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Evaluating future powertrain and recycling technologies and their impact on the life cycle assessment and costing of mid-size and large passenger vehicles

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Electrification is needed to improve the environmental performance of the transportation sector but must be accompanied by technological innovations that eliminate ecological hotspots over the vehicle's lifetime. Additionally, the lifetime costs of electric vehicles must be reduced. This study uses the battery electric vehicle sustainability impact assessment model (BEVSIM) to analyse the effects of potential changes in recycled plastic content and its waste treatment, electricity grid, and the price of battery electric vehicles. Using recycled content in plastic parts improved their environmental performance in all categories except land use; e.g. using 40% recycled plastic reduced 107 kg CO_2 eq. per passenger vehicle. Applying a combination of mechanical and pyrolysis technologies for plastic recycling improved several impact categories. However, a trade-off exists for a few impact categories. Finally, using 2030 and 2050 forecasts of electricity grids revealed lifecycle CO_2 emissions for BEVs to be 50% and 56% lower than ICE, respectively.

Climate change mitigation requires solid plans specific to different sectors, such as construction, agriculture, aviation, and transportation. Transport remains a key obstacle to achieving the EU's climate targets[1]. For example, of Europe's total greenhouse gas emissions in 2018, 15% were produced by passenger cars, whereas 4% were produced by aviation. A shift to battery electric vehicles (BEVs) and a sustainable energy supply has been implemented in several countries and can potentially reduce emissions[2]. The potential advantages of BEVs are their powertrain efficiency, low maintenance requirements, and zero exhaust emissions. In practice, the environmental impact of a vehicle should be assessed throughout the vehicle's life cycle and all major impact categories should be considered using a full life cycle assessment (LCA)[3]. Assuming a 150,000 km vehicle lifetime, EVs powered by the European electricity mix save 26% to 30% GWP relative to gasoline-powered vehicles and 17% to 21% GWP relative to diesel-powered vehicles[4]. The GWP savings depend heavily on the electricity source used for BEVs. Research has shown that it can be counterproductive to use EVs in regions where electricity is primarily produced from fossil sources, such as lignite, coal, or even heavy oil combustion[4]. Battery electric vehicles can perform worse than ICEs in terms of human toxicity, freshwater ecotoxicity, freshwater eutrophication, and metal depletion impacts originating from the production phase[4,5]. The inferior performance of BEVs in terms of greater environmental impact can potentially be managed using "ambitious material efficiency strategies", such as lightweighting (LW), using recycled materials[6] and increasing the recycling rate at the end of life of vehicles[7].

The European targets for the reuse and recycling of end-of-life vehicles (ELVs), as well as the reuse and recovery of components, are 85% and 95%, respectively[8]. The use of recycled plastic in ICEs and BEVs has been proposed but not implemented[9]. The main reasons for this lack of implementation are cultural, regulatory, economic and technical barriers[10]. The recycling design is also an important factor in achieving circularity for the sector. Many challenges have been reported for designing recycling in the automotive industry, such as cost, design, technology, data and information, as well as EoL management and value chain management[11]. The recycling of end-of-life (EoL) plastics in the automotive industry has only been studied recently[7]. This study proposes a sustainable waste management scheme consisting of novel sorting, mechanical recycling, pyrolysis, and plastic upgrading. It is projected that using this scheme will result in a reduction in carcinogens, noncarcinogens, global warming impact categories and nonrenewable energy of 138%, 100%, 42%, and 114%, respectively, compared with state-of-the-art technologies[10,12,13].

The environmental impacts of recycling and the use of recycled content in automobiles have still not been thoroughly investigated. To complement

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environmental and economic impact assessments, i.e., life cycle assessment (LCA) and life cycle costing (LCC), the circularity of materials can be measured via the material circularity index (MCI) developed by the Ellen MacArthur Foundation[14]. The MCI index was estimated to be 0.23 for tyre production on a 0 to 1 scale. Reducing the energy recovery fraction by 14% and using 10% biobutadiene in the production line was predicted to increase the MCI to 0.49. This value can be considered an achievable goal for the coming years[15].

The LCC analysis has been applied to private vehicle ownership to assess the effect of the electricity price on the cost efficiency of electric vehicles compared with that of ICE vehicles[16]. Over the entire life cycle of a vehicle, the electricity price must be below the so-called critical electricity price for the ICE for the BEV to be more cost-efficient than the ICE. The life cycle costs of a BEV are then below those of an ICE. Among the four BEVs considered in the literature, this critical electricity price has been reached for only two BEVs. In the respective study, the vehicle life cycle costs were expressed as the net present value (NPV) to make a fair comparison, because current and future costs may differ between alternatives. The same conclusion was reached by Ayodele and Mustapa[17] in a review of LCC studies from the customer's perspective. As expected, the two important cost components are the vehicle purchase and operating costs. Khoonsari[18] carried out the only LCC study from the producer's perspective. However, battery costs are important to producers of BEVs, where the battery production cost per kWh capacity will be reduced in the future[19]. However, lowering the battery price per kWh may have rebound effects by driving a producer to install a larger battery to increase the mileage range of a vehicle, which can result in a very strong customer preference[20].

The aim of this study is to assess the environmental and economic impacts of BEV and ICE vehicles via LCA and LCC. We use the battery electric vehicle sustainability impact assessment model (BEVSIM)[21] to carry out a full LCA and LCC and perform an MCI assessment of the two vehicle types. The BEVSIM tool is sector- and application-specific, web-based and has a simple design. The BEVSIM tool contains LCA models for materials production, processing, the use phase, EoL fate and recycling processes. The results of the LCA and MCI are presented for ranges of various parameters, including (1) recycled content (plastics), (2) electricity grid mix and (3) alternative EoL scenarios for plastics involving pyrolysis and mechanical recycling.

An LCC for a BEV can be carried out from the perspective of the owner (consumer) or producer of the vehicle. In this study, we considered the perspective of the producer, which aligns with the focus on material and vehicle production of the ALMA project (ALMA 2023), of which the BEVSIM is part. Data from the producer's perspective are considerably more scarce than those from the consumer's perspective. A report by Element Energy Limited (2021)[20] can be used to obtain data from the consumer's perspective.

