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Assessing Life Cycle Impacts and the Risk and Uncertainty of Alternative Bus Technologies

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ABSTRACT

Low-emission alternative bus technologies are of increasing interest to bus fleet operators due to the reduced environmental impact and potential for lower operating costs. However, with uncertainty regarding the total cost of ownership of new technologies and life cycle impacts beyond the typical well-to-wheel boundary, stakeholders may not have the necessary specific tools or evidence to evaluate life cycle impacts. The aim of this paper is to develop a novel framework to assist decision-makers in assessing the uncertainty of the life cycle impacts of alternative bus technologies. The Technology Impact Forecasting methodology was employed, integrating a life cycle model, to investigate whole life cycle impacts in an exploratory assessment environment, allowing for the analysis and trade-off evaluations of alternative drivetrain technologies and operational scenarios. This research provides a comprehensive novel framework for addressing uncertainty in whole life cycle costs and GHG emissions for the manufacture, use, maintenance and infrastructure phases of diesel and battery electric buses. Eleven scenarios are assessed in the framework, evaluating combinations of battery technologies, well-to-tank pathways, charging infrastructure and auxiliary demands. For every battery electric bus scenario, there is an 80% confidence that life cycle GHG emissions are mitigated by 10% to 58% compared to the baseline diesel bus, but life cycle costs are 129% to 247% higher. Opportunity charged electric buses employing a lithium-titanate battery are the most effective scenario for mitigating GHG emissions per additional cost of the new technology to the operator. The framework highlights a key trade-off between dependence on battery capacity and high-power charging infrastructure for battery electric bus technologies. The framework enables stakeholders to make technology adoption and resource allocation decisions based on the risk of a scenario and provides a level of confidence in a technologies' ability to mitigate whole life cycle impacts.

KEYWORDS

Risk and Uncertainty, Technology Impact Forecasting, Life Cycle Assessment, Life Cycle Costing, GHG Emissions, Alternative Bus Fleets.

1. INTRODUCTION

1.1 Background

The 2015 Paris Agreement signalled a global commitment to mitigate the effects of climate change caused by anthropogenic greenhouse gas (GHG) emissions [1]. The EU has also committed to reducing GHG emissions by 80-95% by 2050 compared to 1990 levels [2]. The transport sector is responsible for ~25% of EU GHG emissions [3], and as part of measures to address this emissions standards have been introduced for heavy duty vehicles, e.g. Euro VI legislation [4]. For large operators of heavy duty vehicles, such as bus fleets, the need to conform to environmental legislation is steering interest towards low emission vehicles. However, market penetration of alternative technologies, e.g. battery-electric buses (BEB), is hindered by higher acquisition costs compared to conventional diesel vehicles [5–7]. There are also concerns about the total cost of ownership (TCO) with uncertainty regarding additional infrastructure, maintenance routines and the sensitivity to energy costs [5–10].

Stakeholders seeking to employ alternative driveline technologies need to evaluate both economic and environmental effects. However, the many factors in a vehicle life cycle lead to high levels of variation in whole life cycle impacts reported in literature, both for a specific technology and when comparing multiple technologies (Figure 1 and 2). Although a decreasing trend can be observed in whole life cycle GHG emissions with increased electrification, there is wide variation in the results, and there is no clear trend for whole life cycle costs versus technology type. Such variation leads to uncertainty when comparing alternative technologies. To assist in this complex decision-making process, there is a need for a rapid assessment environment to evaluate whole life cycle environmental and economic impacts of alternative bus driveline technologies (and varying operational conditions) and to quantify the potential uncertainty and assess key sensitivities of the vehicle life cycle.

1.2 Aim and Focus of Paper

The aim of this paper is to develop a novel framework to assist decision-makers in assessing the uncertainty of the life cycle impacts of alternative bus technologies. This paper focuses on conventional Euro VI diesel and theoretical battery electric bus technologies. A BEB provides a good case study; in terms of the degree of vehicle electrification, conventional diesel and battery-electric vehicles are on contrasting ends of the scale [30]. Some knowledge of alternative propulsion systems is assumed and will not be covered in this paper, as many review studies cover these topics extensively, e.g. [7,31–35]. Note that the concepts of risk and uncertainty can differ depending on the field of research e.g. economics [36]. In the context of this paper, uncertainty is a state of limited knowledge, where possible states or outcomes can be quantified by assigning probabilities to these states or outcomes [37]. Risk is therefore the quantified probability of an outcome occurring [37].

1.3 Life Cycle Modelling of Bus Technologies

Life cycle analysis (LCA) methods (Table 1) are typically used to compare alternative technologies. There are three common types of LCA: process-based, economic input-output (EIO-LCA) and a combination of the two, hybrid-LCA. Process-based LCA considers the inputs (energy, materials, etc.) and resultant outputs (emissions, waste, products, etc.) of each unit process over a product's life cycle i.e. a bottom-up approach [38]. EIO-LCA uses monetary transactions between economic sectors to characterise the product's supply chain, including all direct and indirect impacts i.e. a top-down approach [39]. EIO-LCA has the potential for use in a design process, but a hybrid-LCA approach is recommended if more precision is required [40]. Although hybrid-LCAs can still include truncation errors inherent in EIO-LCAs, they can yield a more complete set of results than a single modelling approach [41].

In the context of buses, lack of available component data (e.g. bill of materials) is often cited as a reason why bottom-up studies don't consider the manufacturing phase in life-cycle modelling studies [14,15]. Hybrid-LCAs can provide the additional fidelity of process-based methods for key sections of a product's life cycle, e.g. the WTW phase [28], with EIO-LCA methods covering the product's raw material extraction and fabrication. Previous work has tended to use EIO-LCA data for standard components [21,42], while process based LCA has been combined with EIO-LCA results to quantify the impacts from the addition or replacement of specific items, e.g. battery [9,17]. The literature review highlighted the following key findings/recommendations regarding the set up a hybrid-LCA bus model:

- The use of aggregated process-based WTT inventories from literature or model databases [43,44] is prevalent in many studies, where the quantifiable environmental impact is used as a model input and not mapped for specific pathways [11,12,16,19–21,45–47]. However, WTT environmental impacts are highly location specific and can vary with time, for example the GHG impacts of electric vehicles are dependent on grid composition, which can fluctuate daily and change significantly annually [7,48]. To be of relevance to stakeholders, it is therefore important that life cycle inventories reflect specific geographical and time-related variations.
- Although life cycle impacts can be significantly affected by operational factors, such as auxiliary loads [14,15,20,46] and drivetrain component replacement frequency [5,6,11,17,24,25,49], many models use static inputs of fuel economy [13,17,21,23,42,50] and battery replacements [9–11,17,46]. However, other research has used vehicle simulation models that are sensitive to varying input parameters to predict the fuel or energy consumption, costs and emissions criteria over a drive cycle [5,6,11,14,46,49]. Further developments have integrated the simulation models into life cycle models to calculate battery replacements [5,6,49]. The use of formulation-approach mathematical models, reflective of how urban buses are used in service, will ensure an appropriate level of modelling fidelity.

- With all LCAs it is important to avoid double counting. This is of particular concern in the quantification of the environmental impacts associated with scheduled repair and maintenance as there is relatively limited literature on the subject [9,10,17,18,46,51]. Previous work used hybrid LCA, but the methodology has not been fully developed and double counting was apparent. Caution is advised to ensure system boundaries are consistent for maintenance and repair as well as all other stages of the life cycle.
- Although WTW assessment is a sound scientific methodology for comparing alternative drivelines, it can be considered a ‘narrow point of view’ as it excludes the impacts of raw material extraction and processing, manufacturing, and decommissioning of the vehicle itself [28]. Of the bus life cycle modelling studies reviewed, only four consider the economic impacts of use, acquisition/manufacturing, maintenance and infrastructure phases [15,24,46,49] and only one study considered environmental impacts of the same system boundary phases [17]. The inclusion of the ‘equipment life cycle’ provides a more comprehensive mapping of the vehicle’s environmental impact and can highlight high impact contributing factors in the life cycle of new technologies, e.g. battery manufacturing for electric vehicles [52].

1.4 Assessing Risk and Uncertainty of Alternative Bus Technologies

Uncertainty can be addressed by conducting sensitivity analysis to find ‘critical’ factors of importance to analysts [29]. Incorporating probabilistic techniques (e.g. Monte Carlo simulation [53]) and ‘what-if’ analysis can further produce a range of possible scenario outcomes [29]. Probabilistic techniques can enhance the understanding of complex comparisons between diesel and electric vehicles [54], but are uncommon in bus life cycle studies; previous research evaluates fuel economy inputs [9] or the acquisition cost, fuel consumption, fuel price and maintenance costs influencing TCO [8]. One methodology that facilitates sensitivity, probabilistic and ‘what-if’ analyses is the Technology Impact Forecasting (TIF) technique, which generates an environment for the exploration and quantitative assessment of a technology’s impact on a baseline system.

