Environmental implications of the ongoing electrification of the UK light duty vehicle fleet.

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Abstract

The light duty vehicle fleet in the UK is being electrified aggressively, with an ambitious target to ban the sale of all new internal combustion engine cars by 2030. At the same time, the electricity grid is also undergoing rapid decarbonization, potentially paving the way for a much greener use phase for electric vehicles. The paper presents a holistic prospective life cycle assessment of the environmental implications of these two interrelated transitions, while also considering an alternative scenario characterised by a gradual shift from traditional private vehicle ownership to shared mobility schemes. The results for both scenarios point to clear benefits in terms of reduced demand for non-renewable energy, carbon emissions and local air quality. However, a decisive behavioural shift towards shared mobility is shown to be crucial in order to offset the increased demand for Li, Co, Ni, Mn and Cu for electric vehicle power trains, and to avoid an otherwise potential increase in abiotic resource depletion and human toxicity impacts.

Keywords:

Electric vehicles; shared mobility; lithium-ion batteries; recycling; life cycle assessment.

1. Introduction

Transport is an integral element of society and the economy, providing market and social connectivity. While transport has brought many benefits, it has also created environmental problems. Since the introduction of motor vehicles in the 20th century, their numbers have increased to nearly 40 million today in the United Kingdom (Department for Transport, 2020), where cars have become one of the most popular forms of transport. This has led to increased congestion, greenhouse gas (GHG) emissions and air pollutants such as nitrogen oxides (NO_x) and particulate matter (PM) (Department for Transport, 2019). Furthermore, most of the chemical energy in the fuels used within the internal combustion engine vehicles (ICEV) ends up being released to the environment as heat through the exhaust or the radiator, while only around a quarter of it is converted to kinetic energy to propel the vehicle (National Research Council, 2011). This is mostly due to the inescapable thermodynamic limitations of the internal combustion engines and drivetrain power losses (National Research Council, 2011; Mahmoudi et al., 2015). As a result, while improvements have been made over the years in combustion, engine control and after-treatment technologies to reduce exhaust pollutants (Reitz et al., 2020), it has not been possible to drastically reduce fuel consumption. Electric vehicles (EVs) started gaining momentum in recent years due to the development of battery technologies, reductions in cost and policy initiatives to reduce carbon emissions and improved air quality (IEA, 2020). EVs are considered among the key technologies to decarbonise the road transport sector, providing improved tank-to-wheel (TTW) energy efficiency by a factor of 3 compared to ICE powertrains, and potentially significantly reducing air pollution, provided that the electricity is generated by efficient grid systems comprising of low carbon technologies (IEA, 2020; Brito et al., 2013).

Previous work has investigated the life-cycle environmental impacts of EVs and compared them to those of conventional vehicles. From some of the life cycle assessments (LCA) studies conducted it was clear that battery manufacturing leads to higher carbon emissions for the production phase of EVs when compared to ICEVs (Kawamoto et al., 2019, Kim et al., 2016, Ellingsen et al., 2016). A study conducted by Kim et al. (2016) estimated a 39% increase in total cradle-to-gate GHG emissions for a compact EV compared to an otherwise similar ICEV. However, the whole life-cycle GHG emissions then decrease compared to ICEVs, especially when EVs are driven further (this was also shown to be dependent on the carbon emissions intensity of the grid). Moro and Lonza (2018) analysed well-to-wheel (WTW) GHG emissions of EVs based on the 2013 grid mixes for various EU states, and results indicated that EVs allow potentially 50–60% GHG emission savings compared to ICEVs. Hoekstra (2019) showed that for a typical large car in Europe, switching from an ICE to an electric power train would reduce GHG emissions by over 60% over the full life-cycle of the vehicle. Furthermore, a study by Cox (2020) on the LCA of vehicles confirmed that EVs with large battery packs are very sensitive to the grid mix composition, and that therefore grid decarbonization is required in order to see a significant reduction in the life-cycle GHG emissions. Hill et al. (2019) also examined the life-cycle GHG impacts of EV adoption in the UK transport sector, and their results indicated that the adoption of EVs along with the decarbonisation of the grid mix is the best way to minimise carbon impact.