Methods

Life cycle assessment (LCA) is a method for quantifying the environmental performance of products or services. The ISO 14040 and 14044 standards are provided by the International Organization for Standardization (ISO) to perform LCAs consistently with high data quality and reproducibility.

The aim of this study is to assess the effects of options, such as recycling, recycled content, and price volatility, on the environmental impacts of BEVs and ICEs. The service life of the studied vehicles is set at 248,500 km[18]. The default geographical location is Europe; therefore, the average European electricity from the grid is chosen as the energy supply for the use phase of the BEV. The studied system boundary is presented in Fig. 1.

An appropriate comparison of a BEV and an ICE requires the use of a consistent system boundary and lifecycle inventories, including all the relevant differences between the two alternatives, such as the powertrain, batteries, and chassis. The full bill of materials (BoM) for the two vehicle types can be found in the Supplementary Tables 1–6. The BoM is generated from the literature[22] and modified using expert opinion. The reader is referred to the literature for more information on the life cycle inventory[21]. For additional list of assumption and limitations See Supplementary Note 1.

Three environmental impact assessment methods are used in the BEVSIM: ReCiPe 2016 Midpoint (H), cumulative energy demand V1.11, and IPCC 2013 GWP 100a version 1.03 (also part of ReCiPe 2016). The material circularity index (MCI) developed by the Ellen MacArthur Foundation is used for circularity assessment[14].

To evaluate the impact of future options on the environment, the use of (1) recycled plastic content, (2) alternative electricity grid mixes for different years and (3) different EoL scenarios are studied (Table 1).

The LCC analysis is a cost-accounting approach that considers the economic costs over the life cycle of a service or a product. We consider the production process of a vehicle to perform a LCC. The LCC includes the costs of designing the vehicle, acquisition of the materials needed, and costs during use of the vehicle for which the producer is responsible, such as those made for take-back actions, and finally, waste-related costs. These costs incurred during different stages of the life cycle are partly based on literautre[23] and are balanced by the revenues gained by selling the vehicles to wholesalers or dealers.

The manufacturing stage in the LCC is related to the processing and production stage in the LCA, whereas the distribution stage encompasses transport of the vehicle to the seller. The EoL stage in both the LCC and LCA involves assessing the effects of disposing of production waste on the environment.

The LCC costs are divided into different lifecycle stages: Design, Manufacturing, Profit of the car Manufacturer, Distribution, Use, and End-of-life. Manufacturing costs are divided into materials, processes, labour, assembly, lighting, HVAC, and missing costs. Design costs are related to all the costs in designing a car model. To align with the LCA system boundary and lifecycle stages, material costs include the extraction, production, transport, and processing costs of the different materials sold in the market, such as plastics, metals, etc. The labour, assembly, lighting, and HVAC are cost components at car manufacturing facilities. For the manufacturing phase, cost data for the materials available in BEVSIM was consolidated by collecting costs data from various literature sources (Supplementary Tables 7–9).

Fig. 1 | System boundary. The system boundary used in this study.

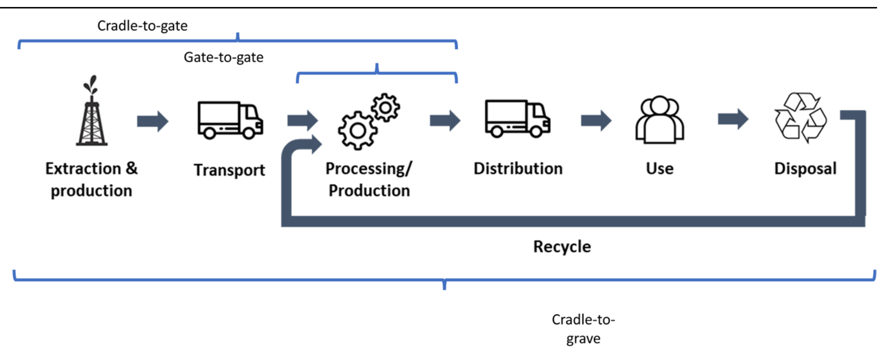

Table 1 | Future parameters assessed in the LCA study

Recycled content (%)	Electricity	EoL scenario for plastic waste (%)			
		Scenario	Mechanical recycling	Pyrolysis	Incineration
0	2021	3a	31	0	69
10	2030	3b	31	33	36
30	2050	3c	44	0	56
40		3 d	44	33	23

'Design' costs are incurred during the process of designing a new car model or redesigning an existing model. However, as no specific open source data on these types of costs are available, a rough estimate of 5% of the sales price was used on the basis of the LCC analysis[24,25]. Users of BEVSIM or other LCC tools can always use their own internal data to determine these design costs. During the manufacturing stage, costs are incurred associated with materials, energy, labour, processing, and equipment[23]. Freight-on-board and other market processes were considered to determine the material costs. For a steel BiW, processing costs were estimated to be 5.2 €/kg using 2021 price levels[18]. Distribution costs are incurred by transporting a stock to car dealers and damage during distribution. A distribution cost of €60 per vehicle was assumed. 'Use' costs do not include costs associated with the use of the vehicle by the consumer but include warrants, take-back and service costs incurred by the producer. No proper data are available for these costs as well; thus, the user should estimate these costs according to their specific LCC study. The costs associated with waste generation during the design and production of vehicle are determined during the EoL stage. The balance of costs is closed by the producer's revenues. Taxes were not included in this LCC.