2. METHODOLOGY

2.1 Overview

The creation of a TIF analysis environment (Figure 3) in this paper follows the detailed methodology outlined in Soban and Mavris [56]. The TIF method includes sensitivities to design goals and constraints to evaluate the effect of a ‘technology scenario’. Using the concept of multiplicative factors (known as k _factors) to model technologies by their effect on key metrics (e.g. vehicle mass), TIF can allow the assessment of existing and undefined technologies. Probabilistic analysis within the TIF environment then permits stakeholders to view not only what could happen, but how likely it is to happen [55]. TIF has previous applications in the aerospace field, but has not been used for analysing bus technologies.

This paper addresses the challenge of synthesizing a compatible framework combining TIF and LCA approaches. However, consideration must first be given to the creation of a suitable life cycle model.

2.2 Life Cycle Model Development

2.2.1 Model Overview

The modelling environment contains multidisciplinary sub models integrated into the life cycle system boundary, consisting of materials processing and manufacturing (for vehicle and infrastructure), vehicle use, and fuel and electricity production phases (Figure 4). The following sections detail how the main multidisciplinary sub models are integrated into the life cycle system boundary. Under the guidance of ISO 14040 [26], comparisons between the two drivetrain technologies are made on the basis of the same function by including the boundary of other systems for battery electric vehicles, e.g. infrastructure requirements, battery replacements and electricity generation scenarios.

2.2.2 Hybrid LCA Model

The hybrid LCA model contains a bespoke EIO-LCA model (Box 1) for vehicle maintenance, manufacturing and infrastructure construction in the UK. Supply and use tables and GHG emissions datasets are UK specific [58,59]. Key calculated outputs of the EIO-LCA model are listed in Table 2.

Process-based LCA data inputs are used to calculate vehicle and battery production GHG emissions, GHG_{manf} (equation 3).

$$GHG_{manf} = 0.630(f_{cap}C_{cap}) + Q_{batt}GHG_{batt} \quad (3)$$

where f_{cap} is the multiplicative capital cost factor, C_{cap} is the capital cost of the baseline vehicle (GBP), Q_{batt} is the energy storage capacity (kWh) and GHG_{batt} is the emissions from the production per kWh of the energy storage system (kgCO₂e/kWh). The vehicle acquisition cost, C_{acq} , is calculated using equation 4.

$$C_{acq} = f_{cap}C_{cap} + Q_{batt}C_{batt} \quad (4)$$

where C_{batt} is the energy storage cost per kWh of capacity (GBP/kWh).

The life cycle cost of infrastructure (C_{LC_infra}), operation and maintenance costs (C_{infra_main}) and the respective GHG emissions (GHG_{infra} , GHG_{infra_main}) are calculated using equations 5 and 6.

$$C_{LC_infra} = C_{infra} + \sum_{y=0}^{Y_s} [(C_{infra}f_{O\&M})(1+r)^y] \quad (5)$$

$$GHG_{LC_{infra}} = 0.539 C_{infra} + \sum_{y=0}^{Y_s} [0.273(C_{infra} f_{o\&m})] \quad (6)$$

where C_{infra} is the capital infrastructure cost per bus, y is the service year (yr), Y_s is the total number of service years (assumed 12 yrs [17]), $f_{o\&m}$ is the yearly infrastructure operation and maintenance cost as a percentage of the capital cost (3% [49]) and r is the discount rate, 3.5% [60]. Emissions coefficients derived from the EIO-LCA model (Table 2) are contained in equation 6.

2.2.3 Use Phase Model

An algorithm was developed for calculating the vehicle energy demand for the drive-cycle for both drivetrain configurations (Appendix A – Figure A1). Key assumptions and constants used are contained in the algorithm key (Appendix A). The BEB mass is calculated by subtracting the mass of a Euro VI engine (500 kg [61]), automatic transmission (329 kg [62]) and after-treatment systems (assumed 200 kg) from the diesel tare mass and adding the mass of the motor-controller (350 kg [63]), and the calculated battery mass. The total drive cycle energy demand, W_{total} (MJ), is calculated using equation 7 and the life cycle fuel or electricity cost in the use phase, C_{use} , is calculated using equation 8.

$$W_{total} = \left(\int_{t=0}^{t=n} W_{trac} \cdot dt / \eta_{drive} \right) + \int_{t=0}^{t=n} W_{idle} \cdot dt + (P_{aux} / \eta_{aux}) \sum_{t=0}^{t=n} T - \left(\int_{t=0}^{t=n} W_{regen} \cdot dt / \eta_{drive} \right) \quad (7)$$

$$C_{use} = \sum_{y=0}^{Y_s} [(3.6 W_{total} C_{f/e} / \eta_{th}) (D_y / D_{DC}) (1 + r)^y] \quad (8)$$

where η_{th} is the engine thermal efficiency (%), $C_{f/e}$ is the fuel or electricity cost per kWh (GBP/kWh), D_y is the annual distance travelled (assumed 50,000 km) and D_{DC} is the drive cycle distance covered. WTW fuel and electricity GHG emissions, GHG_{WTW} , are calculated using equation 9.

$$GHG_{WTW} = Y_s (D_y / D_{DC}) (W_{total} / \eta_{th} \eta_{ch}) (GHG_{WTT} + GHG_{TTW}) \quad (9)$$

where η_{ch} is the BEB charging efficiency, 97% [64], GHG_{WTT} is the WTT GHG emissions factor (gCO_{2e}/MJ) and GHG_{TTW} is the TTW GHG emissions factor (gCO_{2e}/MJ) (Table 5). For the BEB configuration, $\eta_{th} = 100\%$ in equations 8 and 9. For the diesel configuration, $\eta_{ch} = 100\%$ in equation 9. Life cycle maintenance costs and GHG emissions account for the number of battery replacements required in the BEB's service life, calculated using equation 10 [5].

$$B_{life} = N_{cyc} Q_{batt} / E_{km} \rightarrow N_{batt} = Y_s D_y / B_{life} \quad \text{where, } N_{batt} \in \mathbb{N} = \{1, 2, 3 \dots N\} \quad (10)$$

where E_{km} is the energy throughput per km (kWh/km), B_{life} is the energy storage life of the battery in terms of kilometres driven (km) and N_{batt} is the number of replacements required throughout the service

life of the BEB. Maintenance life cycle costs (C_{LC_main}) and GHG emissions (GHG_{LC_main}), are calculated using equations 11 and 12 respectively.

$$C_{LC_main} = \sum_{y=0}^{Y_s} [C_{main} D_y (1+r)^y] + N_{batt} C_{batt} Q_{batt} \quad (11)$$

$$GHG_{LC_main} = 0.247 (Y_s C_{main} D_y) + N_{batt} C_{batt} Q_{batt} \quad (12)$$

where C_{main} is the maintenance cost per km, excluding battery replacements (GBP/km). The maintenance emissions coefficient derived from the EIO-LCA model (Box 1) is contained in equation 12.

2.2.4 Life Cycle Model Outputs

The total cost of ownership, C_{TCO} , and life cycle GHG emissions, GHG_{LC} , were calculated using equations 13 and 14 respectively.

$$C_{TCO} = C_{acq} + C_{LC_infra} + C_{use} + C_{LC_main} \quad (13)$$

$$GHG_{LC} = GHG_{manf} + GHG_{LC_infra} + GHG_{WTW} + GHG_{LC_main} \quad (14)$$

2.3 Creation of the TIF Environment

2.3.1 Baseline System

The baseline system in this study was a conventional Euro VI double decker bus with an assumed tare mass of 11,000 kg, a passenger load of 2176 kg (32 passengers at 68 kg [65]) and a 4 kW auxiliary load. The London Urban Bus (LUB) drive-cycle, a 16.1 km real world test cycle incorporating rural and urban cycles [66], was used in all of the simulated scenarios.

2.3.2 Selection of Candidate Technology Scenarios, Variables and Responses

Three battery technologies suitable for road transportation application [67] were selected for assessment: lithium nickel–cobalt–manganese (NMC), lithium-iron phosphate (LFP) and lithium-titanate (LTO) (Table 3). Unless otherwise stated, all BEBs were simulated with a 300 kWh battery pack and assumed overnight depot charging. Two WTT electricity generation pathways were selected: the yearly average UK electricity supply mix (for 2016) and electricity generated solely from wind power. UK grid emissions are assumed to decrease proportionally with projected UK energy supply GHG emissions (calculated at 3.5% per year from 2016 to 2028) [71]. Opportunity charging scenarios (assuming 300 kW fast charging infrastructure [72]) were simulated for the three battery pack technologies, downsized to 60 kWh. Finally, an additional 4 kW heating, ventilation and air-conditioning (HVAC) unit was simulated for the diesel and LFP buses.

k _factors with both positive and negative effects on the baseline and the technology scenarios were identified from the review of previous models in literature (Table 4). As ICE and battery electric drivelines vary significantly in terms of upstream fuel and energy conversion processes and on-board propulsion energy conversions [75], the metrics selected for the TIF environment are variables in at least one technology scenario. The fuel and electricity costs used in applicable scenarios are shown in Table 5. Finally, the responses of interest were identified that quantify the whole life cycle impacts of the technologies to be assessed (bottom of Figure 4).

