the body of literature reviewed to determine the relative environmental sustainability of circularity strategies within the framework of existing Norwegian and European policy, via use of LCA.

To determine how to account for different circularity strategies in an LCA, an evaluation was carried out utilizing the knowledge gained from the literature/policy review. The aim of this was to determine how each relevant CE principle over a vehicle's life cycle (production, use, and EoL – presented earlier in Figure 3.1) affects the parameters in the foreground system[21]. The resulting overview is given in Table 5.1.

Table 5.1: Overview of key circularity factors over various life cycle stages and their relation to LCA parameters.

Life cycle stage	Key circularity factors	LCA parameters
Processing and production	Design/repairability	Lifetime
		Maintenance resource and energy needs
	Modular	Lifetime
		Maintenance resource and energy needs
	Ecofriendly materials and manufacturing	Material and resource use
	Lightweighting	Material and resource use
		Operational energy use
		Recyclability
	Logistics	Transport mode and distribution options
Use	Fuel efficient and eco driving behaviour	Operational energy use
	User numbers (including e.g. shared mobility effects)	Vehicle utility
	Capacity (e.g. passenger seats, luggage space)	

[21] In LCA the foreground system refers to the processes that are directly within the control or decision-making scope of the study practitioner, such as production and manufacturing processes. The background system consists of processes that are outside the control but are necessary for the LCA (such as electricity generation), typically modelled using external databases.

Life cycle stage	Key circularity factors	LCA parameters
	Use characteristics (e.g. distance, speed, terrain)	
	Fuel type (powertrain)	Operational energy use
		Powertrain technology
		Material and resource use
	Predictive maintenance (including damage prevention), servicing and repair	Lifetime
		Maintenance resource and energy needs
	Optimized vehicle routing and logistics (telematics)	Operational energy use
End of (first) life	Second life applications for used batteries and vehicle parts	Vehicle and battery EoL options (reuse, repurposing, recycling)
		Allocation between multipurpose utilities (i.e. first and second life)
	Closed loop systems to reuse recovered materials in vehicle and battery manufacturing	Material and resource use
	Effective recycling and material recovery systems (vehicles and batteries)	Material and resource use
Supporting infrastructure	Extend producer responsibility through policy	Will contribute to the background system and assumptions
	Consumer awareness and education	
	Reverse logistics for efficient systems for collecting EoL vehicles/batteries	
	Innovation and research	

Overall, nine key LCA parameters were identified including:

- Vehicle and battery lifetime
- Material and resource use
- Maintenance resource and energy needs
- Operational energy use
- Recyclability
- Transport mode and distribution options
- Vehicle utility
- Vehicle and battery EoL options (reuse, repurposing, recycling)
- Allocation between multipurpose utilities (i.e. first and second life)

Having reviewed the LCA literature in this report, we consider how the LCA literature covers the identified LCA parameters and how the upcoming LCA-work in CELECT may cover these, see Table

5.2. In the table, cells with pale green background indicate research gaps that will be a focus area within the CELECT LCA work.

Table 5.2: Evaluation of LCA parameter coverage in LCA literature and potential research gaps that may be covered in the LCA-work in CELECT.

LCA parameter	Evaluation of coverage in LCA literature	Potential research gaps to cover in the CELECT LCA-work
Vehicle and battery lifetime	These parameters are handled in a generic manner, where typically vehicle lifetime is set to a certain number of years or mileage, while battery lifetime is often considered as equal to vehicle lifetime. Furthermore, differences in vehicle lifetime between powertrain technologies or model year are generally not considered. Some LCA studies address the battery lifetime uncertainty through an uncertainty analysis considering battery replacement, but this is not always the case.	Obtain lifetime data (relating to vehicle km driven and years life) for different vehicle powertrains, model years, and batteries from CELECT partners. If data is available, examine which factors affect lifetime.
Material and resource use	Material and resource use associated with production is commonly modelled assuming a generic glider (vehicle without powertrain) and with specific powertrain components. In research, lack of specific vehicle size and component data, makes for rudimentary modelling of various sizes/segments, powertrains, and components. Due to lack of detailed vehicle data, research studies often handle vehicle sizing by linear extrapolation and lightweighting is dealt with in an ad-hoc manner where a certain share of steel and iron is typically replaced by light alloys.	General information about vehicle trends in terms of powertrains, vehicle sizes/segments, and material use may be obtained from CELECT partners, but access to more detailed information about material and resource use is limited within the CELECT project as none of the partners produce vehicles.
Maintenance resource and energy needs	Maintenance resource and energy needs are often omitted or modelled in a generic manner using data from databases. If included, current LCA studies do not sufficiently distinguish the needs for different powertrain technologies or vehicle sizes/segments.	Obtain data regarding material and resource needs for planned maintenance as well as repairs for the various powertrain technologies from project partners. If possible, differentiate between powertrains technologies and size/segments.
Operational energy use	Operational energy use is often modelled with an assumed energy use, sometimes using test data for one or several specific vehicle models on the market (e.g., WLTP energy use). Most studies tend to focus on medium sized passenger vehicles, while very few consider energy use of different vehicle sizes/segments.	Collect both test data for vehicles as well as real-life energy use for various powertrains and sizes/segments from relevant CELECT partners. Look into differences in energy use across model years as well.
Recyclability	Most LCA studies considering EoL treatment use generic data from databases or literature with no differentiation between powertrain technologies, size/segments, or model year. Furthermore, statistics about recycling and recovery rates or reuse of	Compile inventory data based on statistics from CELECT partners dealing with EoL. Where possible, distinguish between various powertrain technologies, size/segments, and model year.

