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# Life cycle greenhouse gas emissions of diesel oil and zero-emission trucks: Systematic review of status and perspectives

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ABSTRACT

The transport sector is a major source of greenhouse emissions, requiring urgent action for transition to cleaner alternatives. In this context, Zero Emission Vehicles (ZEVs) are essential for decarbonizing the transport sector and reducing air pollution. To estimate the potential benefits and burdens of these novel powertrains, the Life Cycle Assessment (LCA) methodology can be used and offer a comprehensive approach. However, when comparing ZEVs to diesel Internal Combustion Engine Vehicles (ICEVs), it is crucial to rely on robust results to ensure comparability. The present study presents a systematic literature review of LCA applications to Heavy-Duty Vehicles (HDVs), employing a rigorous methodology to provide an unbiased and comprehensive overview. The study aims to: (1) investigate the state-of-the-art methodological approaches in LCA implementation to HDVs (i.e., qualitative assessment) and (2) quantify the carbon footprints of current ICE-HDVs and ZEVs (i.e., quantitative assessment). After a screening step, a sample of 24 publications, comprising both scientific articles and industrial reports, has been extracted from several databases. Quantitatively, the analysis highlights a significant variability of the carbon footprint results, ranging between 431–1780 tons $CO_2$eq for diesel ICEVs, 175–1626 tons $CO_2$eq for battery electric vehicles, 366–2944 tons $CO_2$eq for fuel cell electric vehicles. Instead, from the qualitative standpoint, the review reveals that, while quite consensus exists on the functional unit, there are still major differing methodological choices that cause significant variability of carbon footprint among studies. Also, the analysis highlights the strong need for improved primary data coverage and greater harmonization of LCA methods and assumptions.

## 1. Introduction

The transport sector is a significant contributor to global energy consumption and greenhouse gas (GHG) emissions. Specifically, in 2020, heavy-duty vehicles (HDVs) accounted for approximately 40 % of global diesel consumption, underlining their critical role in both energy use and GHG emissions within the transport sector (IEA, 2020). In EU, HDVs are, in fact, responsible for nearly 25 % of all transport-related $CO_2$ emissions despite representing only a small fraction of the total vehicle fleet (EU Council, 2024). This can be attributed to the significant reliance of the HDV fleet on fossil fuels, predominantly diesel.

Life Cycle Assessment (LCA) has emerged as a crucial methodology for evaluating the environmental impacts of HDVs across their entire life cycle, from raw material extraction and manufacturing to operation and end-of-life disposal (ISO 14040, 2006; ISO 14044, 2018). LCA studies provide comprehensive insights into the various stages of the HDV lifetime, identifying key areas where emissions can be mitigated. For instance, the operation phase of diesel HDVs is particularly significant, contributing the majority of life cycle GHG emissions due to the combustion of diesel oil (Hawkins et al., 2013). Reducing the carbon footprint of HDVs requires a shift towards zero emission options. Zero-emission HDVs are vehicles without an internal combustion engine or with an internal combustion engine that emits no more than 3 g $CO_2$/(tons-km) or 1 g $CO_2$/(passenger-km), such as Battery Electric Vehicles (BEVs) and Fuel Cell Electric Vehicles (FCEVs) (European Commission, 2024). The integration of zero-emission technologies into transport systems introduces new layers of complexity that necessitate robust LCA to adequately evaluate potential environmental trade-offs and benefits. Traditionally, environmental evaluations in the transport sector have focused predominantly on operational emissions—such as fuel consumption and tailpipe emissions—while often overlooking upstream and downstream stages of the product life cycle (Wiegmans and Janic,

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2019). This limitation has stimulated increasing interest in the development of standardized LCA methodologies that encompass the entire life cycle of transport-related products and services (Cavallaro et al., 2024). However, a globally harmonized LCA framework for transport is still lacking. In response, the European Commission proposed the CountEmissionsEU framework to create a unified methodology for calculating well-to-wheel GHG emissions across transport services (European Commission, 2023a). Furthermore, the European Union has adopted Regulation (EU) 2023/1542, which introduces LCA-based mandatory carbon footprint declarations for electric vehicle batteries to be placed on the EU market from 2025 onwards (European Parliament, 2023). Lastly, policy frameworks, such as the European Green Deal (European Commission, 2023b), reinforce the EU's goal of achieving climate neutrality by 2050. This evolving policy landscape is also reflected in the changing valuation of environmental costs: for instance, the estimated social cost of carbon dioxide increased from 20 €/t$CO_2$ to 100 €/t$CO_2$ (Cavallaro and Nocera, 2022), while GHG emissions represent around 14 % of EU transport externalities (EEA, 2020). Beyond GHG emissions, road freight transport also generates substantial externalities including air pollution, noise, congestion, and infrastructure degradation, which together impose significant economic and societal burdens (Janic and Vleugel, 2012; Meers et al., 2018).

## 2. State of art and contribution of the present study

LCAs are increasingly used to inform transport research due to their capacity to assess full life cycle impacts beyond tailpipe emissions (Curran, 2016; Hawkins et al., 2013). Nevertheless, the analysis of existing studies reveals substantial heterogeneity in scope, methodology, and core assumptions, resulting in significant lack of comparability within the literature results. For instance, while some analyses adopt a well-to-wheel perspective (Bhardwaj and Mostofi, 2022; Cooper, 2019; Lozanovski et al., 2020), focusing solely on fuel production and vehicle operation emissions, others employ a more comprehensive cradle-to-grave framework, from raw material extraction to the end-of-life treatment (Booto et al., 2021; Burul, 2020; Hanesch et al., 2022; Lommahadthai, 2022; Middela et al., 2022; Osorio-Tejada et al., 2022; Palm, 2020; Ramshankar et al., 2023; Sacchi et al., 2021). This divergence in system boundaries introduces considerable inconsistencies in reported results, limiting the potential for direct cross-study comparisons. Further complicating this issue is the lack of standardization in defining key parameters that are frequently omitted or differently reported, such as vehicle classification, mission type, annual mileage, and vehicle lifetime. For instance, vehicle lifetimes range from 10 (Lommahadthai, 2022) to 15 years (Alonso-Villar et al., 2022), or from 160,000 km (Lyu et al., 2023) to 1,050,000 km (Sacchi et al., 2021). Annual mileage also varies significantly based on the assumed region and vehicle application. Furthermore, differing mission profiles, such as urban, regional, or long-haul, can substantially influence emissions and energy use, as explored in (Sacchi et al., 2021). Regarding vehicle classification, some studies address specific vehicle categories (Baral et al., 2021; El Hannach et al., 2019; van den Oever et al., 2023; Wang et al., 2022), while others rely on generalized assumptions, further hindering the establishment of a consistent analytical basis. Moreover, methodological variability in software tools, background databases, and Life Cycle Impact Assessment (LCIA) methods also contributes to discrepancies in outcomes. Such variations stem from the use of different database versions (e.g., Ecoinvent in (Ricardo-AEA, 2020), GREET in (Wolff et al., 2020)) or LCIA frameworks (e.g., ReCiPe in (Balboa-Espinoza et al., 2023; Baral et al., 2021; Burul, 2020; Cooper, 2019; Hanesch et al., 2022; Lommahadthai, 2022; Osorio-Tejada et al., 2022; Ramshankar et al., 2023; Tahir and Hussain, 2020), or ILCD in (Ferreira et al., 2020; Wolff et al., 2020)).

In the literature, most vehicle-related LCA reviews have primarily focused on alternative fuels (AFs) and powertrains, examining their environmental impacts across the life cycle. For example, (Liu et al., 2025) conducted a systematic review of LCAs on hydrogen as an alternative fuel, with particular attention to electrolysis and natural gas reforming methods. Similarly, (Liu et al., 2023) provided a broad overview of current LCA research on various AFs—such as ammonia, bioethanol, CNG, and hydrogen—highlighting the LCA methodologies and assumptions used in these studies. (Chen et al., 2025) also reviewed LCAs related to hydrogen production, focusing on energy sources and feedstocks. In parallel, (Scrucca et al., 2025) analyzed LCA frameworks applied to lithium-ion batteries for electric vehicles, exploring environmental impacts across the different life cycle stages. Likewise, (Tournaviti et al., 2025) concentrated their review on LCAs of batteries in automotive context. By contrast, (W. Zhang et al., 2024) expanded the scope by reviewing LCAs in the transport sector as a whole, covering multiple modes—road, rail, air, and water. For road transport, despite this work considered not only fuels and powertrains but also broader vehicle-level analyses, methodological framework and carbon footprint output are lacking.

The present work involves a systematic review of 24 publications, encompassing both scientific articles and industrial reports focused on the LCA of HDVs. This systematic literature review makes a timely and significant contribution by providing a detailed synthesis of current applications of LCA to HDVs in freight transport. It clarifies existing methodological inconsistencies, highlights the sensitivity of results to varying assumptions, and underscores the challenges in comparing outcomes across studies. A key novelty of this work is the quantification of representative carbon footprint values for contemporary HDV technologies, based on a critical analysis of recent literature. Furthermore, the review expands the scope of conventional assessments by incorporating studies on zero-emission HDVs, particularly battery electric and hydrogen fuel cell vehicles, offering insight into the environmental implications of emerging technologies. In doing so, this study not only addresses a critical gap in current research but also provides researchers, policymakers, and industry stakeholders with a clearer understanding of the current landscape and future direction of HDV sustainability assessments.

## 3. Methodology

This review has been carried out in line with the systematic review checklist for assessing and reporting reviews of life cycle assessment data (Zumsteg et al., 2012), as well as the more general guidance for conducting research literature reviews (Fink, 2014). A scheme of the main steps needed to conduct a systematic literature review are depicted in Fig. 1 to give an overview of the methodology at a glance. The stages of this process include:

- research problem definition,
- systematic research strategy,
- data collection,
- results analysis

### 3.1. Research problem definition

The first step in a systematic review is to define the research problem in order to clearly frame the issue under investigation. Formulating specific research questions helps identify the key drivers and scope of the study. This review focuses on heavy-duty vehicles (HDVs) with a twofold objective. First, it investigates recent developments in the application of LCA to HDVs, with the aim of tracing the evolution of LCA methodologies within the automotive sector. Second, it examines the carbon footprint results reported in the literature on Internal Combustion Engine (ICE)-HDVs. The review also extends to Zero-Emission HDVs, highlighting current progress and offering insights into future directions.