2.3.3 Creation of Surrogate Models

k _factors were set as inputs into a design of experiments (DOE) table to assess the ‘cause and effect’ relationships between design variables and system responses. A central composite design was selected with one centre point, generating 159 computational runs. Performing variable combinations in the DOE table created response surface equations (RSEs), e.g. a polynomial equation representing the sensitivity of system inputs to outputs. The RSEs produce surrogate models of the entire system, capturing the empirical relationships and masking sensitive proprietary information. The RSEs were then mapped into an interactive visual profiler in the statistical analysis software package JMP® [85]. This prediction profile formed the basis of the TIF environment (Figure 5).

2.3.4 Probabilistic Analysis

To account for uncertainty a shape distribution function (e.g. Gaussian, uniform, triangular, etc.) is assigned to each k _factor. The shape functions used for all scenarios and the rationale for selection of k _factor ranges are contained in Appendix B. An example of the shape distributions applied to each k _factor for the LFP BEB (UK grid) scenario is shown in Figure 6. The shape functions were selected through consultation with industry practitioners and relevant literature (e.g. battery characteristics shown previously in Table 3). Monte Carlo simulation was conducted for 20,000 iterations and the results of each simulation run are presented as cumulative distribution functions (CDFs).

3. RESULTS

CDF plots of TCO and life cycle GHG emissions show both the likelihood and range of values that can occur for the LFP BEB (UK grid) scenario (Figure 7). The CDFs can be interpreted as the confidence of achieving a certain value, e.g. for a probability of 0.2, the TCO response of the LFP (UK grid) is 95% greater than the baseline conventional diesel result. Similarly, a probability of 0.8 corresponds to a 214% TCO increase (a 119% difference between the confidence levels). This result is interpreted as ‘given the assumptions made in the probability distributions, there is an 80% confidence that the TCO of the LFP BEB scenario will be no greater than 214% compared to the diesel baseline’. The same

probabilities for the life cycle GHG emissions plot yield 36% and 24% less emissions respectively. There is a 100% likelihood that life cycle GHG emissions are lower than the diesel baseline.

The baseline values for the conventional diesel Euro VI double decker bus are shown in Table 6 along with the 80% confidence response values for all eleven technology scenarios. This confidence has been selected as a reasonable risk level used by previous TIF studies [55,56].

Despite a 38% to 41% decrease in energy costs, the TCO of the three battery scenarios operating from the UK grid is between 221% and 247% higher than the baseline (Table 6). This is primarily due to the high acquisition and battery replacement costs, with the LFP scenario remaining the cheapest option. WTW GHG emissions reductions of up to 65% are observed for all three battery scenarios due in part to the anticipated decrease in future UK energy supply emissions [71]. Even with increases in GHG emissions from the manufacture of the vehicle and infrastructure, life cycle emissions are up to 30% lower than the diesel baseline for the three battery scenarios. The LTO BEB scenario, despite having the most expensive battery pack and highest TCO, has the lowest additional cost increase per kg of GHG emissions mitigated. The high charge-discharge cycle life of LTO batteries, leading to fewer battery replacements, is the primary reason for this result.

The WTT wind electricity generation scenarios demonstrate a major advantage of BEBs: for every scenario, WTW GHG emissions are reduced by up to 97% compared to the baseline for the 80% confidence band (Table 6). A decrease in electricity cost per kWh for the wind-based scenarios yield significant energy cost reductions compared to the diesel baseline (up to 66%) and noteworthy TCO reductions (from 13-19%) compared to the UK grid-based BEB scenarios.

For the opportunity charging scenarios, the reduction in vehicle energy consumption is due to reduced battery capacity, resulting in a decrease in vehicle mass and up to a 47-49% reduction in energy costs (Table 6). A smaller battery pack also yields cheaper acquisition costs for the BEB scenarios compared to the overnight charging scenarios. More frequent battery replacements are needed, but due to the lower pack cost, life cycle maintenance costs are lower than the overnight charging scenarios. Despite fast charging infrastructure costing approximately five times that of the overnight charging infrastructure, results show up to an additional 8-9% TCO saving over the overnight charge scenarios. TCO is predicted to be over double that of the baseline diesel bus. Of all of the scenarios simulated, the opportunity charge LTO BEB yields the lowest additional cost increase per kgCO_{2e} mitigated compared to the baseline diesel vehicle for the 80% probability band, aligning with previous research [49].

Results of the HVAC scenarios (Table 6) show that the more efficient drivetrain of the LFP BEB results in a 9% energy cost increase with a HVAC unit compared to the LFP BEB operating on the UK grid. In comparison, the diesel vehicle with the HVAC unit has 19% higher energy costs than the baseline.

The additional life cycle energy demand results in more battery replacements and higher maintenance costs and GHG emissions compared to the non-HVAC BEB. The LFP BEB (HVAC) has the highest TCO of all scenarios and still produces lower life cycle GHG emissions compared to the diesel baseline.

4. DISCUSSION

4.1 Informing Decision Makers using the Life Cycle Framework

By calculating both diesel and BEB impacts simultaneously, it can be inferred that the TIF environment can assess contrasting driveline technologies with relative ease, provided the k _factors included sufficiently cover the input requirements of all propulsion technologies evaluated. The results can be examined by decision makers to evaluate whether the technology has an acceptable risk level.

Comparison of the overnight and the opportunity charging scenarios reveals a trade-off between dependence on battery capacity and high-power charging infrastructure. Although the framework does not address the range implications of BEBs, these results do provide stakeholders with some answers to the complexities of designing and planning for the implementation of electric bus technologies. Given the current costs of charging infrastructure and battery packs, operators will find difficulties in justifying the purchase of BEBs purely from an economic perspective. By assigning additional resources to a technology and revisiting the assumptions in the shape distributions, the uncertainty bands would potentially decrease. The expected reduction in future battery pack costs could be one instance to revisit these scenarios with new assumptions [86]; increased mass-market production of lithium ion packs is expected to reduce battery acquisition costs, hence allowing for commercially viable BEBs.

Life cycle GHG emissions from BEBs are highly dependent on the electricity generation source, as demonstrated by the wind scenarios. BEBs in operation should be carefully selected and appraised with respect to emissions from electricity generation in the region of operation [17]. The life cycle framework first and foremost evaluates the impact of the vehicle itself, however it can be a powerful tool in assessing hypothetical or notional ‘what-if?’ electricity generation portfolios, as well as combinations of technology and operational conditions. The TIF environment is a powerful tool in realising what impact a conceptual design will have on whole life cycle costs and GHG emissions. Traditional TIF studies tend to evaluate technologies that are expected to benefit performance characteristics of a baseline. However, the HVAC scenarios demonstrate that the environment is also capable of assessing operational scenarios which have an adverse effect on vehicle performance.

4.2 Assessing Risk and Uncertainty using the combined LCA-TIF Framework

The novel LCA-TIF framework addresses some of the uncertainty and associated risks surrounding BEBs in terms of whole life cycle impacts via probabilistic analysis techniques, giving stakeholders confidence in the level of risk of a technology scenario. Users should be aware of the capabilities of the

framework and the effect of shape function selection. Soban and Mavris [56] caution users of the TIF methodology: ‘because the CDFs are entirely dependent on the shape distributions assigned and the technology scenarios defined, care needs to be taken that the shape functions are not inadvertently used to determine specific desired results’. The implication is that manipulating shape functions that are not representative of the technology could produce misleading life cycle results. Justifying shape function selection with additional data and consultation with disciplinary experts will ensure credible, traceable, and relevant technology assessments for stakeholders.

4.3 Recommendations for Future Life Cycle Model Developments

The life cycle model developed for this study may contribute to additional levels of uncertainty in the framework. For example, GHG emissions from the manufacturing and infrastructure phases rely heavily on the capital cost factors, an input to the EIO-LCA sub-model. Although EIO-LCA has been noted as a useful benchmarking tool for rapidly estimating life cycle emissions [21,40], identifying process improvements is difficult due to highly aggregated data. The use of process-based LCA data in bus life cycle studies is not new; previous research has used process-based LCA data for the manufacturing, maintenance and end-of-life phases of bus technologies [18,20,50,51,87]. However, the inclusion of additional process-based LCA data in the framework would help make more informed decisions and improve fidelity of the models used for the manufacturing, maintenance and end-of-life phases.

Life cycle impacts beyond the typical WTW or vehicle operation boundary can be distorted by the temporal scope and physical process pathways selected. This model, for example, does not include expected cost reductions of battery packs even though future pack replacements may be significantly cheaper and potentially have lower production emissions due to mass production [86]. Likewise, future energy costs remain static in the model, but including diesel or electricity cost prediction scenarios (e.g. [88]) would address this modelling gap. Some cost fluctuation can be accounted for by applying appropriate shape functions in the model and future BEB scenarios could address the cheaper overnight ‘off peak’ electricity tariffs and lower environmental impacts compared to ‘peak’ scenarios [48].

This model conducted a bus-to-bus comparison rather than a fleet-based approach, assuming that diesel and BEB technologies can complete the same route regardless of the range capabilities. However, Lajunen [5] found that additional BEBs are required for equivalent fleet operation, making the implementation of new fleets more expensive. Unique drive-cycles, auxiliary and passenger loads all affect the range and fleet size of plug-in vehicles. Therefore, a fleet-based approach would provide a more appropriate functional unit, as emissions reductions of a BEB fleet could potentially be less pronounced than vehicle-by-vehicle comparisons with diesel buses [89]. Fleet analysis would assist in the sizing and additional impact assessment of new infrastructure needs for plug-in vehicles. Finally, diesel and BEBs are not the only option available to operators; future model developments should appraise HEBs as ‘stepping stones [7]’ towards full electrification of urban bus transportation.

