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# Comparative Life Cycle Assessment of Electric, Hybrid, and Gasoline Passenger Vehicles in the Greek Context

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## Featured Application

This assessment supports the comparison of environmental trade-offs among battery electric, conventional hybrid, and gasoline passenger-vehicle configurations using a Greek electricity supply for BEV charging. It provides a baseline for integrating life cycle environmental indicators into vehicle-selection and procurement assessments tailored to specific operating conditions and transport-service requirements.

## Abstract

Transport electrification can reduce climate impacts, but its environmental performance depends on vehicle production, energy supply, operation, and end-of-life treatment. This study compares battery electric vehicles (BEVs), non-plug-in gasoline hybrid electric vehicles (HEVs), and gasoline internal combustion engine vehicles (ICEVs) using Greek electricity supply and explicitly defined vehicle operating parameters. A cradle-to-grave life cycle assessment was implemented in openLCA using the ReCiPe 2016 endpoint (H), with a functional unit of 150,000 vehicle-kilometres. Climate impacts are reported as ReCiPe endpoint damage to human health and ecosystems. Both electrified configurations exhibited lower impacts than the ICEV across all 18 indicators with positive results. The BEV achieved reductions of 43.2% in global-warming damage to human health, 44.8% in fine particulate-matter formation, and 45.1% in terrestrial acidification relative to the ICEV. The HEV achieved reductions of 51.5% and 55.8% in fossil and mineral resource scarcity, respectively. Each electrified technology recorded the lowest value in nine positive indicators. The BEV–HEV ranking-margin analysis identified equal-impact thresholds of 9.05% for global-warming damage to human health and 4.25% for human carcinogenic toxicity. Negative water-consumption indicators were retained without interpretation as physical water savings. The findings identify category-specific trade-offs and support coordinated vehicle-efficiency and electricity-decarbonization measures without establishing overall environmental superiority.

**Keywords:** electric vehicles (EVs); life cycle assessment (LCA); greenhouse gas emissions; Greece; transport sector; decarbonization; battery electric vehicles (BEVs); hybrid electric vehicles (HEVs); environmental impact assessment; sustainable mobility

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## 1. Introduction

Reducing greenhouse gas (GHG) emissions from road transport requires the consideration of vehicle technologies and the energy systems supporting their operation [1]. Electrification offers a pathway towards transport decarbonization, but its environmental effectiveness depends on the complete vehicle life cycle. Differences in assessment methods influence how electric vehicle impacts are quantified [2], while comparisons with combustion engine vehicles reveal benefits and limitations that extend beyond driving emissions [3]. Vehicle manufacturing introduces burdens before operation begins [4], and the energy supply chain influences subsequent energy consumption and emissions [5]. In Greece, transport electrification and charging-infrastructure development are relevant components in national energy and climate planning [6]. Evaluating alternative passenger-vehicle technologies within this geographical context is therefore important for understanding their potential contribution to sustainable mobility.

Battery electric vehicles (BEVs) operate without exhaust emissions during driving, whereas hybrid electric vehicles (HEVs) combine an internal combustion engine with electric assistance and regenerative braking. Differences in powertrain architecture and battery requirements affect their operation and production [7]. External charging requirements further distinguish plug-in vehicles from conventional hybrids [8]. The present study considers non-plug-in gasoline HEVs, which receive no external charging electricity; this distinction is necessary when interpreting comparisons across vehicle technologies [9]. The absence of BEV exhaust emissions does not eliminate burdens from electricity generation, material extraction, battery manufacturing, or end-of-life (EoL) treatment [10]. Atmospheric impacts associated with electrification depend on the supporting energy system [11], and differences in electricity generation can alter both GHG and air-pollutant comparisons between countries [12]. Consequently, lower climate impacts do not necessarily imply improvements in all pollution-related categories.

Life cycle assessments (LCAs) provide a framework for examining these relationships across raw material extraction, vehicle production, energy supply, operation, and EoL treatment. Combining the vehicle and fuel cycles is necessary to account for both manufacturing- and energy-related GHG emissions [13]. Reported vehicle carbon footprints vary between countries [14], while explicitly defined energy pathways and assessment boundaries are essential for interpreting comparisons between propulsion technologies [15]. Beyond climate change, comparative vehicle assessments identify trade-offs involving toxicity, ecotoxicity, eutrophication, and resource usage [16]. Vehicle rankings can therefore differ between environmental indicators, even within the same geographical setting [17]. Battery and component production remains relevant because electrification changes material requirements and manufacturing processes [18]. A comparison restricted to operational emissions or climate change can consequently overlook burdens elsewhere in the life cycle.

The transferability of existing findings is limited by differences in electricity supply, vehicle efficiency, lifetime mileage, and modelling assumptions. The climate benefits of passenger-car electrification depend on geographical and technological conditions [19], particularly electricity-related emissions and vehicle energy consumption [20]. Alternative electricity and fuel pathways can further change comparative outcomes [21]. These dependencies create a methodological challenge: propulsion technologies must be compared using consistent assumptions while preserving the characteristics of each configuration. Process-level inventory documentation helps distinguish the influence of individual production steps and scale-up assumptions on environmental results [22]. A common functional basis is necessary to compare propulsion alternatives consistently [23]. Explicit documentation of material inputs, energy requirements, and product-system boundaries further supports transparent interpretation of LCA results [24]. Fuel pathways and assessment

periods must also be specified because they influence the conditions represented within the comparison [25].

EoL management introduces an additional challenge because environmental performance depends on treatment processes and the handling of recovered materials. Recovery activities require assessment of their associated resource and energy inputs alongside their potential benefits [26]. For lithium-ion batteries, recycling methods and treatment locations influence environmental outcomes [27]. Biological recovery processes provide alternative routes for extracting critical metals from spent batteries [28], while recycling and second use involve different material-supply and climate implications [29]. Their potential benefits cannot be assumed solely from the inclusion of EoL treatment within a system boundary. Treatment inventories, allocation rules, and recovery assumptions must therefore be stated explicitly.

For Greece, the analytical question is how BEVs, conventional gasoline HEVs, and gasoline internal combustion engine vehicles (ICEVs) compare across environmental impact pathways within a common framework. Uncertainties in vehicle LCA inputs affect the interpretation of technology differences [30]. Comparative environmental assessments in the Greek context also illustrate the importance of defining system boundaries and energy-recovery assumptions consistently [31]. The present study focuses on selected vehicle configurations instead of claiming a universal ranking or measured representation of the Greek fleet. Its geographical specification includes the Greek electricity supply, whereas vehicle consumption, charging efficiency, battery capacity, and lifetime distance are explicitly defined modelling inputs.

The objective of this study is to compare the cradle-to-grave environmental impacts of gasoline ICEVs, non-plug-in gasoline HEVs, and BEVs over a common lifetime distance of 150,000 vehicle-kilometres. The assessment is implemented in openLCA using the ReCiPe 2016 endpoint (H), with individual damage pathways reported separately for human health, ecosystems, and resource scarcity. Its contributions are a comparison under consistent modelling conditions, transparent documentation of foreground inventories and lifetime calculations, and identification of category-specific trade-offs. A BEV–HEV ranking-threshold analysis additionally quantifies the numerical separation between characterized results without treating it as parameter sensitivity or probabilistic uncertainty. The findings support discussion of sustainable mobility in Greece without aggregating the indicators into a single overall technology ranking.

## 2. Materials and Methods

### *2.1. Goal and Scope Definition*

This study applies LCAs to compare a gasoline ICEV, a non-plug-in gasoline HEV, and a BEV using explicitly defined vehicle configurations and operating parameters, with the Greek electricity supply representing BEV charging. The methodological framework follows the four stages described in ISO 14040 and ISO 14044: goal and scope definition, life cycle inventory analysis, life cycle impact assessment, and interpretation [32,33].

A cradle-to-grave approach was adopted, covering vehicle production, fuel or electricity supply, vehicle operation, and EoL management. The comparison examines differences between propulsion technologies across multiple environmental impact pathways while using a common lifetime distance and consistent system boundaries.