The results of this paper are supposed to answer the following

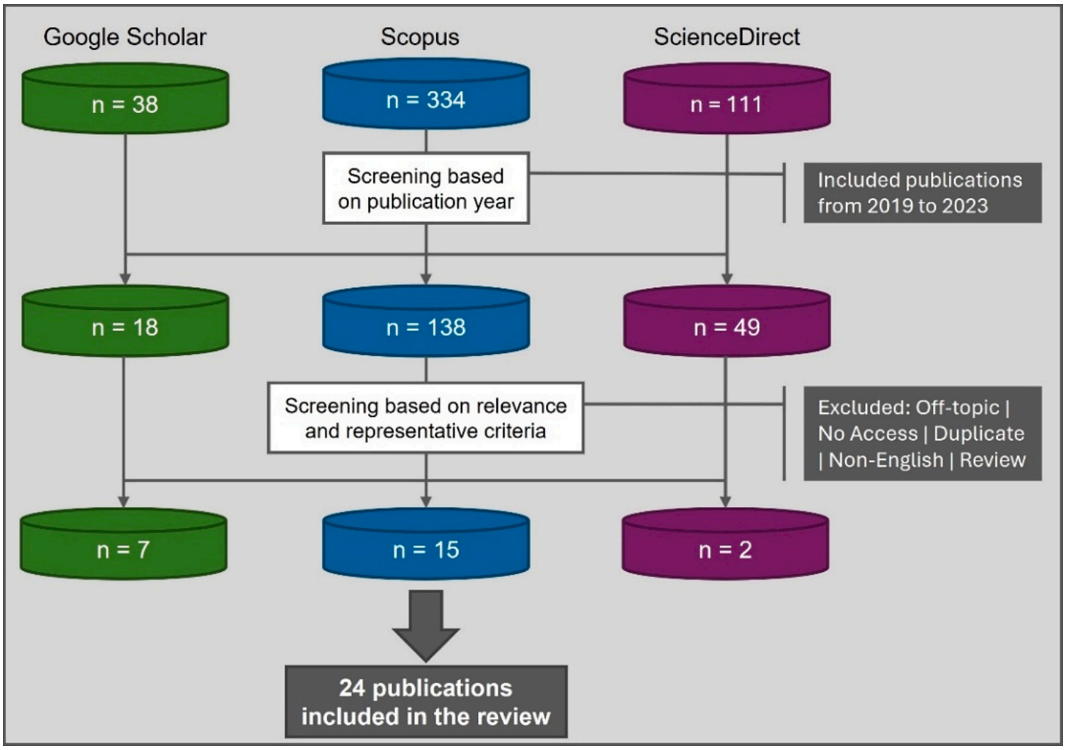


**Fig. 1.** Systematic literature review scheme.

questions:

Q1"What is the state-of-the-art of LCA implementation to HDVs?''

Q2"What are the assumptions, tools and metrics currently used to address the environmental impacts of HDVs in a life cycle perspective?"

Q3"What are the carbon footprint results reported in the latest literature concerning the LCA of HDVs?".

### 3.2. Systematic research strategy

Regarding the strategy, a systematic search was employed using targeted keywords across major scientific platforms. The literature search was conducted using three primary databases: Scopus, ScienceDirect, and Google Scholar. These platforms were selected for their broad coverage of peer-reviewed scientific and technical publications, ensuring a comprehensive and representative review. The search focused on publications related to LCA and HDVs, using the following Boolean keyword combinations: "LCA and heavy and duty and vehicle", "LCA and truck", "life cycle assessment heavy duty vehicles" and "life cycle assessment trucks".

This initial search yielded a total of 483 articles: 334 from Scopus, 111 from ScienceDirect, and 38 from Google Scholar. Two practical criteria were applied to filter the results: time and relevance. The time criterion restricted the selection to studies published between 2019 and 2023. This screening process identified 205 recent publications. The relevance criterion ensured alignment with the study's objectives. Articles were excluded for the following reasons: off-topic focus, lack of access, duplication, non-English language, or classification as review papers. This last screening process resulted in a final selection of 24 relevant publications for in-depth analysis.

### 3.3. Data collection

After applying the aforementioned systematic strategy, 24 filtered publications were selected for analysis. The data extracted from these studies were organized into four main categories:

- Bibliometric
- Technical
- Methodological
- Environmental

Bibliometric data were analyzed using bibliometric techniques, which allow for a structured examination of scientific output and trends within a given field (Donthu et al., 2021). This included the publication year, and keyword frequency—offering insight into research focus and evolution over time.

Technical data were analyzed to extract detailed information about the vehicles under study, including fuel type (e.g., diesel, hydrogen), powertrain configuration (e.g., ICE, fuel cell, battery electric), and vehicle classification. Vehicle classification is particularly relevant in the context of EU Regulation 2017/2400 (European Commission, 2017) which divides N2 and N3 freight vehicles into 18 groups (0–17) based on specifications such as maximum permissible weight. These classifications are critical, as they directly influence emission levels and determine regulatory compliance requirements.

Methodological data refers to how each study applied LCA, including definitions of system boundaries, functional units, impact assessment methods, LCA software, and databases. These methodological choices critically influence the reliability and comparability of LCA results.

Environmental data covers the outcomes of LCA studies, specifically regarding climate change impact. This is typically quantified as Global Warming Potential (GWP) and expressed in kilograms of $CO_2$ equivalent (kg $CO_2$eq), based on greenhouse gas emissions over a vehicle's entire life cycle (IPCC, 2023).

The findings are presented and discussed in the following sections according to these four data categories: bibliometric (Section 3.1), technical (Section 3.2), methodological (Section 3.3), and environmental (Section 3.4).

## 4. Results analysis

### 4.1. Bibliometric results

In this study, bibliometric data have been comprehensively integrated into the analysis to provide a detailed understanding of the research topic. The bibliometric data encompass several significant elements:

- Temporal distribution of publications
- Keyword occurrence.

Fig. 2 illustrates the temporal distribution of publications over the selected time horizon. It reveals a growing academic and industry interest in LCA topic with a significant peak in the years 2022 and 2023. This sharp increase reflects the growing relevance of environmental sustainability and climate change mitigation, particularly in the context of transport systems.

For the keyword analysis, VOSviewer was employed to construct and visualize bibliometric networks (van Eck and Waltman, 2023). Fig. 3 displays the network of co-occurring keywords extracted from the analyzed literature. In this visualization, each keyword is represented by a label and a circle, the size of which corresponds to its frequency of occurrence. A total of 44 keywords were identified. Among these, “life cycle assessment” emerged as the most frequently used term, appearing in eight instances. Other commonly used keywords include “climate change” (4 times), “hydrogen” (3 times), and “greenhouse gas emissions”, “total cost of ownership”, “sustainable freight transportation”, and “biofuels”—each appearing twice. Keywords with only one occurrence were not included in the network visualization to maintain clarity and focus. Overall, the bibliometric findings provide valuable insights into the evolution and current focus areas of LCA research in transportation. The convergence of frequent terms around sustainability, emissions, and energy alternatives underscores a shift toward more integrated, multi-dimensional approaches in transport sustainability studies.

### 4.2. Technical results

This literature review aims to compile and analyze technical data to deepen understanding of the defining characteristics of HDVs assessed LCAs, and to evaluate how variations in technical parameters influence environmental impacts. A key focus of the analysis was the classification of the vehicles studied. Despite the importance of vehicle classification in enabling meaningful comparisons across studies, only 3 out of the 24

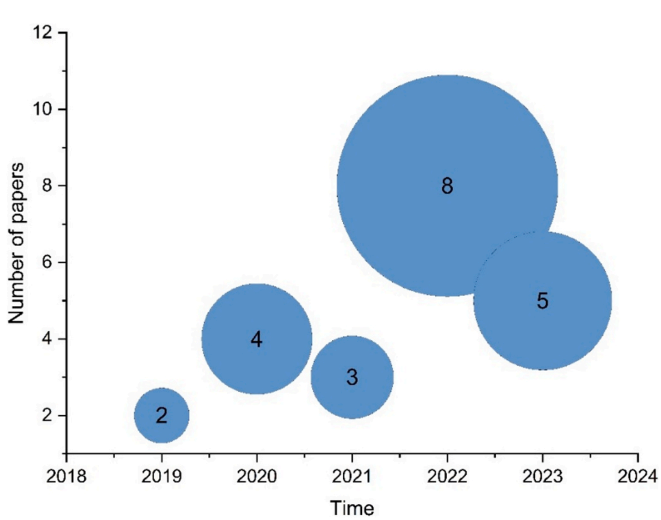


**Fig. 2.** Published papers on LCA application to HDVs from 2019 to 2023.

reviewed publications explicitly specified the vehicle class under consideration. Among these, two studies focused on class 8 trucks—the heaviest segment of commercial vehicles—while one investigated class 3–5 trucks, representative of the medium-duty segments. The general absence of vehicle classification across the majority of studies represents a significant methodological limitation. Without clear classification, it becomes difficult to contextualize results or ensure comparability across studies, as vehicle class is a major determinant of fuel consumption, emissions, and regulatory relevance. In fact, (Sacchi et al., 2021) highlight that variations in key parameters, such as driving range, vehicle size, and payload capacity, substantially influence the results (e.g., the GHG emissions per ton-km reduce as vehicle size and load factor increase).

Special attention was also given to the types of powertrain technologies and the fuels or energy sources utilized in the vehicles analyzed. Fig. 4 illustrates the distribution of the reviewed papers according to the powertrain technologies investigated. The vertical axis indicates the number of studies while the horizontal axis categorizes the powertrains into diesel ICEVs, BEVs, FCEVs, and a grouped “Others” category (i.e., natural gas, liquefied natural gas, biodiesel, hybrid electric systems, biofuels, and plug-in hybrid electric powertrains). Diesel ICEVs and BEVs dominated the literature, appearing in 20 and 19 studies respectively, while FCEVs were covered in only 8. This highlights a research gap, especially given the growing interest in hydrogen-powered transport. As (Camacho et al., 2022) noted, only 95 publications between 2012 and 2022 addressed FC trucks at all—focusing mainly on policy, hydrogen supply chains, and drivetrain technologies. (Usai et al., 2021) attribute this gap to the limited commercial maturity of fuel cell systems, which makes robust LCA difficult.