5. CONCLUSIONS

Meta-analysis of cost and GHG emissions from literature demonstrates high levels of variation and potential uncertainty when comparing life cycle impacts of bus technologies and operational conditions. This paper outlines the development of a novel framework combining TIF and LCA approaches to assist decision makers in addressing the uncertainty surrounding the life cycle impacts of the manufacture, use, maintenance and infrastructure of diesel and BEB technologies. The framework is realised as a rapid assessment environment capable of stochastic simulation to quantify this potential uncertainty. The framework evaluates positive and negative impacts of a bus technology and its response to ‘what-if?’ operational scenarios. The framework enables stakeholders to make technology adoption and resource allocation decisions based on the risk of a scenario and provides a level of confidence in a technologies’ ability to mitigate whole life cycle impacts. The main findings and lessons learned from the development and use of this framework are as follows:

- For every battery electric bus scenario, there is an 80% confidence that life cycle GHG emissions are mitigated by 10% to 58% compared to the baseline diesel bus, but life cycle costs are 129% to 247% higher.
- An opportunity charged LTO BEB is the most cost-effective scenario for mitigating GHG emissions per additional increase in cost to the operator. Stakeholders may wish to pursue this promising technology by assigning additional resources and revisiting assumptions made in the framework.
- A trade-off between dependence on battery capacity and high-power charging infrastructure is apparent between overnight and opportunity charging scenarios. The evaluation of a narrow system boundary (i.e. a WTW assessment) would overlook these key interactions in the life cycle.
- A hybrid-LCA approach provides a useful first insight into the direct and indirect impacts of alternative bus technologies. However, additional process-based modelling of manufacturing, maintenance and end-of-life phases would facilitate more informed early stage design decisions.
- Temporal aspects of the framework are limited; future model iterations should account for uncertainty scenarios of future energy mix pathways and component cost variations.
- The evaluation of a fleet-by-fleet assessment would offer a more appropriate functional unit and address the range limitations of BEBs. Additional drivetrain technologies should be integrated into the framework to enable side-by-side comparisons with conventional diesel buses.

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SUPPORTING INFORMATION

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.rser.2018.08.045.

APPENDIX A

The algorithm for calculating drive cycle energy consumption is shown in Figure A.1, where i = current time step (-), t = time (s), T = difference between timesteps (s), D = distance (m), v = vehicle velocity (m/s), a = vehicle acceleration (m/s²), α = road gradient (rad), e = elevation (m), F_{trac} = vehicle tractive force (N), F_{aero} = vehicle aerodynamic drag force (N), F_{rr} = vehicle rolling resistance force (N), F_{cr} = vehicle climbing resistance force (N), m = vehicle mass (kg), ρ_{air} = density of air, 1.225 kg/m³, C_d = drag coefficient (-), A_f = vehicle frontal area (m²), g = acceleration due to gravity, 9.81 m/s², C_{rr} = coefficient of rolling resistance (-), P_{brake} = braking power (kW), W_{brake} = braking energy (MJ), P_{trac} = tractive power (kW), W_{trac} = tractive energy (MJ), v_{low} = lower velocity bound for regeneration, 5km/s [64], a_{max} = maximum deceleration bound for regeneration, -3 m/s² [64], W_{regen} = braking energy regenerated (MJ), W_{idle} = idling energy (MJ), CV = calorific value of diesel fuel, 42.92 MJ/kg [78], FC_{idle} = idle fuel consumption rate (L/h), ρ_{fuel} = density of fuel, 0.839 kg/L [78], W_{total} = total energy demand for drive cycle (MJ), η_{drive} = drivetrain efficiency, η_{aux} = auxiliary system conversion efficiency (%), P_{aux} = auxiliary electrical power demand (kW) and η_{regen} = regeneration system efficiency (%).

APPENDIX B

Shape functions used in each technology scenario are shown in Table B.1. Dashes in Table B.1 indicate that the input has been left at the baseline value.

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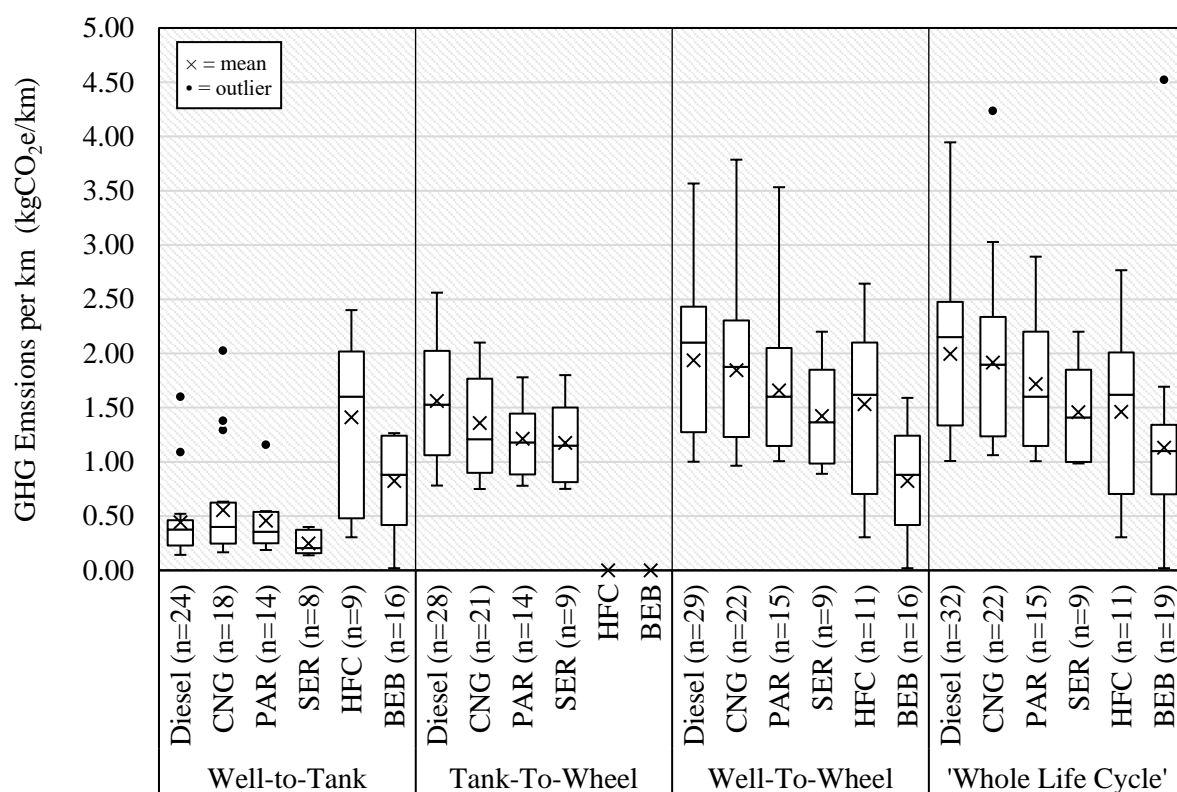


Figure 1. Box and whisker plots of GHG emissions of key life cycle phases for 6 of the most common drivetrain technologies evaluated in literature: diesel, compressed natural gas (CNG), parallel hybrid (PAR), series hybrid (SER), hydrogen fuel cell (HFC) and a battery electric bus (BEB) technologies. Sources: [7,9,11–23] converted to a functional unit of 1 vehicle-km. 'Whole Life Cycle' refers to the entire system boundary considered by each cited source. Table 1 provides descriptions of the Well-to-Tank, Tank-to-Wheel and Well-to-Wheel terms. See supplementary material for data.