However, although EVs address a series of environmental concerns and show large potential for carbon emission reductions, they also raise questions related to energy use and environmental impacts during their manufacturing, in particular as relates to the ecological and human toxicity associated with the increased demand for metals. Girardi et al. (2015) carried out an LCA of EVs in 2013 and 2030 scenarios, and the results showed that EV perform well in most environmental indicators except for human toxicity and eutrophication, mainly due to the manufacturing of EV batteries. Further concerns are related to resource availability for the manufacturing of EVs. Hernandez et al. (2017) examined the impacts related to metal depletion and found that the overall depletion impact is largely due to the large mass of the battery pack. Olivetti et al. (2017) raised concerns about the growing demand for batteries, as some of the materials are not mined in large amounts, are limited by the number of reserves currently available, or are mined in countries with high geopolitical risk Dunn et al. (2012) focused on the cradle-to-gate energy consumption and emission impacts of different recycling methods used for the production of lithium-ion batteries (LIBs) with lithium manganese oxide (LMO) as the cathode material. They estimated that a closed-loop recycling scenario can reduce energy consumption during material production by up to 48%. Ahmadi et al. (2017) carried out an LCA of EVs with the focus on battery recycling and reuse, and their results indicated that although battery manufacturing still dominates the environmental impacts, there is significant reduction to be seen in GHG emissions by extending the battery lifetime to a second life in grid storage application. Other studies identified battery recycling and reuse as possible mitigation strategies to reduce some of the associated environmental burdens (Winslow et al., 2018; Martins et al., 2021).

The UK transition to EVs is set to develop rapidly. Policies are in place to ban the sale of fossil-fuelled vehicles in 2030 (UK Government, 2021), purchase incentives for EVs are available, the cost of batteries is decreasing steadily, and increasing numbers of charging points are being installed around the country. The UK transport sector is now approaching a set of simultaneous transitions that are expected to drastically re-shape mobility services in the next few decades. These include (1) a rapid growth of electrical mobility, with special emphasis on light duty vehicles (LDVs); (2) the co-evolution of the electricity grid, with a range of on-going deep decarbonization efforts, and (3) a possible gradual transition from traditional personal vehicle ownership to various shared mobility solutions, also collectively referred to as "Transport-as-a-Service" (TaaS).

The purpose of TaaS is to provides a shared and flexible mode of travel, whereby users can access mobility services based on travel needs with a concomitant reduction in the requirement for privately-owned cars (MaaS Alliance; Foresight, 2019). In this paper, TaaS is used to refer to shared mobility schemes provided by cars and small vans. Amatuni et al. (2020) examined the environmental impacts associated with car sharing, which were shown to be sensitive to the characteristics of the transport system, such as the average occupancy of car sharing, the electricity grid mix and the total demand for kilometres travelled per passenger. Previous studies on some of the schemes for TaaS identified potential environmental benefits, in terms of air quality, energy consumption and reductions in GHG emissions (Pan et al., 2021; Jung and Koo, 2018; Sheppard et al., 2021).

Understanding the implications of future prospective mobility scenarios requires fully assessing a broad spectrum of environmental impacts, to help inform strategies for environmentally sustainable pathways. To date, however, hardly any studies have attempted to expand the boundaries of assessment to the degree necessary to capture all the interlinkages and all the factors at play in a single, methodologically coherent analysis, including careful assessments of the shifting demand for a range of critical metals, of the key role of end-of-life (EoL) battery management strategies, and of a potential shift towards shared mobility.

This paper aims to fill this critical knowledge gap by presenting a holistic prospective LCA of the evolution of the LDV fleet in the UK over the next three decades (i.e., 2020 to 2050), with special focus on the transition to electrical mobility, the evolving electricity grid mix and the pivotal roles of battery recycling and shared mobility.

2. Materials and Methods

2.1. Goal and scope

The overarching goal of the study is to quantify the environmental impacts of the transport evolution of LDV fleet in the UK. This includes the expected major shift from internal combustion engine (ICE) to electrical power trains, the gradual increased penetration of renewable energy into the grid mix, and the possible large-scale uptake of shared mobility.