LCA parameter	Evaluation of coverage in LCA literature	Potential research gaps to cover in the CELECT LCA-work
	components/materials are not considered specifically.	
Transport mode and distribution options	Transport and distribution of vehicles is often ignored or modelled in a generic manner. The limited focus on this life cycle stage is justified as its contribution to life cycle impacts has been found to be of insignificant size.	Due to its scarce contribution to life cycle impacts, transport and distribution will not be a focus point in the LCA work but information about it will be collected from relevant CELECT partners.
Vehicle utility	LCA studies considering passenger vehicles do not tend to focus on user numbers (e.g. shared mobility) or capacity but rather on technologies. LCA studies tend to report impacts in terms of vehicle-kilometer (or per vehicle) while passenger-kilometer is primarily considered in studies comparing different transport modes (e.g., buses, planes, ferries). Effects of use characteristic variation (e.g. distance driven) are sometimes studied in LCA sensitivity analyses, but are typically not based on real data regarding usage patterns.	CELECT will determine how variation in user numbers/vehicle capacity, as well as use characteristics (e.g. distance driven annually) affects relative environmental impacts. This may be considered using different functional units (e.g., passenger-kilometre or seat-kilometre) or in sensitivity analyses. This data will be based as much as possible on real data regarding vehicle use collected within the project from relevant CELECT partners
Vehicle and battery EoL options (reuse, repurposing, recycling)	A growing body of literature is considering battery reuse, repurposing, and recycling. The access to real-life battery data is limited though and studies often base inventory data and analysis on coarse assumptions about battery life, as well as repurposing needs and second life application and durability.	Collect real-life data for battery EoL options from CELECT partners. Additionally, consider and compile data for reuse options for other vehicle components/materials as CELECT partners have statistics/information about this.
Allocation between multipurpose utilities (i.e. first and second life)	Various allocation alternatives may be used in an attributional LCA to ascribe benefits of multipurpose use of components. Physical (i.e., energy) and economic allocation are particularly relevant, but LCA studies tend to set the allocation keys based on assumptions of use and lifetime, rather than real-life data.	Collect real-life battery data (e.g., state-of-health, value of used battery, cycles numbers, expected cycles/lifetime in second life) from EoL CELECT partners to make for more data-driven allocation procedures.

In summary, we identify seven research gaps that we will focus on in the LCA-work in CELECT. While the other three research gaps are not main focus areas, these may be considered to some degree as they are still relevant for the LCA-work and CELECT at large. To the extent that is possible, data will be collected for these areas from CELECT project partners who include the Norwegian Electric Vehicle Association, the Norwegian Automobile Federation (NAF), the Norwegian Car Industry Association (NBF), the Norwegian Public Roads Administration (SVV), and key players from Norwegian industry including Fremtind Forsikring AS, Bertel O. Steen AS, Autoretur AS, Batteriretur AS, and ECO STOR AS. Scientific literature will supplement the data and provide a unifying basis.

6 The road ahead

To achieve a more sustainable circular transport system and live within planetary boundaries, circularity principles should be considered to reduce environmental impacts and keep materials in the system for as many cycles as possible. These principles should be evaluated by LCA to minimize environmental burdens and problem shifting. The limited literature considering the relative environmental impacts of vehicle circularity principles does not consider differences among powertrains, size/segment, and model year. As these three aspects affect environmental sustainability, CELECT will therefore perform LCA of passenger vehicles considering all powertrains and sizes/segments as well as model year (as far as possible), from which recommendations will be developed to optimize circularity and environmental performance.

In this report, we considered the circularity principles alongside current LCA findings. This allowed us to map out the interplay between these sustainability perspectives. The review led to identification of key circularity factors and LCA parameters (Table 5.2) as well as an evaluation of how these parameters are covered in the LCA literature and what the potential research gaps that we may cover in the CELECT LCA work (Table 5.2). Out of nine identified research gaps, we aim to focus the LCA work in CELECT on:

1. Vehicle and battery lifetime
2. Maintenance resource and energy needs
3. Operation energy use
4. Vehicle utility
5. Recyclability
6. Vehicle and battery EoL options
7. Allocation between multipurpose utilities

For these seven focus areas, we will collect and compile real-life vehicle and battery information and statistics from CELECT partners to compile detailed and comprehensive life cycle inventories for the LCA work.

References

ACEA. (2021). ACEA Position Paper - Life Cycle Assessment in the automotive industry.