It is important to consider the cost of a lithium-ion battery pack for a BEV. The price of this type of batteries was nearly 1200 $/kWh in 2010 but decreased to 137 $/kWh in 2020 because of technological development and an increase in production[26]. A further price decline is expected on the basis of further improvements in battery technology and a considerable increase in the cumulative production volume. This type of price development can be predicted by so-called learning curves[27]. Learning curves can be used to estimate future cost reduction considering the increased cumulative production capacity for a technology or a process using historical cost reduction data. Good-quality production quantities for the NMC 111 battery were not available, but time–price series were available[26,28]. Thus, we fit the series data with a power-law to derive a relationship between the time and battery price[29]. As the penetration rate of the batteries changed after 2014, the relationship was derived for 2014 onwards. The conversion from USD to EUR was based on the average yearly exchange rate. Given the inherent uncertainty in these predictions, the time horizon was limited to 2030. This study excludes the evaluation of novel or emerging battery technologies, such as post-lithium or all-solid-state batteries, as these technologies have achieved limited or no large-scale commercial production. Solid-state batteries (SSBs), lithium-sulfur batteries (LSBs), and lithium-air batteries (LABs) are currently produced at lab or pilot scale, which leads to challenges in accessing reliable cost data for LCC analysis. Excluding emerging battery technologies is a limitation of this study.

The steel cost is also critical for vehicles. Steel prices have fluctuated over the years and are expected to remain important in the future. The steel and plastic markets are mature markets for which learning curves would not be relevant because we did not consider potential changes in the steel market, such as the development of advanced high-strength steel and oxysteel combined with carbon capture. Changes in market-level supply and demand play a critical role in the economic cycle of these commodities. Therefore, future prices were based on the main cost drivers and the prediction of their future values. The main cost drivers for steel are raw materials (65%), CO_2 taxes (28%) and energy prices (17%). Relationships were established among the prices of historic steel, iron ore, coke and hard coal. The steel price from 2010 to 2020 was found to be positively correlated with the price of coal (correlation factor: 0.43) and slightly negatively correlated with the prices of coke (correlation factor: −0.053) and iron ore (correlation factor: −0.049).

Results

The cradle-to-grave GWP results for BEVs and ICEs show that the use phase constitutes the dominant contribution (at 85%) for ICEs to vehicle lifetime greenhouse emissions, whereas material production, especially battery manufacturing, is the largest contributor (at 38%) for BEVs to the life cycle GWP. Figure. 2 shows the cradle-to-grave GWP results for the BEV and ICE vehicles per life cycle stage per functional unit, i.e., one vehicle (BEV or ICE). For this (default) scenario (see the results presented in the first row of Table 1), it is assumed that the plastics used in the vehicle have no recycled content.

Temporal variations in electricity grid mixes are an important consideration for a full LCA of BEVs[30]. In this analysis, the impact of the future electricity grid mix is studied to assess the use-phase performance of BEVs in 2030 and 2050. The application of forecasts for future (2030 and 2050) electricity grid mixes reveal a considerable reduction in the CO_2 emissions of BEVs. The BEV emissions are 28%, 50%, and 56% lower for 2021, 2030 and 2050, respectively, than those of the ICE (Fig. 2). Generally, BEV LCA studies available in the literature consider average values of GHG emissions of certain regions, typically corresponding to the average of the total electricity mix[31]. However, this approach does not consider the impact of the actual increase in the demand and charging times in a given electricity network[32]. The impact of the extra energy consumed by BEVs on the CO_2 emissions from electricity generation is considered marginal emissions. The marginal emissions differ from the yearly average grid mix GHG emissions. Using average emissions values of the energy generation systems might lead to an underestimation of the environmental impacts of the BEVs in the present and future scenarios. Using the ReCiPe 2016 midpoint (H) method shows that the BEV out performs the ICE in terms of the GWP, stratospheric ozone depletion and fossil resource scarcity categories. This is shown in Fig. 3 where all the impact categories are normalized by the respective values for the ICE reference. As can be seen the ICE outperforms the BEV for all other impact categories, in line with past findings[5]. For all three impact categories in which ICE performs below the BEV, the use phase burdens are the dominant factor (Fig. 2). For BEVs, mineral resource scarcity is a crucial issue that can be mitigated through recycling or deploying innovative products with lower contributions to mineral resource scarcity. The major contributors to mineral resource scarcity are cobalt sulfate, lithium hydroxide, and copper, which can be recovered in the forms of Me(Ox), Li_3PO_4 and $Cu_3(PO4)_2$[33]. According to Duarto Castro et al.[33], overall recycling credits are obtained for terrestrial toxicity, human noncarcinogenic toxicity, toxicity, and mineral resource scarcity but not for the GWP. However, in the BEVSIM model, the net GWP for battery recycling is negative. This contradiction results from different acids being used in the waste battery treatment process modelled in the BEVSIM and in leaching. Acetic acid is reportedly used for the leaching process, whereas sulfuric acid is applied in BEVSIM, leading to a lower burden in waste treatment in terms of the GWP. According to the Ecoinvent database version 3.6, the GWPs of acetic acid and sulfuric acid are 1.6 and 0.12 kg CO_2 eq., respectively, although the use of recycled citric acid lowers emissions by up to four times[33]. Battery leaching is mostly performed using inorganic acids, typically hydrochloric acid, sulfuric acid, nitric acid, and phosphoric acid, whereas organic acids include