The functional unit of the study was set as the operation of, and the net changes to, the whole LDV fleet in the UK over the course of one year. Figure 1 shows how the overall LCA model was structured to comprise: (1) the manufacturing of those LDVs that are newly registered in each year of analysis; (2) the use phase of all LDVs on UK roads in the same year, including their maintenance and the supply chains of the required energy carriers, i.e., petrol and diesel for internal combustion engine vehicles (ICEVs) and electricity for EVs; (3) the decommissioning of those LDVs that reach their EoL in the same year of analysis. The co-evolution of the UK grid mix was considered by means of a dedicated LCA sub-model (*cf.* section 2.5).

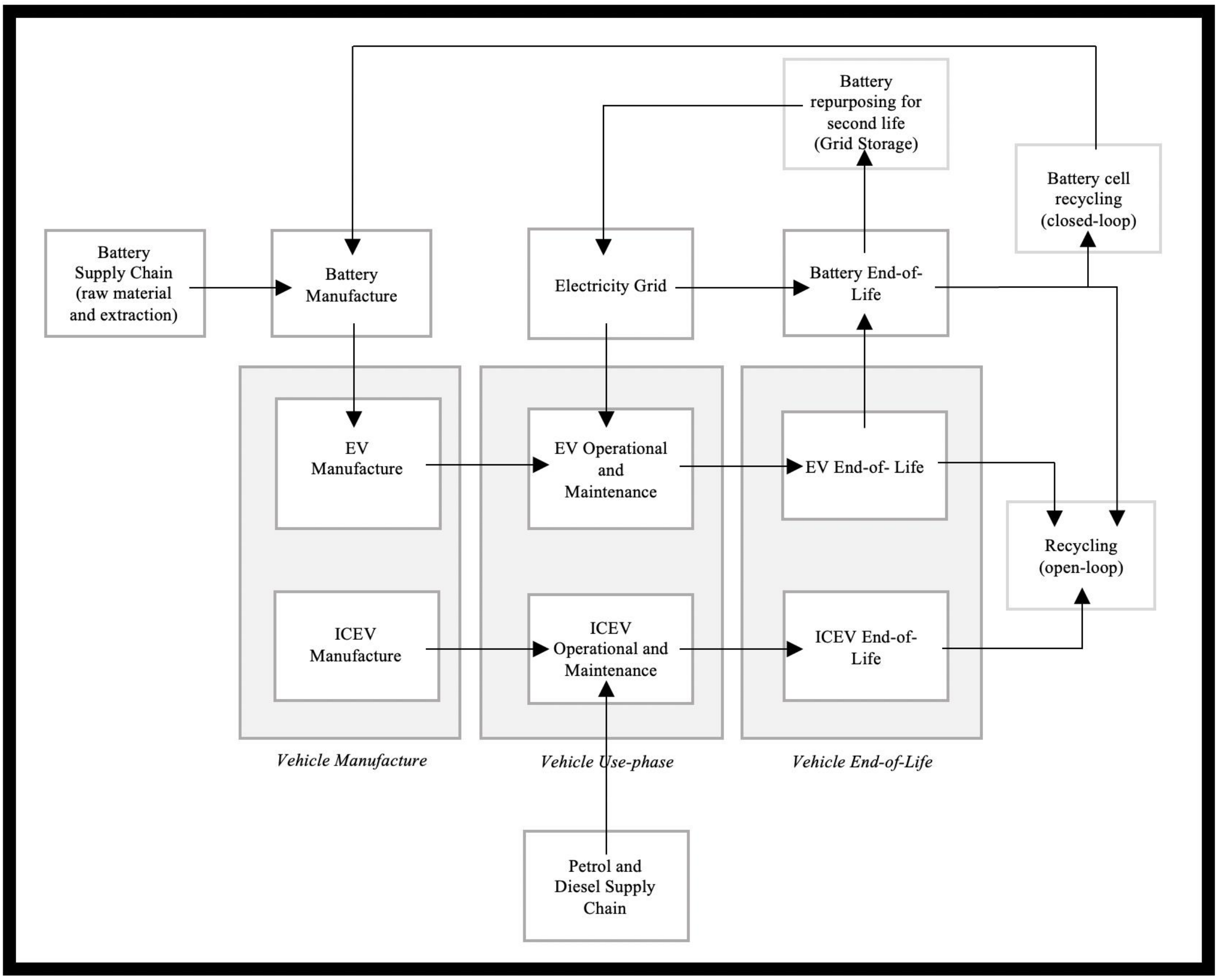


Figure 1 – Structure of the LCA model, with identification of the individual sub-models used for each of the key processes comprising the analysed system.