Aguilar Esteva, L. C., Kasliwal, A., Kinzler, M. S., Kim, H. C., & Keoleian, G. A. (2020). Circular economy framework for automobiles: Closing energy and material loops. Journal of Industrial Ecology, 25(4), 877–889. https://doi.org/10.1111/jiec.13088

Ahmed, A. A., Nazzal, M. A., Darras, B. M., & Deiab, I. M. (2023). A Comprehensive Sustainability Assessment of Battery Electric Vehicles, Fuel Cell Electric Vehicles, and Internal Combustion Engine Vehicles through a Comparative Circular Economy Assessment Approach. Sustainability (Switzerland), 15(1). https://doi.org/10.3390/su15010171

Bauer, C., Hofer, J., Althaus, H.-J., Del Duce, A., & Simons, A. (2015). The environmental performance of current and future passenger vehicles: Life Cycle Assessment based on a novel scenario analysis framework. Applied Energy, 1–13. https://doi.org/10.1016/j.apenergy.2015.01.019

Baum, Z. J., Bird, R. E., Yu, X., & Ma, J. (2022). Lithium-Ion Battery Recycling–Overview of Techniques and Trends. ACS Energy Letters, 7(2), 712–719. https://doi.org/10.1021/acsenergylett.1c02602

Bein, T., Cerdas, F., Lbf, F., Hill, N., & Raugei, M. (2023). TranSensus LCA. 101056715.

Bobba, S., Mathieux, F., Ardente, F., Blengini, G. A., Cusenza, M. A., Podias, A., & Pfrang, A. (2018). Life Cycle Assessment of repurposed electric vehicle batteries: an adapted method based on modelling energy flows. Journal of Energy Storage, 19, 213–225. https://doi.org/10.1016/j.est.2018.07.008

Bonsu, N. O. (2020). Towards a circular and low-carbon economy: Insights from the transitioning to electric vehicles and net zero economy. Journal of Cleaner Production, 256. https://doi.org/10.1016/j.jclepro.2020.120659

Bourguignon, D. (2016). Closing the loop: new circular economy package. Briefing to European Parliament January 2016. https://www.europarl.europa.eu/RegData/etudes/BRIE/2016/573899/EPRS_BRI%282016%29573899_EN.pdf

Bruno, M., & Fiore, S. (2023). Material Flow Analysis of Lithium-Ion Battery Recycling in Europe: Environmental and Economic Implications. Batteries, 9(4). https://doi.org/10.3390/batteries9040231

Casals, L. C., Amante García, B., & Canal, C. (2019). Second life batteries lifespan: Rest of useful life and environmental analysis. Journal of Environmental Management, 232, 354–363. https://doi.org/10.1016/j.jenvman.2018.11.046

Choi, Y., & Rhee, S. W. (2020). Current status and perspectives on recycling of end-of-life battery of electric vehicle in Korea (Republic of). Waste Management, 106, 261–270. https://doi.org/10.1016/j.wasman.2020.03.015

Circle Economy Foundation. (2024). CIRCULARITY GAP REPORT. https://www.circularity-gap.world/2024#download

Climate Action Tracker. (2024). Target overview. https://climateactiontracker.org/countries/norway/targets/

Daimler AG. (2011). Environmental Certificate for the new B-Class.

Daimler AG. (2019). 360 Environmental Check Mercedes-Benz EQC.

Daimler AG. (2022). 360 ° Environmental Check Mercedes-Benz C 300 e Plug-in Hybrid.

Daimler AG. (2023). 360° Environmental check Mercedes-Benz EQE SUV.

Danilecki, K., Eliasz, J., Smurawski, P., Stanek, W., & Szlęk, A. (2021). Modeling inventory and environmental impacts of car maintenance and repair: A case study of Ford Focus passenger car. Journal of Cleaner Production, 315(April 2020). https://doi.org/10.1016/j.jclepro.2021.128085

de Oliveira, C. T., & Oliveira, G. G. A. (2023). What Circular economy indicators really measure? An overview of circular economy principles and sustainable development goals. Resources, Conservation and Recycling, 190. https://doi.org/10.1016/j.resconrec.2022.106850

Dolganova, I., Rödl, A., Bach, V., Kaltschmitt, M., & Finkbeiner, M. (2020). A review of life cycle assessment studies of electric vehicles with a focus on resource use. Resources, 9(3), 4–8. https://doi.org/10.3390/resources9030032

Dornoff, J., & Rodríguez, F. (2024). Euro 7: The new emission standard for light-and heavy-duty vehicles in the European Union. https://data.consilium.europa.eu/doc/document/ST-16960-2023-REV-1/

EEA. (2023). New registrations of electric vehicles in Europe. https://www.eea.europa.eu/en/analysis/indicators/new-registrations-of-electric-vehicles?activeAccordion=546a7c35-9188-4d23-94ee-005d97c26f2b

Ellen MacArthur Foundation. (2013). Towards the Circular Economy vol 1: Economic and business rationale for an accelerated transition. https://www.ellenmacarthurfoundation.org/towards-the-circular-economy-vol-1-an-economic-and-business-rationale-for-an

Ellen MacArthur Foundation. (2019). Circular economy systems diagram. www.ellenmacarthurfoundation.org

Ellingsen, L. A.-W., Thorne, R. J., & Figenbaum, E. (2022). Comparative life cycle assessment of mid-sized electric, diesel, and gasoline passenger vehicles. EVS35 Symposium.