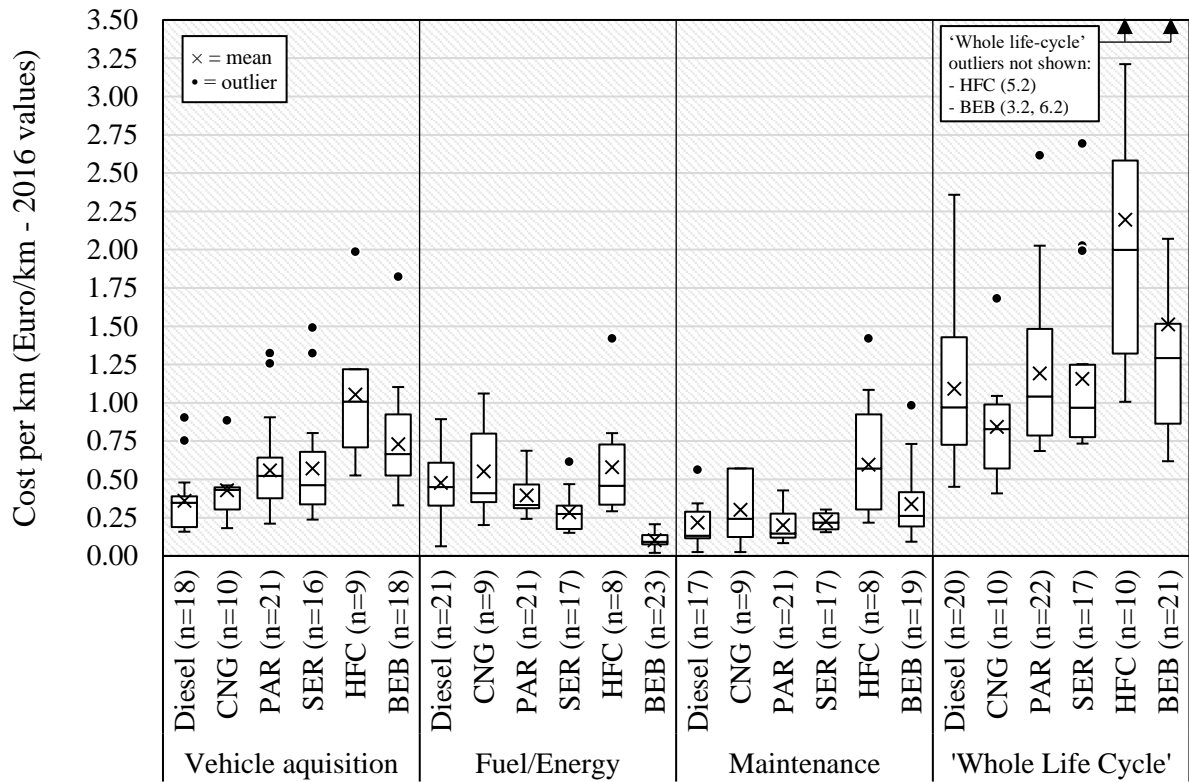


Figure 2. Box and whisker plots of costs (converted to 2016 equivalent values) of key life cycle phases for 6 of the most common drivetrain technologies evaluated in literature: diesel, compressed natural gas (CNG), parallel hybrid (PAR), series hybrid (SER), hydrogen fuel cell (HFC) and a battery electric bus (BEB) technologies. Sources: [5–9,15,21,24,25]. 'Whole Life Cycle' refers to the entire system boundary considered by each cited source. Table 1 provides descriptions of the Well-to-Tank, Tank-to-Wheel and Well-to-Wheel terms. See supplementary material for data.

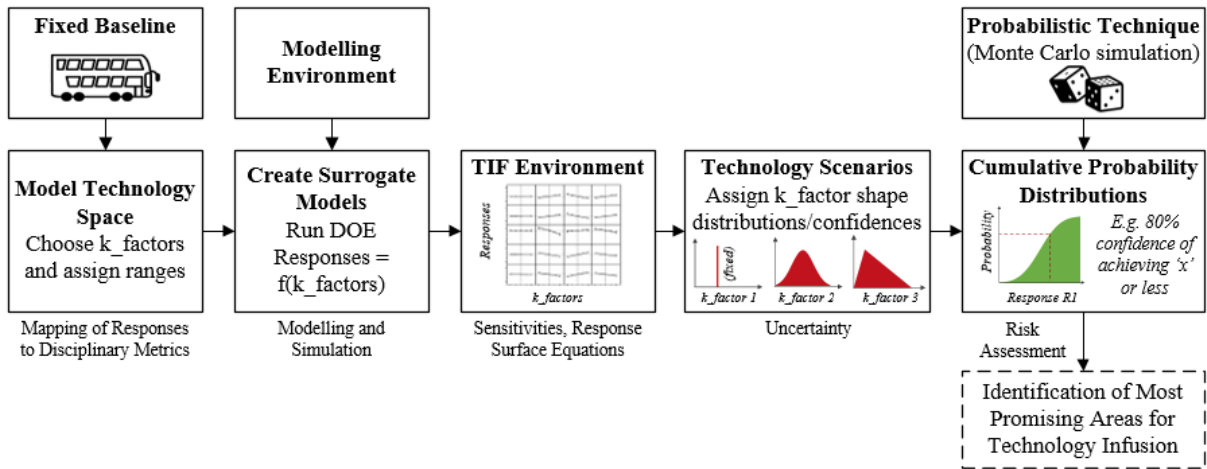


Figure 3. Process of creating a Technology Impact Forecasting analysis environment for assessing technology scenarios. Adapted from [55,56].

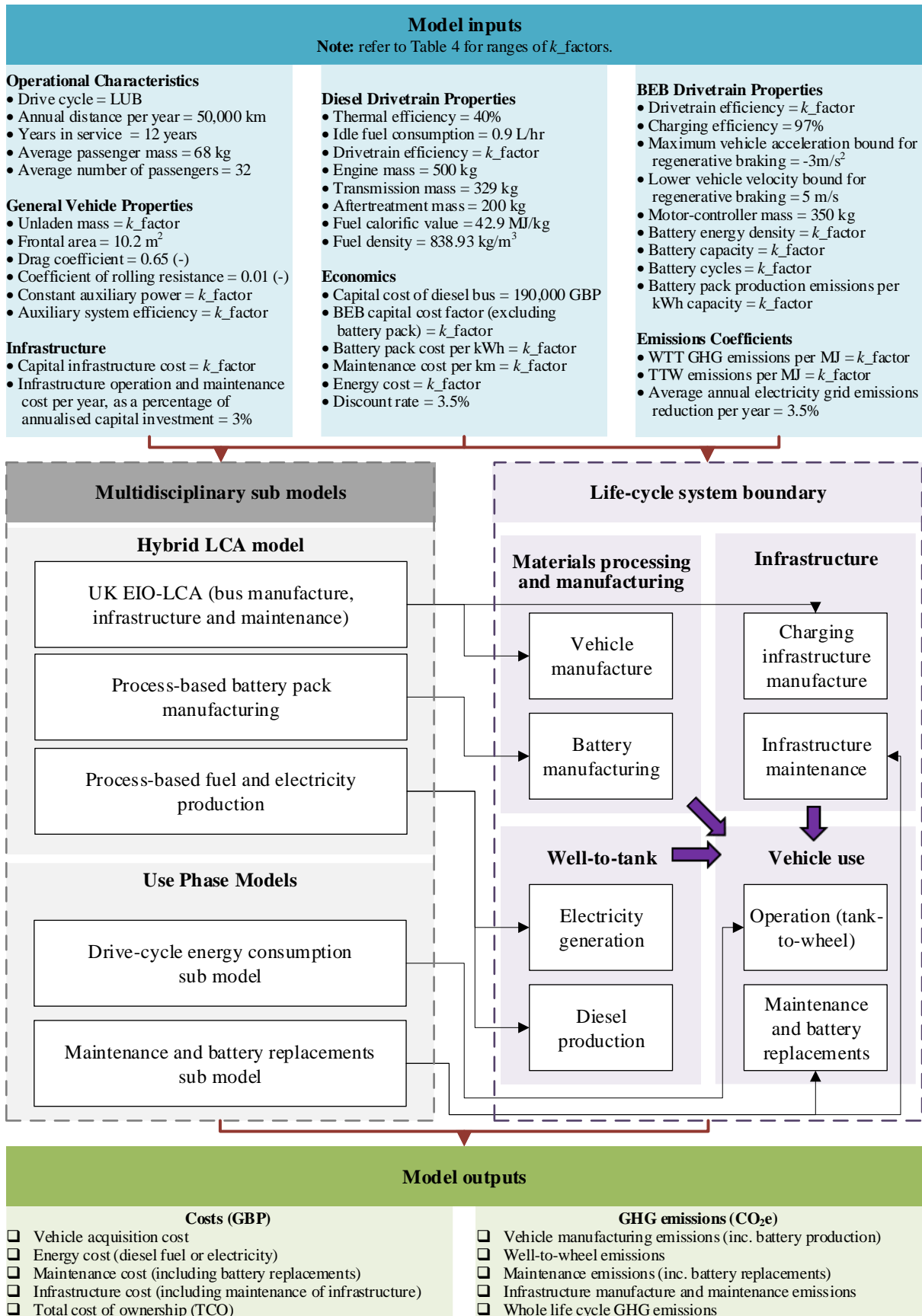


Figure 4. Life-cycle model.