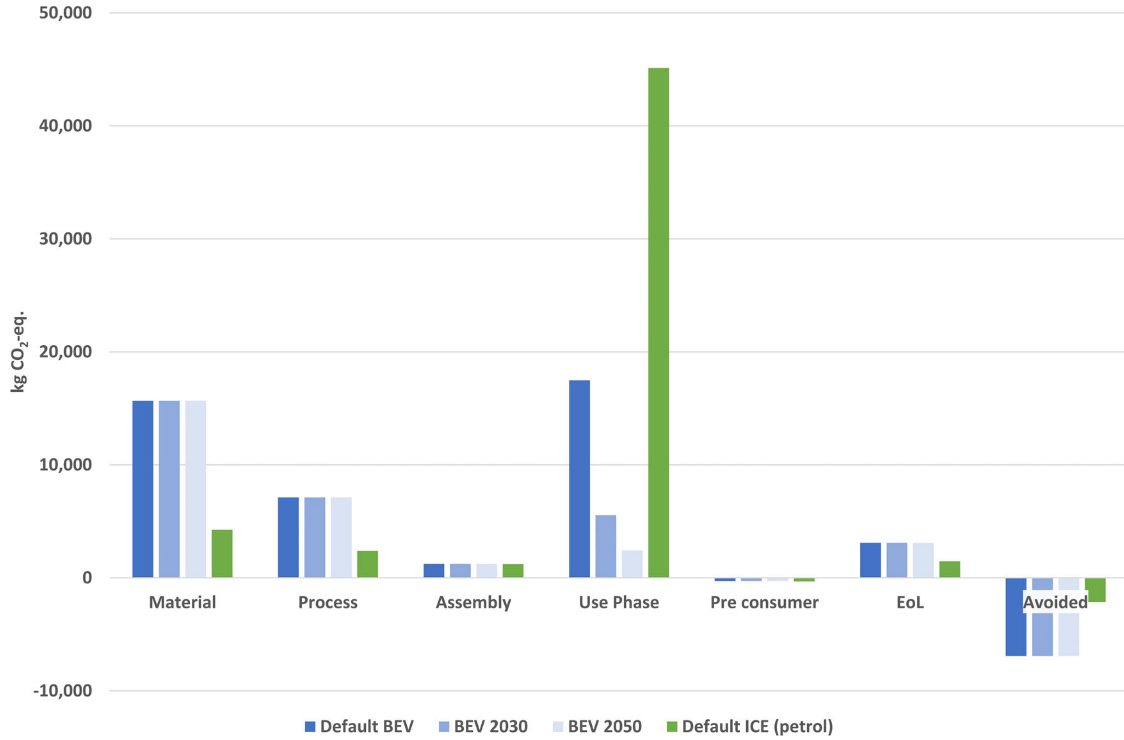

Fig. 2 | IPCC GWP 100. IPCC 2013 GWP 100a for the BEV default, 2030 and 2050 electricity scenarios, and ICE vehicle per life cycle stage. The figures show the GWP from cradle to grave. The functional unit is one passenger vehicle (BEV or ICE Class C and/or D) throughout its life cycle.

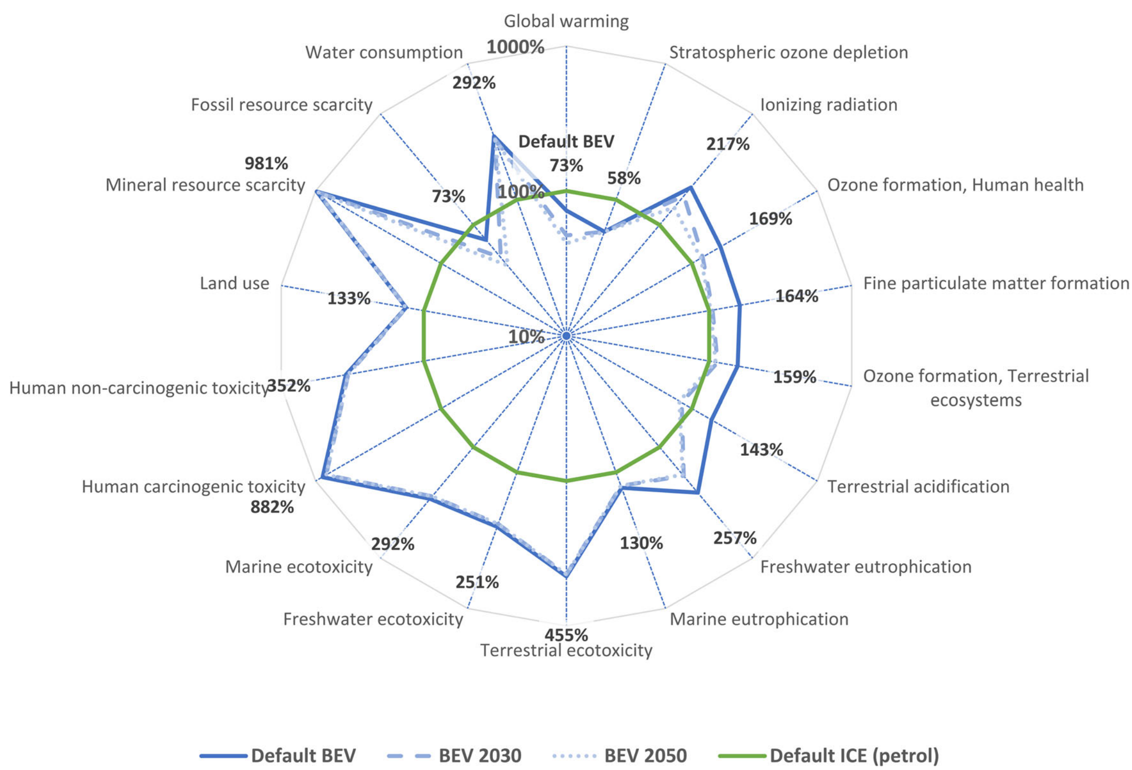

Fig. 3 | ReCiPe midpoint H method. Comparison of the default current BEV, future alternatives to the BEV and the current ICE vehicle using the ReCiPe midpoint H method. "Default" in the legend refers to the default scenario in which EoL treatment is a combination of mechanical recycling and incineration (3a in Table 1) with zero recycled content. A logarithmic scale is used and values per impact categories are normalized by dividing by the reference (ICE) values.

citric acid, oxalic acid and tartaric acid[21]. Thus, the assumed battery waste treatment plays an important role in the analysis and can influence the results considerably. The production of recycled batteries contributes considerably to the impact category of marine ecotoxicity.

