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Comparative Life Cycle Assessment of Electric Bikes for Commuting in the UK

Yue Huang [a, *], Like Jiang [a], Haibo Chen [a], Kaushali Dave [a], Tony Parry [b]

[a] Institute for Transport Studies, University of Leeds, 34-40 University Road, Leeds LS2 9JT, UK

*Corresponding author; Tel: +44 (0)113 343 2254; Email: y.huang1@leeds.ac.uk

[b] Nottingham Transportation Engineering Centre, University of Nottingham, Nottingham NG7 2RD, UK

Abstract

Electric bikes (e-bikes) represent an increasing share of urban mobility due to their small sizes and clean fuels. This study developed an environmental life cycle assessment model to evaluate e-bikes powered by lithium-ion battery, compared with a petrol car and a battery electric vehicle (BEV) for commuting. System boundary included vehicle lifecycle and fuel lifecycle. The model also included emissions from tyre and brake wear, and noise impacts in a case study in the UK. Results showed that BEVs and e-bikes reduce CO_2, and this reduction can increase with change in electricity mix and battery recycling. Advantages of e-bikes are not proportional to the weight of the vehicle. Non-exhaust emissions are significant to respiratory effects and human toxicity, so are noise reductions due to the use of electric vehicles. The model and data should help to conduct similar studies elsewhere in the world and to develop policies for electric vehicles.

Keywords: Life cycle assessment, electric bike, battery electric vehicle, atmospheric emissions, urban commuting

1. Introduction

Sustainable future transport systems must not only be safe, secure, quiet and green, but also accessible, affordable, inclusive and publicly accepted. Electrification of road vehicles has brought a number of benefits to the environment and society, mostly in urban air quality and reduced dependence on fossil fuels. Electric vehicles emit no carbon dioxide (CO_2) or other pollutants such as nitrogen oxides (NO_x), volatile organic compound (VOC) and particulate matter (PM) at the point of use (tank-to-wheel). L-category vehicles, i.e. two- or three-wheelers and quadricycles (see Table 1), are lighter and smaller alternatives to cars for urban mobility. Electric L-category Vehicles (ELVs) combine the benefits of small size and clean fuels. Moreover, they are highly compatible and interoperable with urban high-capacity public transport (e.g. MRT - mass rapid transit), and seen as a complement to the existing public transport by providing seamless door-to-door mobility for both people and goods. A modal shift from cars to ELVs such as electric bikes (e-bikes) is beneficial and suitable for short- and medium-distance travel, such as commuting. Weiss et al. (2015) reviewed the environmental, economic and social benefits of electric two-wheelers, demonstrating that these vehicles are generally more energy efficient and less polluting than fossil fuel powered vehicles. Meanwhile, Santucci et al. (2016) found the distance range, cost and regenerative braking capability are the barriers to wider use of ELVs.

From a lifecycle perspective, ELVs (and electric vehicles in general) require electric power and batteries that may give rise to other environmental problems in their production (well-to-tank) and disposal. These impacts need to be assessed over the lifecycle considering the entire supply chain for vehicles and fuels. A study by Ma et al. (2012) has challenged the widely held belief that battery electric vehicles (BEVs) can lead to a significant greenhouse gas (GHG) reduction from personal transport in the short- to medium-term. According to the International Energy Agency (IEA, 2020), fossil fuels will continue to dominate power generation in many markets up to 2030. Previous studies comparing ICEV (internal combustion engine vehicle) with BEV, suggested that the magnitude of the GHG reduction is determined by the cleanness of the regional electricity mix, and the efficiency of electricity transmission and distribution (Gao et al., 2013). However, some studies set the boundary to tank-to-wheel to include fuel lifecycle only (Huo et al., 2015), or focus on GHG emissions only (Qiao et al., 2017, Wu et al., 2018). Besides, the significance of battery manufacture and end-of-life (EOL) disposal needs to be understood. Life cycle assessment (LCA) adopts a holistic approach that is able to study many environmental impacts of using alternative vehicles and fuels. A LCA model is developed in this study, to evaluate the impacts of electric bike (e-bike) powered by lithium-ion (Li-ion) battery, and compare it with a petrol car and a battery electric vehicle (BEV) for commuting. It represents, to the authors' best knowledge, the UK electricity mix, vehicle make and usage, along with the supply chain and travel behaviour.

1.1 Technical background

Table 1 presents an overview of L-category vehicles, max dimension: 4m (L) x 2m (W) x 2.5m (H).

Table 1. Overview of L-category vehicles

No of wheels	Category code	Category name	Note (variation)		
2-wheels	L1e	Light two-wheel powered vehicle	L1e-A Powered cycles Power⩽1 kW; Max speed 25 km/h	L1e-B Two-wheel moped Power⩽4 kW; Max speed 45 km/h	
	L3e	Motorcycles	L3e-A1 Power⩽11 kW	L3e-A2 Power⩽35 kW	L3e-A3 Power>35 kW
3-wheels	L2e	Three-wheel moped	L2e-P (passenger), L2e-U (utility) Mass⩽270 kg; Power⩽4 kW; Max speed 45 km/h		
	L5e	Powered tricycle	L5e-A Mass⩽1000 kg, Max 5 seats Max speed 55 km/h	L5e-B Mass⩽1000 kg, Max 2 seats Max speed 55 km/h	
4-wheels	L6e	Light quadricycle	L6e-A Max speed 45 km/h; Mass⩽425 kg; Power⩽4 kW	L6e-A Max speed 45 km/h; Mass⩽425 kg; Power⩽6 kW	
	L7e	Heavy quadricycle	L7e-A, L7e-B, L7e-C Mass⩽450 kg (passenger), ⩽600 kg (goods); Power⩽15 kW; Max speed 90 km/h		

Electric cars (7.2 million in stock) accounted for 2.6% of global car sales and about 1% of global car stock in 2019 (IEA, 2020). BEVs are one of the main types of electric vehicles in use, making up about 1% of new vehicle registrations in the EU in 2018 (ICCT, 2019). There were 93,000 BEVs on UK's roads in 2019, about 2.6% of the entire fleet of 35,168,000 (SMMT, 2020). By 2020, BEVs had a 6.6% share of the UK market for new sale, compared with plug-in hybrids at 5.5%, diesel at 14.9% and petrol at 49.5% (Charlton, 2021). Lithium-ion batteries have good potential for vehicle use for a balanced performance and cost (Hu et al., 2017).

According to the International Energy Agency (IEA, 2020), the estimated stock of electric two/three-wheelers in 2019 was 350 million, the majority of which are in China. Fast growth of e-bike sales is also seen in Europe, where annual sales increased 17 fold between 2006 and 2016, and is projected to increase to 62 million in 2030 (Niestadt and Bjørnåvold, 2019). Mintel (2020) reported that around 100,000 e-bikes were sold in the UK in 2019, up from an estimated 73,000 in 2018, an increase of nearly 37% . UK consumers bought 2.5 million bikes in 2019 with about 6% of cyclists currently owning an e-bike. In a survey conducted to estimate the electric vehicle use across several countries in Europe, it was found that most people would be willing to switch to an e-bike as their main mode of transport, with the lower environmental impacts from e-bikes as the main reason driving this change (Chen et al., 2020). E-bikes are also an important factor in mitigating reduced mobility (McQueen et al., 2019), although poor cycling infrastructure poses a barrier to their increased uptake (Leger et al., 2019).

1.2. Impacts on airborne emissions

Belalcazar et al. (2016) indicated that if a bus rapid transit (BRT) system changes fuel type from diesel to electric, CO_2 and $PM_{2.5}$ emissions would reduce by 86% and 88%, respectively. In other studies, L-category vehicles are found to give rise to emissions of particulates. For instance, Giechaskiel et al. (2019) measured the SPN23 (particulates with size larger than 23 nm) emissions of one moped and eight motorcycles using a dilution tunnel. Results showed that some of these two-wheelers were close to, or exceeded, the SPN23 limit for passenger cars (6 x 10^{11} particles/km). Kontses et al. (2020) measured the particulate emissions from 30 L-category vehicles registered in Europe between 2009 and 2016, using a chassis dynamometer test. Results showed that L-category vehicles (e.g. motorcycle, moped) were a significant contributor to SPN23 emissions, which can be several times higher than the Euro 6 passenger car limit. Moreover, vehicle emissions are not limited to the in-use phase. Battery charging for ELVs for instance, demands electricity and the production of batteries generates CO_2. The quantity CO_2-eq./kWh depends on how the electricity is produced, although Ekman believed there are synergies between electric vehicle uptake and renewable energy (Ekman, 2011). A commonly held view, represented by Astegiano et al. (2019), is that due to the relatively small size of the ELV fleet, using ELVs will hardly lead to a noticeable improvement in urban air quality.