Ellingsen, L. A.-W., & Hung, C. R. (2018). Part 2: Research for TRAN Committee - Resource and climate aspects of lithium-ion traction batteries and battery electric vehicles. https://doi.org/10.2861/944056

Ellingsen, L. A.-W., Singh, B., & Strømman, A. H. (2016). The size and range effect: lifecycle greenhouse gas emissions of electric vehicles. Environmental Research Letters, 11(5), 1–8. https://doi.org/10.1088/1748-9326/11/5/054010

European Commission. (2020). COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS: A new Circular Economy Action Plan For a cleaner and more competitive Europe.

European Commission. (2024a). Circular economy. https://environment.ec.europa.eu/topics/circular-economy_en

European Commission. (2024b). CO_2 emission performance standards for cars and vans. https://climate.ec.europa.eu/eu-action/transport/road-transport-reducing-co2-emissions-vehicles/co2-emission-performance-standards-cars-and-vans_en

European Commission. (2024c). REPowerEU: Affordable, secure and sustainable energy for Europe. https://commission.europa.eu/strategy-and-policy/priorities-2019-2024/european-green-deal/repowereu-affordable-secure-and-sustainable-energy-europe_en

European Council, & Council of the European Union. (2024). European Green Deal. https://www.consilium.europa.eu/en/policies/green-deal/fit-for-55/

European Parliament. (2023a). Fit for 55: deal on charging and fuelling stations for alternative fuels. https://www.europarl.europa.eu/news/en/press-room/20230327IPR78504/fit-for-55-deal-on-charging-and-fuelling-stations-for-alternative-fuels

European Parliament. (2023b). New EU regulatory framework for batteries: Briefing - EU Legislation in Progress. https://www.europarl.europa.eu/RegData/etudes/BRIE/2021/689337/EPRS_BRI(2021)689337_EN.pdf

European Parliament, & Council of the European Union. (2000). DIRECTIVE 2000/53/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 18 September 2000 on end-of life vehicles. https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=celex%3A32000L0053

European Parliament, & Council of the European Union. (2005). Directive 2005/64/EC of the European Parliament and of the Council of 26 October 2005 on the type-approval of motor vehicles with regard to their reusability, recyclability and recoverability and amending Council Directive 70/156/EEC. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A32005L0064

European Parliament, & Council of the European Union. (2006). Directive 2006/66/EC of the European Parliament and of the Council of 6 September 2006 on batteries and accumulators and waste batteries and accumulators and repealing Directive 91/157/EEC. https://eur-lex.europa.eu/eli/dir/2006/66/oj

European Parliament, & Council of the European Union. (2019). REPORT FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE, THE COMMITTEE OF THE REGIONS AND THE EUROPEAN INVESTMENT BANK on the Implementation of the Strategic Action Plan on Batteries: Building a Strategic Battery Value Chain in Europe. https://eur-lex.europa.eu/legal-content/en/TXT/?uri=CELEX%3A52019DC0176

European Parliament, & Council of the European Union. (2022). DECISION (EU) 2022/591 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 6 April 2022 on a General Union Environment Action Programme to 2030. https://environment.ec.europa.eu/strategy/environment-action-programme-2030_en

European Parliament, & Council of the European Union. (2023a). Directive (EU) 2023/1791 of the European Parliament and of the Council of 13 September 2023 on energy efficiency and amending Regulation (EU) 2023/955. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=OJ%3AJOL_2023_231_R_0001&qid=1695186598766

European Parliament, & Council of the European Union. (2023b). Directive (EU) 2023/2413 of the European Parliament and of the Council of 18 October 2023 amending Directive (EU) 2018/2001, Regulation (EU) 2018/1999 and Directive 98/70/EC as regards the promotion of energy from renewable sources, and repealing Council Directive (EU) 2015/652. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A32023L2413&qid=1699364355105

European Parliament, & Council of the European Union. (2023c). REGULATION (EU) 2023/1542 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 12 July 2023 concerning batteries and waste batteries, amending Directive 2008/98/EC and Regulation (EU) 2019/1020 and repealing Directive 2006/66/EC. https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX:32023R1542

European Parliament, & Council of the European Union. (2024). REGULATION OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL on type-approval of motor vehicles and engines and of systems, components and separate technical units intended for such vehicles, with respect to their emissions and battery durability (Euro 7), amending Regulation (EU) 2018/858 of the European Parliament and of the Council and repealing Regulations (EC) No 715/2007 and (EC) No 595/2009 of the European Parliament and of the Council. https://data.consilium.europa.eu/doc/document/PE-109-2023-INIT/en/pdf

European Platform on LCA | EPLCA. (2003). https://eplca.jrc.ec.europa.eu/aboutUs.html

Eurostat. (2020). Energy, transport and environment statistics : 2020 edition. European Union. https://ec.europa.eu/eurostat/web/products-statistical-books/-/ks-dk-20-001

Faessler, B. (2021). Stationary, second use battery energy storage systems and their applications: A research review. Energies, 14(8). https://doi.org/10.3390/en14082335