Two alternative scenarios are considered, respectively named "Baseline" and "TaaS", where the latter assumes an increased adoption of shared mobility, leading to a gradual reduction in the total number of vehicles required to meet the same overall demand for personal mobility (i.e., total distance travelled per year). The expected future growth in the total distance travelled each year is based on the UK Department for Transport Road Traffic Forecasts "reference" scenario (Department for Transport, 2018), which projects a 34% increase in total annual distance travelled by cars from 2015 to 2050. The original projection was limited to Wales and England, but it is here assumed to apply to the UK as a whole.

A potential uptake in shared mobility is expected for the future transport system (Möller et al., 2019; Foresight, 2019). While still at an early stage, there are a growing number of shared mobility schemes in the UK, such as Zipcar, Drivy, liftshare, Ubeeqo, BlaBlaCar (Getaround, 2021; Ubeeqo, 2021; BlaBlaCar, 2021; Liftshare, 2021; Ipsos MORI, 2019). Operators of shared mobility are also early adopters of EVs, and therefore the overall uptake of TaaS is likely to be mostly electric, with associated benefits in terms of reductions in energy consumption and CO_2 emissions (Baptista et al., 2014).

In the "TaaS' scenario, two types of shared mobility schemes are considered: car sharing and ride sharing, and in both cases a mileage of 64,000km/year is assumed, based on literature studies (Amatuni et al., 2019; OECD/ITF, 2017; Crabtree, 2019), which results in a lifetime mileage of 190,000km in 3 years. Accordingly, TaaS vehicles fall on the lower end of the scale in terms of yearly mileage covered and are assimilated to rental cars which are typically replaced every 3 years (Cho and Rust, 2008; Mont, 2004). Also, these vehicles not expected to have second-hand market as the batteries may need to be replaced, leaving the vehicles with little residual market value.

Privately-owned cars are assumed to cover 13,700 km/year over a service life of 14 years. Thus, the same overall lifetime mileage of 190,000km is assumed for both types of vehicles. The two scenarios are also the same in assuming a linear increase in total distance travelled (from 420 billion km/year in 2020 to 530 billion km/year in 2050), a gradual phase-out of ICEVs (consistently with the UK government's target to ban sales of new light-duty ICEVs by 2030), and a linearly increasing collection rate for EoL EV LIBs, destined to recycling. Tables 1 and 2 report the key parameters defining the two scenarios, respectively. More detail on all the other assumptions underpinning the scenarios was published previously (Kamran et al., 2021).

Year	**2020**	**2025**	**2030**	**2035**	**2040**	**2045**	**2050**
New ICEVs	2,060,000	830,000	9,000	0	0	0	0
New EVs	410,000	1,160,000	2,670,000	2,780,000	2,400,000	2,830,000	2,800,000
Demand of EV Battery (tonnes)	133,222	373,793	862,899	899,060	774,298	913,257	903,217
% of new EVs for TaaS	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
EoL ICEVs	2,230,000	1,740,000	2,420,000	1,880,000	710,000	0	0
EoL EVs	0	0	0	630,000	1,400,000	2,530,000	2,490,000
Total ICEVs in circulation	29,740,000	26,550,000	16,360,000	6,520,000	1,230,000	0	0
Total EVs in circulation	850,000	5,280,000	15,500,000	27,950,000	34,640,000	37,330,000	38,850,000
EoL EV LIB collection rate	80%	83%	86%	89%	93%	96%	99%
Total Collected EoL LIBs from EVs (tonnes)	0	0	0	181,027	420,157	783,481	795,579

% of collected LIBs sent to second life	0	0	0	26%	20%	10%	9%

Table 1. Key model parameters for "Baseline" scenario (vehicle numbers rounded to nearest 10,000).