The MCI for each material is 0.627. Steel and aluminium have high circularity indices (0.8 and 0.69) because of reuse and recycling. By contrast, plastics in the current EoL scenario (31% mechanical recycling and incineration for the remaining fraction) have an MCI score of only 0.33. This

Table 2 | Results obtained using the ReCiPe midpoint method to determine the effect of the recycled content of plastics on different impact categories

Recycled content %	Global warming potential	Stratospheric ozone depletion	Ionizing radiation	Ozone formation, human health	Fine particulate matter formation	Ozone formation, terrestrial ecosystems	Terrestrial acidification	Freshwater eutrophication	Marine eutrophication	Terrestrial ecotoxicity	Freshwater ecotoxicity	Marine Ecotoxicity	Human carcinogenic toxicity	Human noncarcinogenic toxicity	Land use	Mineral scarcity	Fossil scarcity	Water consumption
0	378.71	0.0011	9.60	0.63	0.23	0.64	0.77	0.015	0.0052	116.27	0.57	0.86	0.71	30.03	0.84	1.15	153.79	10.25
10	351.87	0.0010	8.86	0.58	0.22	0.59	0.72	0.014	0.0047	112.74	0.52	0.79	0.68	28.32	1.77	1.12	140.53	9.55
20	325.03	0.0009	8.11	0.53	0.21	0.54	0.67	0.013	0.0043	109.22	0.47	0.71	0.64	26.61	2.70	1.09	127.26	8.85
30	298.18	0.0008	7.37	0.48	0.19	0.50	0.62	0.012	0.0038	105.70	0.41	0.64	0.60	24.90	3.63	1.07	114.00	8.15
40	271.34	0.0007	6.62	0.44	0.18	0.45	0.57	0.012	0.0034	102.17	0.36	0.56	0.57	23.19	4.56	1.04	100.74	7.45

The numbers are the plastic production by the BEV corresponding to the recycled content used to manufacture the BEV.

result is alarming, given industry efforts to use less metal and more plastic and composite parts. Replacement of steel with composites must be coupled with a suitable infrastructure for the reuse and recycling of these components. Moreover, 31% recycling rate is difficult to achieve based on the recent report by Lase et al. in 2018 in automotive sector a recycling rate of around 10% was achieved which might increase in 2030 up to 50% if chemical recycling and mechanical recycling processes are optimized. That is when they serve as complementary technologies to treat waste streams that otherwise would be landfilled or incinerated.

Plastics in the automotive industry are reported to consist of 0–20% recycled materials[6,34]. Although this value is on the higher side based on the forecast done in a recent study, showing a maximum of 26% in 2030[35]. Changing the recycled content of the plastics from zero to 40% improves the GWP of the plastic parts by 28% overall (Table 2). The results in Table 2 provide valuable information on potential environmental savings that can be achieved with different recycled content for plastics, thereby supporting implications of policy decisions such as the new proposed End-of-Life Vehicles (ELV) Directive by the European Commission. The proposal for an ELV Regulation builds on and replaces two existing Directives: Directive 2000/53/EC on end-of-life vehicles and Directive 2005/64/EC. The proposed new regulation covers all aspects of a car, from its design and placement on the market until its final treatment at the end of its life, to ensure that at least 25% of plastic used for new car models placed on the marker comes from recycling, of which 25% is from recycled ELVs.

According to recent studies, increasing the recycling rate from 3% to 50% improves the levels of carcinogens, noncarcinogens, global warming and nonrenewable energy by 138%, 100%, 42% and 114%, respectively[7,36]. A 33% pyrolysis share was chosen for the EoL treatment because currently, PO can potentially be pyrolyzed, whereas PU and adhesives can only be incinerated, resulting in a maximum 44% share of mechanical recycling of plastics in EoL treatment[7]. Table 3 shows the results of 18 impact categories determined using ReCiPe Midpoint H for the four studied scenarios. A 57% reduction in the GWP (considering only emissions related to EoL treatment) is achievable by 33% pyrolysis and 44% mechanical recycling compared with state-of-the-art methods (31% mechanical recycling and 69% incineration). This value was reported to be 42% by Cardamone et al[7]. The difference between the two values mainly originates from different assumptions being made about the composition of plastics of the vehicles and the inclusion of dissolution-based recycling by Cardamone et al.[7]. Improvements were not observed in only three categories, namely, freshwater eutrophication, terrestrial ecotoxicity and land use. An overall comparison of all impact categories for all four scenarios (3a–d) did not yield conclusive results; instead, a trade-off among the scenarios was obtained.

First, it must be stressed that almost no public cost data for the production of passenger cars are available. Thus, the results shown here are only an indication of the actual costs. On average, car producers make a profit of approximately 7.5%. Data from the leading car manufacturers in the European market reveal an average margin of 6.8% over the period of 2021–2023 (see Table 4). The relatively high standard deviation indicates considerable variability in the margin. Of course, the margin for the car producer is directly reflected in the price that the car dealer pays, and the consumer price also includes the costs and margin of the car dealer. As we only knew the consumer prices, we subtracted the costs and margins of the car dealer to obtain the revenues for the car producer. As an estimate, we subtracted 5% of the consumer price based on literature analysis on dealer margin, inputs from auto industry expert and expert judgement of authors[37] (Supplementary Table 10). The manufacturer's suggested retail price was based on that of the Ford Mustang Mach-E (€60,119) for the BEV and on the Ford Active X 1.0 l EcoBoost (€34,800) for the ICE. Our initial results revealed a very large profit, showing that our costs were underestimated. To solve this issue, 'missing costs' were added to the LCC. These missing costs were 17–20% for the BEV and approximately 32% for the ICE. These missing costs for the ICE were quite uncertain but were needed to correctly balance the consumer price.

