

UC Berkeley

UC Berkeley Previously Published Works

Title

Cost, emissions, and material flow implications of designing electric vehicle components for recyclability

Permalink

<https://escholarship.org/uc/item/8wz5552h>

Journal

Environmental Research Infrastructure and Sustainability

ISSN

2634-4505

Authors

Huang, Tai-Yuan

Bose, Baishakhi

Gu, Grace

et al.

Publication Date

2026-09-11

DOI

10.1088/2634-4505/aea641

Copyright Information

This work is made available under the terms of a Creative Commons Attribution License, available at <https://creativecommons.org/licenses/by/4.0/>

Peer reviewed



PAPER • OPEN ACCESS

Cost, emissions, and material flow implications of designing electric vehicle components for recyclability

To cite this article: Tai-Yuan Huang *et al* 2026 *Environ. Res.: Infrastruct. Sustain.* **6** 045003

View the [article online](#) for updates and enhancements.

You may also like

- [Lightweight Composites for E-Mobility Applications](#)
Sarabjit Singh, Mahapara Abbass and Shalom Akhai
- [Recell: A Closed-Loop Battery Recycling Model](#)
Jeffrey Spangenberg, Linda Gaines and Qiang Dai
- [Lightweight Materials for Automobiles](#)
Nina Busarac, Dragan Adamovic, Nenad Grujovic et al.

ENVIRONMENTAL RESEARCH
INFRASTRUCTURE AND SUSTAINABILITY

PAPER

OPEN ACCESS

RECEIVED

18 May 2026

REVISED

25 August 2026

ACCEPTED FOR PUBLICATION

11 September 2026

PUBLISHED

14 October 2026

Original content from
this work may be used
under the terms of the
[Creative Commons
Attribution 4.0 licence](#).

Any further distribution
of this work must
maintain attribution to
the author(s) and the title
of the work, journal
citation and DOI.

Cost, emissions, and material flow implications of designing
electric vehicle components for recyclabilityTai-Yuan Huang^{1,2,3} , Baishakhi Bose³ , Grace X Gu⁴ , Brett A Helms^{2,5} , Jay D Keasling^{2,6,7}
and Corinne D Scown^{1,2,3,7,*} ¹ Energy & Biosciences Institute, University of California, Berkeley, CA, United States of America² Joint BioEnergy Institute, Emeryville, CA, United States of America³ Energy Analysis Division, Lawrence Berkeley National Laboratory, Berkeley, CA, United States of America⁴ Department of Mechanical Engineering, University of California, Berkeley, CA, United States of America⁵ Materials Sciences Division, Lawrence Berkeley National Laboratory, Berkeley, CA, United States of America⁶ Department of Bioengineering, University of California, Berkeley, CA, United States of America⁷ Biological Systems and Engineering Division, Lawrence Berkeley National Laboratory, Berkeley, CA, United States of America

* Author to whom any correspondence should be addressed.

E-mail: cdscown@lbl.gov**Keywords:** life-cycle assessment, techno-economic analysis, material flow analysis, polydiketoenamine, electric vehicle battery enclosure, polymer recyclingSupplementary material for this article is available [online](#)

Abstract

The electric vehicle (EV) industry is growing rapidly; however, compared to conventional automobiles, the increased weight of EVs contributes to their more limited range. To reduce total EV weight, higher-end vehicles use materials such as aluminum, which are more costly and energy-intensive than steel. This presents a challenge to the industry: there is a need to transition toward both lightweighting and material circularity. Improved materials are needed not only to reduce weight and improve efficiency but also to facilitate reclamation of metals and polymers from end-of-life vehicles. This study provides and demonstrates a framework for evaluating lightweight polymer alternatives for conventional EV components from a circularity perspective. The study identifies replacement materials for bumpers, door panels, interior consoles, and battery enclosures by evaluating material properties, environmental impacts, and economic metrics. Next, the study explores the battery enclosure as a deep-dive case study, establishing four scenarios: steel (S1), carbon fiber reinforced epoxy (S2), glass fiber reinforced epoxy (S3), and glass fiber reinforced polydiketoenamine (PDK) resin (S4). The life-cycle assessment and techno-economic analysis include recycling processes for the various scenarios. Results indicate that current materials in bumpers (polypropylene) and interior consoles (ABS) are functionally, economically, and environmentally favorable, while polymer composites such as fiber reinforced PDK can improve upon some economic and environmental metrics for door panels and battery enclosures relative to current materials. Results show that battery enclosure replacement demonstrates the highest potential for weight reduction and improved circularity without compromising durability. Specifically, the S1 and S4 scenarios achieve comparatively lower levelized recycling costs (LRCs). Between S1 and S4, use-phase CO₂e emissions are lower for S4 due to the lighter-weight PDK composite material. The viability of PDK composites depends on recycling practices (dismantling vs non-dismantling) and the final selling price of recycled PDK resin. Dismantling enables greater overall recovery of battery materials, achieving an LRC of \$93 per enclosure, whereas not dismantling increases the LRC to \$188. If recycled PDK can sell for prices equivalent to higher-cost virgin PDK, the LRC per enclosure can be -\$297 to -\$175. This study offers a framework for systematically evaluating EV lightweighting and circularity strategies using polymer composites.

1. Introduction

The global demand for electric vehicles (EVs) is rising rapidly, with a 35% year-on-year increase in 2023. In the U.S., new EV registrations now account for 10% of total vehicle sales (IEA 2025). To meet consumer expectations for longer ranges and higher performance, manufacturers are increasingly focusing on lightweight vehicle designs (Gotthold *et al* 2019). Because EV battery systems are significantly heavier than internal combustion engines (ICEs), lightweight materials are crucial to offset this added mass. For example, the Cadillac LYRIQ's battery accounts for 24% of its total weight (Ritchie 2023). The Nissan Leaf Plus EV weighs 1749 kg and the similar car model Nissan Versa ICE version weighs 1202 kg (Shaffer *et al* 2021). See supporting information (SI) section S1 for details. Relative to ICEs, EVs are typically heavier and capable of faster acceleration, which increases electricity consumption and associated greenhouse gas (GHG) emissions, and also raises non-exhaust emissions such as particulate matter from tire wear, and from brake wear where regenerative braking does not fully substitute for friction braking (Liu *et al* 2021). Lightweighting EVs therefore improves energy efficiency, extends driving range, and reduces tire wear and braking requirements (Zhang and Xu 2022, Choi *et al* 2025). However, materials commonly used for vehicle lightweighting can also impact the potential for recovery and recycling, raising the question of whether these objectives are at odds or if there are opportunities to achieve improvements in both lightweighting and circularity.

Improving material circularity is another key objective for automakers. When end-of-life (EoL) vehicles are shredded, approximately 75%–84% of the resulting auto parts are recyclable metals that can be directly recovered, while around 16%–25% becomes automotive shredder residue (ASR) and is typically landfilled (Okon Recycling 2025). EVs tend to yield a larger ASR fraction than ICEs, due in part to their high plastics and composite content (Clarkson *et al* 2024). According to the U.S. Environmental Protection Agency, about 4.5 to 5 million tons of ASR are disposed of in landfills each year (Tsuji *et al* 2006, Auto Recycling World 2021). Of this residue, 25% consists of heavy materials such as metal fines and glass, while the remaining 75% is made up of lighter materials including plastics, textiles, rubber, and foam (Vijayan *et al* 2022). Removing bulky auto parts such as bumpers, cushions, dashboards, and windows before shredding can significantly reduce the amount of ASR generated (Buekens and Zhou 2014, Auto Recycling World 2021). If EV components could be separated prior to shredding or crushing, the purity of recycled materials could be improved and environmental impacts reduced; however, the main challenge is the cost associated with dismantling and treatment. Several studies also recommend dismantling and reclaiming valuable auto parts to enhance material circularity (U.S. Environmental Protection Agency 2017). Determining effective strategies for dismantling and recycling lightweight materials will be an important consideration for the next generation of EVs.

Modern vehicles are constructed from a variety of materials, including metals and alloys such as aluminum and steel, as well as plastics, composites, and carbon fiber/graphite. Next-generation lightweight materials for automobiles can be broadly categorized into light alloys (e.g. magnesium and aluminum alloys), high-strength steel, composites (such as synthetic polymers and natural fibers), and other advanced materials (Zhang and Xu 2022). As automotive technologies and new materials have improved, there has been a trend toward materials that reduce overall EV mass without compromising safety. In 2023, steel accounted for 40.0% of the average mid-size EV's total weight, followed by aluminum (14.4%), plastics and composites (10.0%), and synthetic rubber/elastomers (10.0%). From 2014 to 2023, the material composition of mid-size vehicles has gradually shifted toward lightweight materials. Steel usage has decreased from 51.5% to 49.4%, while the use of lightweight aluminum and polymer composites has increased from 9% to 11.4%. Similarly, the application of plastics and composites has grown from 360 lb/vehicle (8.9%) in 2014 to 426 lb/vehicle (9.6%) in 2023, representing an average annual growth of 2% in plastics use (Rose-Glowacki *et al* 2024). In terms of weight reduction potential, use of magnesium alloys can achieve a component-level 30%–70% mass reduction, high-strength steel can reduce mass by 10%–28% compared to their traditional counterparts, and carbon fiber composites like carbon fiber reinforced plastics offer a 50%–70% reduction (Gibbs *et al* 2013). Besides metals and alloys, plastics and composites contribute to vehicle lightweighting, also have significant potential for recycling, which can further promote material circularity and reduce environmental impacts.

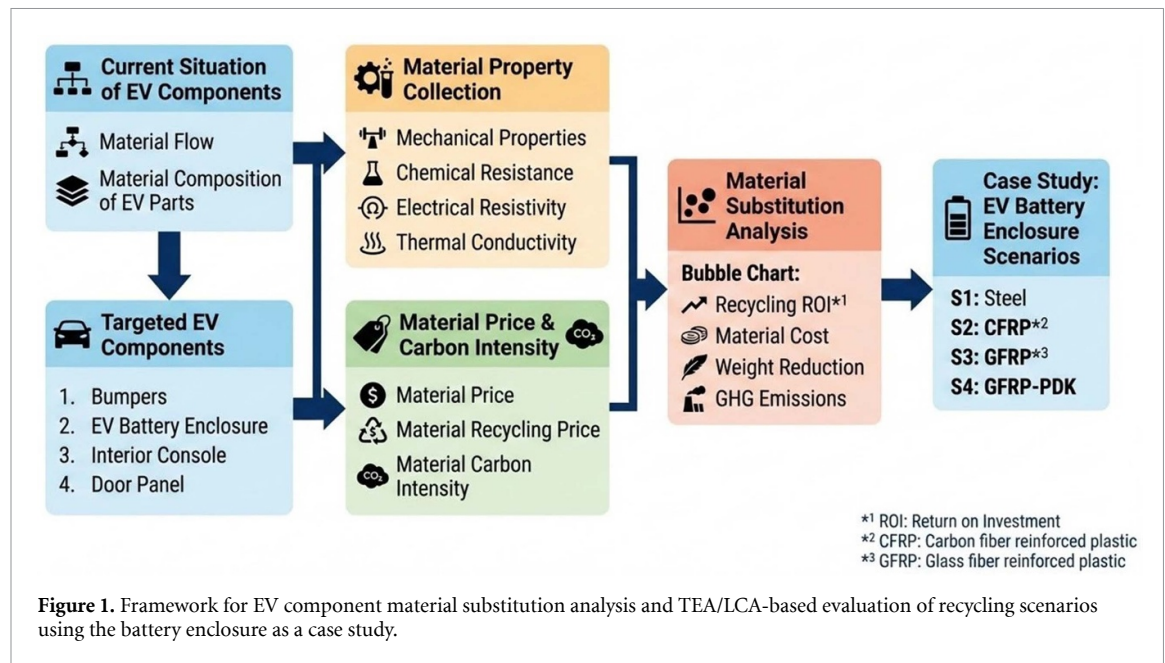
Polymers such as epoxy resin, when combined with organic, inorganic, or metallic fibers, are among the most common polymer composites used in EVs, accounting for about 65% of total polymer use on both the interior and exterior of vehicles. These materials offer multiple advantages, including low weight, high strength, durability, safety factor, corrosion resistance, and lower carbon emissions (Zhang and Xu 2022). Increasingly, EV manufacturers are incorporating polymer composites with natural or synthetic fibers embedded within the plastic matrix. For example, bamboo fibers have been added to

polyurethane (PU) composite door panels to increase wall thickness and improve sound absorption without substantial increase in the weight of material used (Rajak *et al* 2019).