Year	2020	2025	2030	2035	2040	2045	2050
New ICEVs	2,060,000	690,000	7,000	0	0	0	0
New EVs	410,000	820,000	1,660,000	1,470,000	910,000	620,000	270,000
Demand of EV Battery (tonnes)	133,221	266,067	535,095	473,794	292,502	199,049	88,073
% of new EVs for TaaS	0.0%	7.5%	15.0%	22.5%	30.0%	37.5%	45.0%
EoL ICEVs	2,230,000	1,740,000	2,420,000	1,760,000	570,000	0	0
EoL EVs	0	60,000	215,000	1,070,000	1,720,000	2,710,000	2,940,000
Total ICEVs in circulation	29,740,000	25,700,000	16,360,000	5,380,000	960,000	0	0
Total EVs in circulation	850,000	4,940,000	13,400,000	23,040,000	27,640,000	27,580,000	25,930,000
EoL EV LIB collection rate	80%	83%	86%	89%	93%	96%	99%
Total Collected EoL LIBs from EVs (tonnes)	0	16,060	59,955	308,433	516,131	839,588	941,588
% of collected LIBs sent to second life	0	4%	47%	22%	14%	9%	7%

Table 2. Key model parameters for "TaaS" scenario (vehicle numbers rounded to nearest 10,000).

From a modelling perspective, this LCA adopts a hybrid approach, where selected consequential elements are combined with attributional elements, consistently with the main goal of the study, which is to analyse the evolution of the environmental impacts of the UK light duty vehicle fleet, with special focus on the growing role of LIBs, their closed-loop recycling and reuse, and the potential environmental benefits of shared mobility. Specifically, the consequential elements in the model are limited to the battery supply chain and the end-of-life recycling of the technology-specific battery metals, i.e., lithium, cobalt, nickel and manganese. Since these metals are used in large quantities and play a key role in the evolution of the transport sector, in some cases potentially posing questions about the continuity of their supply, we found it necessary to model them in a consequential way. Conversely, an attributional approach is adopted when modelling the recycling of other metals such as copper, aluminium or steel, as the overall markets for these metals are not as significantly impacted by the uptake of EVs.

2.2. ICEV and EV sub-models

The ICEV and EV sub models are divided into three phases: manufacturing, use phase and EoL. The models for both vehicle types were originally developed for a compact (C-segment) passenger car (Raugei et al., 2015). For the ICEV model, the powertrain was based upon a 44kW internal combustion engine used in the Volkswagen Polo and Golf production models. For the EV model, the focus was put entirely on battery electric power trains (thereby disregarding hybrid vehicles as they are likely to play only a relatively minor and temporary role in the fleet, over the first few years of the considered time frame), and an electric motor of the same power was assumed; the LIB was modelled separately (*cf*. section 2.3). Both vehicle models were then scaled up to represent the average LDV fleet on UK roads (based on Raugei et al., 2018): the fleet-average kerb mass was taken as 1370 kg, and fleet-average use-phase energy consumption figures for ICEVs and EVs were taken respectively as 8 litres of fuel (assuming 50% petrol and 50% diesel) per 100km and as 19 kWh of electricity per 100km. Both ICEVs and EVs were assumed to have a service life of 190,000 km (Ricardo-AEA, 2015). The vehicle manufacturing phase comprises the following sub-assemblies, modelled using Ecoinvent database (Ecoinvent, 2020) processes: body and chassis (consisting of all steel components), powertrain, electrical system and trim. In recent years, the LDV fleet on UK roads has comprised a mix of vehicles manufactured domestically and in Europe, with the latter mainly represented by Germany (Department for Transport, 2020; European Automotive Manufacturers Assocation, 2019). However, considering the upcoming change regarding the motor vehicle trade between the UK and EU, also in light of Brexit, it is likely that the share of domestic vehicles will increase in the future, and hence for the sake of simplicity vehicle manufacturing was modelled as taking place in the UK. The vehicle use phase accounts for both direct and indirect emissions. Among the former are the tailpipe emissions due to petrol consumption in ICEVs, and the non-tailpipe emissions due to tyre, brake-pad and tarmac wear which apply to both ICEVs and EVs. Indirect emissions are to do with the fuel supply chains for ICEVs, and with electricity for battery charging for EVs (*cf.* section 2.5). It was further assumed that both vehicle types undergo scheduled maintenance every 30,000 km, which includes the replacement of tyres, brake pads, vehicle lubrication, as well as 5% of worn-out trim. Also, all vehicles were expected to sustain 10% outer body panel replacement due to impact damage over their full-service lives. ICEVs were assumed to undergo one lead-acid battery replacement, whereas the factory-installed LIBs in EVs were assumed to remain in service until the final decommissioning of the vehicles, consistently with current estimates (EDF, 2020; Xu et al., 2016). The EoL of all steel parts was modelled to reflect the legislation in Europe for EoL vehicles (Directive 2000/53/EC), which mandates 85% mass recycling, and steel recycling was assumed to be open-loop. For the Al alloys used in engines and wheels, a lower 75% open-loop recycling rate was assumed, consistent with the data reported by the Aluminium Federation (2013), while for the higher-value metals (excluding LIB metals, which were modelled separately) 100% recycling was assumed, respectively open-loop (for Cu) and closed-loop (for Pt used in catalytic converters). EoL environmental credits for all recycled metals were calculated assuming the displacement of, respectively, primary metal supply chains for closed-loop recycling, and the industry average mix of primary and secondary supply chains for open-loop recycling. Finally, all plastic parts were assumed to be incinerated with energy recovery credits.