Table 3 | Results obtained using the ReCiPe midpoint method to determine the effects of EoL treatment of plastics on different impact categories

EoL scenario	Global warming potential	Stratospheric ozone depletion	Ionizing radiation	Ozone formation, human health	Fine particulate matter formation	Ozone formation, terrestrial ecosystems	Terrestrial acidification	Freshwater eutrophication	Marine eutrophication	Terrestrial ecotoxicity	Freshwater ecotoxicity	Marine Ecotoxicity	Human carcinogenic toxicity	Human noncarcinogenic toxicity	Land use	Mineral scarcity	Fossil scarcity	Water consumption
	E + 01 kg CO2	E − 04 kg CFC11	kBq Co60	E − 02 kg NOx	E − 03 kg PM2.5	E − 02 kg NOx	E − 02 kg SO2	E − 04 kg P	E − 04 kg N	E + 02 kg 1,4 DCB	E − 02 kg 1,4 DCB	E − 01 kg 1,4 DCB	E − 01 kg 1,4 DCB	kg 1,4 DCB	M2a crop	E-01 kg Cu	E + 01 kg oil	M³
3a	6.46	−1.94	−0.95	−5.61	3.76	−5.92	−4.44	2.36	−5.43	0.95	5.72	1.23	6.72	7.85	5.03	−4.79	−3.51	−2.69
3b	5.55	−1.83	−0.97	−6.22	1.87	−6.61	−5.24	6.00	−5.62	1.40	3.62	1.21	5.56	8.71	5.51	−4.76	−4.02	−2.56
3c	3.76	−2.57	−1.08	−7.30	9.28	−7.68	−4.13	7.27	−8.28	1.01	−3.52	0.46	5.79	7.67	7.17	−4.90	−3.91	−2.96
3 d	2.77	−2.48	−1.11	−7.97	7.57	−8.42	−4.92	11.1	−8.56	1.45	−2.32	0.41	4.61	8.52	7.72	−4.88	−4.43	−2.85

The numbers are the results for plastic treatment (including only emissions and the avoided burden for the EoL treatment of plastics) for the BEV corresponding to the recycled content used to manufacture the BEV.
The worst and best results for each impact category are shown in red and green, respectively.

For the BEV, the estimated material cost of €25,930 is the most important part of the manufacturing cost. For the ICE, the estimated process costs of €11,016 (see Fig. 4) is the most important part of the manufacturing cost, and the estimated material costs are €5,135. The NMC battery (€21,265) dominates the material costs for the BEV, followed by the electric engine (see Fig. 5). As battery costs have been dropping and will continue to drop, the development of the battery price was further analysed. A modified learning rate was derived to estimate the time range over which the battery price would drop. From 2014 onwards, the battery price steeply declines for some years and then decreases more gradually. Mauler et al.[26] reported (but did not analyse) this apparent change trend. Battery prices, but not production volumes, are available from 2010 onwards. Thus, the learning curve was modified to use the number of years after 2014 ([26,28]) instead of the production volumes. The following relation was thereby found:

$$Price_y = 156.08 \times Y^{-0.62} \tag{1}$$

$$Price_y = Price(EUR/kg)\mathit{in\ the\ year}\ y,\ Y = \mathit{number\ of\ years\ after}\ 2014$$

The battery price predicted using Formula [1] for the year 2030 was 20 €/kg (see Fig. 6). The battery price predicted using Formula [1] for the year 2030 was 20 €/kg (see Fig. 6). In this study, we assume on one particular battery type with no major change in power density which allows us to use the Euro/kg correlation instead of the Euro/kWh correlation. This assumption though simplifies our analysis is also a major limitation of the LCC performed in this work. Further, the cost elements in this LCC has been to aligned with the LCA LCI datasets used for materials and processes. Even with the forty percent drop in the NMC battery price from 34 EUR/kg in 2021 to 20 EUR/kg in 2030, the battery remains by far the most important cost item of the BEV in 2030. For the other materials, it was not possible to estimate the 2030 prices, and we had to assume that the other cost items would not change by 2030. In this case, the predicted total cost for BEVs will decrease by 20% in 2030; however, the total costs still exceed those of a comparable petrol ICE vehicle (see Fig. 4)[2].

The battery price predicted using Formula [1] for the year 2030 was 20 €/kg (see Fig. 6). In this study, we assume on one particular battery type with no major change in power density which allows us to use the Euro/kg correlation instead of the Euro/kWh correlation. This assumption though simplifies our analysis is also a major limitation of the LCC performed in this work. Further, the cost elements in the LCC have been aligned with the LCA LCI datasets used for materials and processes.

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Discussions

We used the battery electric vehicle sustainability impact assessment model (BEVSIM) to assess the effects of the recycled plastic content and plastic recycling (via pyrolysis and mechanical recycling) on the environmental performance of both BEVs and ICEs, as well as the type of the electricity grid mix. Cradle-to-grave GWP results for BEVs and ICEs show that the use phase is the dominant contributor (85%) to vehicle lifetime greenhouse emissions of ICEs and that material production (especially battery manufacturing) is the largest contributor (38%) to the life cycle GWP of BEVs.

Stallkamp et al.[38] assessed the economic and environmental impact of automotive plastic waste end-of-life (EoL) options such as energy recovery versus chemical recycling. The results showed that chemical recycling leads to a lower net climate change impact (0.57−0.64 kg CO2e/kg waste input). The authors did not discuss whether the effect of chemical recycling as an EoL option is tangible throughout the entire lifecycle. Moreover, the use of

Table 4 | Margins of Ford and other main European car suppliers over the period 2021–2023 and the corresponding average margin per supplier and total average margin with the standard deviation

Company	2021 (%)	2022 (%)	2023 (%)	Average (%)	Term in report	Reference
Ford	7.3	6.6	5.9	6.6	EBIT margin	34
VW group	6.2	5.7	5.6	5.8	Return on sales after tax	41
Stellantis NV	9.1	9.8	8.8	9.2	Adjusted operating income margin	22
Renault	2.8	5.5	7.9	5.4	Operating margin	23,24
Average	6.4	6.9	7.1	6.8 (± 1.8)		