A fixed baseline scenario was assessed. Future electricity-generation scenarios and alternative vehicle lifetimes were not evaluated. An additional analysis of comparative ranking thresholds was performed using the exported characterized results, as described in Section 2.9.

### 2.2. Functional Unit and System Boundaries

The functional unit was defined as 150,000 vehicle-kilometres of passenger-car travel provided by one vehicle. The same lifetime distance was applied to all three technologies. All environmental results are reported per vehicle over this lifetime. Vehicle occupancy was not included in the calculations; consequently, the comparison refers to vehicle travel rather than transport per passenger.

The configurations were selected to provide a transparent baseline comparison of three propulsion types using a common lifetime distance. They are defined by the inventories, energy-consumption parameters, and battery capacities, as reported in Table 1 and Supplementary Table S1, rather than by three fully specified commercial vehicle models. Vehicle mass, passenger capacity, and performance were not harmonized to establish equivalent passenger-transport service. Consequently, differences reflect both the propulsion technologies and particular configurations represented.

**Table 1.** Vehicle parameters and methodological settings adopted for comparison.

| Parameter | ICEV | HEV | BEV | Source or Modelling Basis |
|---|---|---|---|---|
| Vehicle type | Gasoline internal combustion | Non-plug-in gasoline hybrid | Battery electric | Defined model configurations |
| Lifetime distance (km) | 150,000 | 150,000 | 150,000 | Common lifetime assumption |
| Gasoline consumption (L/100 km) | 7.0 | 4.5 | — | ICEV, assumed value within the published range of 6.0–7.0 L/100 km [34]; HEV, consistent with a published compact full-hybrid configuration [35] |
| Lifetime gasoline consumption (L) | 10,500 | 6750 | — | Calculated from gasoline consumption and lifetime distance |
| Battery-side electricity consumption (kWh/100 km) | — | — | 17.0 | Consistent with the 0.17 kWh/km battery-depletion parameter used in published EV modelling [36] |
| Charging efficiency (%) | — | — | 90 | Assumed constant efficiency; experimental evidence of vehicle- and operating-condition dependence is provided in [37] |
| Electricity supplied for charging (kWh/km) | — | — | 0.1889 | Calculated using Equation (4), including charging losses |
| Lifetime electricity supplied for charging (kWh) | — | — | 28,333.3 | Calculated from charging-electricity demand per kilometre and lifetime distance |
| Traction-battery capacity (kWh) | — | 1.3 | 50 | HEV, consistent with [35]; BEV, assumed baseline configuration |
| External charging during operation | No | No | Yes | Defined powertrain configurations |
| Direct fossil $CO_2$ factor (kg $CO_2$/L gasoline) | 2.31 | 2.31 | — | Generic gasoline combustion $CO_2$ factor consistent with published road-transport modelling [38]; excludes upstream fuel-supply emissions |
| Lifetime direct fossil $CO_2$ emissions (kg) | 24,255 | 15,592.5 | 0 | Calculated using Equation (3); BEV has no direct combustion emissions |

Note: Operating parameters define the assessed baseline configurations and are not measured Greek fleet averages. Supporting references document comparable configurations, parameter ranges, or methodological evidence. BEV charging electricity is represented by the ecoinvent 3.12 "market for electricity, medium voltage" dataset for Greece (GR), using the Allocation, cut-off by classification system model. Charging losses are applied separately to battery-side electricity consumption. "—" indicates not applicable.

The system boundaries included vehicle production, energy supply and operation, and EoL treatment. Production comprised the material and manufacturing inputs represented in the vehicle inventories, including traction-battery production for the BEV and HEV. Energy supply included upstream gasoline production and distribution for the ICEV and HEV, as well as electricity supply for the BEV. Operation included energy consumption and the modelled direct combustion emissions. EoL management comprised the treatment-service exchanges reported in Supplementary Table S3.

Road construction, charging-station construction, tyre replacement, and other routine maintenance consumables were excluded from the foreground inventory. Direct non-exhaust emissions from tyre wear, brake wear, road-surface wear, and road-dust resuspension were not explicitly parameterized. Infrastructure and supporting activities embedded in background datasets remained within their respective dataset boundaries.

For vehicle technology $v$ and impact category $c$, total lifetime impacts were calculated as (Equation (1)):

$$I_{lifetime,v,c} = I_{production,v,c} + L \cdot I_{use,v,c} + I_{EoL,v,c}, \quad (1)$$

where $L$ is the lifetime distance of 150,000 km; $I_{use,v,c}$ is the operational impact per kilometre; and the production and EoL terms represent impacts per vehicle. The use-phase term includes energy supply and the operational emissions represented in the model.

### 2.3. Software, Background Database, and Model Structure

The models were developed in openLCA version 2.6.1. Background inventory data were obtained from ecoinvent version 3.12, using the Allocation, cut-off by classification system model. Background processes followed the allocation rules and geographical coverage embedded in the selected datasets.

Ecoinvent 3.12 was released in 2025, while the underlying inventories retain dataset-specific geographical and temporal coverage. The database release year is therefore distinguished from the reference years of the background data. Vehicle and material-production inventories retain the geographical coverage of their selected providers.

Separate foreground processes were established for production, use, and EoL management for each vehicle technology. These processes were linked to form three cradle-to-grave product systems. Production and EoL processes were defined per vehicle, whereas use-phase processes were parameterized per kilometre.

In the kilometre-based aggregation processes, production and EoL inputs were scaled by the reciprocal of the lifetime distance. The complete product systems were evaluated for 150,000 vehicle-kilometres, thereby including production and EoL treatment once and scaling operational inputs to the full lifetime.

The principal model parameters comprised lifetime distance, gasoline consumption, BEV electricity consumption, charging efficiency, and traction-battery capacity. These parameters define the assessed baseline configurations rather than the measured averages for the Greek vehicle fleet. ICEV gasoline consumption was set to 7.0 L/100 km, which was within the 6.0–7.0 L/100 km range used in a published European vehicle assessment [34]. The HEV combines gasoline consumption of 4.5 L/100 km with a traction-battery capacity of 1.3 kWh, consistent with a published compact full-hybrid configuration [35]. BEV battery-side electricity consumption was set to 17.0 kWh/100 km, equivalent to the 0.17 kWh/km driving-consumption parameter used in published EV modelling [36]. Charging efficiency was assumed to be 90%, simplifying the vehicle- and operating-condition dependences that have been documented experimentally [37]. Direct gasoline combustion $CO_2$ emissions were calculated using 2.31 kg $CO_2$/L, consistent with a published petrol combustion factor [38]. Table 1 distinguishes the modelling bases of these parameters from calculated lifetime quantities.

### 2.4. Vehicle and Battery Production

Vehicle production was represented using simplified inventories with common material categories but technology-dependent quantities. For example, the BEV, HEV, and ICEV inventories contain 1000, 950, and 900 kg of steel; 200, 160, and 150 kg of aluminum; and 40, 30, and 25 kg of copper, respectively. Production electricity requirements also differ, as reported in Supplementary Table S1.

BEV battery production was represented using a 50 kWh Li-ion battery-pack proxy. One pack was included for the BEV, and 0.026 pack was included for the 1.3 kWh HEV battery. This linear capacity-based representation is distinct from electricity consumption during driving. This approximation does not explicitly resolve differences in battery chemistry, power requirements, or manufacturing characteristics between HEV and BEV batteries.

Capacity-based scaling was used to represent both traction batteries through a common battery-production proxy within the simplified inventory framework. It provides a transparent calculation of the assigned battery-production burden but assumes proportionality to nominal energy capacity. The approach does not establish technological equivalence between HEV and BEV batteries.

One initial traction battery was included for each electrified vehicle, without replacement during the 150,000 km lifetime. This defines the baseline service-life scenario rather than a prediction that every battery will remain serviceable over that distance. Battery degradation, failure probabilities, and replacement schedules were not modelled.