1.3. Impacts on road traffic noise

Road traffic noise is a major environmental issue. In Europe, more than 20% of the population, and more than 50% of urban residents are exposed to high levels of road traffic noise that exceed 55dB during the day-evening-night period (EEA, 2020). Prolonged exposure to noise is associated with sleep disturbance, cardiovascular diseases and cognitive impairment (WHO, 2018), and traffic noise has been classified as the second most significant environmental threat to public health in western Europe, following the emissions of fine particulate matter (Hänninen et al., 2014). ELVs have the potential to reduce road traffic noise. Propulsion noise from electric vehicles is significantly lower than from conventional internal combustion engine vehicles (Campello-Vicente et al., 2017). In addition, L-category vehicles normally travel at low speeds, e.g., below 45km/h (Table 1), so the rolling noise (from tyre-road interface) is low (Mitchell, 2009). Thus, ELVs are unlikely to cause noise nuisance, although the silence can be a safety concern for other road users (Cocron et al., 2011).

Despite methodological development in life cycle assessment (LCA) to quantify the health impairment from road traffic noise (Müller-Wenk, 2004, Althaus et al., 2009, Franco et al., 2010), the authors are not aware of any main life cycle inventory for ELVs that has systematically included noise impact. Research (Meyer et al., 2019, Ongel, 2015) showed that including noise impact could significantly change LCA results of road transport, and fleet-based analysis would be preferred to individual vehicle-based analysis.

According to UK's Department for Transport, cars and bicycles together accounted for 64% of all commute trips in England in 2019 (DfT, 2020b). The percentage of BEVs and e-bikes in vehicle sale and stock will increase further in the foreseeable future. This calls for scientific evidence of the advantages of these vehicles over their production supply chain and lifetime use. Exclusion of the aforementioned key concerns may lead to biased policy and ill-informed public purchase behaviour. A ban on the registration of new fossil-fuelled vehicles by 2030 (GOV.UK, 2020), and the characteristics of short-to-medium distance, urban commute in the UK make it necessary to carry out a LCA of vehicles comparable for that trip purpose.

2. LCA of electric L-category vehicles - literature review

Publications were reviewed with regard to the following aspects of LCA of electric L-category vehicles, namely: 1) passenger transport, 2) two-wheelers, 3) trade-off between impacts, vehicles and fuels, and 4) process-based vs. input-output based LCA.

[Passenger transport] A life cycle assessment was conducted by Bastos et al. (2019) for Lisbon that compared six commuting modes (i.e. car, bus, train, subway, motorcycle and bicycle) powered by conventional fuels for eight impact categories. Results demonstrated the need for understanding the trade-off between modes and for holistic approaches to avoid problem shifting. Electric bikes and scooters were studied by Cherry et al. (2009) who developed an emission inventory for e-bikes in China, which showed that the emissions varied significantly with the fuel types for power generation. However, e-bikes in this study were powered by lead-acid battery; the battery and bike weighed 10.3 kg and 41.3 kg, respectively, much heavier than a lithium-ion battery (2.6 kg) and the e-bike (24 kg) currently in use. Belalcazar et al. (2016) calculated the lifecycle emissions from a Bus Rapid Transit (BRT) system and compared it with other modes of passenger transport in Bogota, Colombia using the well-to-wheel system boundary. The study focused on the fuel lifecycle and used the Ecoinvent database, assuming a 92% (hydro) and 8% (coal) split for generating electric power. Results indicated that fossil fuel powered motorcycles are associated with high emissions of $PM_{2.5}$, and results are sensitive to vehicle occupancy.

[Two-wheelers] Cox and Mutel (2018) studied the environmental impacts of four motorcycle size categories and three powertrain types (i.e. petrol, battery electric and fuel cell electric) using a number of fuel supply chains and technologies. They found that smaller motorcycles and urban driving (low speed) are associated with lower emissions. The CO_2-eq. emissions can be reduced when the vehicles are powered by electricity generated from renewables, natural gas or even coal, compared to conventional motorcycles. Hollingsworth et al. (2019) studied the impacts of e-scooters in rental use, and found that daily collection and charging accounted for 43% of the GHGs, second only to manufacturing (50%) of e-scooters. The results proved to be highly sensitive to e-scooter lifetime. For instance, a two-year lifetime would give approximately the same amount of CO_2-eq. as conventional scooters.

[Trade-off between impacts, vehicles and fuels] Ma et al. (2012) found that BEVs have lower overall GHGs emissions than ICEVs, and the difference gets larger at lower speed (e.g. urban) driving and when the vehicles are lightly loaded (e.g. driver only, no accessary). However, vehicle lifecycle emissions (associated with vehicle manufacture and disposal) are higher for BEVs due to the GHG emissions associated with the battery. The lifecycle inventories of bicycle and electric bicycle manufacture were established by Leuenberger and Frischknecht (2010) by expanding the Ecoinvent datasets. Bucher et al. (2019) showed that GHG emission reductions between 10% and 17.5%, compared with diesel or petrol cars for commuting, were possible as a result of using electric bikes. His study also concluded that long distance and adverse weather are limiting factors for further uptake. A similar conclusion was drawn by Elliot et al. (2018) who suggested that the efforts to change travel behaviour should be focused on specific users (e.g. people who are most likely to cycle if it is safe) and their needs (e.g. long and physically demanding terrains that need power assistance). Mellino et al. (2017) compared the lifecycle impacts of electric bikes powered by Li-ion battery and those powered by a hydrogen fuel cell for freight transport, using ICEVs as a benchmark. An interesting finding was that the conclusion changed with a change of system boundary, i.e. whether or not the production of the vehicle is included.

[Process-based vs. input-output based LCA] Most LCA studies for BEVs and e-bikes are process-based (also known as bottom-up, quantifying impacts from individual processes and then compiling results), although efforts have been made to compare the different vehicles and fuels, using the

Input-Output (IO) method (also known as top-down, using inter-sectoral economic data matrix for estimating). Results by Dave (2010) who used Economic Input-Output analysis (EIO-LCA) showed that electric bikes use less than 10% of the energy required to power a passenger car and emit 90% fewer pollutants per passenger mile travelled, than a bus operating at off-peak hours. A hybrid LCA method (using process-based LCA and IO-based LCA for different lifecycle stages) was used by Dai et al. (2005) to compare petrol-powered motorcycles with electric bikes. Results showed that the majority of energy consumption and emissions was at the vehicle use phase. Emissions of CO_2-eq., CO and PM_{10} from e-bikes were lower, whilst emissions that contribute to Acidification were higher in e-bikes. However, the justification of boundary and the details of underlying datasets are not found in the paper.

So far, the LCA studies that involved electric vehicles are not representing UK's current electricity mix or energy efficiency of electric vehicles, and the functional unit has not been defined for a typical trip purpose, e.g. commute, in the UK. These make it difficult for researchers and policy makers to evaluate the benefits of using electric bikes for passenger transport. This LCA study is conducted to fill the above knowledge gap, and to provide evidence for devising the penetration strategy for electric bikes and other L-category vehicles.

3. LCA model overview

This study developed a LCA model for passenger road transport in the UK, which compares airborne emissions associated with conventional internal combustion engine vehicle (ICEV), battery electric vehicle (BEV) and electric-bike (e-bike). This model included vehicle lifecycle and fuel lifecycle. Noise impact was included in the inventory and impact assessment. The model components and methodological choices are presented in section 4. It is followed by a case study to test and calibrate the model, in which an e-bike (L1e-B) is compared with a petrol car and a BEV for commuting. In 2019, UK commuting took an average of 2,700 miles (4,345 km) a year (DfT, 2020a). The following assumptions were made in developing the LCA model:

- Each commuter uses one commuting mode.

- Commuting distances vary by mode, a distance of 5 miles (8 km) is selected for the case study in section 5.2, based on statistical data for commuting trips (DfT, 2020b).
- Single occupancy is assumed in all three vehicles.