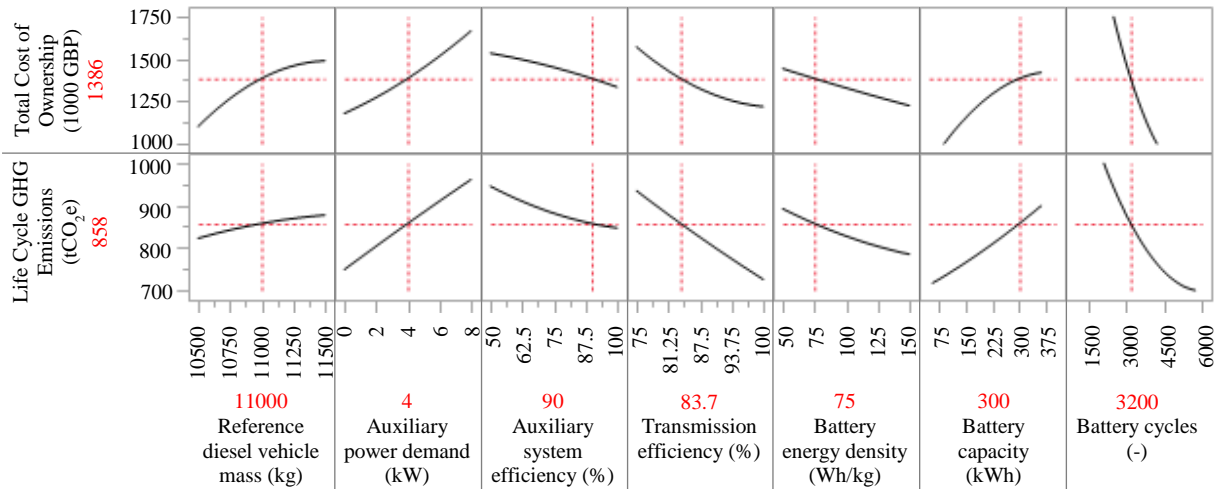


Figure 5. Sample of the TIF prediction profile environment generated for this study in JMP® [85].

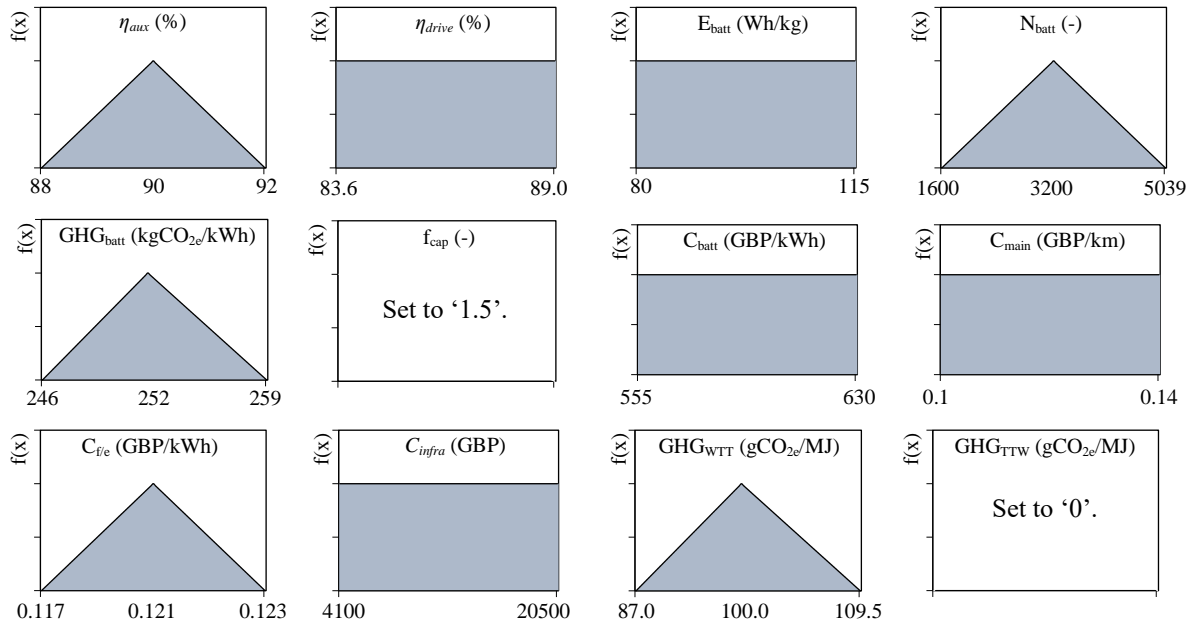


Figure 6. k _factor distributions for the LFP BEB (UK grid) technology scenario.

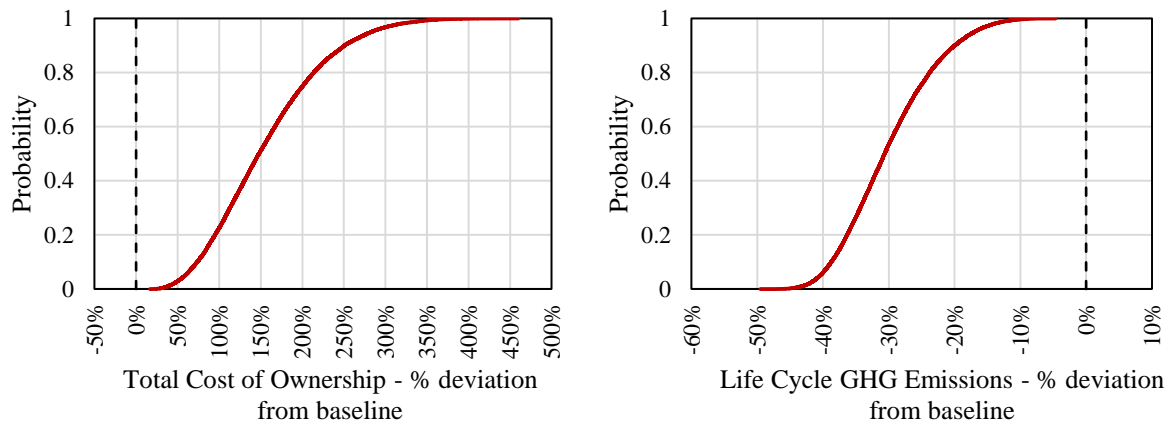


Figure 7. CDF plots for the LFP BEB (UK Grid) technology scenario.

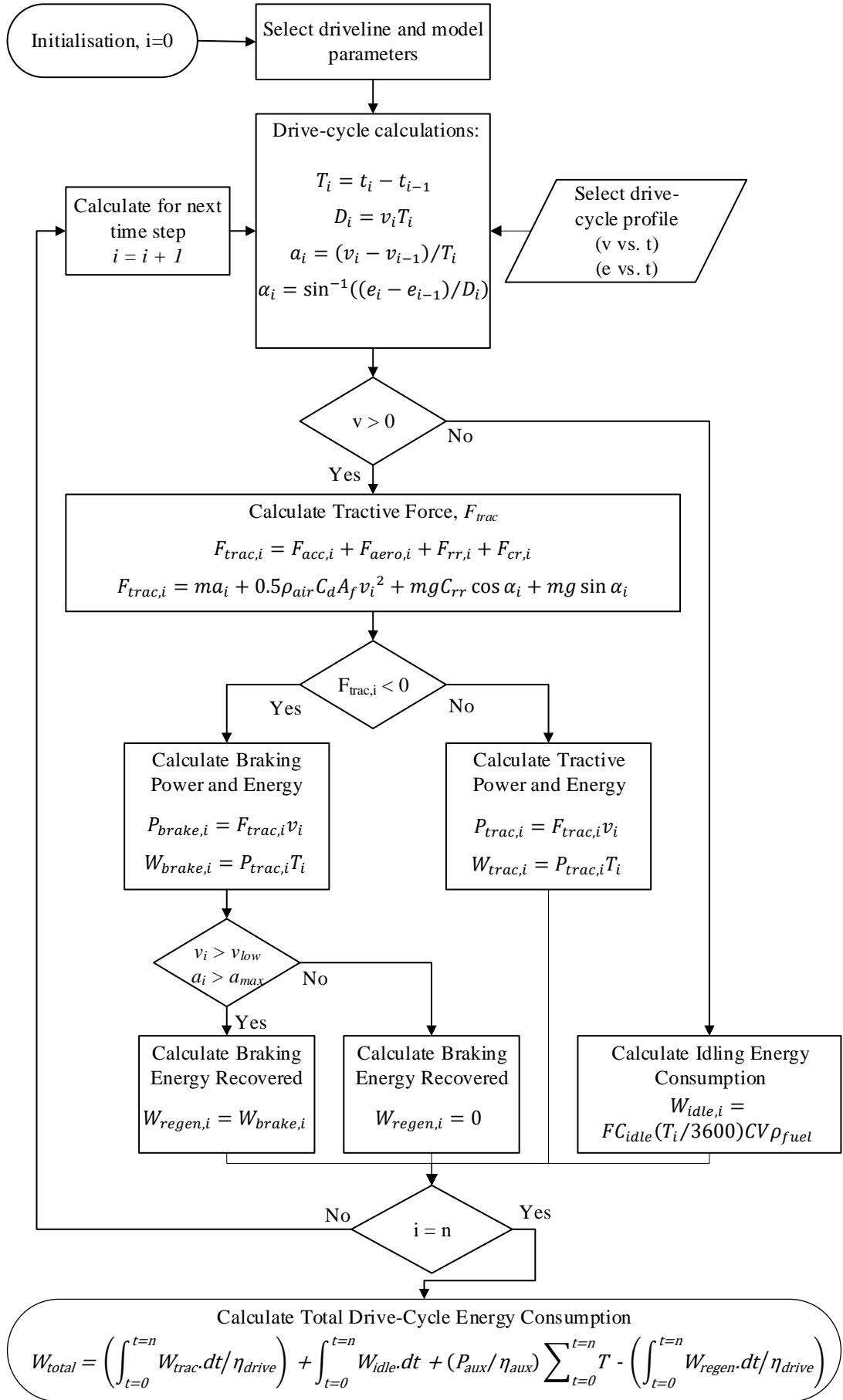


Figure A1. Algorithm for calculating drive-cycle energy demand. Terms defined in Appendix A.