Despite their benefits, recycling polymer composites is both technically and financially challenging. While many pure polymers are relatively easy to recycle, polymer composites are difficult to reclaim due to the presence of multiple, tightly bound material phases (Schade *et al* 2024). The three main recycling methods for polymer composites are mechanical recycling, thermal recycling, and chemical recycling, each with its own limitations and technical barriers (Pierri *et al* 2024, Versteeg *et al* 2026). Mechanical recycling is the simplest and most mature method using shredding and grinding techniques to recover both polymers and fibers. The reclaimed fibers are significantly degraded with shortened lengths and remain in the polymer matrix. The most common thermal recycling includes pyrolysis and fluidized-bed gasification. Carbon and metallic fibers can be partially recovered with some loss of mechanical properties, whereas other fibers such as fiberglass are completely degraded (Kroisová *et al* 2023) after thermal treatment. Chemical recycling can be ideal for retaining complete polymers (or constituent monomers), fibers, and other additives, but these processes can also be expensive and solvent-intensive (Cousins *et al* 2019, Almushaikh *et al* 2023, Maury *et al* 2023). The cost of chemical recycling can reach \$300 to \$1 000 per ton of plastics, compared to only \$50–\$200 per ton for mechanical recycling (Herrmann and Flock 2023). The high cost and technical limitations of chemical recycling, along with the fact that not all conventional plastics are compatible with these processes, underscore the need for new approaches to material design and EoL management (Thomas *et al* 2024). To address these limitations, researchers are developing new classes of polymers designed for easier, more cost-effective, and more efficient recycling.

Beyond conventional petroleum-derived polymer composites, bioplastics such as bio-polypropylene, and bio-nylon are emerging as a promising alternative that can significantly reduce carbon emissions. Bioplastics can be produced from renewable biomass sources such as corn, sugarcane, and other agricultural waste. Some bioplastics are even biodegradable under specific composting conditions, allowing for EoL options that do not compromise their performance in EV applications but enable easier degradation without harming the environment. Some polymers can also be recycled chemically through processes such as depolymerization, which enables the recovery of fibers and flame retardants from composite materials. For example, polydiketoenamines (PDKs) are promising bio-thermoplastic resins that offer circularity advantages. Although the production of PDK remains more costly than other more conventional polymers, it can be chemically recycled back into its monomers with minimal material losses (approximately 5%) and competitive costs, allowing the polymer to be remanufactured into virgin-quality PDK (Christensen *et al* 2019). Importantly, this process enables nearly 100% recovery of both fibers and flame retardants embedded in the resin, facilitating a closed-loop recycling system. After the initial recycling, the cost and carbon intensity of PDK production can be substantially reduced, as no new raw materials are required (Vora *et al* 2021, Demarteau *et al* 2022b, 2023). However, there is currently no systematic framework for matching new materials with candidate automotive components for replacement, nor is there a method by which the economic and environmental tradeoffs can be weighed.

This study aims to identify the suitability of different EV components for replacement with materials that contribute to lightweighting and/or circularity. In addition to a vehicle-wide analysis, we explore EV battery enclosures and the potential for fiber-reinforced PDK replacements as an illustrative case study. The reason is that battery enclosures account for a significant proportion of total vehicle weight due to their large volume and mass, making them an ideal target for lightweighting strategies. In addition to reducing weight, battery enclosures must meet stringent requirements for electrical insulation, mechanical strength, and fire resistance to ensure the safety of battery systems. Traditionally, steel and aluminum have been the primary materials used for battery housings (Bonnen Battery 2025). However, recent trends show a gradual shift toward polymer composites such as carbon fiber-reinforced polymer (CFRP) and glass fiber-reinforced polymer (GFRP), which offer advantages in weight reduction, high strength, and fire resistance (Edah *et al* 2025). Despite these benefits, recycling polymer composites remains a significant challenge, as carbon fiber and fiberglass are difficult to reclaim through processes such as pyrolysis and mechanical recycling, as discussed previously. Most EV literature has focused on the battery itself, with only a few studies addressing materials for battery enclosures (Li *et al* 2024, De Sio *et al* 2025). To evaluate both the economic and environmental implications of material substitution, this study employs techno-economic analysis (TEA) and life-cycle assessment (LCA). By incorporating state-of-the-art circular polymers, this study evaluates the effectiveness of these materials in reducing GHG emissions and enhancing circularity. Our aim is to provide insights that enable the EV industry to reduce vehicle weight and improve material circularity.



2. Method

The methodology comprises two parts. First, we conduct a material substitution analysis for selected EV components. Second, EV battery enclosure is used as a case study to further screen and compare potential recycling scenarios. This case-study evaluation integrates economic feasibility and sustainability considerations using TEA and life-cycle GHG inventories (see figure 1).

2.1. EV part material substitution analysis

2.1.1. Identification of EV components and current status

This study first conducts a material flow analysis (MFA) to characterize material use across EV components, spanning exterior parts (e.g. bumpers and tail lights) and interior parts (e.g. air ducts, seats, and seat belts). The MFA quantifies both (i) the types of materials used and their distribution within the vehicle and (ii) the mass of each material associated with major vehicle sections. For analysis, components are grouped into exterior, interior, under-the-hood, battery system, chassis & powertrain, electrical system, thermal system, tires & suspension, and miscellaneous categories. Because each category has distinct functional requirements, materials are selected to satisfy different property targets. For example, bumpers are commonly manufactured from polypropylene (PP) with steel or aluminum reinforcement to meet strength and energy-absorption requirements. Battery enclosures, in contrast, must meet structural and safety requirements and provide electrical isolation and shielding; steel and aluminum are widely used due to their mechanical performance and manufacturability. The MFA dataset is compiled from published literature on EV material composition and related environmental and economic analyses (Hawkins *et al* 2012, Sato *et al* 2018, Singh *et al* 2018, Moore *et al* 2024, Rose-Glowacki *et al* 2024).

After establishing the material distribution across EV parts, we screen components for material-substitution potential using three criteria: component weight, structural complexity, and dismantling difficulty. Component weight and complexity serve as proxies for (i) the magnitude of material demand and (ii) the quantity of recoverable material available at EoL. Dismantling difficulty reflects the practical and economic barriers to recovery, including labor requirements and dismantling costs, which ultimately influence whether recycling is feasible.

2.1.2. Hard constraints on material selection for EV parts

There are several existing and emerging materials suitable for use in EV components. Traditionally, the automotive industry has relied on metals such as conventional steel, high-strength steel, aluminum, magnesium alloys, and copper, as well as polymers including rubber, PP, nylon (PA), acrylonitrile butadiene styrene (ABS), and other plastics (table 1) (MatWeb 2021, MakeltForm 2026). Data sources and processing steps are summarized in the SI section S2. This study evaluates the potential for substituting conventional materials in selected EV parts with advanced polymers and bioplastics, including PEEK (polyether

Table 1. Properties of Potential Materials for EV Parts.

Material	Model	Density (g cm ⁻³)	Young's Modulus (GPa)	Thermal Conductivity (W/m·K)	Flammability *	Electricity Resistivity (MΩ·m)	Chemical Resistance (over A-C ratio)	Recyclability / Biodegradable	Material/ Scrap Price (\$USD/kg)	Material CI ** (kg CO ₂ e/kg)
Steel	SM400	7.87	188	50.2	non-flammable	2×10^{-14} to 3×10^{-13}	Good	Excellent	0.5-0.8 / 0.396	1.77
High strength steel	HT690	7.87	590	19–52	non-flammable	2×10^{-14} to 3×10^{-13}	Good	Excellent	5.35 / 0.68	2.5
Aluminum	A1050	2.7	125	205	non-flammable	2.83×10^{-14}	Fair	Excellent	2.5 / 1.5	10.26
Magnesium alloy	AM60	1.8	130	156	combustible	1.15×10^{-13}	Fair	Excellent	2.41 / 0.5	52.39
Copper	1/8 hard	8.96	70	358	non-flammable	1.72×10^{-14}	Fair	Excellent	10.9 / 7.1	4.1
Polymer Material										
Rubber	Natural rubber	0.95	1.5	0.19	V-0	10^7	Fair	Limited	171 / 0.26	13
CFRP (epoxy resin)	ZC-100	1.6	150	0.23	HB/V-0	10^{-4} to 10^5	Excellent	Limited	33.1 / 0.0	20
GFRP (epoxy resin)	G10/FR4	1.85	17	0.04	HB/V-0	10^6	Excellent	Limited	75 / 0.0	4.07
Polypropylene (PP)	Homopolymer	0.95	1.26	0.15	HB/V-0	10^{-5} to 10^{10}	Excellent	Limited	0.82 / 0.005	1.9
Polyurethane (PU)	PU Foam	1.25–1.66	52–79	0.02	HB	10^2 to 10^7	Good	Fair	1.53 / 0.12	5.26
Nylon (PA)	Nylon 66	1.14	3.6	0.24	HB/V-2	10^3 to 10^7	Good	Fair	2.95 / 0.80	6.3
ABS	High impact	0.9–1.53	1.8–3.2	0.13-0.19	HB/5VA	1.5×10^{-5} to 10^{10}	Good	Fair	1.9 / 0.32	3.2
Polycarbonate (PC)	Unfilled PC	1.2	2.38	0.16-2.0	HB/5VA	1.02×10^{-4} to 10^{10}	Good	Fair	3.15 / 1.8	4.2
PVC	PVC-P	1.4	3.24	0.17	HB/5VA	2×10^{-2} to 10^7	Excellent	Fair	0.83 / 0.0	2.5
HDPE	high-density	0.92–0.995	1.04	0.49	HB	10^{-3} to 10^9	Excellent	Good	0.95 / 0.28	1.93
LDPE	low-density	0.927	0.3	0.32	HB	10^7 to 10^8	Poor	Fair	1.1 / 0.004	1.75
Phenol-formaldehyde (PF)	–	1.4–2.1	4.1–8.6	0.94	HB/V-0	10^3 to 10^5	Excellent	Limited	1.6 / 0.0	0.072

(Continued.)

Table 1. (Continued.)