3.1 Functional unit

Due to the difference in capacity of the vehicles compared, the functional unit (FU) must ensure fairness and transparency. A service life of eight years and a yearly distance of 2,400 km were estimated for an average commuter bike by European Cyclists' Federation (ECF, 2011). Survey results by Winslott Hiselius and Svensson (2017) indicated that the potential for e-bikes to replace car trips is as great in rural areas as it is in urban areas. Kroesen (2017) found that e-bike ownership reduced the use of conventional bicycle, but also, to a lesser extent, reduced car and public transport use. EMPA (Materials Science and Technology) studied the environmental impacts of e-bikes powered by Lithium-on (Li-ion) batteries, which assumed 15,000 km life expectancy and 0.01 kWh/km efficiency for e-bikes, and 150,000 km life expectancy and 0.2 kWh/km efficiency for electric cars (Del-Duce, 2011). In 2020, more than 57.6% of passenger cars in the UK had an engine size of up to 1.4L (Eurostat, 2020). Considering the methodological choices made in the literature, such as driving condition (45 km/h, urban), load condition (diver only), vehicle size (small-to-medium passenger car) and accessory use, the functional unit for the case study is defined as five miles (eight kilometres) travel at a maximum speed of 45 km/h in a single occupancy vehicle.

3.2 Data and inventory

LCA modelling is carried out in accordance with ISO14040 (ISO, 2006). The main sources of data include Ecoinvent (sub-licensed to SimaPro) and GREET (Greenhouse gases, Regulated Emissions and Energy use in Transportation) model, developed by Argonne National Laboratory (GREET, 2020). GREET includes inventory data for automobile parts and complete vehicles, as well as various fuels for vehicle use. An example of the inventory calculations is presented in Eq1.

$$LCI = LCI_{vehicle} + \sum_i \left(LCI_{haulage_i} \times distance_i\right) + LCI_{use} \times vehicle\ life\ mileage + LCI_{EOL} \qquad \text{Eq1}$$

a) LCI: life cycle inventory (LCI) including vehicle lifecycle and fuel lifecycle

b) $LCI_{vehicle}$: inventory associated with materials, manufacturing and assembly of the vehicle

c) $LCI_{haulage}$: inventory associated with transport of the vehicle to user (e.g. shipping, trucking). It is calculated by summing the inventory of all modes of transport.

d) LCI_{use}: inventory associated with use of the vehicle, e.g. commuting.

- For ICEV, inventory associated with the petrol fuel, well-to-wheel
- For BEV and e-bike, inventory associated with the production, transmission and distribution of electricity used for travel, dividing the inventory as in the database (g emission/kWh) by energy efficiency (km/kWh) of the vehicle

e) LCI_{EOL}: inventory associated with disposal of the vehicle, including battery for BEVs and e-bikes

3.3 System boundary

The system boundary is illustrated in Fig.1.

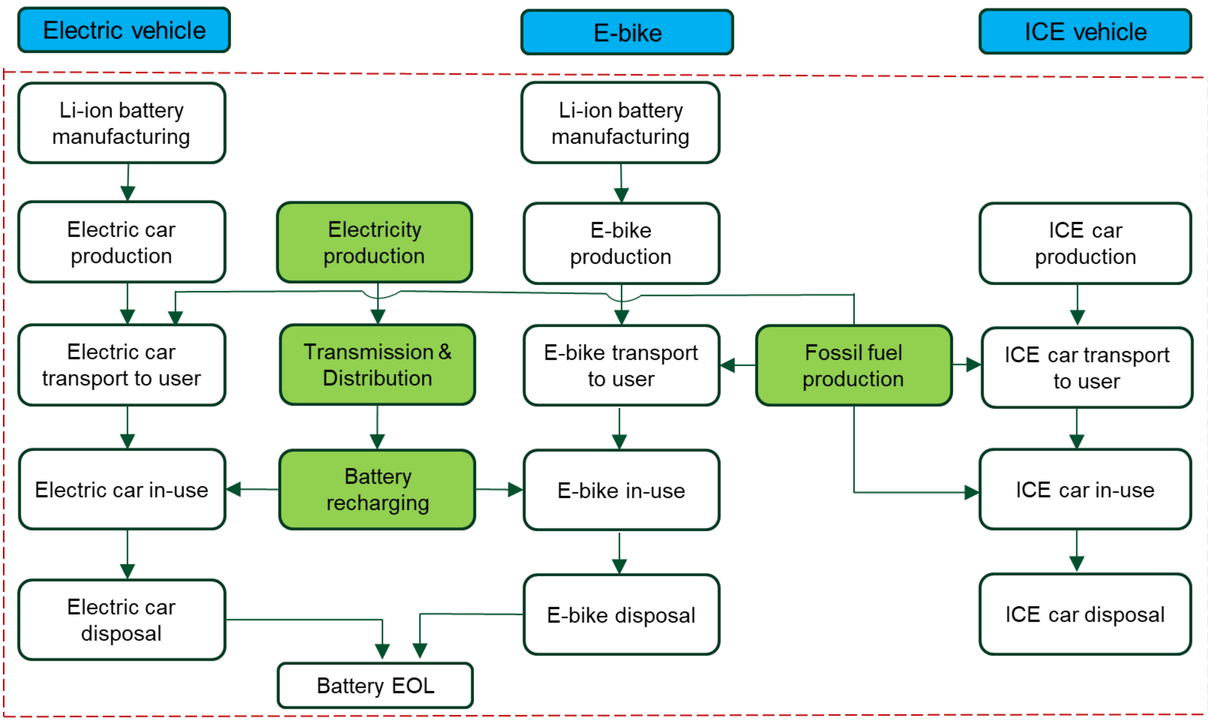


Fig.1 System boundary for comparing vehicles

The system boundary consists of vehicle lifecycle and fuel lifecycle. Airborne emissions from vehicle manufacturing, fuel and power production, vehicle operation and disposal are taken into account. The system boundary is defined to represent urban driving and energy mix in the UK, and made transparent such that different data, assumptions and methodological choices can be compared for sensitivity check. Methodological choices are known to affect the LCA results substantially (Huang et al., 2013). The fuel lifecycle in relation to BEV and e-bike include electricity generation, transmission and distribution (T&D), and vehicle recharging. Details of assumptions and methodological choices made in each phase are described in section 4. The provision, maintenance and disposal of capital equipment and infrastructure, e.g. charging station, is excluded due to insufficient data.

4. LCA model structure

4.1. Vehicle manufacturing phase

Vehicle specifics are described below:

- Vehicle production includes five elements: 1) battery, 2) fluids, 3) other components, 4) assembly, and 5) disposal and recycling. It covers the acquisition of raw materials, manufacturing and distribution of parts, vehicle assembly and end-of-life disposal. Each element includes its own supply chain and replacements (e.g. tyres). Vehicle maintenance and repair is scaled to a fraction of the production in literature (Ma et al., 2012), which is excluded from this study.
- Battery manufacturing, a lithium-ion battery pack consists of 125 kg battery cells, a steel box (case), cables and cooling equipment. The GHG emissions include 1,525 kg CO_2, 2.880 kg CH_4 and 0.033 kg N_2O, which convert to CO_2-eq of 1,615 kg. For comparison, the embodied GHGs for a LiNiMnCo (Lithium Nickel Manganese Cobalt Oxide) battery (253 kg by weight) was found by Ellingsen et al. (2014) to be 4,580 kg. Different weights of Li-ion batteries are found in literature, such as Harper et al. (2019).
- The ICEV is different from BEV mainly in the powertrain system. The main components of an e-bike include the frame, the gears and tyres, an electric motor and a 2.6 kg Li-ion battery pack.

- Vehicle weight (including battery): 1,200 kg for petrol car (PC); 2,070 kg for BEV, which takes the average of two popular models in the UK, i.e. Tesla Model 3 (2,139 kg) and Nissan Leaf (1,995 kg); 24 kg for E-Bike.

4.2 Vehicle transport (to user) phase

- Petrol cars are manufactured in the UK, a transport distance of 100 km to users by heavy-duty vehicles (HDV) with 19 tonnes payload is assumed. Boundary of the fuel data is well-to-wheel (WtW) including oil extraction, refining, distribution and vehicle use.
- BEVs are assumed to be manufactured and assembled (including battery) in the USA (Coffin and Horowitz, 2018), and transported for 3,633 nautical miles by large container to the UK (from Baltimore to Southampton), which is converted (x 1.852 km/nautical mile) to 6,728 km.
- Electric bikes are assumed to be imported from China (EU, 2019), and transported for 11,714 nautical miles by large container to the UK (from Shanghai to Southampton), which is converted (x 1.852 km/nautical mile) to 21,694 km.