Figenbaum, E., & Nordbakke, S. (2019). Battery electric vehicle user experiences in Norway's maturing market. https://www.toi.no/getfile.php?mmfileid=50956

Flash Battery. (2024). European battery regulation: A concrete step towards a sustainable future. https://www.flashbattery.tech/en/new-european-battery-regulation/

Global Footprint Network. (2024, April 17). About Earth overshoot day. https://overshoot.footprintnetwork.org/about/

Hatrem, H. (2021, December 22). Batterikollaps i åtte år gammel Tesla Model S: Dette sier importørene om levetid på batteriet (in Norwegian). VG. https://www.vg.no/forbruker/bil-baat-og-motor/i/a7qdX2/batterikollaps-i-aatte-aar-gammel-tesla-model-s-dette-sier-importoerene-om-levetid-paa-batteriet

Hawkins, T. R., Gausen, O. M., & Strømman, A. H. (2012). Environmental impacts of hybrid and electric vehicles—a review. The International Journal of Life Cycle Assessment, 17(8), 997–1014. https://doi.org/10.1007/s11367-012-0440-9

Hawkins, T. R., Singh, B., Majeau-Bettez, G., & Strømman, A. H. (2012). Comparative Environmental Life Cycle Assessment of Conventional and Electric Vehicles. Journal of Industrial Ecology, 17(0), 53–64. https://doi.org/10.1111/j.1530-9290.2012.00532.x

Heath, G. A., Ravikumar, D., Hansen, B., & Kupets, E. (2022). A critical review of the circular economy for lithium-ion batteries and photovoltaic modules–status, challenges, and opportunities. In Journal of the Air and Waste Management Association (Vol. 72, Issue 6, pp. 478–539). Taylor and Francis Ltd. https://doi.org/10.1080/10962247.2022.2068878

Illa Font, C. H., Siqueira, H. V., Machado Neto, J. E., Santos, J. L. F. dos, Stevan, S. L., Converti, A., & Corrêa, F. C. (2023). Second Life of Lithium-Ion Batteries of Electric Vehicles: A Short Review and Perspectives. Energies, 16(2). https://doi.org/10.3390/en16020953

Jerome, A., Helander, H., Ljunggren, M., & Janssen, M. (2022). Mapping and testing circular economy product-level indicators: A critical review. In Resources, Conservation and Recycling (Vol. 178). Elsevier B.V. https://doi.org/10.1016/j.resconrec.2021.106080

Jerome, A., Ljunggren, M., & Janssen, M. (2023). Is repair of energy using products environmentally beneficial? The case of high voltage electric motors. Resources, Conservation and Recycling, 196. https://doi.org/10.1016/j.resconrec.2023.107038

Jiang, S., Hua, H., Zhang, L., Liu, X., Wu, H., & Yuan, Z. (2022). Environmental impacts of hydrometallurgical recycling and reusing for manufacturing of lithium-ion traction batteries in China. Science of the Total Environment, 811. https://doi.org/10.1016/j.scitotenv.2021.152224

Kallitsis, E., Korre, A., & Kelsall, G. H. (2022). Life cycle assessment of recycling options for automotive Li-ion battery packs. Journal of Cleaner Production, 371. https://doi.org/10.1016/j.jclepro.2022.133636

Kelly, J. C., Kim, T., Kolodziej, C. P., Iyer, R. K., Tripathi, S., Elgowainy, A., & Wang, M. (2024). Comprehensive Cradle to Grave Life Cycle Analysis of On-Road Vehicles in the United States based on GREET. SAE International Journal of Alternative Powertrains.

Keoleian, G. A., & Sullivan, J. L. (2012). Materials challenges and opportunities for enhancing the sustainability of automobiles. MRS Bulletin, 37(4), 365–372. https://doi.org/10.1557/mrs.2012.52

Kifor, C. V., & Grigore, N. A. (2023). Circular Economy Approaches for Electrical and Conventional Vehicles. In Sustainability (Switzerland) (Vol. 15, Issue 7). MDPI. https://doi.org/10.3390/su15076140

Kim, H. C., Keoleian, G. A., Grande, D. E., & Bean, J. C. (2003). Life cycle optimization of automobile replacement: Model and application. Environmental Science & Technology, 37(23), 5407–5413.

Kim, H., McMillan, C., Keoleian, G. A., & Skerlos, S. J. (2010). Greenhouse Gas Emissions Payback for Lightweighted Vehicles Using Aluminum and High-Strength Steel. Journal of Industrial Ecology, 14(6), 929–946. https://doi.org/10.1111/j.1530-9290.2010.00283.x

Kim, H.-J., Raichur, V., & Skerlos, S. J. (2008). Economic and Environmental Assessment of Automotive Remanufacturing: Alternator Case Study. ASME 2008 International Manufacturing Science and Engineering Conference, Volume 1, 33–40. https://doi.org/10.1115/MSEC_ICMP2008-72490

Kirchherr, J., Reike, D., & Hekkert, M. (2017). Conceptualizing the circular economy: An analysis of 114 definitions. In Resources, Conservation and Recycling (Vol. 127, pp. 221–232). Elsevier B.V. https://doi.org/10.1016/j.resconrec.2017.09.005