Engineering Polymers										
PEEK	Polyether ether ketone	1.32	3.38	0.17-0.85	V-1/V-0	4.9×10^8	Good	Excellent	39.7-70.5 / 0.003	13.7
PPS	Polyphenylene sulfide	1.18–2.26	4.0	0.25-13	HB/V-0	10^6 to 10^9	Excellent	Fair	4.5-6.8 / 0.47	5.43
PET/PBT	unreinforced	0.07-2.5	1.57–5.2	0.19-0.29	HB/5VB	5×10^{-2} to 10^{10}	Excellent	Good	1.54 / 0.026	2.7
TPU	Thermoplastic PU	1.2	0.029	0.19	V-2	10^3 to 10^7	Limited	Fair	0.81 / 0.003	0.072
Bio-based Polymers										
PDK	–	1.65	1.2–2.0	0.1-0.3	N/A	N/A	Good	Excellent	45 / 1.5	86 / 1.6
Bio-PA	Nylon-66	1.14	1.5–3.7	0.24	HB/V-2	10^3 to 10^7	Good	Limited /non-biodegradable	6.49-19.5 / 0.8	1.3
Bio-PE	–	0.94–0.97	1.13	0.49	HB	10^{-3} to 10^9	Good	Fair /non-biodegradable	3.25-3.79 / 0.28	-0.75
Bio-PP	–	0.95	1.5–2.0	0.15-0.24	HB/V-0	10^{-5} to 10^{10}	Excellent	Limited	3.79 / 0.005	0.9
Bio-PET	–	1.38	1.57–5.2	0.19-0.29	HB/5VB	5×10^{-2} to 10^{10}	Excellent	Good /low biodegradable	2.44 / 0.026	2.1-3.0

* Flammability rating systems assess a material's ignition resistance, burning rate, and flame spread. This table reports the UL 94 ratings for plastics: 5VA/5VB are the highest (no dripping; hole formation allowed). V-0 stops burning within 10 s, V-1 within 30 s (with dripping allowed), and V-2 within 30 s (flaming drips allowed). HB indicates slow horizontal burning.

** CI: carbon intensity.

ether ketone), bio-based PA, bio-based PP, bio-based PET, and PBAT (polybutylene adipate terephthalate). Information on dismantling difficulty and time requirements is taken from car service manuals, which provide a qualitative ranking of dismantling difficulty as well as labor-hour estimates (CHARM 2014, LEMON 2026).

Given the comparatively limited material restrictions in the U.S. automotive industry (relative to, for example, the European Union), we assume only that any candidate substitute must provide comparable or improved performance relative to the incumbent material for the intended application. We screen candidate materials using (i) technical performance attributes and (ii) environmental and economic metrics. Technical screening includes key mechanical properties (tensile strength and Young's modulus), electrical behavior (conductive vs. insulating performance, depending on part function), chemical resistance, flammability/safety considerations, and recyclability. Mechanically, tensile strength is used as a basic feasibility check (materials with clearly insufficient strength or brittle behavior for the intended loading are excluded), while Young's modulus is used to reflect stiffness-driven design: if a substitute has a lower modulus than the original, additional thickness (and therefore mass) may be required to achieve comparable stiffness. For safety, materials with unfavorable flammability characteristics are flagged and, where applicable, we account for the need for flame-retardant formulations (e.g. additives) to meet enclosure or interior safety expectations. Chemical resistance is evaluated based on compatibility with automotive-relevant exposures (e.g. moisture, road salts, cleaners, oils/coolants, and—where relevant—battery-related chemicals), since degradation, swelling, or stress cracking can limit durability and service life. Electrical properties are evaluated to ensure that electrically conductive parts (e.g. conductors and grounding components) retain adequate conductivity, while parts intended to provide insulation or isolation maintain sufficiently high electrical resistivity and dielectric performance.

Recyclability classifications follow Hendrickson *et al* (2024), which groups materials into four recyclability categories. For each candidate material, we also compiled literature values for cradle-to-gate GHG emissions and unit prices to support trade-off analysis between cost and environmental performance. Please see SI figure S4 for reference. Together, these criteria define the feasible substitution set used in the material substitution framework and the downstream TEA/LCA scenario modeling.

2.1.3. Screening-level economic, emissions, and lightweighting decision metrics

In addition to the 'hard' screening criteria, i.e. whether a candidate material meets the chemical and mechanical property requirements for a given automotive component, this study evaluates potential substitutions using sustainability-oriented metrics, including economic impact, life-cycle GHG emissions, and lightweighting potential. Material substitution affects component mass because candidate materials differ in density and mechanical properties. To estimate the mass of a component made from a new material while maintaining equivalent mechanical performance to the current material, we apply the Ashby material selection approach (equation (1)). Components are assigned a loading index n based on the dominant design constraint: $n = 1$ for strength-limited members (tension/compression), e.g. interior console; $n = 2$ for stiffness-limited plates, e.g. door panels; and $n = 3$ for stiffness-limited panels/beams, e.g. bumpers and battery enclosures. Considering the component mass contribution, structural/composite complexity, and ease of dismantling for recycling, we focus on four components with high substitution potential: the EV battery enclosure, bumpers, door panels, and interior consoles.

$$m_{new,i} = m_{current,i} \times \left(\frac{\rho_{new,i}}{\rho_{current,i}} \right) \times \left(\frac{E_{current,i}}{E_{new,i}} \right)^{\frac{1}{n}} \quad (1)$$

where $m_{new,i}$ and $m_{current,i}$ are the estimated masses of the component i using the candidate material and current material, respectively. ρ is material density, and E_{new} and $E_{current}$ represent as individual Young's modulus. n is the loading index defined in the previous paragraph. The lightweighting benefit is then calculated as the relative mass change in equation (2),

$$\Delta m_i = \frac{m_{new,i} - m_{current,i}}{m_{current,i}} \quad (2)$$

where Δm represents weight difference percentage, $m_{new,i}$ & $m_{current,i}$ denote the weights of potential new and current materials for auto part i .

To capture the economic and environmental implications in a form suitable for preliminary screening, we use simplified industry-relevant parameters. Economic performance is represented by (i) material cost and (ii) an adjusted recycling return on investment (R-ROI) (equation (3)):

$$R - ROI = W_{recycled,j} \times M_{recycled,j} \times Q_{i,j} - T_{dismantle,i} \times Cost_{labor} \quad (3)$$

where $W_{\text{recycled},j}$ is the mass of recycled material j , $M_{\text{recycled},j}$ is the recycled material j unit value, $Q_{i,j}$ is the quantity of material j recovered from component i , and $T_{\text{dismantle}}$ and $\text{Cost}_{\text{labor}}$ represent dismantling time of component i and labor cost, respectively. GHG emissions are estimated by multiplying the component mass by the material-specific emission factor (kg CO₂e per kg material).

2.2. Deep-dive analysis on EV battery enclosures

After careful evaluation of candidate materials for individual automotive components, we identified the EV battery enclosure as a promising target for improving economic performance, reducing GHG emissions, and achieving weight reduction through material substitution. Accordingly, this study further evaluates recycling pathways for EV battery enclosures as a dedicated case study. Current battery enclosures are commonly manufactured from steel to provide robust protection for the battery pack; however, several lightweight alternatives are available, including CFRPs, GFRPs, and other polymer-based materials. Based on this screening, we compare four enclosure material options: steel, CFRP, GFRP, and glass fiber reinforced PDK resin. The candidate enclosure materials also differ in their level of automotive performance validation. Steel and fiber-reinforced epoxy composites have established structural applications, whereas PDK remains an emerging matrix material requiring further validation of crashworthiness, thermal and fire performance, fatigue, and long-term durability; a qualitative comparison of key performance requirements is provided in table S1 in SI.

2.2.1. Bill of materials of EV battery enclosures

EV battery pack design strongly influences enclosure size and material selection. We use the Argonne Battery Performance and Cost (BatPaC) model to define the battery pack and enclosure design (Argonne National Laboratory 2025). In BatPaC, steel and CFRP are provided as default enclosure options, reflecting cost- and mass-focused design choices, respectively. In this study, the masses of the steel and CFRP enclosures are taken directly from the BatPaC outputs (Knehr *et al* 2022). For the GFRP and GFRP–PDK enclosures, BatPaC does not provide default mass estimates; therefore, we estimate enclosure mass using an Ashby material selection approach based on Young's modulus and density (i.e. stiffness- and density-driven scaling). We assume the composite formulation comprises 50% fiber reinforcement (carbon fiber or fiberglass), 20% flame retardant, and the remaining ~30% polymer matrix (epoxy resin or PDK resin). We based these composite formulation values on proprietary industry specifications for EV battery enclosure designs, although the composition is similar to published values (Höhne *et al* 2023, De Sio *et al* 2025, Cahill *et al* 2026). These bill-of-materials fractions are then used to simulate material usage across manufacturing, use, and EoL (recycling) phases.

2.2.2. Process modeling for recycling scenarios

Using the bill of materials generated by the BatPaC model and Ashby method, we evaluate the effectiveness of substituting EV battery-housing materials by simulating alternative EoL recycling pathways. This section presents battery enclosure recycling scenarios used to validate the material-substitution framework. We consider four enclosure material systems, each with two EoL handling options: (A) dismantling of the enclosure prior to recycling, and (B) no dismantling (i.e. the enclosure is processed without prior separation). The enclosure material scenarios include: S1) steel enclosure (A/B), S2) CFRP enclosure (A/B) with epoxy resin, S3) GFRP enclosure (A/B) with epoxy resin, and S4) GFRP–PDK composite enclosure (A/B), where PDK is used as the polymer matrix resin. Material and energy flows for each recycling pathway are simulated in SuperPro Designer v.14. Practical implementation of these recycling pathways would require integration with existing EV collection and battery-pack removal workflows. Material labeling or digital product passports could support rapid identification and sorting of PDK-containing enclosures at dismantling facilities, reducing the risk of misrouting to conventional shredding streams. Because enclosure dismantling adds labor and processing time relative to bulk shredding, appropriate incentives may be needed to support additional dismantling and handling to become economically viable, such as material take-back credits or recycler payments tied to recovered resin value. Unlike conventional thermoset composites, PDK resin can be chemically separated from co-mingled plastics and contaminants through depolymerization, reducing the need for stringent feedstock purity prior to processing; however, some degree of contamination control (e.g. removal of metal fasteners or inorganic debris) would still be beneficial to protect downstream equipment and ensure efficient reagent use. These considerations represent practical requirements for implementing the modeled recycling pathways. Detailed process descriptions, assumptions, and unit-operation settings for each scenario are provided in the SI section S4. Table 2 summarizes the scenario definitions and the recycling technologies applied in each case.

Table 2. EV battery enclosure scenario summary.

Scenario	Dismantling scenario	Scenario 1- steel	Scenario 2- CFRPs	Scenario 3- GFRPs	Scenario 4- GFRP-PDK
Material Content	–	Steel 100%	Carbon fiber: 44% Epoxy resin: 36% Flame retardant: 20%	Glass fiber: 50% Epoxy resin: 30% Flame retardant: 20%	Fiberglass: 50% PDK resin: 30% Flame retardant: 20%
Weight (kg pk ⁻¹)		85.9	18.4	32.8	34.8
Common recycling strategy	Dismantling	Mechanical recycling	Pyrolysis	Mechanical recycling	Chemical recycling
	Non-dismantling	Pyrolysis with rest of battery	Pyrolysis	Mechanical recycling	Chemical recycling
Recyclable Items	Dismantling	Steel scrap	Reclaimed carbon fiber *	Fiberglass powder **	PDK, fiberglass, flame retardant
	Non-dismantling	Steel scrap, Al foil, Cu scrap, iron scrap	Reclaimed carbon fiber, Al foil, Cu scrap, iron scrap	Fiberglass powder	PDK, fiberglass, flame retardant

* Reclaimed carbon fiber means second-life carbon fiber to be proposed to reuse.