### 2.5. Use-Phase Inventory

#### 2.5.1. Gasoline ICEV and HEV

Gasoline consumption was converted from litres per 100 km to litres per kilometre (Equation (2)):

$$q_{fuel,v} = C_v/100, \quad (2)$$

where $C_v$ denotes gasoline consumption in L/100 km. The ICEV consumption of 7.0 L/100 km corresponds to 0.070 L/km, yielding a lifetime gasoline requirement of 10,500 L. For the HEV gasoline consumption of 4.5 L/100 km, the corresponding quantities are 0.045 L/km and 6750 L over the vehicle's lifetime.

Gasoline inputs were connected to background fuel-supply processes to account for upstream extraction, refining, transport, and distribution. The HEV was modelled as a conventional non-plug-in hybrid, with no external charging electricity assigned to its operation.

Direct fossil $CO_2$ emissions were calculated using a generic gasoline combustion factor of 2.31 kg $CO_2$/L, consistent with the gasoline emission factor used in a recent road-transport emission model [38]. This factor was adopted as a generic model parameter rather than a Greek-specific fuel measurement. Upstream fuel-supply impacts were calculated separately through the background gasoline-supply processes. Lifetime direct $CO_2$ emissions were obtained by multiplying the gasoline consumption per kilometre by lifetime distance and the combustion factor, as expressed in Equation (3):

$$M_{CO2,v} = L \cdot q_{fuel,v} \cdot f_{CO2}, \quad (3)$$

where $f_{CO2}$ is the direct combustion emission factor of 2.31 kg $CO_2$/L gasoline. For the ICEV, this calculation gives 24,255 kg $CO_2$ over 150,000 km. For the HEV, with a gasoline consumption of 4.5 L/100 km, the corresponding emissions are 15,592.5 kg $CO_2$. These quantities represent direct fossil $CO_2$ emissions and exclude upstream fuel-supply impacts.

The interpretation of other atmospheric impact categories was restricted to emissions represented in the selected foreground and background inventories. Category-level results were not used to infer the inclusion or magnitude of individual exhaust pollutants.

2.5.2. Battery Electric Vehicle

Battery-side BEV electricity consumption was set to 17.0 kWh/100 km, consistent with the 0.17 kWh/km driving-consumption parameter used in published EV modelling, where battery depletion and charging efficiency are treated separately [36]. Charging efficiency was assumed to be 90%; this fixed parameter simplifies the dependence of charging efficiency on vehicle and operating conditions documented experimentally [37]. Electricity supplied for charging was calculated by accounting for charging losses separately (Equation (4)):

$$q_{electricity} = C_{BEV}/(100 \cdot \eta_{charging}), \quad (4)$$

where $C_{BEV}$ is the vehicle-level electricity consumption and $\eta_{charging}$ is the charging efficiency. The resulting electricity-supply requirement was 0.1889 kWh/km, corresponding to approximately 28,333.3 kWh over 150,000 km.

Electricity supply for BEV charging was represented using the ecoinvent 3.12 Allocation, cut-off dataset "market for electricity, medium voltage" for Greece (GR). According to the database documentation, electricity-market mixes for EU member states are based on Eurostat statistics, with 2022 used as the reference year [39]. Thus, the database's release year and the electricity reference year are distinct. The dataset was applied unchanged throughout the assessed vehicle lifetime, without modelling subsequent changes in the Greek electricity mix.

No direct exhaust emissions were assigned to the BEV. Its modelled use-phase impacts arose from the electricity supply, with charging losses incorporated through the efficiency parameter. Changes in electricity consumption associated with battery degradation or operating conditions were not modelled separately.

### *2.6. End-of-Life Management and Allocation*

EoL management was represented separately for each vehicle technology using the material quantities and treatment-service exchanges reported in Supplementary Table S3. The foreground inventory included landfill and incineration services; no material-recovery rates were inferred from these treatment quantities.

EoL treatment was included once per vehicle in the lifetime calculation. Second-life applications and separate battery-recovery scenarios were not evaluated.

Background treatment processes followed the allocation rules of the ecoinvent Allocation, cut-off by classification system model. No additional foreground substitution credits for avoided primary-material production were applied. The assessment therefore accounted for treatment burdens and the allocation rules embedded in the selected background processes.

Accordingly, the model compares the specified disposal-treatment services and does not evaluate a vehicle- or battery-material recovery strategy. The results cannot quantify recycling benefits, compare recovery technologies, or establish the effects of secondary-material substitution.

### *2.7. Life Cycle Impact Assessment*

Environmental impacts were assessed using the ReCiPe 2016 endpoint (H), adopting the hierarchist perspective [40]. The method was applied consistently to the three vehicle inventories using the common functional unit of one vehicle over a lifetime distance of 150,000 km.

Characterized results were reported separately for individual endpoint impact pathways. Human-health damage was expressed in disability-adjusted life years (DALY), ecosystem damage in species·year, and resource scarcity in USD2013, as applicable. These quantities are characterized potential damage indicators, not measured health outcomes, observed species losses, or masses of emitted pollutants. Endpoint characterization links

inventory exchanges to modelled damage pathways. Indicators must therefore be compared within the same pathway and unit rather than interpreted as interchangeable measures of emissions or added across different units.

Vehicle technologies were compared within each impact indicator to identify category-specific rankings and environmental trade-offs. Absolute values expressed in different units were not compared directly.

No normalization or weighting was applied. Results were therefore interpreted separately for each indicator and without aggregation into a single environmental score or an overall technology ranking.

The global-warming indicators express characterized damage to human health and ecosystems. Their values are distinct from GHG emission inventories expressed in kg $CO_2$-equivalent.

### *2.8. Result Processing and Interpretation*

LCIA results were exported to Excel and organized by vehicle technologies and endpoint impact indicators. All results were interpreted on the common basis of one vehicle over a lifetime distance of 150,000 km.

Comparative interpretation examined the direction and magnitude of differences within each indicator. For the 18 indicators with positive values, the percentage difference for vehicle technology $v$ relative to the ICEV in impact category $c$ was calculated as $100 \times (I_{v,c}/I_{ICEV,c} - 1)$. Negative values indicate lower characterized impacts than the ICEV. This relative scaling was used for comparison and graphical presentation and does not constitute LCIA normalization or weighting.

The number of indicators in which each technology recorded the lowest value was reported descriptively, accounting for relationships between damage pathways. Contributions from particular materials, processes, or life cycle stages were attributed only where they were supported by the results of inventory or contribution analyses.

### *2.9. Comparative Ranking-Threshold Analysis*

A ranking-threshold analysis was performed using the exported LCIA results to quantify the separation between the BEV and HEV within each positive indicator. For each indicator, the lower characterized result was increased while the higher result was held constant. The percentage increase required to produce equality was calculated as (Equation (5)):

$$Tc = 100\ (I_{higher,c}/I_{lower,c}) - 1), \quad (5)$$

where Tc is the equality threshold for indicator c; and $I_{higher,c}$ and $I_{lower,c}$ are the higher and lower BEV–HEV results, respectively.

Equality occurs at the calculated threshold. A larger increase reverses the pairwise ranking under the specified one-sided perturbation. Thresholds were calculated from unrounded results and displayed to two decimal places.

The analysis was restricted to the 18 indicators with positive values. Negative water-consumption results were excluded because the proportional threshold formulation does not provide the same interpretation for those values. The additional information provided is the proportional separation between the two characterized totals, identifying comparisons with smaller or larger numerical margins. The analysis does not estimate whether such changes are plausible under real operating conditions.

### 2.10. Methodological Limitations

The assessment represents fixed vehicle configurations and operating parameters. Future electricity-generation scenarios, alternative lifetime distances, inventory-parameter sensitivity analysis, and probabilistic uncertainty analysis were not evaluated.