2.3. LIB manufacturing sub-model

There have been various studies conducted on battery technology roadmaps (Thielmann, 2013; Edström, 2020; Faraday Institution, 2020; Element Energy, 2016; Hill et al, 2020). A recent report by Battery 2030 provides a comparison of the performance targets set by the battery manufacturers, which imply an expected increase in solid state batteries with Li metal anodes by 2035 (Edström, 2020). However, although Li metal anodes allow higher theoretical capacities than graphite anodes, they still suffer from safety and performance barriers that have so far prevented their use in EVs (Brotchie, 2016, Ricardo, 2020). Current lithium anode cells have shown to deliver 300 Wh/kg, but they still suffer from poor cycle life, and further improvements are required before these cells can replace the well-established lithium-ion chemistries. (Energy.gov. 2020). Hence, the roadmap targets for solid state Li batteries may seem ambitious when taking account of the current barriers (Evarts, 2018; Automotive News Europe, 2021; Ma, J., 2021). Sodium-ion is another possible future battery technology, with cathodes replacing Li with Na, and containing titanium, nickel and manganese metal oxides, but without the presence of cobalt (EVreporter, 2019; Abraham, 2020). Hence, in terms of resource availability sodium-ion batteries show high superiority. A study conducted by Ricardo projects a large future deployment of sodium-ion batteries, along with solid states batteries, through to 2050 (Hill et al, 2020). However, although sodium-ion batteries are already being used in stationary storage applications and show potential for low-speed transport applications such as e-scooters and e-bikes (Faradion, 2021), they still require further work in relation to service life, energy density and safety issues for their use in EVs (Yang et al., 2021; Abraham, 2020; Stringer, 2020; Delmas, 2018). Accordingly, the Faraday Institution estimates a more conservative shift from mainly Li-ion batteries with NMC 622 and NMC 111 cathodes to similar chemistries but with NMC 811 and NMC 955 cathodes by 2050, with leads to a reduced proportion of cobalt in the battery pack (Faraday Institution, 2020).

Considering that NMC batteries have already established themselves as the leading battery technology in the EV market, and in light of the large uncertainties regarding the other battery chemistries and their possible commercialisation in the EV sector, a similar conservative assumption is made in this paper, too, whereby NMC LIBs remain the mainstream technology all the way to 2050. The battery manufacturing model assumes an average battery pack mass of 323kg, which initially equates to a usable energy storage capacity of 50kWh per vehicle, based on current EV battery technology used (Electric Vehicle Database, 2020; Battery University 2020; Raugei et al., 2018). The battery pack mass is then assumed to remain constant, while the usable capacity is expected to change in the coming years as the technology improves. A linear increase in energy density is assumed up to year 2035 due to the expected shift from NMC 622 to NMC 811 (IEA 2020; Element Energy 2016). Further improvements in batteries assumed to be due to the reduction in weight of the battery pack casing and ancillary, this reduction is used to increase the energy density further while keeping the mass constant though to 2050 as to achieve an increasing vehicle driving range.