** Fiberglass powder could be downcycled into filler in concrete, asphalt, or artificial wood.

2.2.3. TEA, LCA & material circularity indicators (MCIs)

The TEA evaluates the economic feasibility of recycling EV battery enclosures across the alternative material options and EoL recycling scenarios described in the section 2.2. The TEA system boundary begins at the EoL collection point of the battery housing and includes all unit operations required to produce recycled material outputs according to each scenario. We assume an *n*th-of-a-kind commercial facility processing 10 packs of EV battery enclosures per batch. Because processing times differ by recycling technology and dismantling choice, annual throughput varies by scenario; the resulting annual capacity spans 11 730–79 200 enclosure packs yr⁻¹. The wide range of processing capacities across scenarios is primarily driven by differences in process configuration and processing time. For example, the CFRP pathway includes pyrolysis and therefore has a longer processing time and lower annual throughput, whereas the steel pathway relies primarily on shredding and separation and can achieve higher throughput. Consequently, throughput-related economies of scale are treated as part of the process-specific economics rather than as an independent assumption. The facility is assumed to be located in California, using region-specific labor, utilities, and material costs, while revenues/credits from recycled scrap metals and recovered composite constituents are based on national market values. Capital and operating costs are obtained from SuperPro Designer's built-in process-based costing framework supplemented with literature data where needed. A discounted cash-flow model is used to compute the levelized recycling cost (LRC), assuming a 30 year plant lifetime and internal rate of return of 10%. The assumed revenues and displacement benefits for recovered materials depend on downstream market acceptance and material quality. For PDK-based composites, chemical depolymerization could enable recovery of the PDK building blocks from composite waste and potentially provide a relatively high-value recycled feedstock for PDK manufacturers or specialized recycling facilities. In contrast, CFRP and GFRP systems use thermoset epoxy matrices that cannot be remelted and readily separated from the reinforcement; therefore, recycling primarily relies on recovering the glass or carbon fibers, with their market value depending on fiber integrity, length, surface condition, and contamination. Recovered fiberglass and carbon fiber may consequently be downcycled to lower-value secondary applications, such as fillers in concrete, rather than directly replacing the virgin fibers. Recovery of flame retardants and other additives would similarly depend on their chemical stability, purity, and separation efficiency. These differences in material recovery pathways and quality requirements introduce uncertainty in the achievable revenue and displacement benefits, and they represent important market constraints for the modeled recycling scenarios. Additional TEA assumptions (e.g. recycling scrap prices) are documented in SI section S5.

We also conducted a life-cycle GHG inventory for different battery enclosure materials. The goal is to compare cradle-to-grave GHG emissions from enclosure production, vehicle use, and enclosure EoL management under the recycling scenarios considered. The functional unit is one EV battery enclosure

pack used in the same vehicle over a 20-year lifetime. The system boundary includes (i) enclosure material production and manufacturing, (ii) the use phase, and (iii) EoL treatment through the modeled recycling scenarios. The life-cycle GHG inventory was conducted using Agile C2G v.1 (Scown *et al* 2020), with additional data sourced from the GREET model (Wang *et al* 2023), Ecoinvent (Sutter 2007), and literature sources (e.g. LCA of flame retardant from Maga *et al* (2024)). The life-cycle inventory (LCI) for enclosure production is adopted from Stegman (2025). Use-phase impacts associated with enclosure mass were estimated based on the additional electricity consumption required to operate a midsize EV over its anticipated lifetime. Unlike ICEs, EVs have no direct tailpipe emissions; therefore, the associated use-phase GHG emissions represent upstream emissions from electricity generation resulting from additional electricity consumption attributable to enclosure mass. The calculation assumes an annual driving distance of 13 500 miles, an energy consumption increase of 0.087 Wh/(km · kg), and an electricity grid carbon intensity of 0.29 kg CO₂e/kWh. Further details on the use-phase assumptions and calculation methodology are provided in the SI section S6. For the EoL stage, the LCI is derived directly from the SuperPro Designer process simulations to ensure consistency with the TEA boundary conditions and operating assumptions. For scenarios generating multiple recovered material streams from shared processing steps, mass-based and economic allocation are applied to consistently distribute the associated environmental burdens among the outputs. Mass allocation reflects the physical contribution of each output, while economic allocation accounts for differences in the relative economic value of the recovered materials and provides an alternative basis for evaluating allocation sensitivity. We use IPCC AR6 global warming potentials to report results as CO₂e over a 100 year time horizon (IPCC 2024).

In addition to TEA and LCA, we evaluate material circularity using the MCI, since material criticality and circularity are important design considerations for automotive components. The MCI is calculated following Rocchi *et al* (2021):

$$\text{MCI} = 1 - \text{LFI} \times F(X_c) = 1 - \left(\frac{V + W_0}{2M} \right) \times F(X_c) \quad (4)$$

where LFI is the linear flow index, V is the virgin material input (mass or mass fraction, consistent with M), W_0 is the unrecoverable waste (mass or mass fraction), and M is the total material mass. $F(X_c)$ is a utility factor that accounts for product lifetime and intensity of use. MCI values closer to 1 indicate greater circularity, while values near 0 indicate a predominantly linear material flow; values below 0 may occur for systems that are ‘worse than linear’ under the MCI methodology.

3. Results

3.1. Current polymer material flows in EVs

Figure 2 presents the material composition of the EV, disaggregated by material class and vehicle subsystem. Aggregating the material flows indicates that the vehicle mass is dominated by ferrous metals, polymers and elastomers (including foams), and non-ferrous metals, which together account for more than 95% of total mass (percentages may not sum to 100% due to rounding). Ferrous metals constitute the largest share, followed by polymers and elastomers, while non-ferrous metals contribute a smaller fraction by mass but remain important for electrified functions. Overall, this distribution is consistent with contemporary EV architectures, in which conventional structural materials remain prevalent alongside increased use of lightweight and electrically functional materials.

Ferrous metals account for approximately 45%–50% of total vehicle mass, with steel providing the dominant contribution within this category. Conventional steel, rolled steel, galvanized steel, and smaller amounts of stainless steel and electric arc furnace (EAF) steel are primarily associated with structural and assembly subsystems, including weld blanks and fasteners, doors, and interior-to-body connections. The persistence of steel in the material flow reflects its favorable mechanical performance, cost-effectiveness, and mature manufacturing infrastructure. From a sustainability standpoint, these results suggest that near-term improvements are likely to come less from wholesale substitution of structural steels and more from reducing steel’s embodied emissions (e.g. increased recycled content and lower-carbon production routes), consistent with the presence of EAF steel in figure 2.

Polymers and elastomers contribute approximately 35%–40% of total vehicle mass, making them the second-largest material category in figure 2. This category is heterogeneous and includes commodity thermoplastics, engineering plastics, elastomers, and foams distributed across interior, exterior, and functional subsystems. PP and PP-based compounds represent the largest share of polymer mass (approximately 35%–40% of total polymer use). These materials are widely used in interior trim components

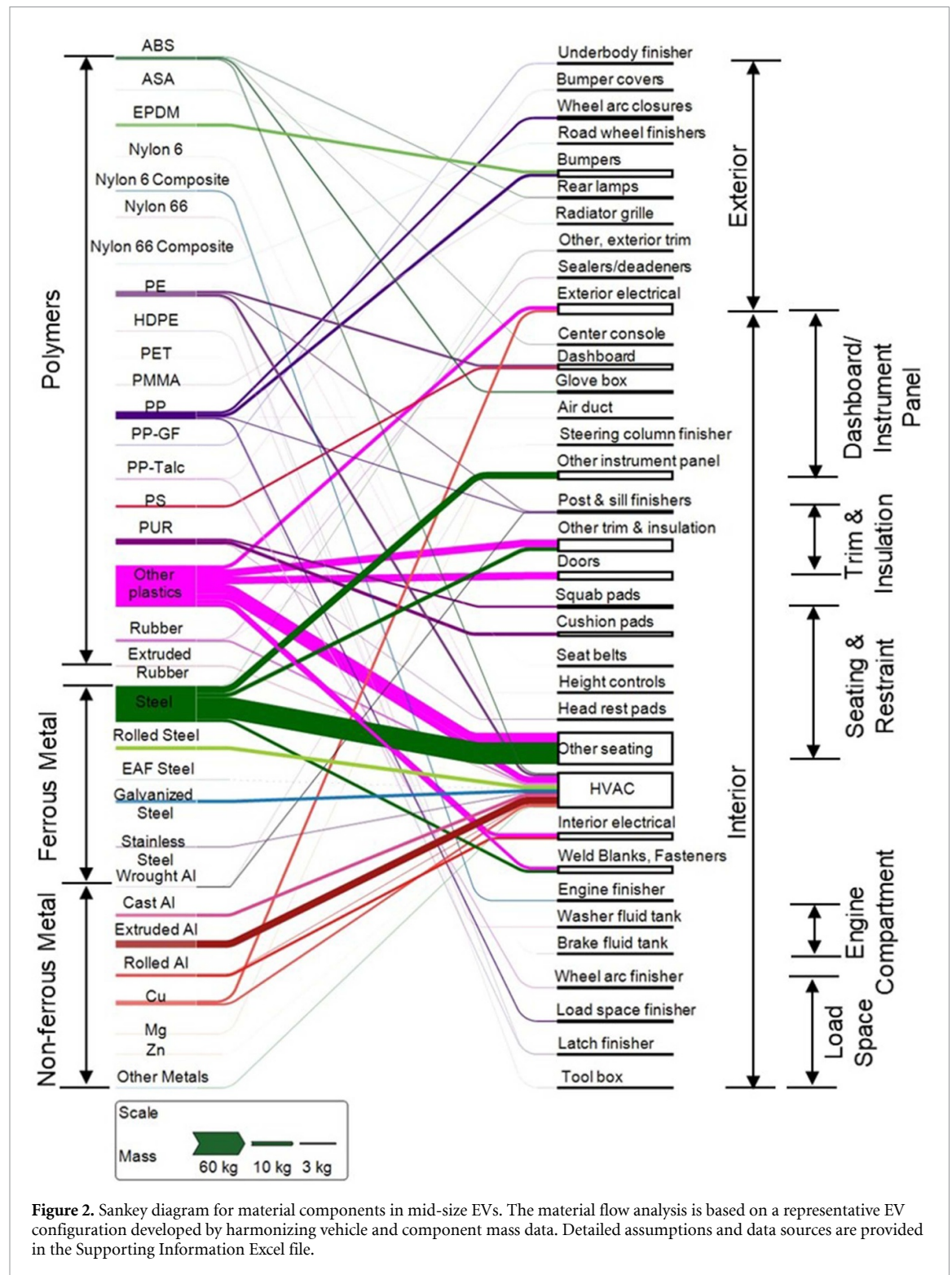


Figure 2. Sankey diagram for material components in mid-size EVs. The material flow analysis is based on a representative EV configuration developed by harmonizing vehicle and component mass data. Detailed assumptions and data sources are provided in the Supporting Information Excel file.

(e.g. dashboards, center consoles, and post and sill finishers) and exterior elements (e.g. bumpers and wheel-arch finishers), reflecting design priorities related to lightweighting, manufacturability, and cost.