It is also important to note that some studies assessed multiple powertrain types within a single analysis, which explains why the total number of powertrain assessments exceeds the number of reviewed publications (n = 24). This trend reflects a growing effort within the research community to compare the environmental performance of emerging ZEVs against conventional diesel-powered alternatives. The increasing stringency of emissions regulations, in fact, is driving a rise in the adoption and development of ZEVs, as well as a greater reliance on LCA methodology. The push towards ZEVs is driven by their potential to eliminate tailpipe emissions and significantly reduce the life cycle GHG emissions of HDVs, aligning with global climate targets (Hawkins et al., 2013).

An essential aspect of evaluating the environmental impact of HDVs within LCAs is the analysis of the use phase, particularly the estimation of representative fuel and energy consumption values. This phase is critical, as it substantially influences the overall environmental performance and operational efficiency of the vehicle. However, despite its significance, there is currently no universally accepted LCA standard or methodology for estimating fuel and energy consumption during vehicle operation. As a result, studies employ a wide range of approaches, resulting in inconsistencies that hinder comparability across literature. This review reveals substantial variability in methodological assumptions. For instance, in ten of the reviewed studies, the methodology used to determine fuel and energy consumption is not explicitly described. In another ten studies, consumption estimates are derived from secondary sources, typically by referencing values reported in prior scientific literature.

The remaining studies rely on the Vehicle Energy Consumption Calculation Tool (VECTO) to derive fuel and energy consumption values. Developed by the European Commission, VECTO is the official simulation tool for assessing the energy consumption and $CO_2$ emissions of HDVs. It plays a crucial role in regulatory frameworks, as it is used for the official declaration of $CO_2$ emissions in compliance with European environmental policies. By simulating vehicle operation under different driving cycles and load conditions, VECTO provides a standardized and policy-aligned approach to fuel and energy consumption estimation (European Commission, 2017).

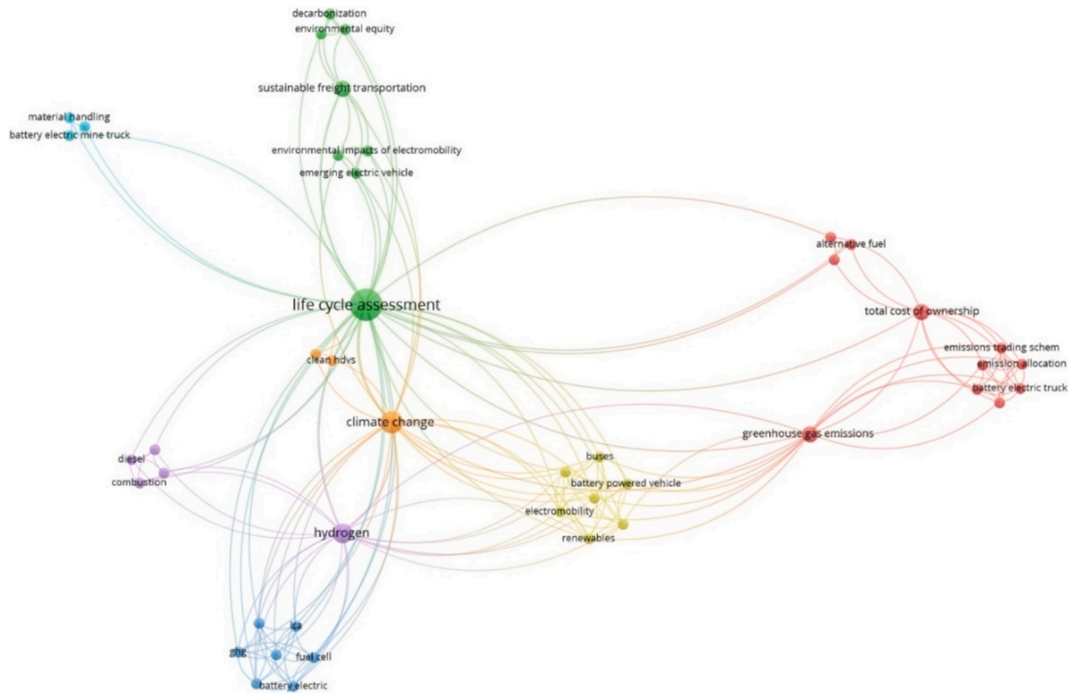


**Fig. 3.** Keyword occurrences and links between co-occurring keywords.

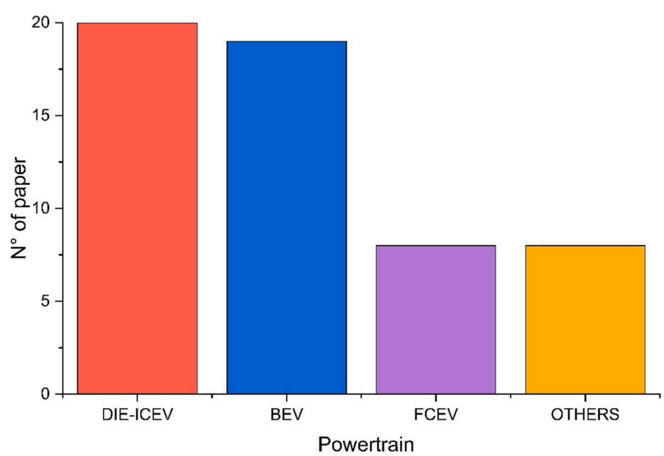


**Fig. 4.** Comparison of the number of papers focused on different powertrain technologies across the reviewed publications.

Table 1 shows $CO_2$ emission values of heavy-duty vehicles produced for the European market during the second half of 2019, determined using VECTO, in line with (European Commission, 2017). The table highlights the average $CO_2$ emissions per vehicle subgroup. These vary significantly as a result of the very different mission profiles, average payloads and annual mileages. Vehicle group 5 (i.e., 4x2 tractors with a laden mass over 16 tonnes and for long haul missions) represents the most significant and best-selling subgroup (i.e., 62.8 % sale share). It shows an average $CO_2$ emissions value of 56.5 g/tkm, representing the largest share—68.2 %—of total $CO_2$ emissions across all the vehicle classes analyzed (ACEA, 2020).

Table 2 shows the fuel consumption and the emission factors for CO, NMVOC, $NO_X$, $N_2O$, $NH_3$, Pb, PM (i.e., considered to be $PM_{10}$), and $CO_2$ per aftertreatment technology and vehicle-type. These average European emission factors were determined by using typical values for driving speeds, ambient temperatures, highway-rural–urban mode mix, trip length, etc (EMEP/EEA, 2024). Fuel consumption and emission factors vary significantly across different vehicle types. As expected, heavier vehicles tend to exhibit higher fuel consumption and $CO_2$ emissions. For instance, $CO_2$ emissions increase from approximately

**Table 1**
Average CO2 performance per VECTO group (ACEA, 2020).

| VECTO GROUP | Sale share | Max laden mass[a] (ton) | Chassis configuration | Mission profile | Configuration | Average $CO_2$ (g/km) | Payload (ton) | Annual mileage (km) |
|---|---|---|---|---|---|---|---|---|
| 4 | 7.9 % | 7.5–16 | Rigid | Regional | 4x2 | 198.1 | 3.2 | 78,000 |
| 4 | 1.9 % | 7.5–16 | Rigid | Long haul | 4x2 | 102.9 | 7.4 | 98,000 |
| 5 | 0.8 % | >16 | Tractor | Regional | 4x2 | 84 | 10.3 | 78,000 |
| 5 | 62.8 % | >16 | Tractor | Long haul | 4x2 | 56.5 | 13.8 | 116,000 |
| 9 | 7.2 % | All weights | Rigid | Regional | 6x2 | 110.9 | 6.3 | 73,000 |
| 9 | 9.2 % | All weights | Rigid | Long haul | 6x2 | 64.7 | 13.4 | 108,000 |
| 10 | 0.1 % | All weights | Tractor | Regional | 6x2 | 84 | 10.3 | 68,000 |
| 10 | 9.7 % | All weights | Tractor | Long haul | 6x2 | 58.6 | 13.8 | 107,000 |

[a] Technically permissible maximum laden mass: the maximum mass allocated to a vehicle on the basis of its construction features and its design performances (European Commission, 2012); the total weight of the vehicle and of the load declared permissible by the competent authority of the country of registration of the vehicle (Eurostat, 2025).

**Table 2**
Fuel consumption and exhaust emission factors per vehicle type (EMEP/EEA, 2024).

| Vehicle type (Rigid) | Aftertreatment technology | FC (g/km) | CO (g/km) | NMVOC (g/km) | $NO_x$ (g/km) | $N_2O$ (g/km) | $NH_3$ (g/km) | Pb (g/km) | $CO_2$ (g/km) |
|---|---|---|---|---|---|---|---|---|---|
| <= 7.5 t | Euro VI A/B/C | 101 | 0.053 | 0.010 | 0.167 | 0.017 | 0.009 | 1.71E-04 | 326.146 |
| <= 7.5 t | EURO VI D/E | 101 | 0.053 | 0.009 | 0.119 | 0.017 | 0.009 | 1.71E-04 | 326.146 |
| 7.5–12 t | Euro VI A/B/C | 155 | 0.086 | 0.016 | 0.249 | 0.017 | 0.009 | 1.71E-04 | 460.900 |
| 7.5–12 t | EURO VI D/E | 155 | 0.086 | 0.015 | 0.192 | 0.017 | 0.009 | 1.71E-04 | 460.90 |
| 12–14 t | Euro VI A/B/C | 155 | 0.099 | 0.018 | 0.264 | 0.035 | 0.009 | 1.71E-04 | 462.848 |
| 12–14 t | EURO VI D/E | 155 | 0.099 | 0.017 | 0.198 | 0.035 | 0.009 | 1.71E-04 | 462.848 |
| 14–20 t | Euro VI A/B/C | 155 (<16 t) 210 (>16 t) | 0.143 | 0.023 | 0.409 | 0.035 | 0.009 | 1.71E-04 | 546.063 |
| 14–20 t | EURO VI D/E | 155 (<16 t) 210 (>16 t) | 0.144 | 0.022 | 0.315 | 0.035 | 0.009 | 1.71E-04 | 546.063 |
| 20–26 t | Euro VI A/B/C | 210 | 0.146 | 0.028 | 0.437 | 0.035 | 0.009 | 1.73E-04 | 676.189 |
| 20–26 t | EURO VI D/E | 210 | 0.146 | 0.026 | 0.324 | 0.035 | 0.009 | 1.73E-04 | 676.189 |
| 26–28 t | Euro VI A/B/C | 210 | 0.166 | 0.028 | 0.452 | 0.035 | 0.009 | 1.73E-04 | 723.332 |
| 26–28 t | EURO VI D/E | 210 | 0.167 | 0.027 | 0.339 | 0.035 | 0.009 | 1.73E-04 | 723.332 |
| 28–32 t | Euro VI A/B/C | 210 | 0.189 | 0.031 | 0.470 | 0.053 | 0.009 | 1.73E-04 | 831.839 |
| 28–32 t | EURO VI D/E | 210 | 0.189 | 0.029 | 0.351 | 0.053 | 0.009 | 1.73E-04 | 831.839 |
| >32 t | Euro VI A/B/C | 251 | 0.159 | 0.032 | 0.445 | 0.053 | 0.009 | 1.73E-04 | 817.309 |
| >32 t | EURO VI D/E | 251 | 0.159 | 0.030 | 0.320 | 0.053 | 0.009 | 1.73E-04 | 817.309 |

326 g/km for rigid trucks with a Gross Vehicle Weight (GVW) lower 7.5 tonnes to over 830 g/km for those with a GVW of 28–32 tonnes. This trend is consistent across most pollutants, with heavier vehicle categories also emitting more CO, NMVOC, $NO_x$ and PM.