Korhonen, J., Nuur, C., Feldmann, A., & Birkie, S. E. (2018). Circular economy as an essentially contested concept. Journal of Cleaner Production, 175, 544–552. https://doi.org/10.1016/j.jclepro.2017.12.111

Kotak, Y., Fernández, C. M., Casals, L. C., Kotak, B. S., Koch, D., Geisbauer, C., Trilla, L., Gómez-Núñez, A., & Schweiger, H. G. (2021). End of electric vehicle batteries: Reuse vs. recycle. In Energies (Vol. 14, Issue 8). MDPI AG. https://doi.org/10.3390/en14082217

Lonca, G., Muggéo, R., Imbeault-Tétreault, H., Bernard, S., & Margni, M. (2018). Does material circularity rhyme with environmental efficiency? Case studies on used tires. Journal of Cleaner Production, 183, 424–435. https://doi.org/10.1016/j.jclepro.2018.02.108

Lone, K. R. (2022). A bottom-up approach to end-of-life management of lithium-ion batteries from passenger electric vehicles in Norway: Masters thesis project work. NTNU.

Mansuino, M., Thakur, J., & Lakshmi, A. (2024). Turning the wheel: Measuring circularity in Swedish automotive products. Sustainable Production and Consumption, 45, 139–157. https://doi.org/10.1016/j.spc.2023.12.016

Martins, A. V., Godina, R., Azevedo, S. G., & Carvalho, H. (2021). Towards the development of a model for circularity: The circular car as a case study. Sustainable Energy Technologies and Assessments, 45. https://doi.org/10.1016/j.seta.2021.101215

Matos, J., Santos, S., Simões, C. L., Martins, C. I., & Simoes, R. (2023). Practical application of circularity micro-indicators to automotive plastic parts in an industrial context. Sustainable Production and Consumption, 43, 155–167. https://doi.org/10.1016/j.spc.2023.11.009

Mikosch, N., Dettmer, T., Plaga, B., Gernuks, M., & Finkbeiner, M. (2022). Relevance of Impact Categories and Applicability of Life Cycle Impact Assessment Methods from an Automotive Industry Perspective. Sustainability (Switzerland), 14(14). https://doi.org/10.3390/su14148837

Moraga, G., Huysveld, S., Mathieux, F., Blengini, G. A., Alaerts, L., Van Acker, K., de Meester, S., & Dewulf, J. (2019). Circular economy indicators: What do they measure? Resources, Conservation and Recycling, 146, 452–461. https://doi.org/10.1016/j.resconrec.2019.03.045

Neef, M., Dettmer, T., Bäuml, G., & Schebek, L. (2023). Decarbonisation Index (DCI): an LCA-based key performance indicator for the automotive industry. International Journal of Life Cycle Assessment, DCI. https://doi.org/10.1007/s11367-023-02247-w

Nobre, G. C., & Tavares, E. (2021). The quest for a circular economy final definition: A scientific perspective. Journal of Cleaner Production, 314. https://doi.org/10.1016/j.jclepro.2021.127973

Nordelöf, A., Messagie, M., Tillman, A.-M., Ljunggren Söderman, M., & Van Mierlo, J. (2014). Environmental impacts of hybrid, plug-in hybrid, and battery electric vehicles—what can we learn from life cycle assessment? The International Journal of Life Cycle Assessment. https://doi.org/10.1007/s11367-014-0788-0

Norsk Elbilforening. (2024). Norwegian EV policy. https://elbil.no/english/norwegian-ev-policy/

Norwegian Ministry of Climate and Environment. (2021). Norway's Climate Action Plan for 2021-2030. https://www.regjeringen.no/contentassets/a78ecf5ad2344fa5ae4a394412ef8975/en-gb/pdfs/stm202020210013000engpdfs.pdf

Norwegian Ministry of Trade Industry and Fisheries. (2022). Norway's battery strategy. https://www.regjeringen.no/contentassets/a894b5594dbf4eccbec0d65f491e4809/norways-battery-strategy_spreads_web.pdf

Nurdiawati, A., & Agrawal, T. K. (2022). Creating a circular EV battery value chain: End-of-life strategies and future perspective. Resources, Conservation and Recycling, 185. https://doi.org/10.1016/j.resconrec.2022.106484

OFV (2024). Bilsalget i september 2024. https://ofv.no/bilsalget/bilsalget-i-september-2024

Pagliaro, M., & Meneguzzo, F. (2019). Lithium battery reusing and recycling: A circular economy insight. In Heliyon (Vol. 5, Issue 6). Elsevier Ltd. https://doi.org/10.1016/j.heliyon.2019.e01866

Pedneault, J., Majeau-bettez, G., Krey, V., & Margni, M. (2021). What future for primary aluminium production in a decarbonizing economy ? Global Environmental Change, 69(July). https://doi.org/10.1016/j.gloenvcha.2021.102316

Pellow, M. A., Ambrose, H., Mulvaney, D., Betita, R., & Shaw, S. (2020). Research gaps in environmental life cycle assessments of lithium ion batteries for grid-scale stationary energy storage systems: End-of-life options and other issues. In Sustainable Materials and Technologies (Vol. 23). Elsevier B.V. https://doi.org/10.1016/j.susmat.2019.e00120