4.3 Electricity production phase

- In 2019, 34% of UK's electricity was generated from fossil fuels including oil, natural gas and coal (ofgem, 2020). This compares to 52% in Europe (UCTE, 2009) and 62% in USA (GREET, 2020). It is pointed out in literature (Ma et al., 2012) that regional electricity mix is important, and different generation technology using the same types of fuel may have different emission profiles. To reflect the UK practice, this study uses the GHG conversion factors provided by UK Department for Environment, Food & Rural Affairs (DEFRA, 2020), see Table 2.

  Table 2. GHG emission factors for electricity production for electric vehicles (EV) and Tyre and Brake (non-exhaust) emissions

	GHG emission factors for electricity				Tyre & Brake emissions			
	EV use (g/mile)	EV T&D loss (g/mile)	Total (g/mile)	Total (g/kWh)*	ICEV (AQEG, 2019), mg/km		BEV (Beddows and Harrison, 2021), mg/km	
CH_4	0.26	0.02	0.28	1.652	n/a	n/a	n/a	n/a
N_2O	0.50	0.04	0.54	3.186	n/a	n/a	n/a	n/a
CO_2	84.11	7.23	91.34	538.9	n/a	n/a	n/a	n/a
PM_{10}	n/a	n/a	n/a	n/a	8.7	11.7	9.3 (ICEV+0.6)	12.3 (ICEV+0.6)
$PM_{2.5}$	n/a	n/a	n/a	n/a	6.1	4.7	7.0 (ICEV+0.9)	6.1 (ICEV+1.4)

* Assuming an energy efficiency of 5.9 miles/kWh (or 9.5 km/kWh), explained in section 4.4.

- The outputs are in *gram emission per kWh electricity use*. This is believed to be an accurate reflection of UK practice. Boundary of the data is well-to-wheel (WtW) including electricity production, transmission and distribution (T&D, a loss of 4.9% is used in GREET) for recharge, and vehicle use (zero emission).
- Defra data only include GHG emissions; therefore, emissions contributing to impact categories other than GHG were calculated using GREEET conversion factors.

4.4 Vehicle operating phase

- Fossil fuel lifecycle includes oil extraction and refining, transportation and distribution of fuels (well-to-tank, or WtT), and vehicle fuel use (tank-to-wheels, or TtW).
- The energy efficiency of e-bike is taking the average of an electric mode (0.025 kWh/mile) and a pedal assistant mode (0.015 kWh/mile) running on slightly hilly terrain (Electrek, 2020), which gives 0.02 kWh/mile (or 50 mile/kWh). This is close to an efficiency of 1.2 kWh / 100km (or 52 mile/kWh) found in literature (Mellino et al., 2017).
- An energy efficiency of 4.6 km/kWh (or 2.9 mile/kWh) for BEVs is found in literature (Bicer and Dincer, 2017). Data in this model are sourced from the EU electric vehicle database (EVDatabase, 2020), taking the average of Tesla Model 3 (0.160 kWh/mile) and Nissan Leaf (0.180 kWh/mile), which gives 6.25 mile/kWh and 5.5 mile/kWh, respectively, for mild weather in a city environment. This is close to the energy efficiency found in another literature (Coffin and Horowitz, 2018), i.e. 4.3 mile/kWh for Tesla Model 3 and 5.0 mile/kWh for Nissan Leaf. An average of 5.9 mile/kWh (or 9.5 km/kWh) is used in the model.

- Exhaust emissions during the use phase for ICEV were obtained from the GREET database. The PM_{10} and $PM_{2.5}$ emissions from tyre and brake wear for ICEV are estimated by Air Quality Expert Group (AQEG, 2019). The regression coefficients derived by Beddows and Harrison (2021) in a recent publication are used to estimate the non-exhaust PM emissions from BEV, see Table 2. Tyre and brake emissions from e-bike were excluded due to insufficient data.
- According to Wu et al. (2018), on-road fuel consumption of ICEVs (in units of litre/100 km) and BEVs is estimated to be 15% and 25% higher than in laboratory conditions, respectively. This difference varies with vehicle class, driving condition (e.g. speed) and auxiliary loading (e.g. use of heating, air conditioning), and was not considered in this study. The electricity loss during recharging was also excluded due to insufficient data.
- Assumed vehicle lifetime mileage: 170,000 miles (273,588 km) for petro car (PC); 120,000 miles (193,121 km) for BEV; 15,000 km for E-Bike.

4.5 Vehicle (including battery) disposal phase

- Vehicle disposal for ICEV and BEV (excluding battery) has been counted in the production phase in GREET, as part of the vehicle ARD (assembly, recycling and disposal). The Ecoinvent inventory for EOL disposal of electric bike (including disposal of battery) was used in this model.
- The inventory for disposal of BEV battery was obtained from GREET, in units of *gram emission per tonne battery cell recycling*, assuming the technique of *direct recycling* was used, which separates the different components by physical (e.g. gravity) processes, and enables recovery of reusable cathode material with minimal treatment (Gaines, 2018). The inventory was scaled to a weight of 125 kg – the weight of battery assumed in section 4.1.

4.6 Summary of data sources

A summary of the data used to build the LCA model is presented in Table 3.

Table 3. Summary of LCA model structure and data source

Vehicle	Model block code	Model block name	Unit	Data source	Note
	PC-p	Production	per vehicle	GREET	Parts, assembly, disposal

Petrol car	PC-t	Transport to user	tonne.km	GREET	By truck, 19t payload
	PC-u	Vehicle in-use	per km	GREET, Ecoinvent, DEFRA	DEFRA data only include GHG
	PC-d*	Vehicle EOL	per vehicle	GREET	Included in production
Battery electric vehicle	EV-p	Production	per vehicle	GREET	Parts, assembly, disposal
	EV-t	Transport to user	tonne.km	GREET	By shipping from USA
	EV-u	Vehicle in-use	per km	GREET, Ecoinvent, DEFRA	DEFRA data only include GHG
	EV-d*	Vehicle EOL	per vehicle	GREET	Included in production
	EV-db	Li-ion battery EOL	per tonne battery	GREET	Scaled to 125 kg
E-bike	EB-p	Production	per vehicle	Ecoinvent	24 kg e-bike
	EB-t	Transport to user	tonne.km	GREET	By shipping from China
	EB-u	Vehicle in-use	per km	GREET, Ecoinvent, DEFRA	DEFRA data only include GHG
	EB-d	Vehicle EOL	per vehicle	Ecoinvent	24 kg e-bike
	EB-db*	Li-ion battery EOL	per vehicle	Ecoinvent	Included in vehicle EOL

* Shaded blocks are excluded to avoid double counting.

One general observation of the data is the difference in inventory data per unit between Econinvent and GREET. While the values in relation to GHGs are close between the two databases, larger differences are found in other emissions. The authors believe this is more a reflection of the method (e.g. boundary, allocation method) used in deriving the data, than the difference in technology between Europe and USA.

4.7 Impact assessment

Four impact categories are included in the study, namely global warming potential (GWP, CO_2 eq.), acidification (SO_2 eq.), respiratory effects (DALY, Disability Adjusted Life Years) and human toxicity (1,4-dichlorobenzene eq., or 1,4-DB eq.). These categories are mostly relevant to airborne emissions and urban traffic environment. The characterisation factors are derived from the CMLCA software (CML, 2016) and listed in Table 4.