For BEVs, the energy supply chain (i.e., well-to-tank (WTT) contribution) often accounts for the largest share of the total carbon footprint. However, the dominance of this stage is highly sensitive to key assumptions, such as the carbon intensity of the electricity mix used, as well as the size and type of vehicle being assessed. These factors play a crucial role in shaping the overall environmental profile of EVs (Ricardo-AEA, 2020). In this research, particular attention is given to analyzing the electricity mix and energy consumption assumed in the publications under study. Fig. 5 shows the country electricity mixes (left graph) and the energy consumption results (right graph). The most used mix is the average European electricity mix, utilized in 8 papers. The German electricity mix is the second most used, appearing in 2 papers. Other regions include British Columbia, Chile, China, Colombia, Iceland, India, Malaysia, Norway, Spain, Thailand, the United Kingdom, and New Zealand. The variation in electricity mixes results in a broad spectrum of environmental outcomes, driven by significant regional differences in carbon intensity and energy sources. These disparities underscore the critical role that the electricity grid's composition plays in shaping the overall environmental impact of EVs. For instance, regions with a greater share of renewable energy in their electricity mix achieve significantly lower GHG emissions for BEVs compared to regions heavily dependent on fossil fuels (Xia et al., 2022). The reported energy consumption values for BEVs exhibit substantial variability, ranging from 0.04 to 2.61 kWh/km. This wide range underscores the significant influence of various factors, including vehicle efficiency, driving conditions, and the methodologies employed in assessments.

Battery degradation and the potential need for replacement during a vehicle's lifetime are critical factors in LCAs of BEVs. Over time, battery degradation leads to a decline in efficiency, resulting in increased energy consumption during the vehicle's operational phase. This degradation

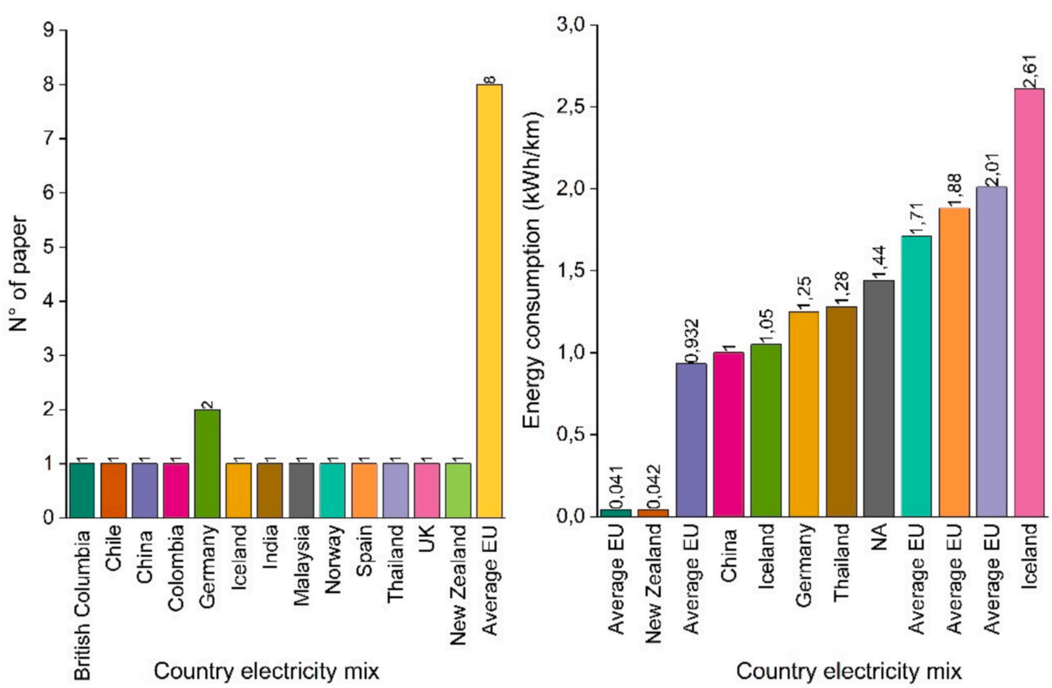


**Fig. 5.** Country electricity mixes of BEVs (left) and energy consumption during operation (right) in the considered publications.

significantly impacts the WTT phase, which is a substantial contributor to emissions in BEVs. A study by (Y. Zhang et al., 2024) demonstrated that when a power battery degrades to 85 % of its original capacity, the fossil energy consumption during the operation phase increases by 73 % for BEVs compared to scenarios without degradation. Despite this importance, battery degradation is often underrepresented in literature. Additionally, battery replacement, required when performance drops below acceptable thresholds, adds further emissions, especially during the battery manufacturing phase. This phase is energy-intensive and involves the extraction and processing of critical raw materials such as lithium, nickel, cobalt, and manganese (Lai et al., 2022). These processes contribute substantially to the environmental burden of BEVs. Despite its significance, battery replacement is not widely considered in existing studies. Among the 19 reviewed publications, only five explicitly account for battery replacement over the vehicle's lifetime (Balboa-Espinoza et al., 2023; Jahangir Samet et al., 2023; Lommahadthai, 2022; Lyu et al., 2023; Sacchi et al., 2021). For instance, (Wang et al., 2022) estimate the cradle-to-gate carbon footprint of a BEV truck equipped with a 260 kWh battery, an electric range of 240 km, and a lifetime of 800,000 km at 99,900 $kgCO_2eq$. In this case, the battery alone accounts for 19 % of the total footprint, based on GWP of 71.5 $kgCO_2eq/kWh$ as reported by (Koroma et al., 2023). When battery replacement is considered, the carbon footprint increases by 19 % at the vehicle level, further highlighting the substantial impact of this factor on overall emissions. Moreover, the environmental impact of charging infrastructure represents another critical yet frequently neglected component in LCA studies. While the development of charging networks is essential to the long-term sustainability and scalability of electric mobility, only two of the reviewed studies incorporate this factor into their assessments.

In the case of FCEVs, both hydrogen consumption and the pathways used for hydrogen production are critical determinants of their overall life cycle carbon footprint. The reviewed studies report hydrogen consumption values ranging from 0.054 to 0.107 kg $H_2$ per kilometer, underscoring substantial variability in energy efficiency across vehicle models and operational contexts.

One of the most influential factors affecting the environmental performance of FCEVs is the method by which hydrogen is produced. Hydrogen can be generated through a range of pathways, each associated with distinct emissions profiles. Among the studies analyzed, three focus specifically on green hydrogen (Alonso-Villar et al., 2022; Lozanovski et al., 2020; Sacchi et al., 2021), produced via electrolysis powered by renewable energy sources, which offers the lowest life cycle GHG emissions and is considered the most environmentally sustainable option. Another two studies investigate grey hydrogen (Baral et al., 2021; Ricardo-AEA, 2020), generated from natural gas through steam methane reforming (SMR), a process that results in significantly higher GHG emissions. Only one study provides a comparative assessment of both green and grey hydrogen pathways (Wang et al., 2022), offering a more nuanced understanding of their respective environmental impacts. According to (Wang et al., 2022), the cradle-to-gate carbon footprint of an FCEV truck is 55,000 $kgCO_2eq$, assuming a total lifetime of 800,000 km and a hydrogen consumption rate of 0.054 kg $H_2/km$. Based on values reported by (Accardo et al., 2025), hydrogen production via SMR, the most widely adopted pathway, emits 16 $kgCO_2eq/kg\ H_2$, while alkaline electrolysis (AE), a leading green hydrogen technology, emits 3.67 $kgCO_2eq/kg\ H_2$. Using these values, the hydrogen production phase accounts for approximately 93 % and 74 % when SMR and AE are used, respectively. However, the broader literature remains limited in its treatment of hydrogen supply chain variability and its influence on the life cycle emissions of FCEVs. In particular, a critical research gap is the insufficient examination of how increasing the share of renewable electricity in the energy mix affects the environmental outcomes of hydrogen production. Given the dynamic nature of energy systems and the growing penetration of renewables, future studies should more comprehensively explore these evolving conditions to accurately assess the long-term sustainability of FCEVs.