Assumptions likely to influence the BEV–HEV comparison include vehicle electricity and gasoline consumption, the electricity supply used for BEV charging, battery capacity and production impacts, vehicle lifetime distance, and the need for battery replacement. Electricity consumption and electricity supply affect the BEV use-phase inventory, whereas gasoline consumption affects the HEV use-phase inventory. Battery capacity, the production proxy, and replacement assumptions influence the production and end-of-life inventories. Their relative importance and the magnitude of their effects on the reported rankings have not been quantified because parameter sensitivity analysis was not performed.

The operating parameters represent baseline configurations supported by published modelling values and assumptions rather than measurements or fleet-average estimates for Greece. Greek geographical specificity is provided by the charging-electricity dataset; fleet composition, vehicle age, annual mileage distributions, and regional driving patterns were not modelled. Differences in vehicle specifications, driving conditions, charging behaviours, and fuel properties may therefore affect the applicability of the results to individual vehicles. The comparison concerns the defined vehicle configurations using a Greek charging-electricity supply and does not establish national fleet performance.

The deterministic ranking-threshold analysis describes numerical separation between alternatives. Its results cannot be interpreted as uncertainty bounds or evidence of robustness to changes in the electricity supply, energy consumption, battery characteristics, or vehicle lifetime.

The use of production proxies and simplified battery representations limits the extent to which differences can be attributed to specific manufacturing technologies or battery chemistries. The exclusion of battery replacement, maintenance consumables, and variations in driving conditions also limits the representation of individual vehicle-use patterns.

Interpretation of traffic-related air pollution is limited to pollutants represented in the inventory. In particular, the particulate-matter results do not quantify the full contribution of direct non-exhaust emissions, and the effects of omitted maintenance and replacement activities on particulate-matter formation and terrestrial acidification remain unquantified. Interpretation of the water-consumption indicators is limited by the absence of exchange-level contribution results; their negative signs are not used to infer physical water savings or comparative environmental superiority.

The findings are conditional on the selected inventories, energy requirements, electricity supplies, battery-production representations, and EoL treatment specifications. They describe the modelled vehicle systems rather than all vehicles within each propulsion category.

## 3. Results

### 3.1. Comparative Environmental Performance

The cradle-to-grave environmental impacts of the three vehicle technologies were compared using the ReCiPe 2016 endpoint (H), considering a vehicle lifetime of 150,000 km. Results are expressed in DALY for human-health damage, species·year for ecosystem damage, and USD2013 for resource scarcity.

Both the BEV and HEV exhibited lower characterized impacts than the ICEV across all 18 indicators with positive values. Their relative performance varied between impact pathways: the BEV recorded the lowest values in nine positive indicators, while the HEV recorded the lowest values in the remaining nine. The HEV also recorded the most negative values in the three water-consumption indicators.

The BEV exhibited lower impacts than the HEV for the three global-warming pathways, fine particulate-matter formation, both ozone-formation pathways, terrestrial acidification, terrestrial ecotoxicity, and marine ecotoxicity. Conversely, the HEV exhibited lower impacts for ionizing radiation, stratospheric ozone depletion, both human-toxicity indicators, freshwater ecotoxicity, freshwater and marine eutrophication, and fossil and mineral resource scarcity.

Category-specific rankings are reported without weighting or aggregation; the number of favourable indicators does not establish an overall ranking. Figure 1 presents four illustrative pathways: global-warming damage to human health, fine particulate-matter formation, fossil resource scarcity, and mineral resource scarcity. They were selected to show climate- and particulate-matter-related human-health outcomes alongside the two resource-scarcity indicators, thereby illustrating categories favouring the BEV and categories favouring the HEV. The selection does not imply that these pathways are more important than toxicity, ecotoxicity, or eutrophication. Figure 2 presents all 18 positive indicators, while Supplementary Table S5 reports the complete results. Figure 2 presents the relative differences for each positive indicator, allowing the HEV and BEV results to be compared against the same ICEV reference within each pathway.

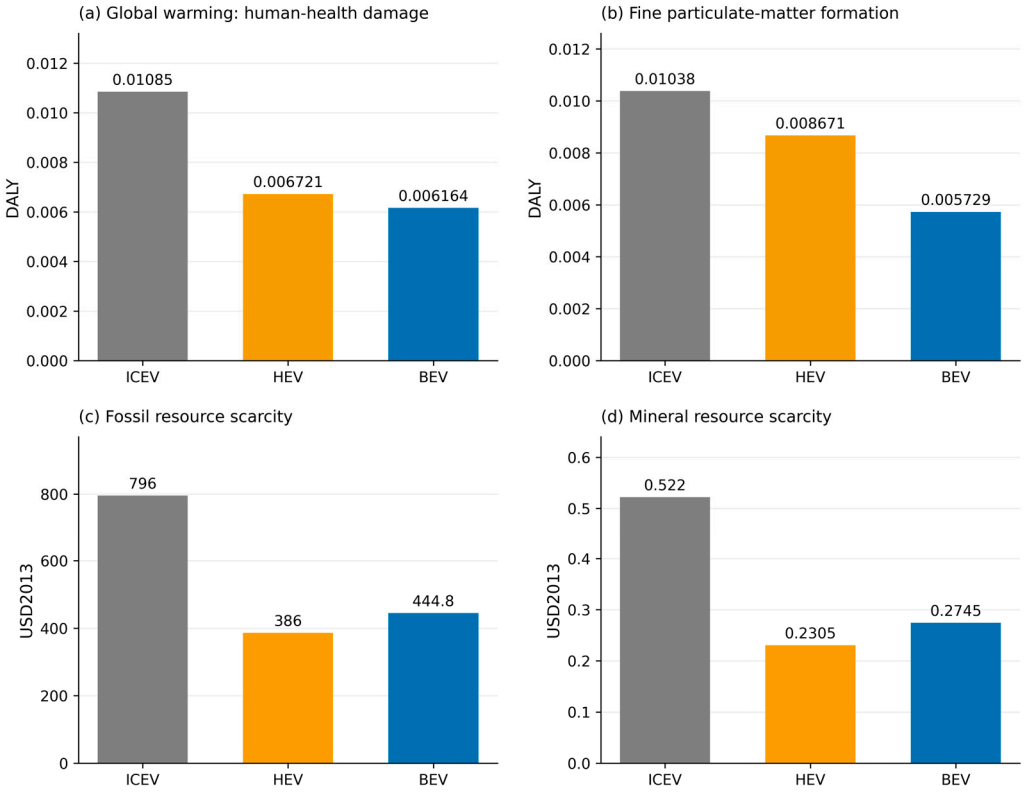



**Figure 1.** The BEV has the lowest modelled global-warming- and particulate-matter-related human-health damage, whereas the HEV has the lowest fossil and mineral resource-scarcity impacts among the assessed configurations. Cradle-to-grave ReCiPe 2016 endpoint (H) results over 150,000 vehicle-kilometres show: (**a**) global-warming damage to human health; (**b**) fine particulate-matter formation; (**c**) fossil resource scarcity; and (**d**) mineral resource scarcity. Each panel retains its category-specific unit. Panel selection is illustrative; Figure 2 presents all 18 positive indicators. Direct non-exhaust emissions were not explicitly parameterized.