PU foams account for a substantial portion of polymer use, contributing approximately 20%–25% of total polymer mass. They are concentrated in seating systems, including cushion pads, squab pads, and headrest pads. Although PU foams are essential for comfort, vibration damping, and noise reduction, their EoL management remains challenging due to limited recyclability and collection/processing constraints. Particularly as EV operational emissions decline with increased shares of low-emission electricity sources, such materials can become more prominent contributors to embodied impacts in cradle-to-grave EV assessments. Engineering plastics—including ABS, ASA, PA6, PA66, and reinforced variants—collectively account for approximately 15%–20% of polymer mass. These materials are

used in instrument panels, heating, ventilation, and air conditioning (HVAC) components, and other functional interior elements where stiffness, thermal stability, and dimensional accuracy are required. Relative to internal-combustion platforms, EV packaging and thermal requirements can shift where high-performance polymers are most advantageous. However, increased use of fiber-reinforced and mineral-filled plastics can also increase material complexity, which may hinder sorting and reduce the feasibility of closed-loop recycling.

Non-ferrous metals contribute approximately 10%–15% of total vehicle mass. Aluminum is the dominant non-ferrous metal, comprising roughly 65%–70% of non-ferrous mass (about 8%–9% of total vehicle mass). It appears in cast, extruded, and rolled forms and is used in both structural and functional applications to support lightweighting—an important strategy for offsetting battery mass and improving efficiency. However, the net environmental benefit of aluminum depends strongly on production pathways and recycled content, given the high energy intensity of primary aluminum production. Copper accounts for approximately 20%–25% of non-ferrous mass, corresponding to about 3% of total vehicle mass. While its mass contribution is modest, copper is functionally critical due to its role in electrical systems, power distribution, and auxiliaries such as HVAC. The copper content visible in figure 2 is consistent with greater electrification of vehicle subsystems relative to conventional vehicles. From an environmental and supply perspective, copper can represent a hotspot because mining and refining are energy-intensive and may face constraints under large-scale EV deployment (Huang *et al* 2020).

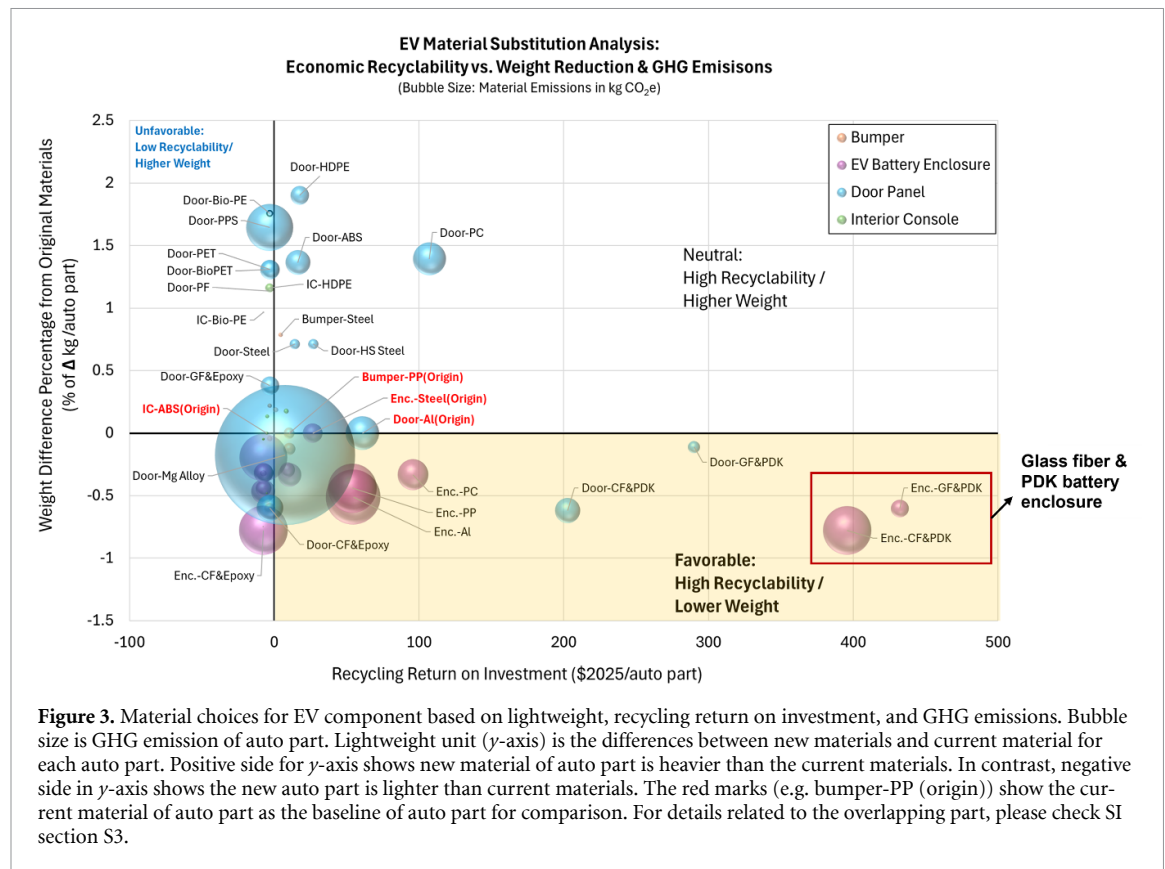
Subsystem-level results reveal pronounced differences in composition across vehicle systems. Interior and trim subsystems are polymer-dominated (often exceeding 70% by mass), consistent with requirements for lightweighting, comfort, and aesthetics, while seating systems are strongly influenced by PU foams, which contribute substantial mass but offer limited circularity potential. In contrast, structural and assembly-related subsystems are dominated by ferrous metals (frequently exceeding 80% steel by mass), underscoring the continued role of steel in safety and structural integrity. Electrical and HVAC subsystems show comparatively higher contributions from copper and engineering plastics, highlighting material hotspots associated with electrification. Overall, these material-flow results reinforce the shift in sustainability challenges from the use phase toward the production and EoL phases. Lightweight polymers and non-ferrous metals can reduce operational energy demand, but they also increase embodied emissions and material diversity, complicating recycling. Accordingly, further improvements in EV sustainability are likely to depend on strategies such as increasing recycled content, reducing unnecessary material variety, and enabling higher-quality EoL recovery, rather than relying solely on additional reductions in operational energy consumption.

3.2. Potential for material substitution across EV components

Material flow trends indicate that polymers contribute significantly to EV composition. We identified battery enclosures, bumpers, door panels, and ICs (interior consoles) as the primary components suitable for material substitution due to their ease of dismantling and lower complexity. Figure 3 compares these components based on weight reduction, R-ROI, and GHG emissions. In this plot, Quadrant 4 represents the optimal zone (high recyclability, low weight), while Quadrant 2 represents the unfavorable zone (unprofitable recycling, high weight). Additional data regarding material costs and GHG emissions is available in the SI section S3.

Evaluations suggest that current materials for bumpers and ICs are already near-optimal. The current material for automotive bumpers is PP, though potential substitutes include steel, recycled steel, magnesium alloy, ABS, PET, bio-PET, and GP/Epoxy resin. However, the current PP bumper already offers a medium R-ROI benefit, and no alternative material achieved a higher R-ROI. Furthermore, existing EVs typically use ABS for the interior console. Including ABS, the majority of potential polymer candidates are not economically feasible for recycling, as their R-ROI is negative. While an IC made from PC would be a viable choice with a positive, higher R-ROI, it would increase the weight by 18% compared to ABS. Consequently, bumpers and ICs occupy a small portion of figure 3 (please see SI figure S5 for an enlarged view). Door panels, currently manufactured from high-strength aluminum, face challenges in material substitution. Replacing aluminum with polymers typically increases weight, as the lower strength of polymers necessitates higher material volumes. Only carbon or glass fiber composites are feasible structural alternatives, with glass fiber composites offering the advantage of lower GHG emissions compared to carbon fiber.

In contrast, battery enclosures offer the greatest potential for improvement. Replacing standard steel enclosures with aluminum or composites (CFRP/GFRP) maintains strength while facilitating lightweighting. While conventional CFRP with epoxy yields the lowest weight (18.4 kg), it incurs negative R-ROI (tipping fees). However, using PDK resin with CFRP/GFRP dramatically improves economic feasibility. PDK allows for simple depolymerization, reducing treatment costs and enabling the reclamation of



monomers, fibers, and flame retardants. Therefore, PDK-based battery enclosures offer an attractive balance of lightweighting, economic recyclability, and reduced GHG emissions. The following section further details the sustainability and economic viability of these enclosures.

3.3. Economic and environmental comparison of EV battery enclosure materials

3.3.1. TEA results

Material selection for EV battery enclosures influences not only recycling economic feasibility, but also environmental performance and the long-term circular availability of materials. The TEA results (figure 4) indicate that the LRC is primarily driven by two factors: (i) recycling process costs (CAPEX/OPEX) and (ii) revenues from recovered products, including co-products (metal scraps), battery/pack resale, and black mass (BM). In dismantling scenarios, the battery pack is detached from the enclosure and sold to secondary handlers, yielding a benefit of \$0.918/kg (Gutsch and Leker 2024). In non-dismantling scenarios, the enclosure and remaining battery materials are co-processed; for most cases we apply battery-related revenue assumptions, while for S1 (steel) and S2 (CFRP) we instead consider BM sales because pyrolysis/refining produces a BM stream with elevated metal content. Because BM prices are volatile, we evaluate both low (\$0.5/kg) and high (\$3.35/kg) BM prices (GLG Network 2021, Shanghai Metal Market 2026).

For S1 (steel), dismantling uses simple mechanical recycling and benefits from battery pack resale, resulting in an LRC of −\$294 per pack enclosure. A more deeply negative LRC means the process is more profitable. The non-dismantling route uses pyrolysis, which increases OPEX due to energy demand, but enables additional co-product recovery (e.g. Al, Cu, Fe scraps) and BM production. Consequently, the non-dismantling LRC is highly sensitive to BM price, ranging from −\$22 (low-price BM) to −\$623 per pack enclosure (high-price BM). For S2 (CFRP/epoxy), there is no mechanical recycling option. In the dismantling scenario for S2, the enclosure is separated, but it is sold alongside the battery for further off-site recycling. This results in a higher processing cost, giving an LRC of −\$87 per pack enclosure. Non-dismantling options in S2 show the same BM-driven sensitivity as S1, with LRC values of −\$21 (low-price BM) and −\$604 (high-price BM) per pack enclosure. For S3 (GFRP/epoxy), dismantling and non-dismantling require only separation and shredding, leading to the lowest processing costs. When the enclosure is shredded, it produces a near-zero-value glass fiber powder, contributing negligible revenue. However, if the enclosure is left on the battery (non-dismantling case), we assume