A critical gap identified in the reviewed literature is the limited analysis of how an increasing share of renewable energy in the electricity mix influences the life cycle impacts of hydrogen production. (Drawer et al., 2024) compared hydrogen production using the European electricity mix versus electricity derived entirely from wind energy. Their findings demonstrate that utilizing 100 % renewable electricity for electrolysis significantly reduces GHG emissions across the entire life cycle. These observations align with those made for BEVs and can be similarly extended to FCEVs. However, key factors that influence the environmental performance of FCEVs remain largely overlooked in current literature. Notably, fuel cell degradation over time has a direct impact on vehicle efficiency, leading to increased hydrogen consumption and higher operational emissions. Despite its relevance, none of the reviewed studies incorporate fuel cell degradation into their analyses, representing a major omission in capturing real-world environmental impacts. According to the findings of (Ahmadi, 2022), a hydrogen vehicle with a degraded fuel cell consumes 14.3 % more fuel than a fresh fuel cell hydrogen vehicle. Fuel cell replacement is another underexplored dimension. As fuel cells degrade and performance deteriorates, replacement becomes necessary over the vehicle's lifetime. This process adds to the overall environmental burden, particularly through the manufacturing and sourcing of critical materials such as platinum, a key component in fuel cell catalysts. Nevertheless, only one of the reviewed studies accounts for fuel cell replacement, highlighting a substantial research gap in this area (Sacchi et al., 2021). For example, (Wang et al., 2022) estimate a cradle-to-gate carbon footprint of 55,000 $kgCO_2eq$ for an FCEV truck with a lifetime of 800,000 km. Based on the GWP of 47.9 $kgCO_2eq/kW$ for fuel cell production reported by (Gentilucci et al., 2024), it can be estimated that the fuel cell accounts for approximately 10 % of the vehicle's total carbon footprint. If one fuel cell replacement is assumed over the vehicle's lifetime, the carbon footprint would increase by 10 % at the vehicle level.

Additionally, the environmental impacts associated with hydrogen refueling infrastructure, including the production, deployment, and maintenance of refueling stations, are seldom addressed in LCAs. Although this infrastructure contributes meaningfully to the total life cycle emissions of FCEVs, none of the studies reviewed included it in their assessments.

### 4.3. Methodological results

In LCA studies, methodological data encompass the fundamental parameters, assumptions, and methodological choices that guide the structure and execution of the analysis. These elements are essential to ensure consistency, transparency, and reliability of the results. In this review, key methodological aspects were systematically collected and analyzed, including geographical coverage, system boundaries, functional unit, annual mileage, vehicle lifetime, the LCA software and databases employed, Life Cycle Inventory (LCI) data sources, the life cycle impact assessment (LCIA) methodologies used, and the scope of impact categories assessed. Each of these components plays a pivotal role in shaping the outcomes and comparability of LCA studies, and they are examined in detail in the following sections.

#### 4.3.1. Geographical coverage/scope

The geographical scope of LCA studies on HDVs underscores the global importance of the transport sector and its environmental impacts. The choice of geographical context is particularly critical because factors such as the electricity grid composition significantly influence environmental outcomes. However, many reviewed studies do not explicitly report their geographical boundaries, limiting transparency and comparability of results. When declared, as shown in Fig. 6, most studies focus on Europe (58 %), reflecting the region's stringent environmental policies, emission regulations, and accelerated adoption of ZEVs. Other regions such as the Americas (21 %), Asia (13 %), Oceania, and the UK (each about 4 %) are less frequently studied, indicating a

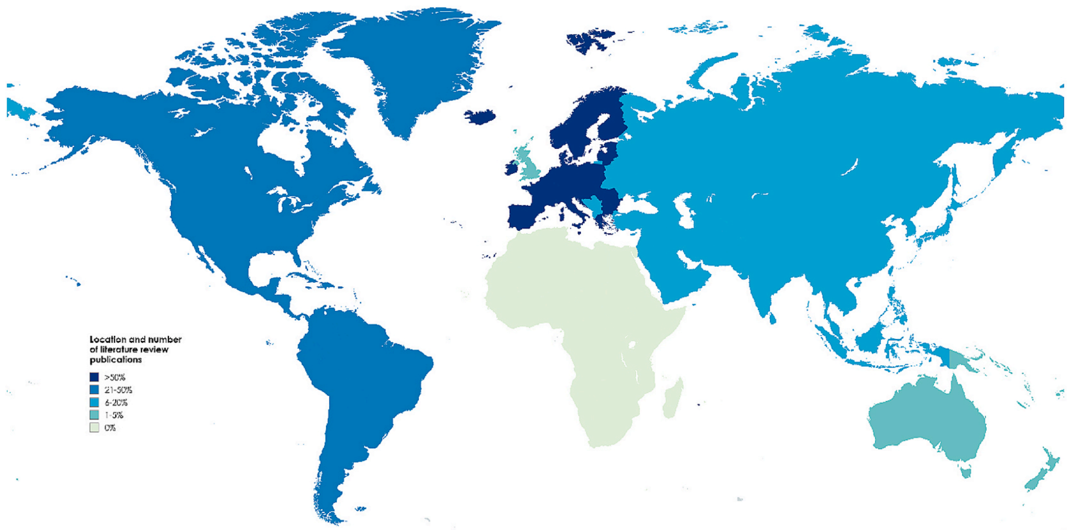


**Fig. 6.** Geographical coverage of literature review publications.

disproportionate concentration of research efforts.

#### 4.3.2. System boundary

In LCA studies, the definition of the system boundary is a pivotal step that delineates which life cycle stages and unit processes are encompassed in the analysis. A systematic review of the literature revealed substantial heterogeneity in system boundary specifications, indicative of the varied objectives and emphases across different investigations. As depicted in Fig. 7, among the considered studies, 10 adopted a cradle-to-grave boundary, thereby encompassing all phases from raw material extraction through end-of-life disposals. Two studies applied a cradle-to-use boundary, covering stages from resource acquisition to the operational phase of the vehicle. Five studies utilized a cradle-to-gate approach, incorporating processes from raw material extraction to manufacturing, while a single study employed a gate-to-gate boundary, focusing exclusively on the manufacturing stage. Moreover, six studies concentrated solely on the use phase, commonly referred to as the well-to-wheel stage, encompassing fuel or energy consumption during vehicle operation. Despite the methodological diversity observed, the cradle-to-grave system boundary remains predominant, underscoring its comprehensive capacity to capture the full spectrum of environmental impacts throughout the vehicle life cycle. Nevertheless, sever methodologies exist to account for the End-of-Life (EoL) emissions. For instance,

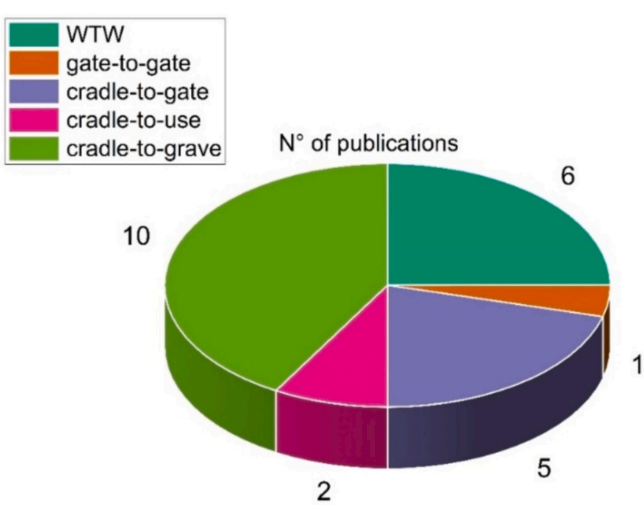


**Fig. 7.** System boundary assumption in the considered publications.

in (Ricardo-AEA, 2020), where boundary encompasses the whole vehicle life cycle, Product Environmental Footprint (PEF) Circular Footprint Formula (CFF) is applied. Materials and impacts from battery recycling are based on GREET data and methodologies. EoL accounting considers implications of second-life batteries, using a credit applied based on assumptions for the avoided use of an equivalent new energy storage battery. In contrast, no other studies take the CFF into account. Yet, the use of CFF offers a robust approach to properly accounting for credits associated with material recycling. For example, (Drawer et al., 2024) applied the CFF to a hydrogen fuel cell truck and reported a notable reduction in environmental impact, with mineral resource scarcity reflecting a credit of 370 kg Cu-eq due to recycling benefits. Moreover, the studies reviewed here that adopt a cradle-to-grave system boundary often lack transparency regarding the allocation methods used, recycling techniques applied, recovery efficiencies, or data sources.

The set of graphs in Fig. 8 provides an overview of various assumptions, tools and metrics used in the LCA studies of HDVs.

#### 4.3.3. Functional unit

The scope of a LCA must clearly define the functions of the system under study. The selection of the functional unit is a fundamental aspect of LCA methodology, as it serves as the reference against which all environmental impacts are normalized, enabling comparability across different studies (ISO 14044, 2018). Given that the objective of the reviewed publications is to evaluate the environmental impacts of HDVs, the chosen functional unit must be consistent with the goal and scope of each study. A variety of functional units were identified in the review of LCA studies on HDVs, reflecting diverse methodological approaches. Fig. 8a illustrates the functional units used. The most frequently reported category was "NA" (not available), followed by "tkm" (ton-kilometers), "km" (kilometers), and "1 truck."

The unit "tkm" refers to the transportation of one ton of goods over one kilometer by the vehicle under analysis, assessed across the vehicle's entire lifetime. This unit, adopted by 13 out of 24 studies, provides a comprehensive metric for transport efficiency, accommodating differences in payload and vehicle capacity (Börjesson, 2015). The unit "km" was used in 7 studies, while "1 truck" was reported as the functional unit in one study. In 4 studies, the functional unit was not disclosed. The lack of consistency in the selection and reporting of functional units presents a significant barrier to the interpretation and comparison of results across studies. This underscores the importance of clearly defining the system boundaries, study objectives, and methodological choices in