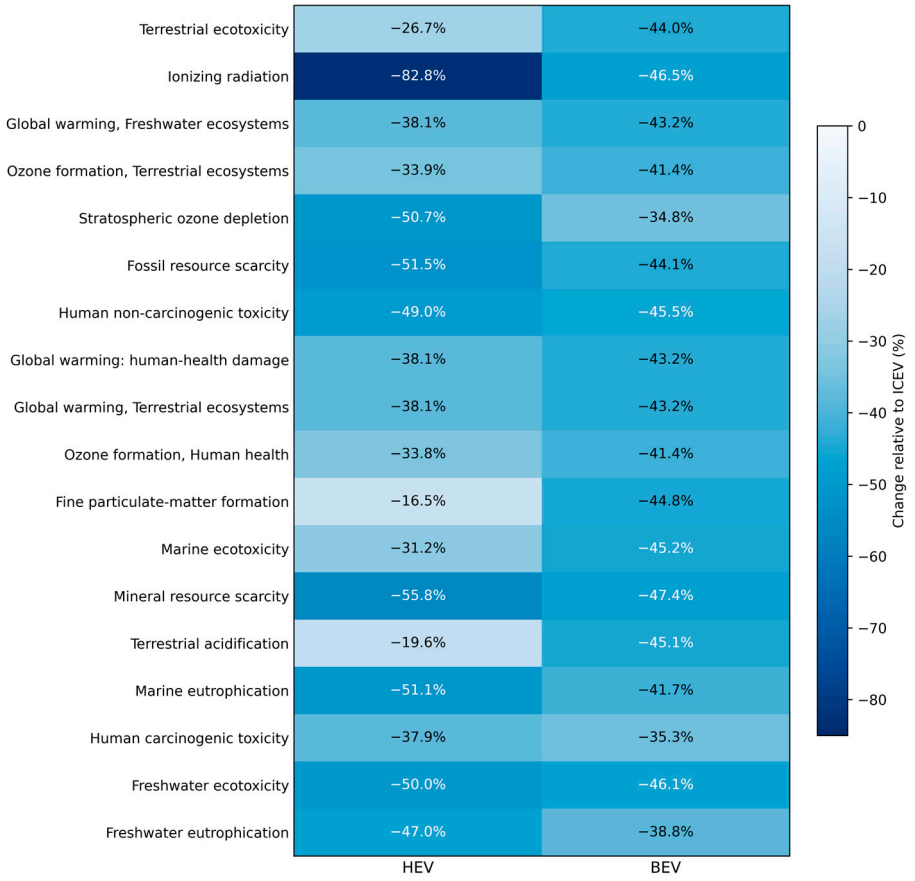



**Figure 2.** Percentage differences in ReCiPe 2016 endpoint (H) impacts for the HEV and BEV, relative to the ICEV, across the 18 indicators with positive results for all three vehicle configurations. Each cell reports the percentage difference for one vehicle and impact indicator, calculated as $100 \times (I_{v,c}/I_{ICEV,c} - 1)$. Negative values indicate lower characterized impacts than the ICEV. Darker blue indicates a larger percentage reduction, while lighter blue indicates a smaller reduction. Values are displayed to one decimal place. The three water-consumption indicators were excluded because their characterized totals were negative. The percentages describe relative differences within each indicator and do not constitute LCIA normalization or weighting. Absolute results and calculations are provided in Supplementary Tables S5 and S6a.

### *3.2. Global Warming and Human-Health Impacts*

Global-warming damage to human health amounted to $1.0853 \times 10^{-2}$ DALY for the ICEV, $6.7214 \times 10^{-3}$ DALY for the HEV, and $6.1638 \times 10^{-3}$ DALY for the BEV. Compared with the ICEV, the HEV and BEV exhibited reductions of 38.1% and 43.2%, respectively.

The same ranking, BEV < HEV < ICEV, was observed for global-warming damage to terrestrial and freshwater ecosystems. These percentages refer to characterized damage within the corresponding endpoint pathways rather than directly calculated reductions in GHG emission mass.

Fine particulate-matter formation showed a larger separation between the BEV and HEV. Results were $1.0380 \times 10^{-2}$ DALY for the ICEV, $8.6709 \times 10^{-3}$ DALY for the HEV, and $5.7294 \times 10^{-3}$ DALY for the BEV. These values correspond to reductions of 16.5% for the HEV and 44.8% for the BEV relative to the ICEV. The particulate-matter percentages refer to characterized life cycle human-health damage and not reductions in directly emitted particulate mass or roadside concentrations.

Ozone-formation damage to human health also followed the order BEV < HEV < ICEV, with reductions of 41.4% and 33.8% for the BEV and HEV, respectively.

Human-toxicity indicators showed a different ranking. Human carcinogenic toxicity amounted to $1.0521 \times 10^{-4}$ DALY for the ICEV, $6.5318 \times 10^{-5}$ DALY for the HEV, and $6.8091 \times 10^{-5}$ DALY for the BEV. The corresponding reductions were 37.9% for the HEV and 35.3% for the BEV.

For human non-carcinogenic toxicity, the HEV and BEV exhibited reductions of 49.0% and 45.5%, respectively, relative to the ICEV.

The HEV also exhibited the lowest impacts for ionizing radiation and stratospheric ozone depletion. Relative to the ICEV, ionizing-radiation impacts decreased by 82.8% for the HEV and 46.5% for the BEV, while stratospheric ozone-depletion impacts decreased by 50.7% and 34.8%, respectively.

### *3.3. Ecosystem Impacts*

The BEV exhibited the lowest terrestrial acidification impacts, with a value of $6.2814 \times 10^{-6}$ species·year, compared with $9.1971 \times 10^{-6}$ species·year for the HEV and $1.1438 \times 10^{-5}$ species·year for the ICEV. These results represent reductions of 45.1% for the BEV and 19.6% for the HEV relative to the ICEV. Ozone-formation damage to terrestrial ecosystems followed the same ranking, with reductions of 41.4% for the BEV and 33.9% for the HEV.

Terrestrial ecotoxicity impacts were $6.0195 \times 10^{-8}$ species·year for the ICEV, $4.4129 \times 10^{-8}$ species·year for the HEV, and $3.3705 \times 10^{-8}$ species·year for the BEV. Marine ecotoxicity similarly favoured the BEV, which achieved a reduction of 45.2% relative to the ICEV compared to 31.2% for the HEV.

In contrast, the HEV exhibited lower freshwater ecotoxicity and eutrophication impacts than the BEV. Freshwater ecotoxicity decreased by 50.0% for the HEV and 46.1% for the BEV relative to the ICEV. Freshwater eutrophication decreased by 47.0% and 38.8%, while marine eutrophication decreased by 51.1% and 41.7%. Thus, the relative performance of the two electrified technologies differed between terrestrial, freshwater, and marine impact pathways.

### *3.4. Resource Scarcity*

Fossil resource scarcity amounted to 796.04 USD2013 for the ICEV, 385.96 USD2013 for the HEV, and 444.85 USD2013 for the BEV. Relative to the ICEV, the HEV achieved a reduction of 51.5% compared to 44.1% for the BEV.

Mineral resource scarcity followed the same ranking. Values were 0.52200 USD2013 for the ICEV, 0.23053 USD2013 for the HEV, and 0.27450 USD2013 for the BEV, corresponding to reductions of 55.8% and 47.4% for the HEV and BEV, respectively. Although both electrified technologies exhibited lower resource-scarcity impacts than the ICEV, the HEV recorded the lowest values in both resource indicators.

### *3.5. Water Consumption*

All three water-consumption damage indicators yielded negative net characterized values. For human health, these values were $-1.1712 \times 10^{-5}$ DALY for the ICEV, $-3.8969 \times 10^{-5}$ DALY for the HEV, and $-6.6980 \times 10^{-6}$ DALY for the BEV. The corresponding terrestrial ecosystem values were $-7.1219 \times 10^{-8}$, $-2.3697 \times 10^{-7}$, and $-4.0731 \times 10^{-8}$ species·year, while aquatic ecosystem values were $-3.1864 \times 10^{-12}$, $-1.0602 \times 10^{-11}$, and $-1.8223 \times 10^{-12}$ species·year.

The HEV exhibited the most negative value in each water-consumption pathway. These indicators represent signed, net characterized results rather than gross water withdrawals. Their calculation sums the contributions of inventory water exchanges according to the implemented characterization factors. Negative totals indicate that negative contributions

outweigh positive contributions. Water-accounting conventions are relevant to this interpretation: return flows can contribute negatively while allocation can alter the balance between water inputs and outputs, as discussed in the ecoinvent implementation documentation [41]. The reported endpoint values therefore should not be read as physical quantities of water saved. Since the aggregate exports do not resolve the individual contributions, no process-specific explanation or environmental ranking is inferred from their negative signs. Percentage reductions and proportional ranking thresholds were consequently not calculated for these indicators.