Pero, F. Del, Delogu, M., & Pierini, M. (2018). Life Cycle Assessment in the automotive sector: a comparative case study of Internal Combustion Engine (ICE) and electric car. Procedia Structural Integrity, 12, 521–537. https://doi.org/10.1016/J.PROSTR.2018.11.066

Picatoste, A., Justel, D., & Mendoza, J. M. F. (2022). Circularity and life cycle environmental impact assessment of batteries for electric vehicles: Industrial challenges, best practices and research guidelines. In Renewable and Sustainable Energy Reviews (Vol. 169). Elsevier Ltd. https://doi.org/10.1016/j.rser.2022.112941

Popien, J. L., Thies, C., & Spengler, T. S. (2022). Exploring recycling options in battery supply chains - A life cycle sustainability assessment. Procedia CIRP, 105, 434–439. https://doi.org/10.1016/j.procir.2022.02.072

Potting, J., Hekkert, M. P., Worrell, E., & Hanemaaijer, A. (2017). CIRCULAR ECONOMY: MEASURING INNOVATION IN THE PRODUCT CHAIN. https://www.researchgate.net/publication/319314335

Prochatzki, G., Mayer, R., Haenel, J., Schmidt, A., Götze, U., Ulber, M., Fischer, A., & Arnold, M. G. (2023). A critical review of the current state of circular economy in the automotive sector. In Journal of Cleaner Production (Vol. 425). Elsevier Ltd. https://doi.org/10.1016/j.jclepro.2023.138787

Ragonnaud, G. (2023). BRIEFING EU Legislation in Progress Proposal for a Regulation of the European Parliament and of the Council on circularity requirements for vehicle design and on management of end-of-life vehicles, amending Regulations (EU) 2018/858 and 2019/1020 and repealing Directives 2000/53/EC and 2005/64/EC. https://www.europarl.europa.eu/RegData/etudes/BRIE/2023/754627/EPRS_BRI(2023)754627_EN.pdf

Rask, N. (2022). An intersectional reading of circular economy policies: towards just and sufficiency-driven sustainabilities. Local Environment, 27(10–11), 1287–1303. https://doi.org/10.1080/13549839.2022.2040467

Regjeringen. (2021). Nasjonal strategi for ein grøn, sirkulaer økonomi. https://www.regjeringen.no/contentassets/f6c799ac7c474e5b8f561d1e72d474da/t-1573n.pdf

Ricardo, ifeu, & E4tech. (2020). Determining the environmental impacts of conventional and alternatively fuelled vehicles through LCA. European Commission, DG Climate Action, 456.

Royo, M., Chulvi, V., Mulet, E., & Ruiz-Pastor, L. (2023). Analysis of parameters about useful life extension in 70 tools and methods related to eco-design and circular economy. Journal of Industrial Ecology, 27(2), 562–586. https://doi.org/10.1111/jiec.13378

Sacchi, R., Bauer, C., Cox, B. L., & Mutel, C. (2022). When, where and how can the electrification of passenger cars reduce greenhouse gas emissions? Renewable and Sustainable Energy Reviews, 162(March). https://doi.org/10.1016/j.rser.2022.112475

Samferdselsdepartementet (2024). Meld. St. 14 (2023–2024) Melding til Stortinget: Nasjonal transportplan 2025-2036. https://www.regjeringen.no/contentassets/aaee20cf5a9e468ea97fd51638c42407/no/pdfs/stm202320240014000dddpdfs.pdf

Sato, F. E. K., & Nakata, T. (2020). Energy consumption analysis for vehicle production through a material flow approach. Energies, 13(9). https://doi.org/10.3390/en13092396

Schaubroeck, S., Schaubroeck, T., Baustert, P., Gibon, T., & Benetto, E. (2020). When to replace a product to decrease environmental impact?-a consequential LCA framework and case study on car replacement. The International Journal of Life Cycle Assessment, 25, 1500–1521. https://doi.org/10.1007/s11367-020-01758-0/Published

Schulz-Mönninghoff, M., Neidhardt, M., & Niero, M. (2023). What is the contribution of different business processes to material circularity at company-level? A case study for electric vehicle batteries. Journal of Cleaner Production, 382. https://doi.org/10.1016/j.jclepro.2022.135232

Schweimer, G. W., & Levin, M. (2000). Life cycle inventory for the Golf A4. Environment and Research, 1–40.