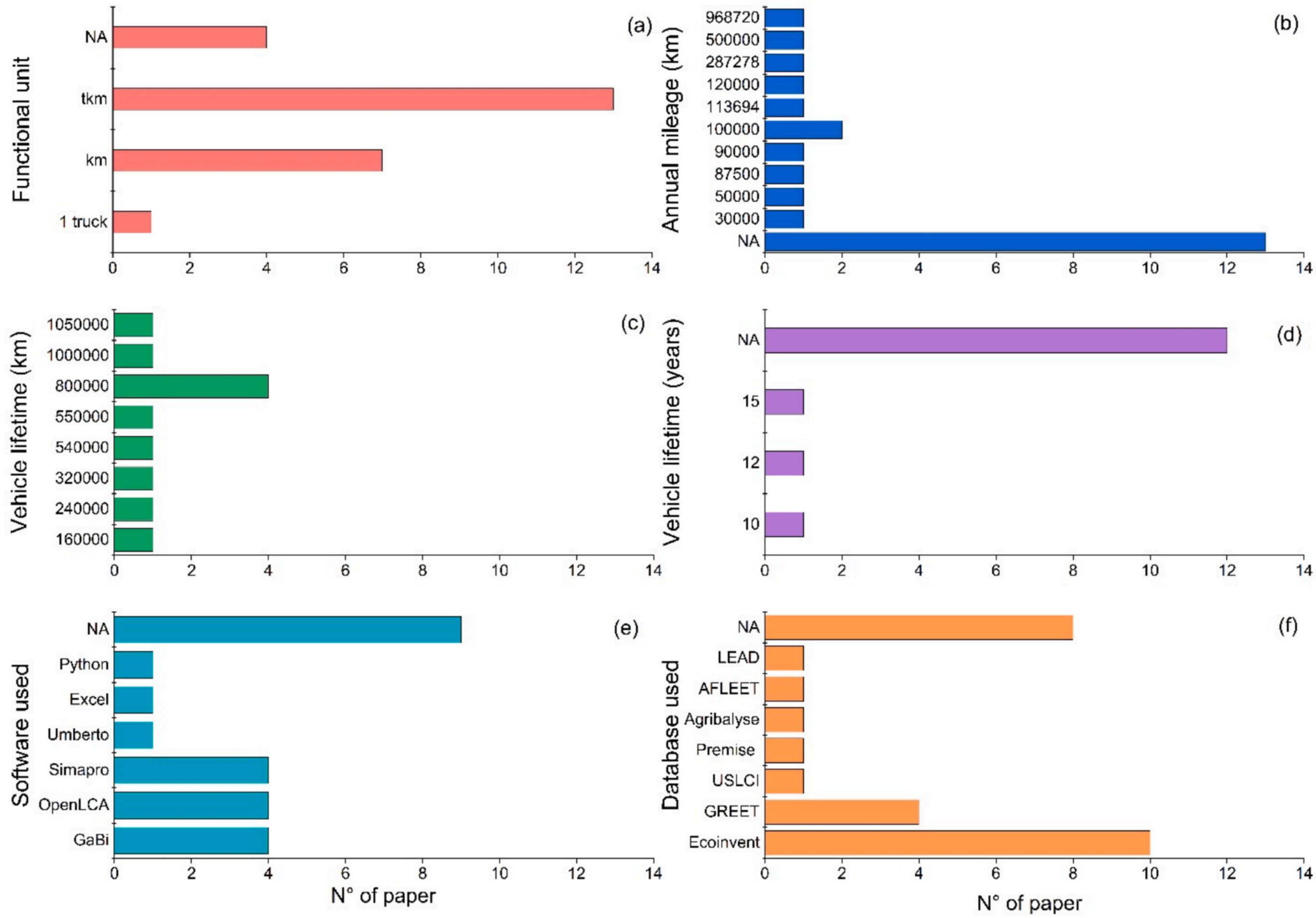


**Fig. 8.** Overall results of the methodological literature review related to the following LCA assumptions, tools and metrics: a) functional unit, b) annual mileage, c) vehicle lifetime, d) vehicle lifetime (years), e) software and f) database.

each LCA. The selection of the functional unit has a direct impact on the assessment of environmental impacts and the identification of improvement opportunities throughout the HDV life cycle.

*4.3.4. Annual mileage*

Annual mileage parameter exhibits substantial variation across different vehicle categories, primarily due to disparities in usage intensity, functional purpose, and operating conditions. As illustrated by the green bar chart in the upper-right quadrant of Fig. 8b, annual mileage is a key determinant of environmental performance outcomes. The literature review revealed a broad range of reported annual mileage values, reflecting significant heterogeneity in operational contexts, powertrain technologies, and vehicle applications. Specifically, annual mileage figures reported in the reviewed studies span from approximately 30,000 to 100,000 km per year.

*4.3.5. Vehicle lifetime*

Vehicle lifetime is typically defined as the total service duration or cumulative distance a vehicle is expected to cover before being decommissioned. It is generally expressed either in terms of years or kilometers. However, both the reported values and the units of measurement vary considerably across literature. Fig. 8c illustrates vehicle lifetime expressed in kilometers, with reported values ranging from a minimum of 160,000 km to a maximum of 1,050,000 km. The most frequently cited value is 800,000 km, which appears in four studies. Notably, the Euro 7 regulation has established 700,000 km as the minimum required lifetime for heavy-duty vehicles (HDVs) (European Parliament and Council, 2024). Despite this regulatory threshold, it has not yet seen widespread adoption in the reviewed literature.

Fig. 8d presents lifetime values expressed in years. Only three studies reported time-based lifespans, specifying 10, 12, and 15 years, respectively. In contrast, the Euro 7 standard assumes a default operational lifetime of 15 years for HDVs (European Parliament and Council, 2024). In the majority of cases, specifically, 13 out of the 24 studies, the vehicle lifetime was not disclosed. In the draft delegated act of the new EU battery regulation (European Parliament, 2023), although it does not provide for a vehicle-level calculation, the use of the commercial warranty is introduced as a possible basis for estimating battery lifetime. This marks a step toward recognizing commercially available warranty data as a valid input for assessing battery durability and long-term performance.

*4.3.6. LCA software*

LCAs are typically conducted using specialized software tools designed to facilitate and standardize the analysis process. These tools are essential for automating the LCIA phase, the third stage of the LCA methodology, which translates LCI data into quantifiable environmental impact categories. The use of dedicated LCA software is critical to ensuring both the accuracy and the reproducibility of results, particularly in studies involving complex systems such as HDVs. Despite the importance of transparency in methodological choices, the literature review revealed that the specific software employed is frequently omitted in published studies. Fig. 8e summarizes the reported software tools used through the reviewed literature. The most commonly cited platforms were GaBi, SimaPro, and OpenLCA, each of which appeared in four studies. In contrast, Umberto NXT and a Python-based calculator developed specifically for truck assessments were each mentioned in only one study.

#### *4.3.7. LCA database*

The selection of scientifically robust and up-to-date databases is a critical component of LCA, as background data significantly influence the accuracy, consistency and credibility of environmental impact results (ISO 14044, 2018). Fig. 8f summarizes the databases referenced across the reviewed studies. Among these, Ecoinvent emerges as the most frequently used, cited in ten papers. Its widespread adoption is likely attributed to its comprehensive, peer-reviewed datasets, which cover a broad range of industrial processes and geographic regions. GREET was cited in four studies and is particularly valued for its detailed modeling of energy systems and transportation-related fuel cycles. Other databases, including USLCI, Premise, Agribalyse, Afleet, and Lead, were each referenced in only one paper, potentially due to their narrower geographic applicability or sector-specific focus. Notably, nine studies did not specify the database used. This lack of transparency presents a significant barrier to reproducibility and highlights the continued need for standardization in LCA reporting practices.

#### *4.3.8. Life cyle inventory*

The LCI represents the second phase of the LCA, during which data collection is carried out for all processes included within the defined system boundaries. Qualitative and quantitative data shall be collected for each unit process that is included within the system boundary. Both qualitative and quantitative data must be gathered for each unit process in order to quantify the respective inputs (e.g., raw materials, energy) and outputs (e.g., emissions, waste) (ISO 14044, 2018). Detailed and transparent documentation of LCI data is essential, as it directly influences the replicability and credibility of the study. However, the reviewed literature revealed a significant limitation: only 6 out of 24 studies reviewed provided sufficiently detailed and clearly reported LCI data to allow for full replicability. This lack of transparency poses challenges for comparative analyses and undermines the reliability of the results presented in the broader context of LCA-based decision-making.

#### *4.3.9. Life cycle impact assessment*

According to (ISO 14044, 2018), the LCIA phase of LCA is intended to understand and evaluate the magnitude and significance of the potential environmental impacts associated with a product system. Specifically, this phase translates inventory data—such as emissions and resource use—into environmental impact indicators linked to defined areas of protection (e.g., human health, ecosystems, and resource availability). The selection of an LCIA method is crucial as it determines the framework for evaluating potential environmental impacts across various categories. Among the LCIA methods identified in the reviewed literature, ReCiPe has emerged as the most frequently applied method, cited in nine studies. ReCiPe provides midpoint and endpoint indicators and offers a broad coverage of impact categories, making it particularly suitable for comprehensive assessments. The IPCC method, which focuses primarily on global warming potential (GWP), was employed in four studies. Two studies each reported the use of the CML and ILCD methods, both of which are established frameworks offering midpoint indicators aligned with ISO standards. The Environmental Footprint (EF) 3.0 method, developed by the European Commission to support policy and product environmental performance analysis, was cited in one paper. Notably, seven publications did not specify the LCIA method employed, limiting the comparability and reproducibility of the findings.

#### *3.3.10. Impact category coverage*

In the LCIA phase, the selected methodology strongly influences the scope and granularity of results. Studies limited to GHG emissions are categorized as carbon footprint assessments, focusing solely on climate change. In contrast, comprehensive LCAs evaluate multiple environmental impact categories, offering a more holistic view and enabling the identification of trade-offs and co-benefits.

The literature review revealed an even split: 12 studies conducted full LCAs using diverse LCIA methods, while the remaining 12 focused exclusively on climate change. A broader approach is essential for capturing environmental burdens beyond GHGs. For example, raw material extraction for batteries significantly affects resource depletion, particularly in the “resource use, minerals and metals” category (European Commission, 2021; Peters et al., 2017). Similarly, (Rial et al., 2021) found that natural gas trucks reduce terrestrial eutrophication and acidification compared to diesel, underscoring the value of multi-category impact assessments.

#### *4.3.11. Uncertainty analysis*

Uncertainty analysis is a critical element of LCA studies. However, only 7 out of 24 reviewed papers addressed this issue. Three out of 24 reviewed papers were transparent in the uncertainty methodology and used the Monte Carlo method to quantify how input variability affects outcomes. For instance, (Baral et al., 2021) ran 10,000 simulations to assess uncertainty in feedstock costs, butanol prices, and emissions. Lyu et al. (2023) also used Monte Carlo analysis, while (Lommahadthai, 2022) combined it with process-based and Economic Input Output LCA (EIO-LCA) models to address data gaps and regional energy variability. Other studies conducted uncertainty analyses without specifying the method. Hanesch et al. (2022) performed a sensitivity analysis focusing on those LCI parameters with the highest impact. Sacchi et al. (2021) organized inputs by powertrain, size class, and year in the European context, using uncertainty distributions for error propagation. Cooper (2019) explored a wide range of fuel consumption values to account for uncertainty in the environmental outcomes, while (Jahangir Samet et al., 2023) examined best- and worst-case scenarios for key variables.