Table 4. Characterisation factors

Impact Category		Indicator	Characterisation Factor	Method
Global warming potential	CO_2	CO_2 eq.	1	IPCC, 2013
	CH_4		28	
	N_2O		265	
Acidification	SOx	SO_2 eq.	1.20	Huijbregts, 1999
	NO		0.76	
	NO_2		0.50	
Respiratory effects	CH_4	DALY	1.28E-08	ECOINDICATOR 99
	SOx		5.46E-05	

	NO		1.37E-04	
	NO_2		8.87E-05	
	VOC		1.28E-06	
	PM_{10}		3.75E-04	
	$PM_{2.5}$		7.00E-04	
Human toxicity	SOx	1,4-DB (dichlorobenzene) eq.	0.10	Huijbregts, 1999
	NOx		1.20	
	PM_{10}		0.82	

5. Results

5.1. Initial results

Inventory and characterisation results are shown in Table 5 and Table 6. It can be seen that:

a) Vehicle in-use phase represents the largest contributor to GHGs, accounting for 90% (or 55.6t) of GHGs for petrol cars. For BEV and e-bike, vehicle use (when including electricity production) accounts for 68% (or 29.1t) and 54% (or 0.3t) of GHGs. The production of ICEV and BEV generates about 6.1t and 6.7t of GHGs, respectively. Results are quite different to Qiao et al. (2017), who found that BEV production emits about 15.0-15.2t of CO_2-eq, in a study based on China's energy mix and vehicle production technology. The reason for lower total GHGs in BEV is that its fuel lifecycle is associated with much less GHGs compared with ICEV. This difference, however, will be dependent on the electricity generation technology in use.

b) It is also obvious that BEVs have higher impacts than ICEVs in two impact categories, namely Acidification and Respiratory Effects. This agrees with a previous study comparing the two vehicles (Hawkins et al., 2013), and raises the question of how to evaluate the trade-off between impacts. The impact of Human Toxicity is similar (difference <2%) between the two vehicles. Li-ion battery manufacture accounts for 24% of GHGs in the lifecycle of BEV, compared with 6% of GHGs in the lifecycle of ICEV from the manufacture of lead-acid battery.

c) Vehicle in-use phase also represents the largest contributor to Human Toxicity for petrol cars. Vehicle production represents the largest contributor to Respiratory Effect. It also represents the largest contributor to Acidification for both types of vehicles. For BEV, impacts from vehicle production are quite close to vehicle in-use phase in Human Toxicity (difference 4%) and Respiratory Effect (difference 3%), but much higher than vehicle transport and disposal.

d) For electric bike, production represents the largest contributor to three impact categories, namely Acidification (51%), Human Toxicity (51%) and Respiratory Effects (53%). Bike disposal accounts for 42%, 34% and 31% of these three impact categories. This indicates the importance of supply chain management and end-of-life management for e-bikes. Recycling (battery, motor) at the end-of-life will preserve finite resources such as cobalt and nickel. Other options are also available such as direct reuse or remanufacture (EEA, 2018). The challenge for LCA practitioners is to obtain data on the designated use, and develop an EOL method such as cut-off or substitution, to account for the second life of these components.

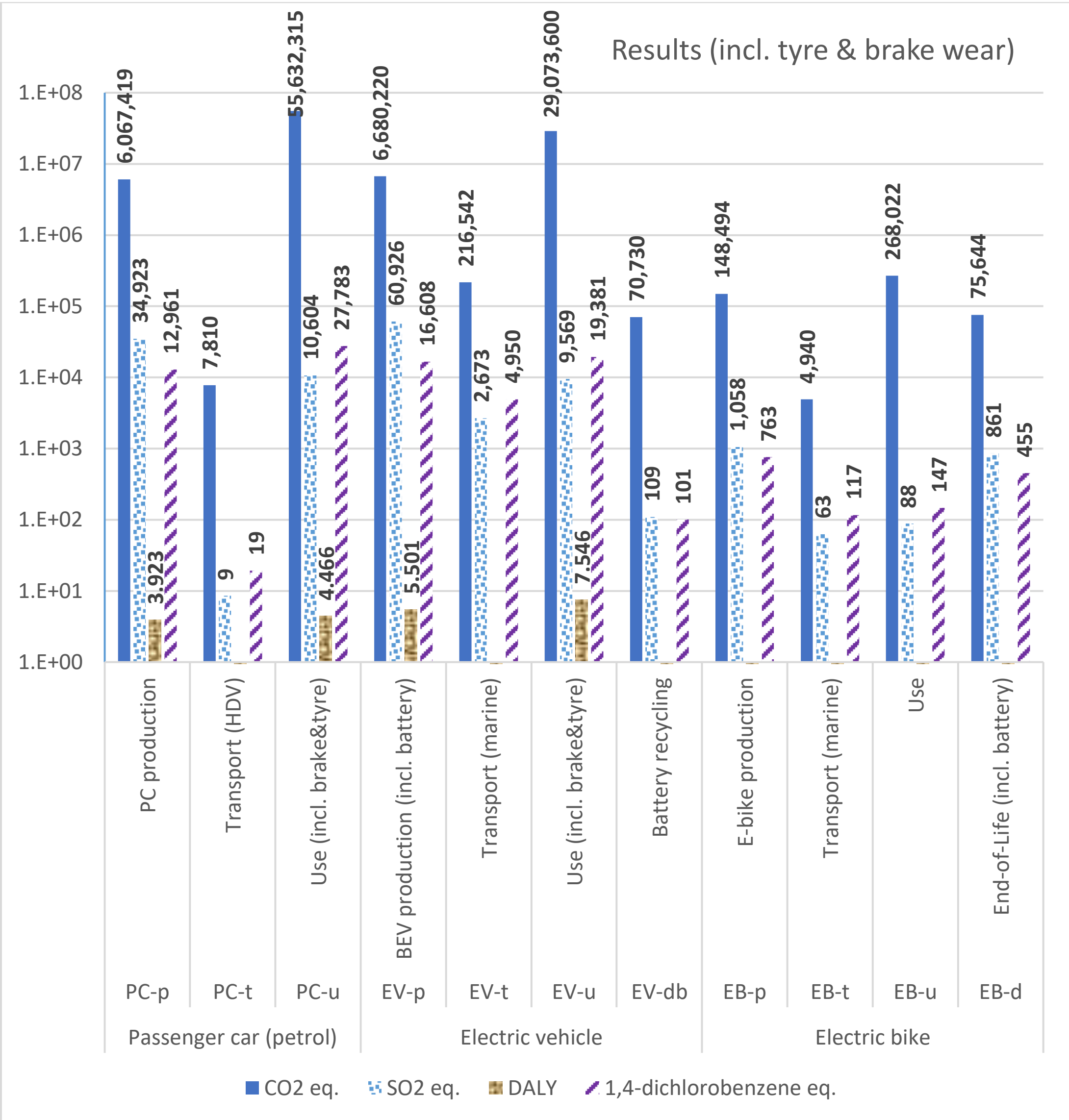
e) Vehicle in-use phase accounts for the highest percentage of GHGs in all three vehicles, which is 9.1 times, 4.4 times and 1.8 times vehicle production for ICEV, BEV and e-bike, respectively. This indicates the importance of reducing tailpipe emissions and, with the fast growing volume of BEVs, the impacts from electricity generation. BEV powered by electricity generated in the UK (see section 4.3) would reduce CO_2-eq. by 41.6% compared with a petrol car over the vehicle lifetime. Transport presents less than 1% of GHGs for ICEV and BEV, and about 1.7% for e-bike. This negates the need to refine the LCA model by adding transport of BEV and e-bike from port to the user.

5.2. Sensitivity analysis

Different data, assumptions and methodological choices can significantly change the LCA results. For instance, Ma et al. (2012) found GHG intensity of marginal electricity (800 g/kWh) is 60% higher than the grid average (518 g/kWh). This is because the incremental electricity brought into the grid to meet the additional demand for BEVs have higher GHG footprint than electricity from the national grid. A study by Berzi et al. (2016) predicted that the dismantling of a scooter is able to recover 85-95% of the metals. Therefore if the EOL recycling is credited to the production of vehicles, it will reduce emissions associated with it. Sensitivity checks are carried out over the issues raised in section 1.2 and 1.3, to gauge the effects of functional unit and boundary setting on the results.

Sensitivity check 1 – Include non-exhaust emissions

The proportion of emissions from tyre and brake wear to the total vehicular emissions have been increasing, largely due to the advancement in engine technology that drives down emissions from exhaust pipes (AQEG, 2019). In this analysis, the tyre and brake emissions (detailed in Table 2) were added to the vehicle in-use phase for petrol cars and BEV. The impact assessment results are illustrated in Fig.2.



Impact Category	Indicator (unit: gram)	PC (with T&B)	PC (w/o T&B)*	BEV (with T&B)	BEV (w/o T&B)*	E-BIKE

GWP	CO_2 eq.	61,707,544	61,707,544	36,041,092	36,041,092	497,101
Acidification	SO_2 eq.	45,534	45,534	73,277	73,277	2,070
Resp. effects	DALY (10^{-3})	8,391	6,696	13,655	11,777	518
Human toxicity	1,4-DB eq.	40,763	38,254	41,039	37,614	1,482

* Results without (w/o) tyre and brake (T&B) emissions are not illustrated in Fig. 2.