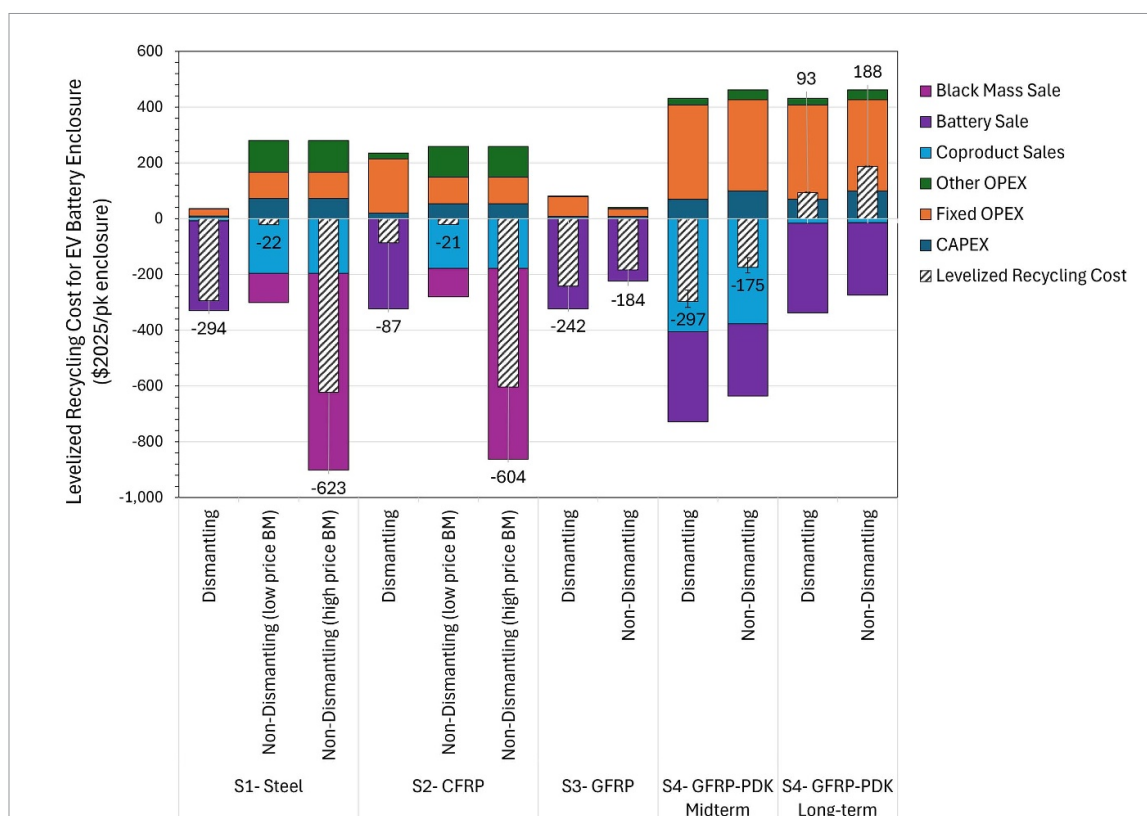
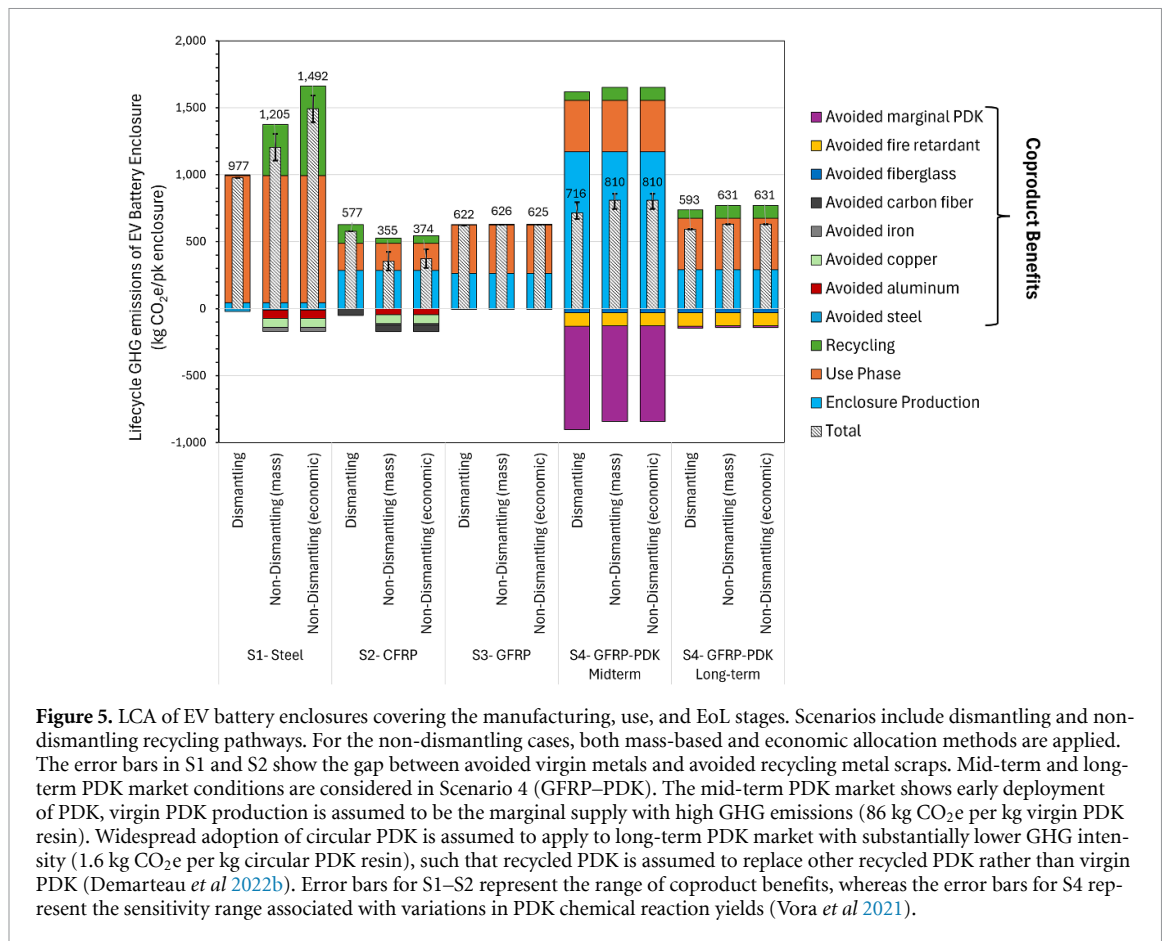


Figure 4. Breakdown of levelized recycling cost for EV battery enclosure scenarios. The coproduct sales include steel, copper, aluminum, iron scraps, reclaimed carbon fiber, fiberglass powder, fiberglass, flame retardant, and PDK. Rest of battery benefit from process could be distinguished as battery sale (0.918/kg (Gutsch and Leker 2024)) and black mass sale (low BM price is \$0.5/kg (Shanghai Metal Market 2026)), and high BM price is \$3.35/kg (GLG Network 2021)) depending on the quality of rest of battery. The battery resale value is held constant because it is primarily determined by battery condition and market factors outside the recycling process, whereas black mass value is more directly affected by the quality of the recovered material and recycling process. The mid-term PDK market represents early deployment of PDK, in which virgin PDK production remains the marginal supply and is associated with high material price (\$45 per kg virgin PDK resin); recycled PDK is therefore assumed to displace virgin PDK on a 1:1 mass basis. In contrast, the long-term PDK market assumes widespread adoption of circular PDK with substantially lower material price (\$1.5 per kg circular PDK resin), such that recycled PDK is assumed to replace other recycled PDK rather than virgin PDK. Error bars represent the sensitivity range resulting from variations in the PDK chemical reaction yields based on the ranges reported in Vora *et al* (2021).

this will result in a lower battery sale price. LRC values are −\$242 (dismantling) and −\$184 (non-dismantling) per pack enclosure, with the higher non-dismantling LRC attributed to partial losses of battery-related value after mechanical processing. For S4 (GFRP–PDK), depolymerization and drying drive the highest operating costs, but these are offset by substantially higher coproduct sales revenue. Non-dismantling is slightly more expensive due to an additional shredding/separation step. LRC is also strongly dependent on the assumed value of recovered PDK. Using a mid-term market (virgin PDK price \$45/kg) versus a long-term market (circular PDK price \$1.5/kg) (Demarteau *et al* 2022b), dismantling LRC shifts from −\$297 to +\$93 per pack enclosure, and non-dismantling shifts from −\$175 to +\$188 per pack enclosure. Overall, most scenarios are revenue-positive under the assumed co-product and battery/BM prices; however, S4 becomes cost-positive (meaning recyclers will lose money) under mature-market circular PDK pricing. This indicates that process cost reductions and/or higher-value outlets for recovered PDK will be important for long-term economic competitiveness.

3.3.2. LCA results

Compared with the TEA, the LCA system boundary extends beyond recycling to include enclosure production and the use phase, thereby capturing additional emissions trade-offs among scenarios. As shown in figure 5, both production and use-phase impacts are strongly mass-dependent; therefore, lightweighting can substantially reduce GHG emissions in these two phases. In contrast, the recycling stage contributes a relatively small share of total GHG emissions across scenarios. For S1 (steel), enclosure manufacturing has comparatively low emissions, but the enclosure's higher mass leads to larger use-phase impacts (quantified as the additional upstream GHG emissions from electricity generation resulting from the additional electricity consumption attributable to the enclosure mass over the anticipated lifetime of a midsize EV). Total impacts are 977 kg CO₂e per pack enclosure for the dismantling scenario and 1205



(mass allocation between enclosure and battery) / 1492 (economic allocation) kg CO₂e per pack enclosure for the non-dismantling scenario. For non-dismantling, we applied both mass and economic allocation to attribute recycling impacts between the enclosure and the rest of the battery. Non-dismantling yields higher GHG emissions primarily due to the additional pyrolysis step and limited co-product credits (e.g. recovered steel, aluminum, copper, and iron scraps), consistent with the recycling discussion in the TEA. S2 (CFRP/epoxy) has the lowest enclosure mass, which reduces use-phase emissions; however, carbon fiber production dominates the enclosure production impacts. The dismantling scenario has higher total emissions (577 kg CO₂e per pack enclosure) than the non-dismantling scenario (355 mass allocation; 374 economic allocation kg CO₂e per pack enclosure) because dismantling eliminates co-product credits from reclaimed metals (Al, Cu, Fe) recovered with the remaining battery. Economic allocation is slightly higher than mass allocation because the recovered metals receive greater weight under the economic allocation method. For S3 (GFRP/epoxy), recycling impacts are negligible; GHG emissions are driven mainly by enclosure production and the use phase. Dismantling and non-dismantling results are similar because the rest of the battery can be removed early during shredding and float-and-sink separation. Total impacts are 622 kg CO₂e (dismantling) and 626 / 625 kg CO₂e (non-dismantling, mass/economic) per pack enclosure. For S4 (GFRP–PDK), we distinguish mid-term and long-term PDK markets using avoided marginal PDK assumptions: mid-term credits virgin PDK displacement (86 kg CO₂e per kg virgin PDK), whereas long-term credits circular PDK displacement (1.6 kg CO₂e per kg circular PDK) (Demarteau *et al* 2022b). In the mid-term case, total impacts are 716 kg CO₂e (dismantling) and 810 / 810 kg CO₂e (non-dismantling, mass/economic) per pack enclosure; production emissions and avoided PDK credits largely offset each other, and the PDK mass balance approaches 95% recovery. Non-dismantling is slightly higher due to one additional separation step. In the long-term case, impacts decrease to 593 kg CO₂e (dismantling) and 631 / 631 kg CO₂e (non-dismantling, mass/economic) per pack enclosure: avoided-credit magnitude is smaller, but enclosure production uses circular PDK with much lower upstream emissions than virgin PDK.

3.3.3. Material circularity indicator results

Based on equation (4), the MCI values are calculated from the recycled material usage and EoL recycling rates obtained from the process modeling. Steel achieves a relatively high MCI of 0.925 because

Table 3. MCI for EV battery enclosure recycling scenarios.