### *4.4. Environmental analysis*

The literature review focused on collecting environmental data, specifically carbon footprint values, for diesel ICEVs, BEVs, and FCEVs. Since environmental impacts are often reported relative to functional units, these values were converted into absolute GHG emissions (i.e., expressed in $tCO_2eq$) to enable meaningful cross-study comparisons. This conversion incorporated factors such as annual mileage, vehicle lifetime, and payload.

Fig. 9 and Table 3 present the GWP results for diesel ICEVs differentiated by system boundaries. Cradle-to-gate values range from 30 to 129 $tCO_2eq$, reflecting emissions from raw material extraction through manufacturing. Extending the boundary to cradle-to-use, emissions increase substantially, ranging from 450 to 753 $tCO_2eq$, as operational energy use and fuel consumption are included. The most comprehensive cradle-to-grave assessments, covering the entire lifecycle including end-of-life processes, report the widest GWP range—from 431 to 1780 $tCO_2eq$. This variability is influenced by differences in system scope, vehicle configurations, operational efficiency, and assumed service life.

HDVs consistently exhibit higher energy demands and emissions compared to lighter vehicles due to their substantial weight, high power output, and intensive use in freight transportation. Consequently, their fuel consumption and combustion-related emissions position HDVs as major contributors to GHG emissions within the transport sector (O’Connell, 2023).

Table 4 and Fig. 10 summarize carbon footprint values for BEVs, reported in tons of $CO_2$ equivalent, showing substantial variation depending on the system boundary applied. For the cradle-to-gate boundary (blue bars), reported values range from 23 to 313 $tCO_2eq$. The cradle-to-use boundary shows a narrower range, from 326 to 519 $tCO_2eq$, while the cradle-to-grave boundary (green bars) reveals the

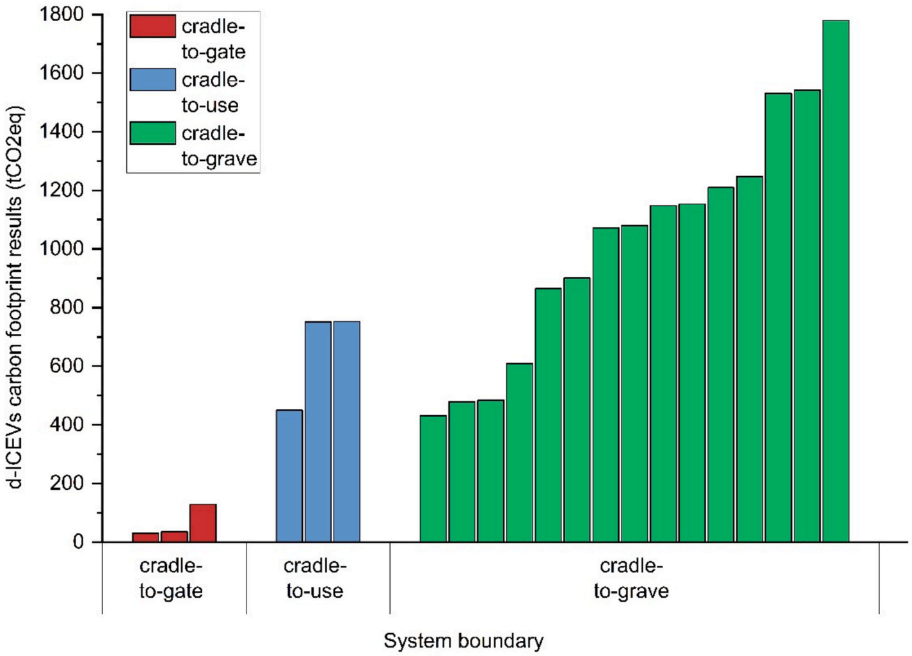


**Fig. 9.** Diesel-ICEVs carbon footprint values in literature.

**Table 3**
Diesel ICEVs carbon footprint values for different system boundaries.

| CARBON FOOTPRINT VALUES OF DIESEL ICEVs ($tCO_2eq$) | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Cradle-to-gate | | | Cradle-to-use | | | Cradle-to-grave | | | | | | | |
| 30 | 35 | 129 | 450 | 751 | 753 | 431 | 479 | 484 | 609 | 865 | 902 | 1072 | 1080 |
| | | | | | | 1148 | 1153 | 1210 | 1247 | 1531 | 1542 | 1780 | |

**Table 4**
BEVs carbon footprint values for different system boundaries.

| CARBON FOOTPRINT VALUES OF BEVs ($tCO_2eq$) | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Cradle-to-gate | | | | | | | Cradle-to-grave | | | | | |
| 29 | 62 | 100 | 115 | 160 | 168 | 313 | 175 | 209 | 236 | 268 | 294 | 300 |
| | | | | | | | 390 | 577 | 760 | 1082 | 1226 | 1626 |

broadest variability, from 175 to 1626 $tCO_2eq$. This wide range highlights the sensitivity of LCA results to methodological choices, particularly system boundaries. Cradle-to-gate assessments cover emissions through manufacturing, whereas cradle-to-grave analyses include use-phase emissions and end-of-life processes. Another key factor influencing results is the electricity mix assumed, as grids dominated by fossil fuels produce significantly higher emissions than those relying on renewable energy sources (Ellingsen et al., 2016). Vehicle lifetime assumptions also affect total carbon footprints; longer lifespans increase cumulative emissions, especially when electricity is carbon intensive. Battery degradation and replacement represent additional critical considerations, given that battery production is among the most carbon-intensive phases of BEV manufacturing. Replacement batteries can substantially raise life cycle impacts, emphasizing the importance of improving battery durability and recycling (Notter et al., 2010). However, only five studies explicitly address battery replacement, indicating a significant research gap and the need for more focused investigations into its environmental implications.

Despite the critical role of battery replacement in shaping the life cycle carbon footprint of BEVs, none of the reviewed studies provide a comparative analysis of carbon footprint with and without battery replacement. For instance, (Sacchi et al., 2021) assume that the battery is replaced once over the 40 tonnes vehicle's lifetime, yet report only the carbon footprint including replacement, which amounts 1,626 $tCO_2eq$. Table 5 shows the variation in carbon footprint estimates for BEVs under different system boundaries and battery replacement scenarios. The results demonstrate that including battery replacement significantly increases the carbon footprint, particularly in cradle-to-grave assessments. These variations are also influenced by differences in energy consumption and assumed vehicle lifetimes. Overall, the table highlights the critical role of battery replacement, energy efficiency, and consistent lifetime assumptions in life cycle assessments of BEVs.

Table 6 and Fig. 11 present the carbon footprint values of FCEVs, expressed in tons of $CO_2$ equivalent, as reported across the literature. A

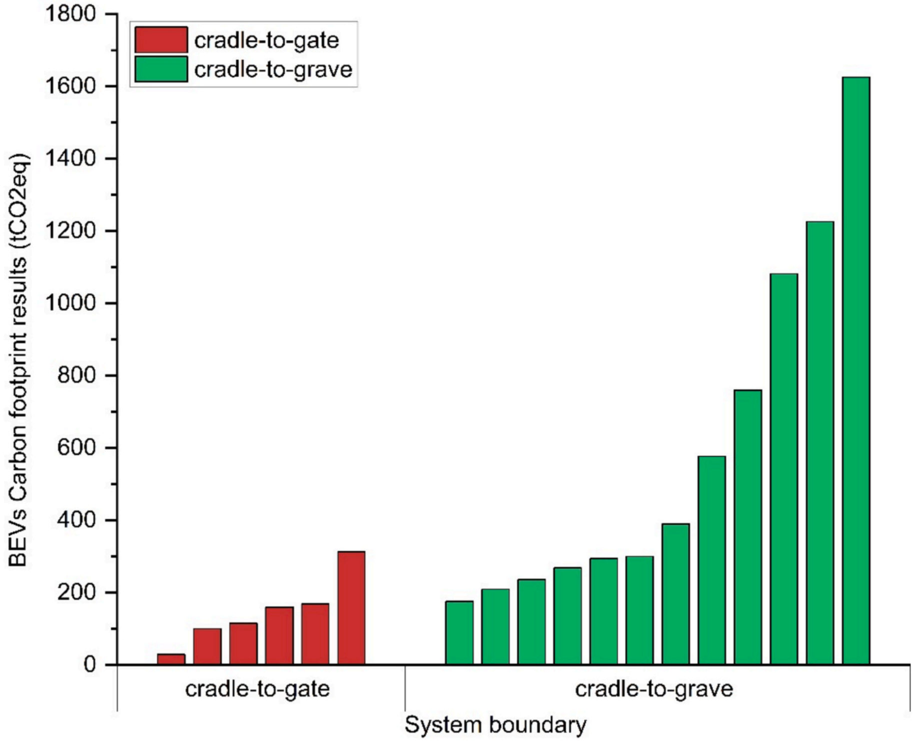


**Fig. 10.** BEVs carbon footprint values in literature.

**Table 5**
BEVs carbon footprint range for system boundary, energy consumption and lifetime with and without battery replacement.

| Battery replacement | System Boundary | Energy consumption (kWh/km) | Vehicle lifetime (km or year) | Carbon footprint range ($tCO_2eq$) |
|---|---|---|---|---|
| Yes | Cradle-to-grave | 1.71–2.01 | 1,050,000 km | 1,082–1,626 |
| No | Cradle-to-grave | – | 500,000–800,000 km or 12 years | 175–760 |
| Yes | Cradle-to-gate | 1.1–1.5 | 800,000 | 100–313 |
| No | Cradle-to-gate | 0.336 | 10 years | 29–62 |

**Table 6**
FCEVs carbon footprint values for different system boundaries.

| | Carbon footprint values of FCEVs ($tCO_2eq$) | | | | | |
|---|---|---|---|---|---|---|
| Cradle-to-gate | Cradle-to-grave | | | | | |
| 55 | 366 | 651 | 990 | 2159 | 2442 | 2944 |

single value is available for the cradle-to-gate boundary (blue bars), amounting to 55 $tCO_2eq$, while cradle-to-grave estimates (green bars) vary widely, ranging from 366 to 2944 $tCO_2eq$. This broad variability reflects the influence of key factors such as the hydrogen production pathway, energy source, the fuel cell replacement and overall vehicle efficiency. Hydrogen production is the dominant contributor to life cycle emissions. When hydrogen is generated via SMR without carbon capture, total GHG emissions can exceed those of conventional diesel vehicles. In contrast, green hydrogen, produced through electrolysis using renewable electricity, significantly reduces overall emissions (Bicer and Dincer, 2018). The efficiency of the fuel cell system also plays a crucial role. Despite offering fast refueling and high energy density, FCEVs generally suffer from lower well-to-wheel efficiency compared to BEVs, due to losses during hydrogen production, compression, storage, and conversion (Ueckerdt et al., 2021). Cradle-to-grave data suggest that FCEVs typically exhibit a higher absolute carbon footprint than BEVs, largely due to the energy-intensive nature of hydrogen-related processes. The only study that considers fuel cell replacement is (Sacchi et al., 2021), which reports the highest carbon footprint values among FCEV studies—ranging from 2159 to 2944 $tCO_2eq$—based on 2 to 4 fuel cell replacements, depending on the application type (i.e., urban, regional, or long-haul delivery). These findings highlight the urgent need to scale up renewable hydrogen production and improve fuel cell system efficiencies to unlock the full decarbonization potential of FCEV.