Silvestri, L., Forcina, A., Silvestri, C., & Traverso, M. (2021). Circularity potential of rare earths for sustainable mobility: Recent developments, challenges and future prospects. Journal of Cleaner Production, 292. https://doi.org/10.1016/j.jclepro.2021.126089

Soo, V. K., Doolan, M., Compston, P., Duflou, J. R., Peeters, J., & Umeda, Y. (2021). The influence of end-of-life regulation on vehicle material circularity: A comparison of Europe, Japan, Australia and the US. Resources, Conservation and Recycling, 168. https://doi.org/10.1016/j.resconrec.2020.105294

SSB. (2024). 05522: Biler vraket mot pant, etter statistikkvariabel og år. https://www.ssb.no/statbank/table/05522/tableViewLayout1/

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., Biggs, R., Carpenter, S. R., De Vries, W., De Wit, C. A., Folke, C., Gerten, D., Heinke, J., Mace, G. M., Persson, L. M., Ramanathan, V., Reyers, B., & Sörlin, S. (2015). Planetary boundaries: Guiding human development on a changing planet. Science, 347(6223). https://doi.org/10.1126/science.1259855

Tarne, P., Traverso, M., & Finkbeiner, M. (2017). Review of life cycle sustainability assessment and potential for its adoption at an automotive company. Sustainability, 9(4), 670. https://doi.org/10.3390/su9040670

The International Organization for Standardization. (2024a). ISO 59004:2024 Circular economy. Vocabulary, principles and guidance for implementation.

The International Organization for Standardization. (2024b). ISO 59020:2024 Circular economy. Measuring and assessing circularity performance.

Toyota. (2013, October 29). Parts recycled from end-of-life vehicles. https://global.toyota/en/download/3785592/

Toyota Europe. (n.d.-a). LCA | Toyota Prius Plug-In Hybrid. Retrieved February 20, 2024, from https://www.toyota-europe.com/sustainability/carbon-neutrality/vehicle-life-cycle-assessments/lca-prius

Toyota Europe. (n.d.-b). LCA | Toyota RAV4 Plug-In Hybrid. Retrieved February 23, 2024, from https://www.toyota-europe.com/sustainability/carbon-neutrality/vehicle-life-cycle-assessments/lca-rav4

Toyota Europe. (n.d.-c). The Prius evolution. Retrieved February 14, 2024, from https://www.toyota-europe.com/sustainability/carbon-neutrality/vehicle-life-cycle-assessments/lca-prius

Velenturf, A. P. M., Archer, S. A., Gomes, H. I., Christgen, B., Lag-Brotons, A. J., & Purnell, P. (2019). Circular economy and the matter of integrated resources. In Science of the Total Environment (Vol. 689, pp. 963–969). Elsevier B.V. https://doi.org/10.1016/j.scitotenv.2019.06.449

Velenturf, A. P. M., & Purnell, P. (2021). Principles for a sustainable circular economy. In Sustainable Production and Consumption (Vol. 27, pp. 1437–1457). Elsevier B.V. https://doi.org/10.1016/j.spc.2021.02.018

Völker, T., Kovacic, Z., & Strand, R. (2020). Indicator development as a site of collective imagination? The case of European Commission policies on the circular economy. Culture and Organization, 26(2), 103–120. https://doi.org/10.1080/14759551.2019.1699092

Volkswagen AG. (2014). Der e-Golf Umweltprädikat – Hintergrundbericht [The e-Golf Environmental Commendation - Background Report].

Volvo. (2021). Carbon footprint report - Battery electric XC40 Recharge and the XC40 ICE.

Watkins, E., & Meysner, A. (2022). European Circular Economy policy landscape overview Institute for European Environmental Policy: A report for PBL Netherlands Environmental Assessment Agency. https://ieep.eu/wp-content/uploads/2022/11/European-Circular-Economy-policy-landscape-overview.pdf

Wautelet, T. (2018). The concept of circular economy: its origins and its evolution. https://doi.org/10.13140/RG.2.2.17021.87523

Wrålsen, B., & Faessler, B. (2022). Multiple Scenario Analysis of Battery Energy Storage System Investment: Measuring Economic and Circular Viability. Batteries, 8(2). https://doi.org/10.3390/batteries8020007

Wrålsen, B., & O'Born, R. (2023). Use of life cycle assessment to evaluate circular economy business models in the case of Li-ion battery remanufacturing. International Journal of Life Cycle Assessment, 28(5), 554–565. https://doi.org/10.1007/s11367-023-02154-0

Wu, W., Lin, B., Xie, C., Elliott, R. J. R., & Radcliffe, J. (2020). Does energy storage provide a profitable second life for electric vehicle batteries? Energy Economics, 92. https://doi.org/10.1016/j.eneco.2020.105010

Xia, X., & Li, P. (2022). A review of the life cycle assessment of electric vehicles: Considering the influence of batteries. Science of the Total Environment, 814, 152870. https://doi.org/10.1016/j.scitotenv.2021.152870

Xiong, S., Ji, J., & Ma, X. (2020). Environmental and economic evaluation of remanufacturing lithium-ion batteries from electric vehicles. Waste Management, 102, 579–586. https://doi.org/10.1016/j.wasman.2019.11.013

Yu, M., Bai, B., Xiong, S., & Liao, X. (2021). Evaluating environmental impacts and economic performance of remanufacturing electric vehicle lithium-ion batteries. Journal of Cleaner Production, 321. https://doi.org/10.1016/j.jclepro.2021.128935

Zhou, L. F., Yang, D., Du, T., Gong, H., & Luo, W. Bin. (2020). The Current Process for the Recycling of Spent Lithium Ion Batteries. In Frontiers in Chemistry (Vol. 8). Frontiers Media S.A. https://doi.org/10.3389/fchem.2020.578044

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