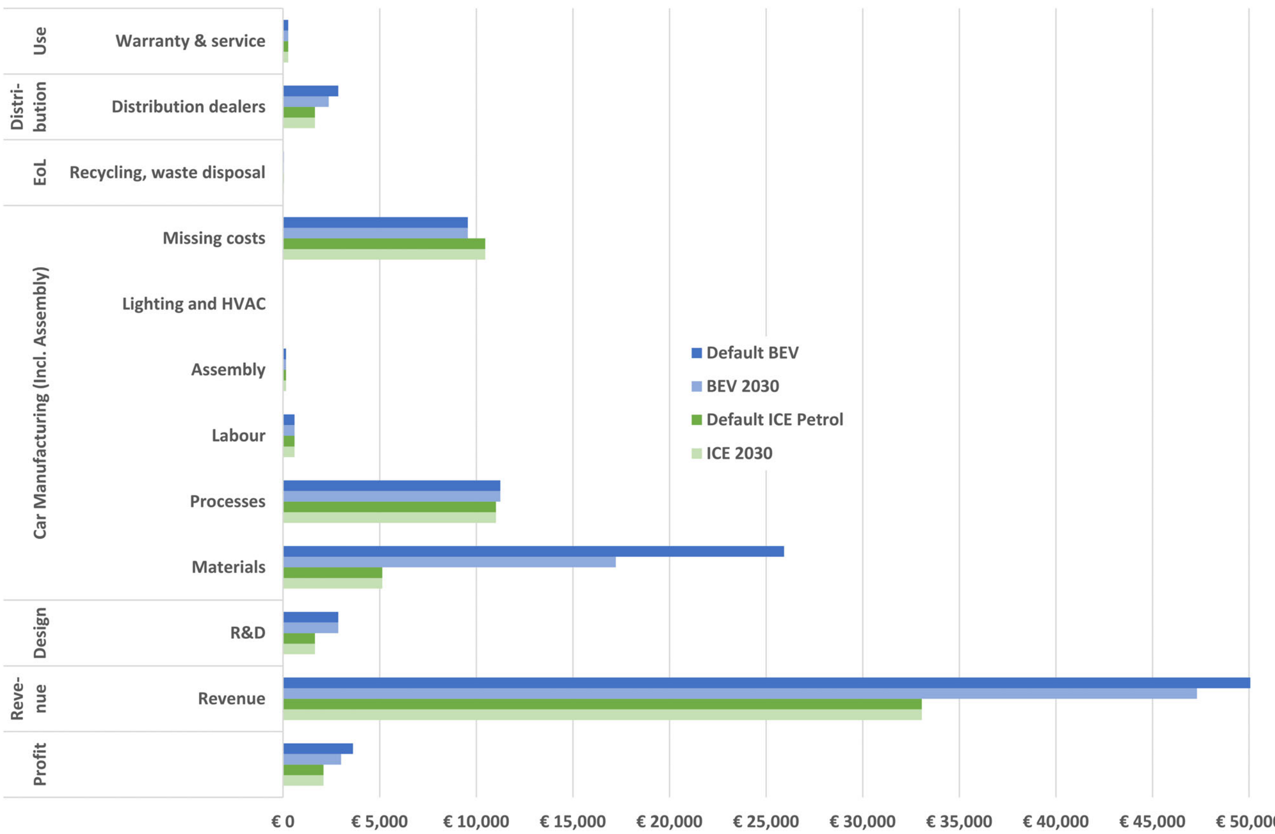

Fig. 4 | Estimated life cycle costs for the producer. Estimated life cycle costs for the producer in EUR for the current and 2030 BEV and the current and 2030 ICE vehicle with a petrol engine. The lighting & HVAC to distribution costs are below €100 and are not shown.

GWP exclusively is not representative, as is shown in the current work; there is a trade-off between different impact categories.

Using a combination of pyrolysis (33%) and mechanical recycling (44%) for plastic waste treatment improved all impact categories except land use, terrestrial ecotoxicity and freshwater eutrophication. An overall comparison of all impact categories for all 4 scenarios (3a–d) did not yield conclusive results; instead, a trade-off among the scenarios was obtained. Other studies[39,40] conducted LCAs for automobile recycling but did not include chemical recycling.

The effect of recycled content of plastics in automotive sectors is rarely studied. Liu et al. discussed the methodological choices on the inclusion of the recycled content of Steel, Magnesium and Aluminum[41]. In this work we apply the same approach to the recycled plastic content. Changing the recycled content of the plastics from zero to 40% improves the GWP of the plastic parts by 28% overall.

In addition to evaluating the environmental performance, the economic performance was assessed as a LCC of vehicle production. The NMC battery is and will be (as of 2030) the dominant contribution to the total life cycle costs for the producer. Even with considerably reduced battery costs in the near future, the production costs of a BEV are higher than those of an ICE vehicle with a petrol engine.

The performances of the BEV and ICE are comparable in terms of the circularity index (MCI = 0. 627). Steel and aluminium both have high material circularity indices (0.8 and 0.69) due to reuse and recycling, whereas plastics within the current EoL scenario (31% mechanical recycling and incineration as the remaining share) have an MCI of only 0.33. It is necessary to increase the recycling rates and recycled content of the plastic parts of vehicles. Automotive engineers must emphasize both lightweightness and circularity for batteries. Considering other impact categories, such as ecotoxicity-related categories, results in recycling having the dominant impact on environmental performance. The current automotive recycling infrastructure has limitations. In the recycling of BEVs and ICEs, there are specific infrastructure requirements for recycling vehicle parts and materials because of the complexities of BEV and ICE vehicles and their supply chains. Therefore, the recycling infrastructure must be redesigned and set up according to the vehicle type, i.e., BEV or ICE. In addition to industry efforts to develop recycling technology and establish recycling infrastructure, specific government legislation on automotive recycling for