## 5. Discussion

The effective implementation of WTW-based CountEmission EU and, more broadly, full vehicle LCA-based policies require resolving both confidentiality concerns associated with data exchange and practical implementation challenges. Key questions remain regarding the optimal approach to carbon footprint data management, specifically, whether suppliers should independently calculate and communicate their emissions data upstream through the supply chain, or whether original equipment manufacturers (OEMs) should assume responsibility for data collection, carbon accounting, and footprint declarations.

From a methodological standpoint, several foundational issues must be addressed before LCA-based regulatory frameworks can be implemented. These include the definition of functional units and consistent calculation rules, for example, whether the vehicle's warranty period can serve as a proxy for its operational lifetime. Furthermore, clear minimum system boundary requirements are essential, such as the extent to which primary data must be collected from upstream suppliers (e.g., tier-2 and beyond). Battery replacement accounting also requires harmonization, including determining whether only routine

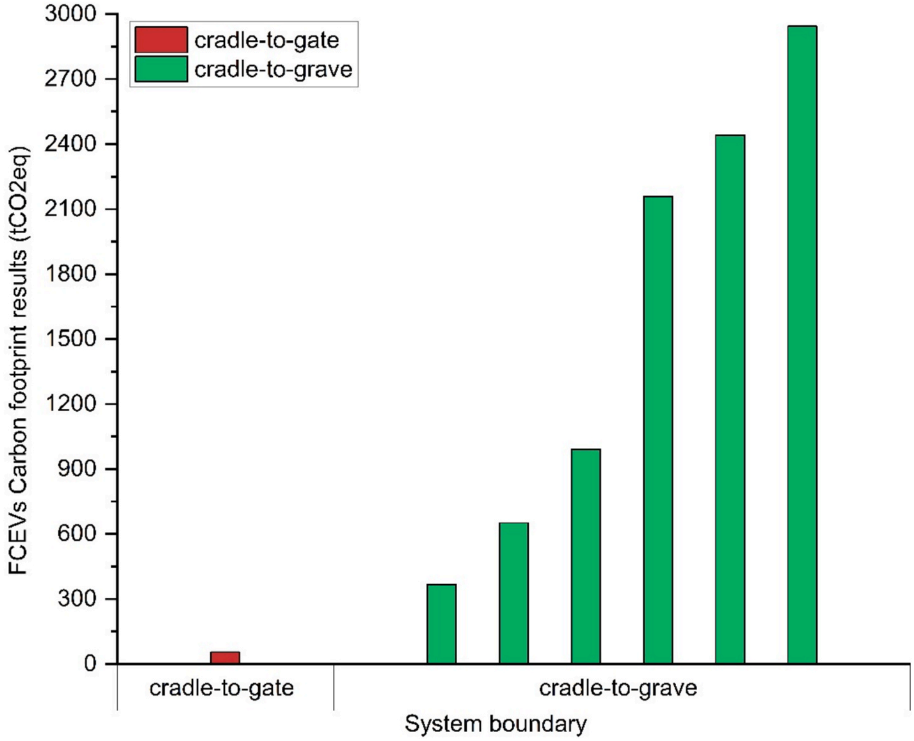


**Fig. 11.** FCEVs carbon footprint values in literature.

maintenance should be considered or if additional replacements must be accounted for in the LCA. Crucial methodological gaps remain, particularly in modeling EoL impacts. It is imperative to determine which approach—avoided burden, cut-off, or CFF (Andreasi et al., 2023) —is most appropriate for full vehicle LCA applications. Ensuring consistency across OEM calculations is equally important, with standardized assumptions for driving cycles, LCIA methods, and uncertainty quantification methods being necessary for credible and comparable results.

Affordability is another critical consideration in the transition to zero-emission vehicles. For BEVs, the average EU electricity cost was 0.33 €/kWh in 2022 (Eurostat, 2022). However, according to (ICCT, 2023), EU charging costs are forecasted to decrease to 0.123 €/kWh by 2030 and €0.120 €/kWh by 2050. For FCEVs, the 2023 hydrogen price ranged from 2.70 to 5.60 €/kg for SMR-based production, with an average levelized cost of €3.76/kg (European Hydrogen Observatory, 2023), compared to €1.54/kg for diesel as of May 2025 (Business Analytiq, 2025). Greener alternatives, such as grid-connected electrolysis or renewable hydrogen, result in significantly higher average costs of 7.94 €/kg and 6.61 €/kg, respectively (European Hydrogen Observatory, 2023). Nevertheless, future projections suggest that hydrogen from electrolysis could decline to 4.12 €/kg by 2030 and 3.16 €/kg by 2050 ( Frieden, 2024). Vehicle choice modeling by (Zhao et al., 2024) highlights that reductions in fuel and electricity prices, combined with improved infrastructure and policy support, are key drivers for accelerating the market adoption of zero-emission medium- and heavy-duty vehicles. This transition will also require coordinated progress in both policy innovation and technological development to ensure a sustainable and long-term transformation of the transport sector (Al-lami and Török, 2025).

In terms of total cost of ownership (TCO), battery electric trucks are expected to become the most cost-effective option by 2030. According to (ICCT, 2023), diesel trucks will be 15 % to 23 % more expensive to own and operate than battery electric trucks, depending on vehicle class. Although hydrogen fuel cell trucks are projected to outperform diesel in cost-effectiveness, they will still be 14 % to 32 % more expensive than their battery electric counterparts. For long-haul applications, battery electric powertrains are expected to achieve TCO parity with diesel between 2025 and 2026, depending on daily mileage and battery size. Fuel cell long-haul trucks are forecasted to reach parity approximately a decade later, between 2035 and 2036 (ICCT, 2023).

## 6. Conclusion

This study presents a comprehensive review of LCA literature focused on trucks, with an emphasis on methodological assumptions and environmental impact outcomes, particularly carbon footprint values for diesel ICEVs, BEVs, and FCEVs. The analysis reveals substantial variability across studies in terms of system boundaries, functional units, vehicle lifetime, energy and fuel consumptions, which collectively contribute to wide-ranging carbon footprint estimates. Values range from 30 to 1780 tons $CO_2$eq for diesel ICEVs, from 29 to 1626 tons $CO_2$eq for BEVs, and from 55 to 2944 tons $CO_2$eq for FCEVs. In the case of ZEVs, also the battery and fuel cell replacements strongly influence the outcomes. These discrepancies highlight the sensitivity of LCAs to input data, methodological choices, and particularly the energy sources assumed, such as the electricity mix for BEVs and hydrogen production pathways for FCEVs. Moreover, the review helped consolidate current methodological assumptions, tools, and metrics used to evaluate the life cycle impacts of trucks. Ten out of 24 studies adopted a cradle-to-grave scope, 13 used tonnes per kilometer as the functional unit, and 12 assumed a 12-year vehicle lifetime, though annual mileage definitions varied widely. Among LCA tools, Simapro, OpenLCA, and Gabi were equally used (4 papers each), with Ecoinvent as the most common database (10 out of 24), and ReCiPe as the leading LCIA method (9 out of 24). Technically, few studies clearly classified vehicle types. Lastly, VECTO emerged as the primary tool for estimating fuel and energy use.

This work underscores the urgent need for harmonized and transparent LCA methodologies, including standardized system boundaries, functional units, and operational parameters, to enhance comparability and reproducibility across studies. Improved availability and incorporation of high-quality primary data are essential to increase the accuracy of environmental impact assessments. Comparison of existing LCAs remains difficult until standardization and data improvements are achieved.

The review also identifies significant research gaps in current literature. While much focus has been placed on diesel and zero-emission technologies for HDVs, the role of alternative fuels, such as natural gas, LNG, biofuels, e-fuels as well as ICE retrofitted to run on hydrogen, remains underrepresented in the literature. Given their potential to substantially reduce carbon emissions in heavy-duty freight transport, future assessments should more comprehensively include these fuels to provide a balanced and realistic evaluation of the full range of decarbonization pathways. Lastly, battery and fuel cell degradation, component replacement, as well as infrastructure-related impacts, such as hydrogen refueling and battery charging stations, remain underexplored in most LCAs.

In terms of metrics, GHG emissions are a key indicator of climate performance. However, a comprehensive LCA should also address other environmental dimensions, including resource depletion, air and water pollution, and ecosystem degradation. In literature, the prevailing focus is on carbon footprint, while other environmental impacts are often excluded. The exclusion of these metrics from many analyses results in an incomplete understanding of the true sustainability profile of trucks.

In summary, while current LCA literature offers valuable insights into the environmental impacts of HDVs, substantial efforts remain necessary to refine methodologies, expand the scope of analyzed fuels and technologies, and develop a robust evidence base to support policy and industry decisions aimed at decarbonizing the heavy-duty transport sector.

## CRediT authorship contribution statement

**Gaia Gentilucci:** Writing – original draft, Methodology, Formal analysis, Data curation, Conceptualization. **Antonella Accardo:** Writing – original draft, Methodology, Formal analysis, Data curation, Conceptualization. **Ezio Spessa:** Writing – review & editing, Supervision.

## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.trip.2025.101563.

## Data availability

No data was used for the research described in the article.

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