### 3.6. Comparative Ranking Thresholds

The BEV–HEV equality thresholds varied substantially across the positive indicators (Table 2). Human carcinogenic toxicity exhibited the smallest threshold, where increasing the HEV result by 4.25%, with the BEV result held constant, would produce equality. The corresponding thresholds were 6.80% for human non-carcinogenic toxicity and 7.86% for freshwater ecotoxicity.

**Table 2.** One-sided percentage increases in the lower BEV–HEV results required to produce equality, with the higher results held constant.

| ReCiPe 2016 Endpoint (H) Indicator | Technology with Lower Result | Increase Required for Equality (%) |
|---|---|---|
| Human carcinogenic toxicity | HEV | 4.25 |
| Human non-carcinogenic toxicity | HEV | 6.80 |
| Freshwater ecotoxicity | HEV | 7.86 |
| Global warming, human health | BEV | 9.05 |
| Global warming, terrestrial ecosystems | BEV | 9.05 |
| Global warming, freshwater ecosystems | BEV | 9.05 |
| Ozone formation, terrestrial ecosystems | BEV | 12.76 |
| Ozone formation, human health | BEV | 12.94 |
| Fossil resource scarcity | HEV | 15.26 |
| Freshwater eutrophication | HEV | 15.34 |
| Mineral resource scarcity | HEV | 19.07 |
| Marine eutrophication | HEV | 19.28 |
| Marine ecotoxicity | BEV | 25.45 |
| Terrestrial ecotoxicity | BEV | 30.93 |
| Stratospheric ozone depletion | HEV | 32.31 |
| Terrestrial acidification | BEV | 46.42 |
| Fine particulate-matter formation | BEV | 51.34 |
| Ionizing radiation | HEV | 211.37 |

For global-warming damage to human health, increasing the BEV result by 9.05%, with the HEV result unchanged, would produce equality. Similar thresholds were obtained for global-warming damage to terrestrial and freshwater ecosystems.

Six of the 18 positive indicators exhibited equality thresholds below 10%: the three global-warming pathways, the two human-toxicity indicators, and freshwater ecotoxicity. These indicators showed comparatively narrow numerical separations between the BEV and HEV under the specified one-sided perturbation.

Larger separations were observed for fine particulate-matter formation and terrestrial acidification. The lower BEV results would require increases of 51.34% and 46.42% to equal the HEV results.

Ionizing radiation exhibited the largest threshold: the lower HEV result would require an increase of 211.37% to equal the BEV result. For fossil and mineral resource scarcity, the HEV results would require increases of 15.26% and 19.07%, respectively, to equal the BEV results.

## 4. Discussion

### *4.1. Climate Performance and Comparison with Previous Vehicle Assessments*

The present assessment indicates that both electrified configurations reduce global-warming damage relative to the gasoline ICEV for the evaluated configurations. The BEV exhibited a 43.2% reduction in global-warming damage to human health compared to 38.1% for the HEV. These findings support the role of vehicle electrification in transport decarbonization while demonstrating that conventional hybridization provides substantial benefits within the modelled comparison.

The direction of the BEV–ICEV comparison agrees with several previous assessments, although the reported magnitudes differ. A multinational vehicle assessment reported life cycle GHG reductions ranging from approximately 30% to 80% for BEVs relative to gasoline vehicles, depending on country-specific electricity supply and vehicle efficiency [5]. Earlier European modelling, using the same lifetime distance of 150,000 km, reported global-warming potential reductions of approximately 10–24% relative to conventional gasoline or diesel vehicles [4]. Chinese assessments reported smaller climate benefits under their respective modelling conditions. One assessment estimated life cycle GHG emissions of approximately 41.0 t $CO_2$-eq for an electric vehicle and 50.0 t $CO_2$-eq for an ICEV under 2015 conditions, corresponding to an 18% reduction [13]. Another assessment projected a life cycle GHG reduction potential of 13.4% for BEVs relative to gasoline ICEVs under its 2020 electricity-system scenario [20]. A more recent assessment reported 12% lower life cycle $CO_2$ emissions for EVs than for ICEVs [11].

These comparisons establish agreement in the direction of the climate benefit rather than numerical equivalence. The present percentages concern ReCiPe endpoint damage, whereas the cited studies generally report GHG emissions or global-warming potential in $CO_2$-equivalent units. Differences in electricity supply, vehicle specifications, manufacturing inventories, and assessment periods further limit direct comparison. Consequently, the larger percentage reduction obtained here should not be interpreted as evidence that electrification necessarily provides greater benefits in Greece than in the countries examined elsewhere.

The published findings also demonstrate that a BEV climate advantage is conditional. A Lithuanian assessment reported that BEVs supplied with the 2015 electricity mix generated 26% and 47% more GHG emissions than petrol and diesel vehicles, respectively, although cleaner future electricity scenarios changed the comparison [42]. Conversely, a Malaysian assessment found that replacing petrol vehicles with electric vehicles reduced life cycle GHG emissions [43]. These contrasting outcomes support the need for geographically specific assessment rather than a universal ranking based solely on propulsion technology.

The BEV–HEV comparison deserves particular attention. In the present study, the BEV global-warming damage to human health was approximately 8.3% lower than the HEV result, a considerably smaller separation than that between the ICEV and either technology. A Chinese cradle-to-grave assessment reported 212 g $CO_2$-eq/km for gasoline HEVs compared to 231 and 279 g $CO_2$-eq/km for BEVs with different driving ranges, demonstrating conditions under which the hybrid configuration outperformed the electric alternatives [44]. The opposite ranking obtained here is therefore compatible with the broader literature: the relative performance of BEVs and HEVs depends on electricity supply, battery size, and vehicle efficiency rather than electrification alone.

### 4.2. Electricity Supply and the Geographical Specificity of the Findings

Electricity supply is central to interpreting the BEV results because it connects vehicle operation to upstream generation processes. Country-level modelling has shown that BEVs outperform gasoline vehicles in many, but not all, geographical contexts. An assessment covering 35 countries found a climate advantage for current BEVs in 26 countries, illustrating the importance of the energy system supporting their use [19]. Reviews of vehicle LCAs similarly identify electricity generation, vehicle efficiency, and lifetime specifications as major sources of variation between the published results [9,14,45].

The present assessment uses a Greek electricity-market dataset and therefore represents the conditions embedded in that dataset. It does not reconstruct electricity generation for the publication year or account dynamically for changes throughout vehicle life. This distinction matters because a vehicle driven over several years may experience a changing electricity supply, whereas the present baseline applies a fixed background representation.

A European assessment of battery charging across 28 countries demonstrated substantial geographical differences in climate, human-health, ecosystem, resource, and water-related indicators [46]. However, this study assessed electricity provision for charging rather than complete vehicle life cycles. Its results therefore support the importance of electricity-related variation without directly validating the cradle-to-grave rankings obtained here.

Scenario-based vehicle research has also demonstrated that incorporating future changes in charging electricity can reduce estimated climate impacts when compared with a static electricity representation [10]. The present BEV results should accordingly be interpreted as a baseline outcome rather than a forecast of its lifetime performance under an evolving Greek energy system. Cleaner electricity could improve its climate performance, but the magnitude of that improvement has not been calculated in this study.

### 4.3. Environmental Trade-Offs Beyond Climate Change

The results do not identify a technology that minimizes every environmental impact. The BEV recorded the lowest values in nine positive indicators, while the HEV recorded the lowest values in the remaining nine. This distribution supports a multi-category interpretation, consistent with reviews identifying potential shifts between operational benefits and production-related burdens [2,3,9]. The aggregate results establish the direction and magnitude of indicator differences but do not establish their distribution among production, operation, and EoL stages. References to electricity supply, battery production, and material requirements below identify plausible explanations informed by the inventory structure and literature; they are not quantified causal attributions from a stage-contribution analysis.