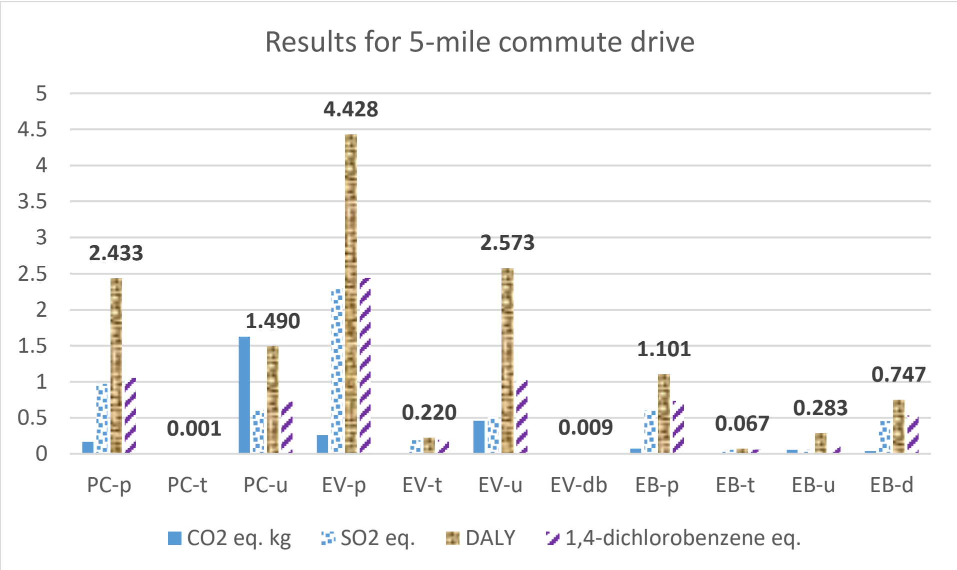
Fig. 2 Summary of impact assessment results with and without tyre & brake emissions

It can be seen from Fig. 2 that when including tyre and brake emissions, the impacts of Respiratory Effect and Human Toxicity increased by 25% and 7% for petrol cars, and increased by 16% and 9% for BEV, indicating the significance of non-exhaust emissions. In both scenarios, the BEV showed higher impacts compared with ICEV, with GHGs being the only exception. The difference will only be larger if the same lifetime mileage is assumed for the two vehicles, as opposed to **4.4**. Investigation into the lifecycle phases revealed that vehicle production is the reason of BEV having high impacts. The higher GHGs associated with BEV production is offset by the vehicle in-use phase, while BEV over its lifecycle is higher than ICEV in the other three impact categories. There is a danger that uninformed shift to BEV where electricity embodied emissions are high will negate the efforts made by limiting tailpipe emissions and shift the problem elsewhere.

The emissions (such as SO_x and CO_2) of electric bike *per passenger.km* derived from the inventory agree well with other researchers (Cherry et al., 2009, Bucher et al., 2019). The tailpipe emission of PM_{10} *per km* is calculated as 32 mg/km (475,784mg / 15,000km), which is between the current Euro 4 (80 mg/km) and Euro 5 (4.5 mg/km) emission standard limits for L-category vehicles. The following questions need to be answered: 1) Will the assumption of vehicle lifetime miles (as in **4.4**) make a difference to the results? 2) What will be the impacts for a 5-mile journey (e.g. commuting) made using these three vehicles in an urban environment?

Sensitivity check 2 – Travel distance and vehicle occupancy

The total vehicle life mileage may not be a fair comparison. For instance, miles commuted by e-bike may be a large proportion of the total life mileage. A petrol car, in comparison, may be used for long recreational journeys (on holiday etc.), thus the miles commuted may be a much lower percentage of the total life mileage. To define a functional unit for passenger transport LCA, it is important to state who the users are and what the purpose of the trip is. As stated in section 3, a typical commute distance of 5 miles (8 kilometres) is used to scale the results from vehicle lifetime mileage (as in **4.4**) to 5 miles. Authors believe there is a good potential that electric bikes can replace cars for this trip purpose. The results are presented in Fig. 3 and those for individual phases are shown in Table 7. It can be seen that when a battery electric car is used to replace a petrol car, GHGs reduce by 59% and more than 1kg of CO_2-eq. is saved for a 5-mile drive. On the other hand, impacts increase by 90%, 84% and 106%, respectively, for Acidification, Respiratory Effect and Human Toxicity. As illustrated in Fig. 3 (Respiratory Effect annotated, CO_2-eq in kg), this is caused by the high impacts associated with BEV production (e.g. 4.428 DALY as opposed to 2.433 DALY by petrol car) and vehicle in-use (2.573 as opposed to 1.490). The counter argument can be that the health risks are different, and BEVs decrease exposure to pollution as their emissions largely come from vehicle production and electricity generation far away from population.



Impact Category	**Indicator** (unit: gram)	**PC**	**BEV**	**E-BIKE**
GWP	CO_2 eq.	1,792.624	733.802	167.325
Acidification	SO_2 eq.	1.585	3.012	1.175
Resp. effects	DALY	3.924	7.231	2.197
Human toxicity	1,4-dichlorobenzene eq.	1.779	3.665	1.430

Fig. 3 Summary of impact assessment results for a 5-mile commuting journey

Compared to a petrol car, using an electric bike for a 5-mile commute emits only a fraction (9%) of the GHGs. However, the difference in other impacts is small (-26%, -44% and -20%, respectively, for Acidification, Respiratory Effect and Human Toxicity). Unlike cars, bike production (1.101g) and disposal (0.747g) are the two largest contributors to respiratory effect, as shown in Fig. 3. This calls for improvement in battery manufacture and higher uptake of reuse/recycling.

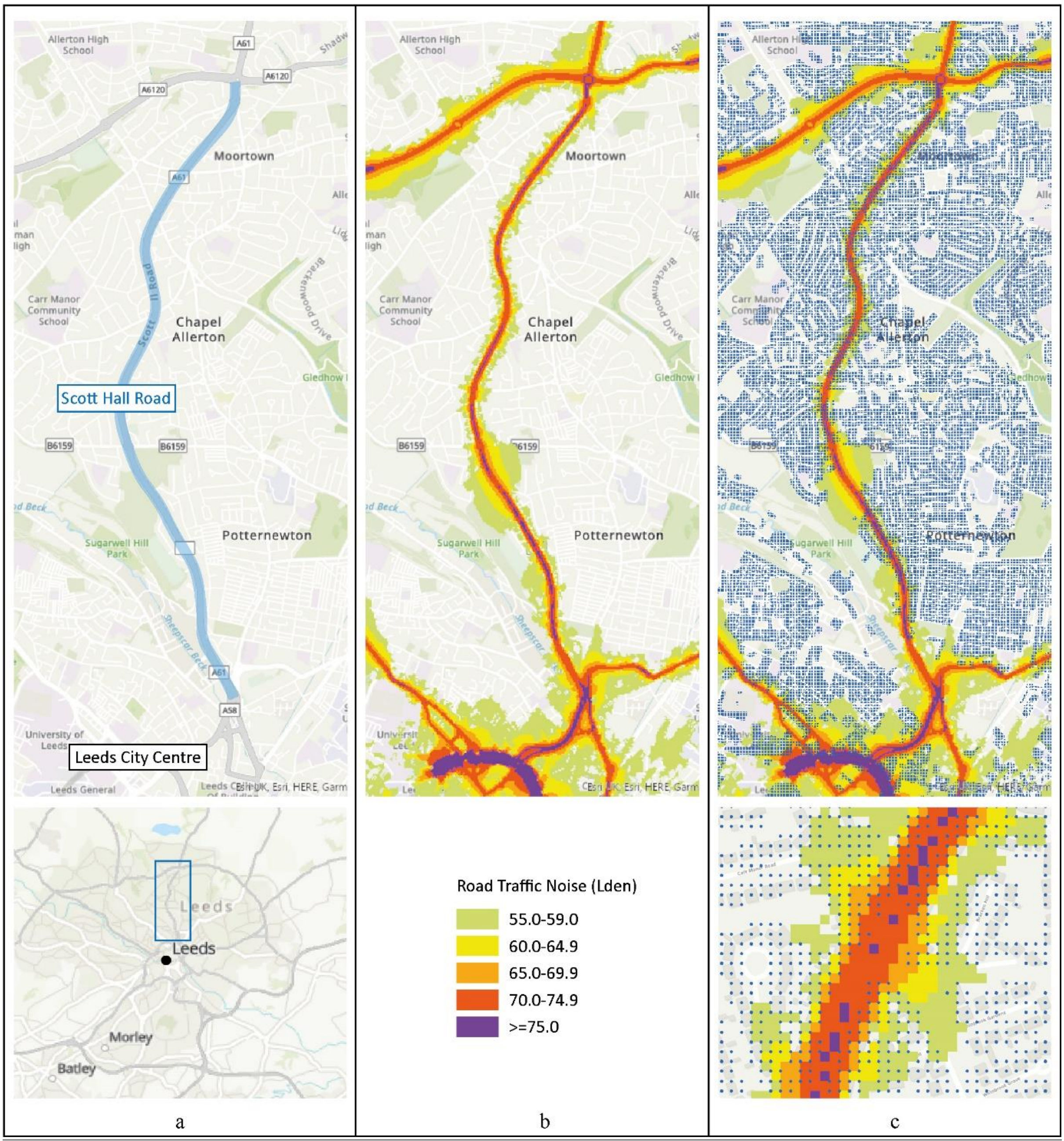
Compared to an electric car, electric bike has lower impacts over all categories. However, it is not proportional to the weight, or occupancy, of the vehicle. It can be calculated from Fig. 3, that three passenger trips on an e-bike would negate all (except the GHGs) benefits compared with driving an electric car. Targeting the right users, for instance those in cars of single occupancy, is therefore important to ensure the use of e-bike will benefit the environment. It also shows the importance of shared mobility and travel demand management in urban transport, which should not be abandoned because of using cleaner and lighter vehicles. Replacing journeys in single-passenger cars with high-occupancy vehicles, improving energy efficiency of electric vehicles and opting for clean electricity, as recommended by International Transport Forum (OECD/ITF, 2020), are all relevant.