Table 1. Life cycle analysis terminology.

Term	Description
Life cycle analysis (LCA)	A methodology which addresses the potential environmental impacts throughout a product's life cycle from raw material acquisition through production, use and end-of-life treatment [26,27].
Well-to-wheels (WTW) analysis	The dominant LCA approach for comparing alternative vehicle technologies. Widely used for policy support in road transport [28]. Focuses on the processes of the energy carrier (i.e. diesel or electricity) used to propel the vehicle during operation. Comprises the well-to-tank (WTT) and tank-to-wheel (TTW) phases.
Well-to-tank (WTT) analysis	Comprises the recovery or production of the feedstock for the energy carrier and subsequent energy conversion, delivery/transmission and storage.
Tank-to-wheels (TTW) analysis	Comprises the on-board energy conversion to drive the vehicle based on the lifetime distance travelled, fuel energy required and vehicle efficiency [29].

Table 2. Values obtained from the UK EIO-LCA model (adjusted for 2016 costs).

SIC	Sector Description	GHG Emissions produced per GBP demand (kgCO ₂ e/GBP)
29	Manufacture of motor vehicles, trailers and semi-trailers	0.630
45	Repair services of motor vehicles and motorcycles	0.247
27	Manufacture of electrical equipment	0.539
33	Rest of repair	0.273

Table 3. *Battery technology characteristics.*

Energy Storage Type	Specific Energy, E_{batt} (Wh/kg)	Cycles, N_{batt}	Estimated Pack Cost per kWh, C_{batt} (USD/kWh)	Estimated Production emissions, GHG_{batt} (kgCO ₂ e/kWh)
NMC	75 – 170 ^a	1000 – 3000 ^a	750 – 850 ^b	248 – 267 ^a
LFP	80 – 115 ^a	1600 – 5039 ^a	900 – 1540 ^{b,c}	246 – 259 ^a
LTO	45 – 100 ^a	2000 – 6800 ^a	1500 – 2000 ^b	254 – 279 ^a

^a Values correspond to electric vehicle traction batteries from [68].

^b Reported battery pack costs for bus applications from [69].

^c Converted at rate of 0.81 Euros per USD from [70].

Table 4. Driveline technology variables, baseline values and ranges of variability studied.

k_factor (variable of interest)	Baseline value	Deviation from baseline	
		Lower	Upper
Reference diesel vehicle tare mass, m (kg)	11,000	0%	+5%
Auxiliary power demand, P_{aux} (kW)	4	0%	+100%
Auxiliary system efficiency, η_{aux} (%)	60% ^a	0%	+53%
Drivetrain efficiency, η_{drive} (%)	76.8% ^b	0%	+16%
Battery capacity, Q_{batt} (kWh)	0 ^c	0	300
Battery specific energy, E_{batt} (Wh/kg)	0 ^c	0	170
Battery cycles, N_{batt} (-)	0 ^c	0	5039
Battery production emissions, GHG_{batt} (kgCO _{2e} /kWh)	0 ^c	0	300
Capital cost factor (exc. battery), f_{cap} (-)	1	0%	+50%
Battery pack cost per kWh, C_{batt} (GBP/kWh)	0 ^c	0	1480
Maintenance cost per km, C_{main} (GBP/km)	0.16 ^d	-38%	+10%
Energy cost per kWh, C_{fe} (GBP/kWh)	0.054 ^e	-7%	+44%
Infrastructure cost per bus, C_{infra} (GBP)	1199 ^f	0%	+9093%
WTT GHG emissions, GHG_{WTT} (gCO _{2e} /MJ)	15.4 ^e	-87%	+878%
TTW GHG emissions, GHG_{TTW} (gCO _{2e} /MJ)	76.2 ^e	-100%	+15%

^a Assumed alternator efficiency [73].

^b Sum of clutch, gearbox, final drive and wheel efficiencies from [64].

^c Traditionally in TIF studies, percentage deviations from the baseline are used, this study also includes variability of inputs applicable to BEBs only.

^d Converted at a rate of 0.82 GBP per Euro from [6],

^e See Table 5.

^f Calculated based on lifetime diesel consumption and operation & maintenance cost used in [74].

Table 5. *GHG Emissions coefficients and energy costs used in the life cycle model.*

	<i>GHG_{WTT}</i> (gCO _{2e} /MJ)			<i>GHG_{TTW}</i> (gCO _{2e} /MJ)			<i>C_{fl/e}</i> (GBP/kWh)		
	Min	Mean	Max	Min	Mean	Max	Min	Mean	Max
Diesel	13.8 ^a	15.4 ^a	17 ^a	75.7 ^d	76.2 ^d	76.5 ^d	0.046 ^e	0.054 ^e	0.061 ^e
Electricity (UK grid)	87.0 ^b	100.0 ^b	109.5 ^b	0	0	0	0.117 ^f	0.121 ^f	0.123 ^f
Electricity (wind)	1.94 ^c	3.06 ^c	15.6 ^c	0	0	0	0.050 ^g	0.067 ^g	0.081 ^g

^a Crude oil production, transport, refining into diesel fuel, distribution and dispensing on site [76].

^b Calculated using 2016 emissions data from [77] and transmission losses in the UK grid [78].

^c Values obtained from [79].

^d Calculated using energy and emission results of a representative Euro VI diesel engine [80].

^e Weekly road fuel price 2016 excluding VAT and a fuel duty rebate of 0.3457 GBP per L [81,82].

^f Average annual 2016 UK electricity cost assuming small consumer size and includes Climate Change Levy [83].

^g Values obtained from [84].

Table 6. Comparison of results with 80% probability for all ten technology scenarios.

Drivetrain/Battery configuration	Baseline result	UK grid scenarios			Wind scenarios			Opportunity charging scenarios			HVAC scenarios	
	Diesel	NMC ^a	LFP ^a	LTO ^a	NMC ^a	LFP ^a	LTO ^a	NMC ^a	LFP ^a	LTO ^a	LFP ^a	Diesel
Costs (1000 GBP)	^dCosts % deviation from baseline value (80% probability)											
Vehicle acquisition	190.0	+147%	+198%	+255%	+147%	+198%	+255%	+69%	+80%	+91%	+209%	+8%
Fuel/Energy	269.5	-41%	-40%	-38%	-66%	-65%	-64%	-49%	-48%	-47%	-32%	+19%
^b Maintenance	119.6	+936%	+822%	+889%	+948%	+821%	+878%	+642%	+546%	+599%	+1032%	+7%
^c Infrastructure	1.2	+2046%	+2050%	+2038%	+2354%	+2340%	+2039%	+9681%	+9682%	+9640%	+2340%	+0%
TCO	580.2	+221%	+214%	+247%	+212%	+201%	+231%	+145%	+129%	+144%	+264%	+13%
GHG Emissions (tCO₂e)	^dGHG Emissions % deviation from baseline value (80% probability)											
Manufacturing	119.8	+115%	+114%	+117%	+115%	+114%	+117%	+63%	+63%	+63%	+124%	+8%
WTW	992.9	-65%	-65%	-63%	-97%	-97%	-96%	-70%	-69%	-69%	-60%	+20%
^b Maintenance	24.3	+1692%	+1021%	+673%	+1714%	+1035%	+669%	+1607%	+935%	+563%	+1225%	+32%
^c Infrastructure	0.3	+4135%	+4144%	+4121%	+4743%	+4716%	+4122%	+19206%	+19208%	+19125%	+4716%	+0%
Life cycle	1137.2	-10%	-24%	-30%	-37%	-51%	-58%	-18%	-32%	-39%	-14%	+19%
Additional cost increase per kgCO ₂ e mitigated (GBP/kgCO ₂ e)	-	+11.29	+4.59	+4.21	+2.96	+2.01	+2.04	+4.15	+2.05	+1.86	+9.33	-0.30

^a Battery technology abbreviations: NMC = lithium nickel–cobalt–manganese, LFP = lithium-iron phosphate, LTO = lithium-titanate.

^b Includes battery replacements.

^c Includes placement, operation and maintenance of the infrastructure.

^d Percentage values correspond to the result at the 80% probability/confidence interval of each scenario compared to the baseline result.

Table B.1 Shape Functions used in each Technology Scenario.