	S1- Steel		S2- CFRP		S3- GFRP	
	Dismantling	Non-dismantling	Dismantling	Non-dismantling	Dismantling	Non-dismantling
Material	Steel	Steel	Carbon fiber	Carbon fiber	Fiberglass	Fiberglass
Recycled material usage (%) - assumption	85%	85%	50%	50%	0%	0%
EoL recycling rate (%) - process modeling	100%	100%	99%	99%	95% downcycling	95% downcycling
Material circularity indicator (MCI)	0.925	0.925	0.331	0.331	0.238	0.238
S4- GFRP-PDK						
Material	Dismantling PDK	Fiberglass	Flame Retardant	Non-dismantling PDK	Fiberglass	Flame Retardant
Recycled material usage (%) - assumption	100%	100%	100%	100%	80%	80%
EoL recycling rate (%) - process modeling	94%	95%	100%	94%	95%	100%
Material circularity indicator (MCI)	0.979		0.909			

of its high recycled material content (85%) and recycling rate (100%) based on the simulation, while CFRP and GFRP have lower MCI values of 0.331 and 0.238, respectively, due primarily to lower recycled material usage and downcycling of recovered fibers. The GFRP-PDK enclosure achieves the highest MCI, with values of 0.979 for dismantling and 0.909 for non-dismantling scenarios. The higher MCI in the dismantling scenario is mainly attributed to the higher fiberglass recovery rate (100% versus 80%) and flame retardant recovery rate (100% versus 80%), demonstrating the benefit of material separation for improving circularity (see table 3).

4. Discussion

This study illustrates how current EV design is simultaneously moving toward lightweighting and improved circularity, and shows how a material-substitution framework can identify where alternative polymers and composites are most likely to deliver net benefits. The screening results suggest that for several existing polymer-dominated components, such as PP in bumpers and ABS in interior consoles, the incumbent material choices are already close to ‘best fit’ when manufacturability, mechanical performance, and cost are considered. In other words, the largest substitution opportunities are not necessarily in parts that already rely on commodity polymers, but rather in heavy structural components where stiffness- and strength-driven design currently favors metals. Consistent with this, the battery enclosure emerges as a high-leverage target: replacing a steel enclosure (85.9 kg) with fiber-reinforced polymer options can substantially reduce enclosure mass (e.g. CFRP: 18.4 kg; GFRP-PDK: 34.8 kg), with potential downstream benefits in vehicle energy use and related life-cycle impacts. This finding aligns with prior lightweighting literature, which has often evaluated substitution through lightweight metals or scaling-based comparisons (Mayyas *et al* 2017, Amasawa *et al* 2020, Burd *et al* 2021, Wolfram *et al* 2021, Nwokolo 2025). At the same time, our results reinforce an important caution raised by Delogu *et al* (2017): lightweight materials can shift environmental burdens toward the production stage, such that use-phase benefits may not fully offset higher embodied impacts depending on the electricity mix, lifetime mileage, and material production pathways.

A central contribution of this work is quantifying the EoL implications of composite substitution using explicit dismantling and non-dismantling recycling scenarios. In particular, the PDK-based composite case (S4) illustrates how resin chemistry can change the cost-GHG-circularity trade space relative to conventional thermoset composites. Unlike mechanical recycling routes that often downcycle

polymers and degrade composite properties, PDK enables chemical recycling with the potential to recover monomers and higher-quality reinforcement fibers (and associated additives) at higher yields (Vora *et al* 2021, Demarteau *et al* 2022b). Under our modeled assumptions, S4 achieves competitive LRC (−\$297 and −\$175 per enclosure pack for dismantling and non-dismantling, respectively) while maintaining substantial lightweighting relative to steel (−\$294 / −\$623 to −\$22). Importantly, the same scenario also provides improved circularity performance relative to conventional composite options, indicating a plausible pathway toward higher-value recovery for structural EV parts rather than persistent downcycling.

Compared with prior enclosure-focused LCA studies, our results are directionally consistent with reports that the use phase can contribute substantially to enclosure life-cycle impacts (particularly when mass differences are large and lifetime mileage is high) (Li *et al* 2024, De Sio *et al* 2025). However, most existing work emphasizes metal-to-metal comparisons (e.g. steel vs. aluminum) and gives limited treatment to fiber-reinforced polymer enclosures, often noting that polymer/composite adoption may increase life-cycle impacts because of higher embodied emissions and more challenging EoL management.

Several barriers may limit the near-term use of PDK composites in automobile applications. PDK composites are still at an early stage of manufacturing readiness (TRL 3-4), and continuous-fiber laminates, automotive components, and production processes with automotive-scale cycle times have not yet been demonstrated (Helms 2022, Demarteau *et al* 2022a). Important properties such as durability in acidic environments and long-term creep also require further validation. In contrast, the potential benefits of PDK chemical recycling have been studied more extensively. As noted above, the large cost gap between virgin PDK and resin produced from recovered monomers provides strong incentives for closed-loop recycling (Vora *et al* 2021, Demarteau *et al* 2022b). However, large-scale recycling would still require appropriate processing infrastructure, waste-stream management, and sufficient feedstock volumes. Compared with other emerging recyclable polymer systems, such as cleavable epoxies, vitrimers, and thermoplastics (Post *et al* 2020), PDK is less mature in manufacturing but distinctive in recovering virgin-quality monomer from mixed and contaminated plastic streams without relying on conventional melt filtration (Christensen *et al* 2019). Our study extends this literature by evaluating polymer and composite substitutions together with recycling process models (consistent TEA and LCA boundaries). This more holistic view indicates that polymer-based lightweighting is not inherently beneficial or detrimental; rather, its net performance depends on (i) stiffness-driven redesign requirements, (ii) the feasibility and cost of dismantling to achieve clean material streams, and (iii) whether an EoL technology exists that can recover high-quality outputs that displace virgin production.

5. Conclusion

This study establishes a comprehensive material substitution framework for the preliminary assessment of potential EV components. Uniquely, this evaluation integrates both sustainability and economic feasibility metrics to provide a fair comparison of candidate applications, addressing a gap in previous research which often treats these metrics in isolation. Significantly, this work presents the first application of PDK in a material evaluation context, supported by a specific LCA and TEA case study of EV battery enclosures. We demonstrate that PDK is a highly promising polymer; it can be effectively compounded with additives like fibers and flame retardants while offering closed-loop recyclability to reclaim all constituent materials. This capability mitigates polymer matrix degradation and ensures high separation efficiency, distinguishing PDK from conventional plastics. Future work should expand upon these findings to address the inherent complexities of automotive manufacturing. First, while this study isolated components to ensure precise evaluation, real-world EV components often involve complex multi-material assemblies. Future research requires collaboration with EV manufacturers to evaluate these assemblies through advanced mechanical modeling. Second, although this study focuses on GHG emissions, other environmental impact categories may introduce important trade-offs among the material and recycling pathways. For example, glass fiber production may contribute to particulate matter and human toxicity impacts, while flame retardants and solvents associated with PDK recycling may introduce additional toxicity and ecotoxicity concerns. Broader multi-impact assessments are therefore needed to provide a more comprehensive evaluation of environmental performance. Finally, while the current TEA focused on the recycling process, the system boundary should be extended to include the battery housing manufacturing phase to provide a holistic view of economic feasibility across the entire product life cycle.

Acknowledgments

The authors gratefully acknowledge our industry partner, Bhavesh Shah and Paul Hernley at General Motors for their valuable technical guidance and constructive feedback throughout this study. Their insights on EV component design considerations, material selection, and EoL practices helped refine the scope, assumptions, and interpretation of results.

Data availability statement

All data that support the findings of this study are included within the article (and any supplementary information files).

SI S1 available at <https://doi.org/10.1088/2634-4505/aea641/data1>.

SI S2. EV & ICE BOM available at <https://doi.org/10.1088/2634-4505/aea641/data2>.

Funding

This material is based upon work supported by the U.S. Department of Energy's Office of Critical Minerals and Energy Innovation (CMEI) under the Transportation Technologies Office (TTO) award number DE-EE0011181. This material is also based upon work at the Joint BioEnergy Institute supported by the US Department of Energy, Office of Science, Biological and Environmental Research Program under Contract Number DE-AC02-05CH11231 between Lawrence Berkeley National Laboratory and the US Department of Energy. The US Government retains, and the publisher, by accepting the article for publication, acknowledges that the US Government retains a nonexclusive, paid-up, irrevocable, worldwide license to publish or reproduce the published form of this manuscript, or allow others to do so, for US Government purposes. Any subjective views or opinions that might be expressed in the paper do not necessarily represent the views of the US Department of Energy or the US Government.

Author contributions

Tai-Yuan Huang  [0000-0001-8767-8452](https://orcid.org/0000-0001-8767-8452)

Conceptualization (equal), Formal analysis (lead), Methodology (lead), Visualization (lead), Writing – original draft (lead), Writing – review & editing (equal)

Baishakhi Bose  [0000-0002-6309-0088](https://orcid.org/0000-0002-6309-0088)

Formal analysis (supporting), Methodology (supporting), Visualization (supporting), Writing – original draft (supporting), Writing – review & editing (supporting)

Grace X Gu  [0000-0001-7118-3228](https://orcid.org/0000-0001-7118-3228)

Conceptualization (equal), Writing – review & editing (equal)

Brett A Helms  [0000-0003-3925-4174](https://orcid.org/0000-0003-3925-4174)

Conceptualization (equal), Writing – review & editing (equal)

Jay D Keasling  [0000-0003-4170-6088](https://orcid.org/0000-0003-4170-6088)

Conceptualization (equal), Project administration (lead), Writing – review & editing (equal)

Corinne D Scown  [0000-0003-2078-1126](https://orcid.org/0000-0003-2078-1126)

Conceptualization (lead), Funding acquisition (lead), Supervision (lead), Writing – review & editing (lead)

References

- Almushaikeh A M *et al* 2023 Manufacturing of carbon fiber reinforced thermoplastics and its recovery of carbon fiber: a review *Polym. Test.* **122** 108029
- Amasawa E, Hasegawa M, Yokokawa N, Sugiyama H and Hirao M 2020 Environmental performance of an electric vehicle composed of 47% polymers and polymer composites *Sustain. Mater. Technol.* **25** e00189
- Argonne National Laboratory 2025 Battery performance and cost model (BatPac) v.5.0 (available at: www.anl.gov/partnerships/batpac-battery-manufacturing-cost-estimation)