The advantages of electric bike when compared on a vehicle life basis (Fig. 2) were reduced when an urban commute is defined as the functional unit (Fig. 3). In terms of the variation in E-Bike's environmental performance across impact categories, results agreed with findings from previous studies, such as Elliot et al. (2018). Nevertheless, this model reflects on the electricity generation mix, vehicle energy efficiency and trip characteristics in the UK. These results will not undermine the potential of electric L-category vehicles in the long term, as battery design and materials are developing at a rapid pace. However, the public need rigorous assessments to avoid making ill-informed behavioural change.

Case study including noise effects

As pointed out in section 1.3, noise impact needs to be estimated on a fleet basis and its impact will depend on the number of receptors and their distance to the source. For this reason, Scott Hall Road (3.3 miles or 5.3 km long), a typical road segment for a commute in Leeds, UK, was chosen as a case study site for the noise impact assessment. A map of the road is shown in Fig. 4a; annual average daily flows by vehicle type on the road in 2019 are given in Fig. 4d. The England Strategic Noise Map 2017 (DEFRA, 2019) was used to find the base (all petrol cars) noise levels, see Fig. 4b and 4c. Since no changes were made to terrains, land uses or meteorological conditions, noise propagation was assumed to remain unchanged when the noise (vehicle fleet) changed at source.

In the base (petrol car) scenario, all cars and taxis on Scott Hall Road were assumed to be travelling at 45 km/h. Using traffic flow data (Fig. 4d) and the CRTN (Calculation of Road Traffic Noise) model (DoT, 1988), the noise level at 10 m from the nearside carriageway edge was calculated. The result is in units of $L_{A10,18h}$ and then converted to 69.3 L_{den} using the equation provided by UK Defra (DEFRA, 2014): $L_{den} = 0.92 \times L_{A10,18h} + 4.2$. This is close to the levels shown along the road on the strategic noise map (Fig. 4b). According to Campello-Vicente et al. (2017), the noise level from electric cars is about 1.1 dB(A) lower than that of petrol cars at 45 km/h, and is about 0.8 dB(A) lower when there are 5% HGVs in both fleets. Considering the 1.1% HGVs, 0.9% buses and coaches, and 10.2% LGVs on Scott Hall Road (Fig. 4d) that will not be electrified, a 1.0 dB(A) reduction in noise level was assumed for the electric car scenario.



d Annual Average Daily Flow on Scott Hall Road in 2019, Leeds, UK (DfT, 2020d)

Pedal cycles	Motor bikes	Cars and taxis	Buses and coaches	Light goods vehicles (LGV)	Heavy goods vehicles (HGV)	All motor vehicles
121 (0.4%)	150 (0.5%)	26,108 (87.3%)	269 (0.9%)	3,050 (10.2%)	329 (1.1%)	29,906 (100%)

Fig. 4 a: Scott Hall Road; b: Strategic Noise Map 2017; c. OpenPopGrid; d. Daily Traffic Flow 2019

For the e-bike scenario, all cars on Scott Hall Road commuting under 5 miles were assumed to be replaced by e-bikes. According to England National Travel Survey (DfT, 2020c), 19.7% of car trips in England in 2019 were for commuting, 57.1% were under 5 miles, and the ratio of car trips to taxi trips was 380:11. Based on these, the number of cars to be replaced by e-bikes on Scott Hall Road is estimated to be: 26108 x (380 / (380+11)) x 0.197 x 0.571 = 2860. Noise from electric engines is ignored in road traffic noise modelling (Patella et al., 2019, Sheng et al., 2016), and the rolling noise from bikes is negligible especially in urban environments. Noise emission from e-bikes was therefore ignored in the e-bike scenario. Using the adjusted traffic data (removing 2860 cars) and the CRTN road noise model, the noise level at 10 m from the nearside carriageway edge was calculated to be 68.9 L_{den}, i.e. 0.4 dB(A) lower than that in the base scenario.

Noise impact was measured in DALY through the impact pathway of annoyance. To do this, the number of people exposed to road noise above 55 L_{den} from Scott Hall Road was calculated in GIS (Table 8), based on the Strategic Noise Map (DEFRA, 2019) and OpenPopGrid (Murdock, 2015). The percentage of people "highly annoyed" by road traffic noise within each noise band in all scenarios was calculated using the dose-response function (Miedema and Oudshoorn, 2001), shown in Eq2.

$$\%\ highly\ annoyed = 0.9868 \times \frac{(L_{den}-42)^3}{10^3} - 1.436 \times \frac{(L_{den}-42)^2}{10^2} + 0.5118 \times \frac{L_{den}-42}{10} \qquad \text{Eq2}$$

Using this method, the total DALY in each scenario was calculated by multiplying the number of "highly annoyed" people with a Disability Weighting of 0.02 (WHO, 2011). Results are shown in Table 8. It can be seen that replacing petrol cars by electric cars can reduce DALY by 8.3%, while the reduction in the e-bike scenario is only 3.4% due to fewer petrol cars being replaced. Larger reductions are possible when travel purposes other than commuting are considered, and other noise impact pathways, such as sleep disturbance, are included. However, a method is needed to make comparison of DALYs between Table 8 and Fig. 3, since the noise impact is assessed on a fleet basis while emission impact is assessed on vehicle basis.

Table 8. Noise impacts and DALY on Scott Hall Road, Leeds, UK

	Noise bands (dB) in base Strategic Noise Map (L_{den})					**Total DALY**	**DALY reduction**
	55.0-55.9	60.0-64.9	65.0-69.9	70.0-74.9	>=75.0		
Number of people within noise band	1,770	520	478	534	55		

% highly annoyed in petrol car scenario	8.16	12.96	20.08	30.25	44.22	9.87	0
% highly annoyed in electric car scenario	7.41	11.84	18.43	27.93	41.09	9.05	-8.3%
% highly annoyed in e-bike scenario	7.85	12.5	19.4	29.31	42.95	9.54	-3.4%

6. Conclusions and Recommendations

Electrification of cars and bikes is increasing in the UK and globally, to improve air quality, reduce carbon footprint and dependence on fossil fuels. L-category vehicles are lighter and smaller alternatives to cars for urban mobility. Electric L-category Vehicles (ELVs) combine the benefits of small size and clean fuels. Their benefits as well as costs to the environment, however, need to be fully understood. Life cycle assessment provides a method to measure the many environmental impacts over the lifecycle considering the supply chain for vehicles and fuels. This study developed a LCA model to evaluate the impacts of electric bike (e-bike) powered by lithium-ion (Li-ion) battery, and compare it with a petrol car and a battery electric vehicle (BEV) for commuting. Four impact categories, namely Global Warming Potential, Acidification, Respiratory Effect and Human Toxicity, are used to characterise the inventory results. The model development and data acquisition presented in this paper should be helpful to conduct similar studies elsewhere in the world. For instance, adoption of 'commute' as an functional unit; need for location specific energy data; case studies for traffic flows and noise impacts; inclusion of tyre and brake wear emissions; importance of end-of-life considerations for batteries. These methodological choices can be further tested and developed in future.