	Baseline value ^a	UK Grid Scenarios			Wind Scenarios			Opportunity Charging Scenarios			HVAC Scenarios	
Drivetrain/Battery configuration	Diesel	NMC	LFP	LTO	NMC	LFP	LTO	NMC	LFP	LTO	LFP	Diesel
Reference diesel vehicle mass, m (kg)	11000	-	-	-	-	-	-	-	-	-	11500	11500
Auxiliary power demand, P_{aux} (kW)	4	-	-	-	-	-	-	-	-	-	Triangular Min: 4 Cen: 8 Max: 8	Triangular Min: 4 Cen: 8 Max: 8
Auxiliary system efficiency, η_{aux} (%)	60	Triang. ^b Min: 88 Cen: 90 Max: 92	Triang. ^b Min: 88 Cen: 90 Max: 92	Triang. ^b Min: 88 Cen: 90 Max: 92	Triang. ^b Min: 88 Cen: 90 Max: 92	Triang. ^b Min: 88 Cen: 90 Max: 92	Triang. ^b Min: 88 Cen: 90 Max: 92	Triang. ^b Min: 88 Cen: 90 Max: 92	Triang. ^b Min: 88 Cen: 90 Max: 92	Triang. ^b Min: 88 Cen: 90 Max: 92	Triang. ^b Min: 88 Cen: 90 Max: 92	-
Drivetrain efficiency, η_{trans} (%)	76.8	Uniform ^c Min: 83.6 Max: 89	Uniform ^c Min: 83.6 Max: 89	Uniform ^c Min: 83.6 Max: 89	Uniform ^c Min: 83.6 Max: 89	Uniform ^c Min: 83.6 Max: 89	Uniform ^c Min: 83.6 Max: 89	Uniform ^c Min: 83.6 Max: 89	Uniform ^c Min: 83.6 Max: 89	Uniform ^c Min: 83.6 Max: 89	Uniform ^c Min: 83.6 Max: 89	-
Battery energy density, E_{batt} (Wh/kg)	0	Uniform ^d Min: 75 Max: 170	Uniform ^d Min: 80 Max: 115	Uniform ^d Min: 45 Max: 100	Uniform ^d Min: 75 Max: 170	Uniform ^d Min: 80 Max: 115	Uniform ^d Min: 45 Max: 100	Uniform ^d Min: 75 Max: 170	Uniform ^d Min: 80 Max: 115	Uniform ^d Min: 45 Max: 100	Uniform ^d Min: 80 Max: 115	-
Battery capacity, Q_{batt} (kWh)	0	300	300	300	300	300	300	60	60	60	300	-
Battery cycles, N_{batt} (-)	0	Triang. ^d Min: 1000 Cen: 1700 Max: 3000	Triang. ^d Min: 1600 Cen: 3200 Max: 5039	Triang. ^d Min: 2000 Cen: 5000 Max: 6800	Triang. ^d Min: 1000 Cen: 1700 Max: 3000	Triang. ^d Min: 1600 Cen: 3200 Max: 5039	Triang. ^d Min: 2000 Cen: 5000 Max: 6800	Triang. ^d Min: 1000 Cen: 1700 Max: 3000	Triang. ^d Min: 1600 Cen: 3200 Max: 5039	Triang. ^d Min: 2000 Cen: 5000 Max: 6800	Triang. ^d Min: 1600 Cen: 3200 Max: 5039	-

GHG emissions per kg-battery, GHG_{batt} (kgCO ₂ e/kWh)	0	Triang. ^d Min: 248 Cen: 254 Max: 267	Triang. ^d Min: 246 Cen: 252 Max: 259	Triang. ^d Min: 254 Cen: 260 Max: 279	Triang. ^d Min: 248 Cen: 254 Max: 267	Triang. ^d Min: 246 Cen: 252 Max: 259	Triang. ^d Min: 254 Cen: 260 Max: 279	Triang. ^d Min: 248 Cen: 254 Max: 267	Triang. ^d Min: 246 Cen: 252 Max: 259	Triang. ^d Min: 254 Cen: 260 Max: 279	Triang. ^d Min: 246 Cen: 252 Max: 259	-	
Capital cost factor (exc. BEB battery), C_{cap} (-)	1	1.56 ^e	1.56 ^e	1.56 ^e	1.56 ^e	1.56 ^e	1.56 ^e	1.56 ^e	1.56 ^e	1.56 ^e	1.72 ^f	1.1 ^f	
Battery cost per kWh, C_{batt} (GBP/kWh)	0	Uniform ^d Min: 555 Max: 630	Triang. ^d Min: 666 Cen: 725 Max: 1140	Triang. ^d Min: 1050 Cen: 1110 Max: 1480	Uniform ^d Min: 555 Max: 630	Triang. ^d Min: 666 Cen: 725 Max: 1140	Triang. ^d Min: 1050 Cen: 1110 Max: 1480	Uniform ^d Min: 555 Max: 630	Triang. ^d Min: 666 Cen: 725 Max: 1140	Triang. ^d Min: 1050 Cen: 1110 Max: 1480	Triang. ^d Min: 666 Cen: 725 Max: 1140	-	
Maintenance Cost per km, C_{main} (GBP/km)	0.16	Uniform ^g Min: 0.1 Max: 0.14	Uniform ^g Min: 0.1 Max: 0.14	Uniform ^g Min: 0.1 Max: 0.14	Uniform ^g Min: 0.1 Max: 0.14	Uniform ^g Min: 0.1 Max: 0.14	Uniform ^g Min: 0.1 Max: 0.14	Uniform ^g Min: 0.1 Max: 0.14	Uniform ^g Min: 0.1 Max: 0.14	Uniform ^g Min: 0.1 Max: 0.14	Uniform ^g Min: 0.1 Max: 0.14	Uniform ^{f,g} Min: 0.1 Max: 0.154	Uniform ^f Min: 0.16 Max: 0.176
Fuel/Energy cost per kWh, $C_{f/e}$ (GBP/kWh)	0.054	Triang. ^h Min: 0.117 Cen: 0.121 Max: 0.123	Triang. ^h Min: 0.117 Cen: 0.121 Max: 0.123	Triang. ^h Min: 0.117 Cen: 0.121 Max: 0.123	Triang. ^h Min: 0.050 Cen: 0.067 Max: 0.081	Triang. ^h Min: 0.050 Cen: 0.067 Max: 0.081	Triang. ^h Min: 0.050 Cen: 0.067 Max: 0.081	Triang. ^h Min: 0.117 Cen: 0.121 Max: 0.123	Triang. ^h Min: 0.117 Cen: 0.121 Max: 0.123	Triang. ^h Min: 0.117 Cen: 0.121 Max: 0.123	Triang. ^h Min: 0.117 Cen: 0.121 Max: 0.123	-	
Infrastructure cost C_{infra} (1000 GBP)	1.119	Uniform ⁱ Min: 4.1 Max: 20.5	Uniform ⁱ Min: 4.1 Max: 20.5	Uniform ⁱ Min: 4.1 Max: 20.5	Uniform ⁱ Min: 4.1 Max: 20.5	Uniform ⁱ Min: 4.1 Max: 20.5	Uniform ⁱ Min: 4.1 Max: 20.5	Triang. ^j Min: 24.1 Cen: 51.7 Max: 110.2	Triang. ^j Min: 24.1 Cen: 51.7 Max: 110.2	Triang. ^j Min: 24.1 Cen: 51.7 Max: 110.2	Uniform ⁱ Min: 4.1 Max: 20.5	-	
Well-to-tank GHG emissions per MJ, GHG_{WTT} (gCO ₂ e/MJ)	15.4	Triang. ^h Min: 87.0 Cen: 100.0 Max: 109.5	Triang. ^h Min: 87.0 Cen: 100.0 Max: 109.5	Triang. ^h Min: 87.0 Cen: 100.0 Max: 109.5	Triang. ^h Min: 1.94 Cen: 3.06 Max: 15.6	Triang. ^h Min: 1.94 Cen: 3.06 Max: 15.6	Triang. ^h Min: 1.94 Cen: 3.06 Max: 15.6	Triang. ^h Min: 87.0 Cen: 100.0 Max: 109.5	Triang. ^h Min: 87.0 Cen: 100.0 Max: 109.5	Triang. ^h Min: 87.0 Cen: 100.0 Max: 109.5	Triang. ^h Min: 87.0 Cen: 100.0 Max: 109.5	-	
Tank-to-wheel GHG emissions per MJ, GHG_{TTW} (gCO ₂ e/MJ)	76.2	0	0	0	0	0	0	0	0	0	0	-	

Abbreviations: Triang = triangular, min = minimum bound, cen = central value (most likely), max = maximum bound.

^a See Table 4.

^b Assumed DC-DC converter efficiency.

^c Assumptions based on Gao et al. [64].

^d See Table 3.

^e Calculated based on the ratio between the purchase cost of an electric bus (without battery) and diesel bus in Lajunen [49].

^f Assuming an additional 10% cost for the HVAC unit.

^g BEB maintenance costs excluding propulsion system converted at 0.74 GBP per USD [90].

^h See Table 5.

ⁱ Converted at a rate of 0.82 GBP per Euro from [70].

^j Calculated from the infrastructure cost per bus for the baseline double decker bus scenarios in Kunith et al. [72].

Box 1. *Creation of an EIO-LCA model [39,57].*

The Leontif inverse equation (equation 1) relates changes in final demand, i.e. the value of a good or service (the price of the bus), to changes in monetary flows in the economy.

$$X = [I - A]^{-1} F \quad (1)$$

The sum of environmental impact discharges is calculated using equation 2.

$$B = R \cdot X = R[I - A]^{-1} F \quad (2)$$

where, X = sum of the total direct and indirect economic activity (GBP), I = identity matrix (-), A = direct requirements matrix (-), F = change in final demand (GBP), B = sum of environmental impact discharges (kgCO_{2e}), R = matrix of environmental discharge coefficients (kgCO_{2e}/GBP).