- Auto Recycling World 2021 Introducing closed-loop circularity for hard-to-recycle automotive plastics (available at: <https://autorecyclingworld.com/introducing-closed-loop-circularity-for-hard-to-recycle-automotive-plastics/>) (Accessed 27 January 2026)
- Bonnen Battery 2025 Understanding lithium battery pack enclosure design for electric vehicles and boats (available at: www.bonnenbatteries.com/understanding-lithium-battery-pack-enclosure-design-for-electric-vehicles-and-boats/) (Accessed 27 January 2026)
- Buekens A and Zhou X 2014 Recycling plastics from automotive shredder residues: a review *J. Mater. Cycles Waste Manag.* **16** 398–414
- Burd J T J, Moore E A, Ezzat H and Kirchain R 2021 Improvements in electric vehicle battery technology influence vehicle lightweighting and material substitution decisions *Appl. Energy* **283** 116269
- Cahill K M, Kim D H and Hueckel A 2026 Composite system for battery enclosure (available at: https://patents.justia.com/patent/20260005361?utm_source)
- CHARM 2014 Operation CHARM: car repair manuals for everyone (available at: <https://charm.li/>) (Accessed 18 June 2025)
- Choi M, Cha J and Song J 2025 Impact of lightweighting and driving conditions on electric vehicle energy consumption: in-depth analysis using real-world testing and simulation *Energy* **323** 135746
- Christensen P R, Scheuermann A M, Loeffler K E and Helms B A 2019 Closed-loop recycling of plastics enabled by dynamic covalent diketoenamine bonds *Nat. Chem.* **11** 442–8
- Clarkson C, Hubbard A, Wang P, Lobodin V and Korey M 2024 Circular economy for automotive shredder residue *Technical report* Oak Ridge National Laboratory
- Cousins D S, Suzuki Y, Murray R E, Samaniuk J R and Stebner A P 2019 Recycling glass fiber thermoplastic composites from wind turbine blades *J. Clean. Prod.* **209** 1252–63
- De Sio P, Gaito M, Esperto V, Cozzolino E, Astarita A and Tucci F 2025 Life cycle assessment of a composite prototype battery enclosure for electric vehicles *Sustainability* **17** 1579
- Delogo M, Zanchi L, Dattilo C and Pierini M 2017 Innovative composites and hybrid materials for electric vehicles lightweight design in a sustainability perspective *Mater. Today Commun.* **13** 192–209
- Demarteau J et al 2022 Circularity in mixed-plastic chemical recycling enabled by variable rates of polydiketoenamine hydrolysis *Sci. Adv.* **8** eabp8823
- Demarteau J et al 2023 Biorenewable and circular polydiketoenamine plastics *Nat. Sustain.* **6** 1426–35
- Demarteau J, Vora N, Keasling J D, Helms B A and Scown C D 2022 Lower-cost, lower-carbon production of circular polydiketoenamine plastics *ACS Sustain. Chem. Eng.* **10** 2740–9
- Edah G O, Atiba J O and Fayomi O S 2025 Advancements in fibre-reinforced polymers: properties, applications (a mini review) *Next Mater.* **8** 100743
- Gibbs J, Joost W and Schutte C 2013 Light-duty vehicles technical requirements and gaps for lightweight and propulsion materials *Technical report* DOE Vehicle Technologies Office
- GLG Network 2021 The economics around lithium-ion battery recycling are strong and growing (available at: <https://glginsights.com/articles/the-economics-around-lithium-ion-battery-recycling-are-strong-and-growing/>) (Accessed 18 June 2025)
- Gotthold D, Kleinbaum S, Wu F, Gibbs J and Singh G 2019 Vehicle lightweighting program progress status- 2019 *Technical report* Department of Energy, Vehicle Technologies Office
- Gutsch M and Leker J 2024 Costs, carbon footprint and environmental impacts of lithium-ion batteries – from cathode active material synthesis to cell manufacturing and recycling *Appl. Energy* **353** 122132
- Hawkins T R, Singh B, Majeau-Bettez G and Strømman A H 2012 Comparative environmental life cycle assessment of conventional and electric vehicles *J. Ind. Ecol.* **17** 53–64
- Helms B A 2022 Polydiketoenamines for a circular plastics economy *Acc. Chem. Res.* **55** 2753–65
- Hendrickson T P, Bose B, Vora N, Huntington T, Nordahl S L, Helms B A and Scown C D 2024 Paths to circularity for plastics in the United States *One Earth* **7** 520–31
- Herrmann I and Flock H 2023 The industry that created plastic may fix its waste problem' (available at: www.oliverwyman.com/our-expertise/journals/energy-and-natural-resources-journal/2024/the-industry-that-created-plastic-may-fix-its-waste-problem.html#text=Economics%20differ,demonstration%20plants%20and%20smaller%20factories.) (Accessed 27 January 2026)
- Höhne C-C, Blaess P, Ilinzeer S and Griesbaum P 2023 New approach for electric vehicle composite battery housings: electromagnetic shielding and flame retardancy of PUR/UP-based sheet moulding compound *Composites A* **167** 107404
- Huang C-L, Xu M, Cui S, Li Z, Fang H and Wang P 2020 Copper-induced ripple effects by the expanding electric vehicle fleet: a crisis or an opportunity *Resour. Conserv. Recycl.* **161** 104861
- IEA 2025 Global EV outlook 2025- trends in other light-duty electric vehicles *Technical report* International Energy Agency
- IPCC 2024 IPCC sixth assessment report *Technical report* The Intergovernmental Panel on Climate Change
- Knehr K W, Kubal J J, Nelson P A and Ahmed S 2022 Battery performance and cost modeling for electric vehicles- a manual for BatPaC v5.0 *Technical report* Chemical Sciences and Engineering Division, Argonne National Laboratory
- Kroisová D, Štěpánka D'a, Knap A and Knápek T 2023 Destruction of carbon and glass fibers during chip machining of composite systems *Polymers* **15** 2888
- LEMON 2026 LEMON manuals: even more car manuals for everyone: 1960-2025 (available at: <https://lemon-manuals.org.ua/>) (Accessed 5 April 2026)
- Li X, Zhang Y, Liao Y and Yu G 2024 Environmental impact assessment of battery boxes based on lightweight material substitution *Sci. Rep.* **14** 2594
- Liu Y, Chen H, Gao J, Li Y, Dave K, Chen J, Federici M and Perricone G 2021 Comparative analysis of non-exhaust airborne particles from electric and internal combustion engine vehicles *J. Hazard. Mater.* **420** 126626
- Maga D, Aryan V and Beard A 2024 Toward sustainable fire safety: life cycle assessment of phosphinate-based and brominated flame retardants in e-mobility and electronic devices *ACS Sustain. Chem. Eng.* **12** 3652–8
- MakeltForm 2026 MakeltForm.com (available at: www.makeitfrom.com/) (Accessed 18 June 2025)
- MatWeb 2021 MatWeb- material property data (available at: www.matweb.com/) (Accessed 18 June 2025)
- Maurty T, Tazi N, de Matos C T, Nessi S, Antonopoulos I, Pierri E, Baldassarre B, Garbarino E, Gaudillat P and Mathieux F 2023 Towards recycled plastic content targets in new passenger cars and light commercial vehicles- technical proposals and analysis of impacts in the context of the review of the ELV directive *Technical Report* European Commission
- Mayyas A, Omar M, Hayajneh M and Mayyas A R 2017 Vehicle's lightweight design vs. electrification from life cycle assessment perspective *J. Clean. Prod.* **167** 687–701

- Moore M G, Belton K, Lan D, Rose-Glowacki H and Sanchez E 2024 Chemistry and automobiles- driving the future *Technical Report* American Chemistry Council
- Nwokolo S C 2025 Assessing the role of material substitution in cost reduction and demand mitigation for sustainable wind energy infrastructure *Cleaner Chem. Eng.* **11** 100203
- Okon Recycling 2025 Automotive shredder residue (ASR) processing: methods, regulations, and environmental impact (available at: www.okonrecycling.com/industrial-scrap-metal-recycling/steel-and-aluminum/automotive-shredder-residue-processing/) (Accessed 27 January 2026)
- Pierri E, Egle L, Milios L and Saveyn H 2024 EU-wide end-of-waste criteria for plastic waste- JRC technical proposals *Technical Report* Joint Research Centre
- Post W, Susa A, Blaauw R, Molenveld K and Knoop R J 2020 A review on the potential and limitations of recyclable thermosets for structural applications *Polym. Rev.* **60** 359–88
- Rajak D K, Pagar D D, Menezes P L and Linul E 2019 Fiber-reinforced polymer composites: manufacturing, properties and applications *Polymers* **11** 1667
- Ritchie H 2023 The weighty issue of electric cars [Part 2] (available at: <https://hannahritchie.substack.com/p/weighty-issue-of-electric-cars-two>) (Accessed 23 February 2026)
- Rocchi L, Paolotti L, Cortina C, Fagioli F and Boggia A 2021 Measuring circularity: an application of modified material circularity indicator to agricultural systems *Agric. Food Econ.* **9** 9
- Rose-Glowacki H, Moore M G, Belton K, Lan D and Sanchez E 2024 Chemistry and automobiles- driving the future *Technical report* American Chemistry Council
- Sato F E K, Furubayashi T and Nakata T 2018 Energy and CO₂ benefit assessment of reused vehicle parts through a material flow approach *Int. J. Automat. Eng.* **9** 91–98
- Schade A, Melzer M, Zimmermann S, Schwarz T, Stoewe K and Kuhn H 2024 Plastic waste recycling- a chemical recycling perspective *ACS Sustain. Chem. Eng.* **12** 12270–88
- Scown C D, Kavvada O and Neupane B 2020 Agile cradle-to-grave (Agile C2G) v.1, (available at: <https://www.osti.gov/biblio/code-52381>)
- Shaffer B, Auffhammer M and Samaras C 2021 Make electric vehicles lighter to maximize climate and safety benefits *Nature* **598** 254–6
- Shanghai Metal Market 2026 Used lithium-ion battery prices (available at: www.metal.com/en/markets/45) (Accessed 18 June 2025)
- Singh H, Kramer D, Fisher A, Paramasuwoom M, Mogal V and Ganesan V 2018 Mass reduction for light-duty vehicles for model years 2017–2025 Report No. DOT HS 812 487 *Technical Report* National Highway Traffic Safety Administration
- Stegman B S 2025 Comparative life cycle assessment of steel, aluminum, and glass fiber composite EV battery boxes *Master Thesis* University of Tennessee
- Sutter J 2007 Life cycle inventories of highly pure chemicals. data v2.0. 2007. *Technical Report* Swiss Centre for Life Cycle Inventories
- Thomas D S, Kneifel J D and Butry D T 2024 The U.S. plastics recycling economy- current state, challenges, and opportunities *Technical Report* U.S. Department of Commerce
- Tsuji A, Nelson Y, Kean A and Vigil S A 2006 Recyclability index for automobiles *Proc. Air & Waste Management Association (A & WMA) 99th Annual Conf. and Exhibition* pp 3876–94
- U.S. Environmental Protection Agency 2017 Processing end-of-life vehicles: a guide for environmental protection, safety and profit in the United States-Mexico border area *Technical Report* U.S. Environmental Protection Agency
- Versteeg F G, de Korte R J, Kulkarni S P, Stefanidis G and Van Geem K M 2026 Recycling strategies for plastics in end-of-life vehicles: a complete overview *Waste Manag.* **211** 115307
- Vijayan S K, Sahajwalla V and Bhattacharya S 2022 Insights into the options of energy and metal recovery from automotive shredder residue: a review *Resour. Conserv. Recycl. Adv.* **15** 200097
- Vora N, Christensen P R, Demarteau J, Baral N R, Keasling J D, Helms B A and Scown C D 2021 Leveling the cost and carbon footprint of circular polymers that are chemically recycled to monomer *Sci. Adv.* **7** eabf0187
- Wang M et al 2023 Summary of expansions and updates in R&D GREET® (available at: www.osti.gov/biblio/2278803)
- Wolfram P, Tu Q, Heeren N, Pauliuk S and Hertwich E G 2021 Material efficiency and climate change mitigation of passenger vehicles *J. Ind. Ecol.* **25** 494–510
- Zhang W and Xu J 2022 Advanced lightweight materials for automobiles: a review *Mater. Des.* **221** 110994