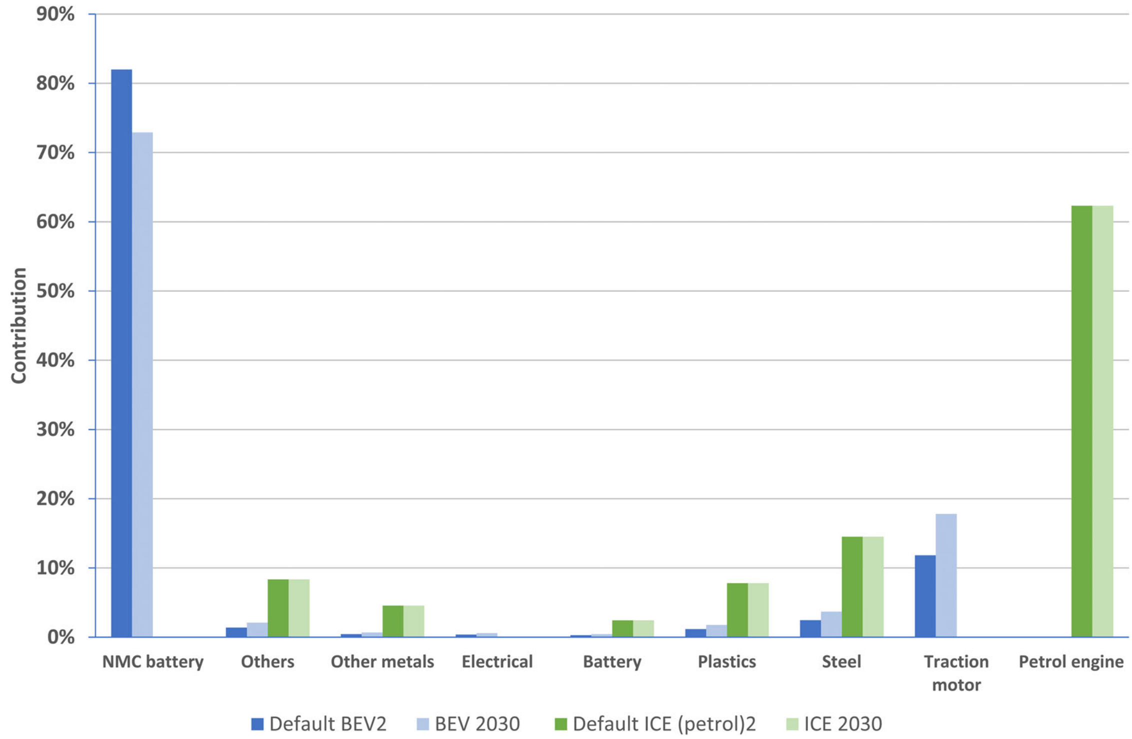

Fig. 5 | Material costs (% of total). The relative contributions of materials and specific parts to the total material costs of the BEV and ICE. The predicted NMC battery prices of 2030 are shown for an alternative BEV.

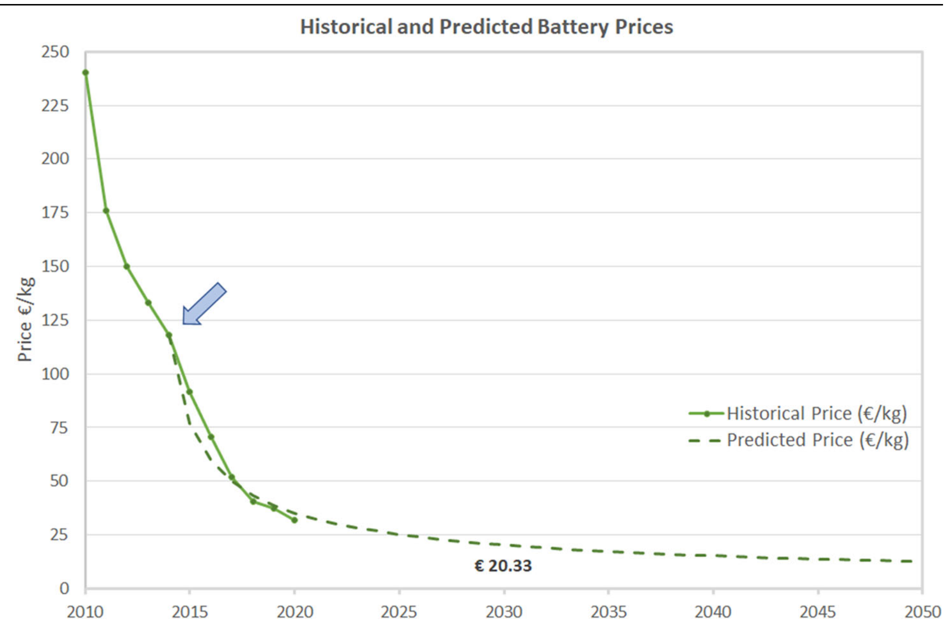

Fig. 6 | Historical and prediced battery prices. Historical and predicted electric vehicle global battery prices (based on published data[34,35]). The arrow shows the year 2014 at which a change in the trend occurs.

each category may be necessary to increase the circularity performance of the automotive sector.

Another consideration is battery cost. On the basis of results in the literature, we showed that battery costs will substantially decrease at the end of this decade. The overall conclusion of this study is that lightweighting of vehicles (both ICEs and BEVs) can be beneficial for environmental performance in combination with recycling technologies for plastics. The MCI could thus be lowered from its current value of 0.33.

The LCC clearly shows that from the producer's perspective, the reliability and availability of cost data are strongly limited, probably due to the highly competitive value of this data for producers. The price of BEVs will decrease because the price of the most important cost item, the battery, will continue to decrease in the near future. However, using lighter materials, such as composites and high-strength steel, in the future to increase the range of vehicles may increase material costs. A reduced vehicle mass translates into lower energy costs for the consumer, potentially reducing life cycle costs for the consumer.

The future electricity grid mix will impact the GWP of BEVs, providing insights into their long-term potential compared to ICEs. While plastics are not currently the dominant material in vehicles, their share is expected to grow due to lightweighting trends. With new EU directives, the use of recycled plastics and their end-of-life treatment will become key research areas. Additionally, LCC analysis can help assess the future costs and benefits of BEVs, supporting informed decision-making in the transition to sustainable mobility.

These findings underscore the need for manufacturers to invest in improved recycling infrastructure, policymakers to enforce higher recycled content mandates, and industry stakeholders to balance lightweighting strategies with recyclability to enhance sustainability across the automotive value chain.

Data availability

No datasets were generated or analysed during the current study.

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Author contributions
Milad Golkaram conducted the LCA Tom Ligthart carried out the LCC Aylin Yildiran and Spela Ferjan supported on the data collection for LCC and LCA, respectively. Rajesh Mehta supported the design of project, and supervised the project Jack J.T.W.E. Vogels and Eugene Someren designed the BEVSIM platform.

Competing interests
The authors declare no competing interests.

Additional information
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