Results showed that BEVs and e-bikes reduce GHGs significantly, and this reduction has room for increase with change in electricity generation mix. Other lifecycle emissions from BEVs are close to, or even higher than, conventional vehicles powered by fossil fuels although the exposure risks are different (Ji et al., 2012). Electric bike has lower impacts than the other two vehicles, but their advantage is not proportional to the weight of the vehicle. Fuel mix as well as the technology used in electricity production, battery manufacturing and recycling are the key areas for reducing emissions from BEVs and ELVs. Also important are the assumptions made in vehicle lifetime mileage, mileage per litre/kWh and battery recycling technique. Different datasets, such as GREET, UK DEFRA and Ecoinvent also affect the results, and they only represent the current and near-future technology. In the case of Li-ion batteries for instance, the design and materials are rapidly evolving, presenting challenges in predicting their durability and EOL options. Batteries in electric bikes are assumed to be replaced in the e-bike's lifecycle but the EOL disposal is not specified. Due to small size and weight, the fate of e-bike batteries is more difficult to predict and it may well be that a lower percentage compared to BEV batteries are recycled. This represents a major challenge to both researchers and policy makers. Sensitivity checks are useful to understand the difference in results caused by data, assumptions and methodological choices.

This study also showed that if the emissions from tyre and brake wear are included in the lifecycle analysis, the impacts of Respiratory Effect and Human Toxicity from petrol cars and BEVs will increase significantly. The proportions of non-exhaust emissions from vehicle fleets are projected to continue increasing, experimental work to quantify tyre and brake emissions will complement the current databases. Very few studies are found on the tyre and brake emissions from e-bikes. Experimental work is needed as the materials (e.g. disc, pads), wear rate and (sometimes) riding surface are different, and therefore the physical modelling results found in literature for car tyre and brake wear, cannot be used to project what the emissions might be from e-bikes.

Noise reduction due to the use of electric vehicles is offsetting some, but not able to negate entirely, the increase in human health impacts. The inclusion of road traffic noise in road LCA has been proposed in the literature. This study confirmed that a fleet-based approach is feasible for a small-scale (scheme level) noise assessment. There is room for developing the methodology in LCA that integrates noise impacts with other environmental impacts. Findings from the study will help automobile manufacturers to identify what the key phases are in the supply chain, as well as help the public to make informed decisions about which vehicle to choose for a specific trip purpose. Investigation of market potential and user shift from ICEV or BEV is relevant but outside the scope of this study. Also excluded are the influence on individual modal choice and the potential brought by decarbonisation of the grid.

Maintenance needs of the vehicles and charging infrastructure are yet to be included in the LCA. The limitations as a result of excluding transport infrastructure have been discussed in the literature (Chester and Horvath, 2009). Including the infrastructure for ELVs however, will be complex as their demand for road space and charging facilities will inevitably have to be assumed and it will be difficult to generalise for another techno-economic model. There are unique features associated with electric bikes such as more frequent replacements of battery and tyres. Figure 3 and Table 7 also demonstrated the production of electric bikes is more significant compared with petrol and electric cars. Vehicle maintenance and parts replacement for electric bikes should be included in future LCA studies. Other types of fuels for L-category vehicles such as diesel, biofuel and fuel cell, are excluded from this study but present interesting research for the future.

Encouraging the uptake of electric L-category vehicles needs the key drivers (e.g. fuel efficiency) and barriers (e.g. range, safety) to be identified. The true and complete environmental impacts of ELVs, as well as a roadmap to reduce them, need to be understood prior to any smart mobility initiatives to promote them. However, battery electric vehicles (cars and bikes) should be encouraged, given the potential of improvement in energy mix and production efficiency in the long term. Low speed and short distance make ELVs suitable for the urban commute, while overcoming range anxiety. Electrification should come with other green transport initiatives such as shared mobility and travel behaviour change. As pointed out in the literature (Weiss et al., 2015), large-scale adoption of electric two-wheelers can reduce traffic noise and road congestion but may necessitate adaptations to urban infrastructure and safety regulations. The undesirable effects of e-bikes on reduced physical activity in European cities (Winslott Hiselius and Svensson, 2017) and on increased risk to vulnerable road users due to the higher speed of e-bikes, have been noted by researchers. It raises an interesting question of where and to whom the electric bikes, or in fact any power-assisted bikes, should be promoted.

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Table 5 – Life cycle inventory results (unit: gram per vehicle)*

Inventory	Petrol car			Battery electric vehicle				Electric bike			
	PC-p	PC-t	PC-u	EV-p	EV-t	EV-u	EV-db	EB-p	EB-t	EB-u	EB-d
CO_2	5.678E+06	7.528E+03	5.525E+07	6.262E+06	2.078E+05	1.096E+07	6.862E+04	1.356E+05	4.739E+03	1.010E+05	6.903E+04
CH_4	1.259E+04	9.845E+00	1.469E+03	1.353E+04	2.189E+02	3.360E+04	6.932E+01	4.621E+02	5.000E+00	3.098E+02	2.362E+02
N_2O	1.385E+02	2.580E-02	1.295E+03	1.500E+02	9.843E+00	6.480E+04	6.528E-01	0.000E+00	2.329E-01	5.974E+02	0.000E+00
SOx	2.622E+04	6.588E-01	3.337E+02	4.752E+04	5.870E+02	4.879E+03	6.070E+01	7.114E+02	1.345E+01	4.497E+01	6.226E+02
NOx	6.909E+03	1.564E+01	2.041E+04	7.801E+03	3.938E+03	7.429E+03	7.303E+01	4.085E+02	9.310E+01	6.848E+01	2.287E+02
VOC	3.317E+04	2.246E+00	2.288E+04	3.280E+04	1.526E+02	7.363E+02	9.338E+00	6.861E+01	3.605E+00	6.788E+00	6.283E+01
PM_{10}	2.626E+03	5.848E-01	9.191E+02	3.274E+03	2.052E+02	8.017E+03	8.636E+00	2.494E+02	4.712E+00	7.391E+01	1.478E+02
$PM_{2.5}$	1.216E+03	3.284E-01	8.130E+02	1.349E+03	1.886E+02	2.481E+03	5.968E+00	1.528E+02	4.329E+00	2.287E+01	9.532E+01

* PC-p/t/u: petrol car production/transport/vehicle use; EV-p/t/u/db: electric vehicle production/transport/vehicle use/battery disposal; EB-p/t/u/d: electric bike production/transport/vehicle use/disposal

Table 6 – Life cycle impact assessment results (unit: gram per vehicle)

Impact	Indicator	Petrol car			Battery electric vehicle				Electric bike			
		PC-p	PC-t	PC-u	EV-p	EV-t	EV-u	EV-db	EB-p	EB-t	EB-u	EB-d
GWP	CO_2 eq.	6,067,419	7,810	55,632,315	6,680,220	216,542	29,073,600	70,730	148,494	4,940	268,022	75,644
Acidification	SO_2 eq.	34,923	9	10,604	60,926	2,673	9,569	109	1,058	63	88	861
Respiratory effects	DALY (10^{-3})	3,923	2	2,771	5,501	590	5,670	17	276	14	52	177
Human toxicity	1,4-DB eq.*	12,961	19	25,274	16,608	4,950	15,957	101	763	117	147	455

*1, 4-DB eq.: 1,4-dichlorobenzene eq.

Table 7 – Impact assessment results for 5-mile commuting (unit: gram)

Impact	Indicator	Petrol car			Battery electric vehicle				Electric bike			
		PC-p	PC-t	PC-u	EV-p	EV-t	EV-u	EV-db	EB-p	EB-t	EB-u	EB-d
GWP	CO_2 eq.	167.378	0.222	1,625.024	261.472	8.668	460.800	2.862	72.951	2.544	54.680	37.149
Acidification	SO_2 eq.	0.974	0.000	0.610	2.305	0.189	0.513	0.006	0.601	0.057	0.061	0.457
Respiratory effects	DALY (10^{-3})	2.433	0.001	1.490	4.428	0.220	2.573	0.009	1.101	0.067	0.283	0.747
Human toxicity	1,4-DB eq.*	1.052	0.000	0.727	2.442	0.197	1.021	0.006	0.734	0.060	0.100	0.536

*1, 4-DB eq.: 1,4-dichlorobenzene eq.