For human carcinogenic and non-carcinogenic toxicity, freshwater ecotoxicity, and freshwater and marine eutrophication, the HEV performed better than the BEV. This pattern may be related to environmental burdens associated with battery and electric-powertrain supply chains, as discussed in previous studies [4,17,18]. However, the present aggregate results do not establish the contribution of these supply chains to the observed differences. Nevertheless, both electrified configurations performed better than the ICEV in these indicators within the present model.

This latter finding differs from studies reporting increased human toxicity, freshwater ecotoxicity, freshwater eutrophication, and metal depletion for electric vehicles relative to conventional vehicles [4]. A comparative Spanish assessment likewise identified the highest ecosystem and human-health toxicity impacts for electric vehicles [17]. These differences should remain explicit: agreement that electrification involves trade-offs does not mean agreement on the direction for each individual BEV–ICEV comparison.

An especially relevant divergence concerns mineral resource scarcity. The present BEV result was 47.4% lower than the ICEV result, whereas several published assessments

associate electrification with increased material requirements. A multiregional material-footprint assessment reported a BEV footprint approximately 60% higher than that of a conventional vehicle under its baseline scenario [47]. However, material footprint measures extracted material quantities, while ReCiPe mineral resource scarcity expresses a characterized consequence of resource extraction. The indicators are therefore related but not interchangeable.

The 47.4% lower BEV mineral resource-scarcity result is specific to the implemented inventories and endpoint characterization. It does not demonstrate lower total metal requirements or lower critical-material demand. Without stage-resolved contributions, the respective influences of vehicle materials, the battery proxy, energy supply, and treatment processes cannot be established. Corresponding midpoint results have not been reported, so the endpoint comparison is not independently corroborated at that level. This finding is therefore retained as a conditional model result and is not used to support general claims of resource superiority.

The lower BEV impacts for particulate-matter formation and terrestrial acidification also contrast with a European comparative assessment reporting approximately doubled particulate matter and acidifying emissions for BEVs, despite lower global-warming impacts [16]. The present reductions of 44.8% and 45.1% therefore represent a substantive difference from that study. Potential explanations include electricity generation, upstream production inventories, operational pollutant coverage, and impact-assessment choices; their individual contributions would require harmonized inventories or process-level contribution analysis. Broader reviews similarly caution that differences in boundaries, battery characteristics, and geographical conditions can change comparative conclusions [48,49].

The particulate-matter comparison is particularly sensitive to the coverage of non-exhaust emissions. Tyre and road wear and resuspended road dust remain relevant for BEVs, while regenerative braking can reduce friction-brake wear. Higher vehicle mass may increase some wear-related emissions; consequently, the net comparison depends on vehicle characteristics and operating conditions [50]. Because these sources were not explicitly parameterized here, the 44.8% reduction represents the assessed life cycle inventory and should not be interpreted as an equivalent reduction in total traffic-related particulate emissions. Including non-exhaust emissions and maintenance could reduce this advantage, although its magnitude and any ranking reversal cannot be established without recalculation. Terrestrial acidification is a distinct impact pathway: tyre and brake particulate emissions do not directly establish the direction of its comparison, while additional maintenance and replacement supply chains could affect its total. More generally, the reported percentage differences concern characterized damage indicators and should not be transferred directly to the emission mass of any individual pollutant.

### *4.4. Vehicle Configuration, Battery Representation, and Service Life*

The comparison concerns specific configurations: a 50 kWh BEV battery; a 1.3 kWh HEV battery; and a gasoline consumption of 7.0 and 4.5 L/100 km for the ICEV and HEV, respectively. The HEV configuration therefore combines a smaller traction battery with approximately 35.7% lower gasoline consumption than the ICEV. These configuration differences provide plausible reasons for the observed comparisons, but the aggregate results do not quantify their individual contributions.

The HEV must also be distinguished from a PHEV. The present hybrid receives no external charging electricity. Findings for plug-in hybrids, which combine grid electricity with fuel use, cannot therefore be transferred directly to this configuration [7,9]. Studies examining renewable fuels and different degrees of electrification have identified circumstances in which PHEVs or renewable-fuel pathways offer favourable climate performance [21,25].

These findings challenge the view that larger batteries or greater electrification invariably minimize life cycle impacts, but they do not directly contradict the present comparison because the assessed fuel pathways and powertrains differ.

Battery production was represented through capacity-based scaling. This approach provides a common modelling basis but does not capture differences in battery chemistry, manufacturing location, production efficiency, or the distinct design requirements of energy-oriented BEV batteries and power-oriented HEV batteries. More detailed, technology-specific battery inventories would help distinguish the influence of battery capacity from that of battery design and manufacturing conditions on the comparative results.

If the HEV battery has a higher production impact per kWh than the shared proxy, capacity scaling would underestimate its battery-production burden; the opposite would occur if its impacts per kWh were lower. The direction and magnitude cannot be determined from capacity alone. Similarly, the simplified vehicle-material inventories do not resolve all powertrain-specific components. These approximations can influence category-specific rankings, particularly where BEV–HEV differences are small.

Service life introduces an additional limitation. The present comparison covers 150,000 km and no battery replacement. Previous vehicle research demonstrated that extending lifetime mileage increases the opportunity for operational savings to compensate for production burdens [4]. Research on stationary battery systems also shows that chemistry and replacement frequency can substantially affect life cycle performance [51]. Although stationary storage is not directly comparable with vehicle propulsion, it reinforces the importance of documenting durability specifications.

A battery replacement would introduce an additional production burden and associated treatment requirements, potentially reducing the advantages calculated for the electrified configurations. Within the capacity-scaled proxy, replacement of the 50 kWh BEV pack would add a larger battery-production inventory than the replacement of the 1.3 kWh HEV pack. However, replacement probabilities and timing are not established here, and their effects on total impacts and rankings have not been quantified. The baseline results are therefore conditional on the no-replacement assumption.

Related fuel-cell vehicle assessments identify energy-production pathways, component durability, and performance losses as influential factors [15,52,53]. These studies provide methodological context rather than evidence for the present BEV–HEV ranking. Similarly, findings for heavy-duty vehicles and waste-transport systems cannot be transferred directly to passenger cars because duty cycles, payloads, and functional units differ [23,54]. The present results therefore describe equivalent vehicle travel for the evaluated configurations rather than the environmental performance of all transport applications.

### *4.5. Interpretation of the Ranking Thresholds and Uncertainty*

The ranking-threshold analysis adds information about the numerical separation between the BEV and HEV. The 4.25% threshold for human carcinogenic toxicity, 6.80% for human non-carcinogenic toxicity, and 7.86% for freshwater ecotoxicity indicate comparatively small differences in the characterized totals. The approximately 9.05% thresholds for the global-warming pathways likewise show that the BEV climate advantage over the HEV is narrower than its advantage over the ICEV.

For example, the 9.05% threshold concerns an increase in the BEV's total global-warming damage to human health, which includes production, electricity supply, and end-of-life treatment. It does not specify the increase in electricity consumption needed to produce equal BEV and HEV impacts.

These thresholds identify comparisons that warrant particular attention in future parameter analysis. They do not demonstrate that the differences are statistically insignificant

or that ranking reversal is probable. Such conclusions would require quantified uncertainty in the inventories and characterization results. Without inventory-parameter sensitivity analysis or quantified uncertainty distributions, the study cannot establish whether the BEV–HEV rankings persist across plausible combinations of electricity supply, consumption, battery-production impacts, and service life. This is particularly consequential for the toxicity, freshwater ecotoxicity, and global-warming pathways with relatively small numerical separations. The reported ordering remains a deterministic baseline result rather than evidence of a reliably preferable technology across operating conditions.

Larger thresholds, including 51.34% for particulate-matter formation, 46.42% for terrestrial acidification, and 211.37% for ionizing radiation, indicate greater numerical separation under the specified one-sided perturbation. They should not be interpreted as proof that these rankings would survive every plausible change in electricity supply, vehicle configuration, or manufacturing conditions. A physical parameter change may affect both alternatives and several life cycle stages simultaneously.

Probabilistic passenger-vehicle assessments have explicitly propagated input uncertainty and evaluated alternative electricity scenarios [30]. Comparable work on freight trucks has also demonstrated that technology rankings can change with vehicle class and assessment period [55]. The present threshold analysis complements baseline comparisons but does not provide the uncertainty distributions or parameter-response relationships established by those approaches.

The most informative extension would therefore combine stage-resolved contributions with variations in electricity supply, fuel and electricity consumption, battery production, and lifetime mileage. The resulting analysis could determine whether the comparatively narrow BEV–HEV differences persist under plausible operational and production conditions.

### *4.6. End-of-Life Management and Circular Material Flows*

EoL management connects vehicle electrification with the recovery of battery and vehicle materials. However, including treatment within a cradle-to-grave boundary does not establish its contribution to the relative performance of competing technologies. The model applies the selected cut-off allocation framework without additional foreground substitution credits. Reductions in the reported impacts should therefore not be attributed to avoided primary-material production without supporting inventory exchanges and contribution analysis.

Published battery assessments demonstrate that recycling technology and geographical location influence the environmental performance of recovered materials [27]. Comparisons of recycling and second use also identify a trade-off between recovering materials for new batteries and retaining residual capacity for stationary storage [29]. Since second use was excluded from the model, its potential benefits fall outside the reported vehicle comparison.

The infrastructure supplying cleaner electricity introduces additional material-management requirements beyond the vehicle systems assessed here. Projections of EoL photovoltaic volumes illustrate the need to plan treatment capacity for electricity-generation equipment [56]. Life cycle assessments of electricity-generation technologies further demonstrate the importance of accounting for manufacturing and EoL processes alongside operational performance [57]. Assessments of photovoltaic-powered EV charging stations extend this accounting to charging-system components [58]. Renewable electricity can thus improve vehicle climate performance while requiring its own circular material-management strategy. Photovoltaic charging remains a possible extension of the current analysis rather than a quantified contributor to its results.

Assessment of circular industrial processes requires consideration of the environmental burdens associated with by-product recovery [26]. Biological processing offers alternative

routes for recovering valuable metals from spent batteries [28]. Waste valorization can also provide secondary material inputs, although the suitability of the recovered materials depends on their properties and intended application [59]. The environmental value of these pathways depends on energy demand, reagent consumption, recovery yields, and the quality of recovered products. Comparative LCAs of hydrometallurgical recovery from spent lithium-ion batteries illustrate these dependencies: organic-acid leaching was not consistently environmentally preferable to inorganic-acid alternatives, while solvent extraction generally generated greater impacts than leaching [60]. These findings show why recovery performance alone is insufficient to establish an environmental benefit. Evaluating alternative recycling routes would require process-specific inventories and consistent treatment of recovered products within the selected allocation framework. This would allow the comparison to determine whether improved recovery reduces life cycle impacts or shifts burdens between environmental categories.

### *4.7. Implications for Sustainable Mobility in Greece*

The assessed baseline illustrates environmental trade-offs that may inform discussion of passenger-vehicle choices involving the Greek electricity supply. The BEV has the lowest modelled global-warming damage, while the HEV performs better in resource scarcity and several toxicity and eutrophication pathways. These findings concern the specified configurations and do not establish the environmental outcome of national fleet electrification.

These vehicle-level outcomes cannot be translated directly into national emission reductions. Transport demand, fleet composition, vehicle replacement, and electricity-system development determine how improvements in individual vehicles affect total emissions [1]. European electricity-demand research indicates that increasing EV deployment introduces additional charging demand whose climate implications depend on electricity generation [61]. Energy-system modelling for Cyprus similarly illustrates the importance of coordinating transport electrification with electricity planning and charging strategies [62].

European econometric research has associated BEV adoption with lower GHG emissions, providing evidence at a different analytical scale [63]. This association, however, does not validate the numerical reductions calculated for an individual vehicle. Transport-policy reviews likewise place electrification and hybridization within a broader combination of infrastructure, technology, and policy measures [64].

For Greece, the findings highlight the importance of vehicle efficiency, electricity decarbonization, and material recovery. These priorities complement the transport-electrification objectives examined in the assessment of Greece's National Energy and Climate Plan [6], while broader charging-infrastructure research highlights the importance of charging provision and grid integration [8]. The favourable HEV results in several indicators also demonstrate the need to distinguish conventional hybrids from fully electric vehicles when assessing environmental performance.

Manufacturing improvements offer a further direction for research. Work on additive manufacturing has demonstrated how component redesign can reduce printing-support requirements and facilitate subsequent production through injection moulding [65]. Although developed for medical equipment, this example illustrates the relevance of assessing design and production choices together. Applying such approaches to vehicle components would require technology-specific inventories and LCAs to establish their environmental implications.

The study identifies category-specific trade-offs for the assessed configurations. National policy evaluation would additionally require fleet composition, vehicle replacement, transport demand, and evolving electricity-supply scenarios. The present results therefore

provide a vehicle-level comparison rather than a quantified recommendation for a national technology mix.

## 5. Conclusions

This study compared the cradle-to-grave environmental impacts of gasoline ICEVs, non-plug-in gasoline HEVs, and BEVs using explicitly defined vehicle configurations and operating parameters, with Greek electricity supply represented for BEV charging. Over a lifetime distance of 150,000 vehicle-kilometres, both electrified configurations exhibited lower characterized impacts than the ICEV across all 18 ReCiPe 2016 endpoint (H)-positive indicators. The BEV provided the lowest impacts in the three global-warming pathways and several air-pollution-related indicators. Relative to the ICEV, global-warming damage to human health decreased by 43.2%, fine particulate-matter formation by 44.8%, and terrestrial acidification by 45.1%. These percentage reductions refer specifically to ReCiPe endpoint damage indicators for the assessed vehicle configurations. The HEV performed better in fossil and mineral resource scarcity, both human-toxicity indicators, freshwater ecotoxicity, and freshwater and marine eutrophication. Its fossil and mineral resource-scarcity impacts were 51.5% and 55.8% lower than those of the ICEV, respectively. The mineral resource-scarcity results are characterized endpoint indicators for the modelled inventories and do not establish the total quantities of metals or critical raw materials required by each vehicle. The ranking-margin analysis identified comparatively narrow BEV–HEV differences in several indicators. Increasing the lower result by 9.05% for global-warming damage to human health or 4.25% for human carcinogenic toxicity would produce equal impacts with the competing value held constant. These thresholds describe numerical separation between the baseline results; they do not quantify parameter sensitivity or the probability of ranking reversal and do not establish that the rankings would remain unchanged under different operating conditions. The findings support a category-specific assessment of vehicle electrification alongside improvements in vehicle efficiency and electricity supply. They remain conditional on the selected inventories, battery representations, lifetime distance, and EoL modelling choices. Future work should examine the stage contributions and variations in electricity supply, energy consumption, battery production, and service life.

**Supplementary Materials:** The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/app162110737/s1, Table S1: Vehicle-production inventory; Table S2: Use-phase parameters and lifetime calculations. References [34–38] correspond to the main-manuscript reference list; Table S3: End-of-life treatment inventory; Table S4: Modelling settings; Table S5: ReCiPe 2016 endpoint (H) results; Table S6a,b: Relative differences, BEV–HEV ranking margins, and absolute impact differences.

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## Abbreviations

The following abbreviations are used in this manuscript:

| Abbreviation | Definition |
|---|---|
| BEV | Battery electric vehicle |
| DALY | Disability-adjusted life year |
| EoL | End of life |
| EV | Electric vehicle |
| GHG | Greenhouse gas |
| GR | Greece, dataset geographical code |
| HEV | Hybrid electric vehicle |
| ICEV | Internal combustion engine vehicle |
| ISO | International Organization for Standardization |
| LCA | Life cycle assessment |
| LCIA | Life cycle impact assessment |
| PHEV | Plug-in hybrid electric vehicle |
| USD2013 | US dollars, expressed using the 2013 monetary basis |

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