Currently China, South Korea and Japan represent the major shares of battery manufacturing worldwide (Eddy et al., 2019). However, there is growing demand for battery production to be close to car manufacturers. Therefore, considering the recent uptake of EVs and the planned ramp-up of battery production capacity in Europe, it was assumed that the cell components and battery packs for the UK fleet will all be manufactured and assembled in Europe. Specifically, LG Chem battery production in Poland is scheduled to increase to 65-70 GWh/year (Reiserer, 2019), SK Innovation production is to reach 30 GWh/year in Hungary (Inside EVs, 2020), and Northvolt further plans production of up to 32 GWh/year in Sweden (Phillips, 2020). These represent the largest battery manufacturing capacities in Europe, and thus a suitably weighted combination of the current Polish, Hungarian and Swedish grid mixes was used in the model to estimate the impacts from the electricity use during battery manufacturing up to the year 2050. As for transportation impacts, it was assumed that the batteries will be transported over an average distance of 2,000 km by large (16-32 tonnes) lorries, from the EU to the UK. The foreground material and energy use inventory for battery manufacturing was informed by a recent report by Argonne Laboratory (Dai et al., 2018), and the average LIB cell-to-pack mass ratio was taken as 0.71.

The primary supply chains of the key cathode metals Li, Co, Mn and Ni were modelled as follows. In 2016, the three main global suppliers of primary lithium were Australia (45% of global supply, from spodumene rocks), Chile (31%, from Li brines) and Argentina (12%, from Li brines) (BGS, 2018; USGS, 2018). Therefore, the focus of the model was on production taking place in those three countries, and then it was assumed that the Li import shares to Europe will reflect the same relative shares of global production capacity. Cobalt is extracted mainly from mining of copper-cobalt ores and, to a lesser extent, nickel-cobalt ores. The Republic of Congo is currently the most prominent producer of Co by far, contributing to 55% of global supply (British Geological Survey, 2018 and US Geological Survey, 2018), and so Co production was modelled as coming from there, with subsequent transport to China for refining (Dai et al., 2018b). Manganese is relatively abundant in the Earth's crust, and is extracted by mining a range of ores, the most important of which is pyrolusite (MnO_2). In 2016, the five main producing countries were South Africa (31%), China (18%), Australia (14%), Gabon (12%) and Brazil (7%) (BGS, 2018; USGS, 2018). Therefore, the focus of the model was on production taking place in those five countries. In 2016, the largest shares of global nickel production were evenly distributed among a relatively large number of countries, among which foremost were the Philippines (16%), Russia (11%), Canada (11%), Australia (10%), New Caledonia (10%) and Indonesia (9%) (BGS, 2018; USGS, 2018). The two main sources of Ni are sulfide mines, which is characteristic of Canada and prevalent (>60%) in Australia, and lateritic deposits. The latter is gradually becoming more prevalent, but unfortunately, detailed life-cycle inventory data for laterite mining is yet unavailable. Also, Canadian production is expected to drop after the planned phasing out of Thompson Mines' operations in Manitoba, Newfoundland and Labrador. Therefore, Australian production of raw Ni from sulfide mines was selected for the model, plus its subsequent transport to Europe. The model also includes processing of all four metals up to the chemical forms in which they are fed to the LIB manufacturing industry, i.e., respectively, LiOH, $CoSO_4$ $MnSO_4$ and $NiSO_4$.

Finally, and very importantly in terms of the results of the overall LCA, the consequential approach adopted for the battery supply chain model dictates that the masses of the recovered metals from recycled EoL EV batteries and from second-life batteries from grid storage directly reduce the quantities of the same metals that are sourced from primary supply chains for the manufacturing of new EV batteries (*cf*. Tables 1 and 2, and section 2.4). The number of batteries that are recycled is determined by the number of EoL EVs, the collection rate, and also by how many batteries are repurposed for second life, and all of these parameters are estimated dynamically within the model and change every year. The resulting shares of metals demand for battery manufacturing that are supplied by recycling in the "Baseline" and "TaaS" scenarios are reported in Tables 3 and 4, respectively.

	% secondary Li	% secondary Co	% secondary Ni	% secondary Mn
2020	0%	0%	0%	0%
2025	0%	0%	0%	0%
2030	0%	0%	0%	0%
2035	10%	9%	8%	19%
2040	37%	61%	32%	61%
2045	70%	92%	67%	93%
2050	77%	78%	79%	79%

Table 3. Shares of metals demand for battery manufacturing that are supplied by recycling in the "Baseline" scenario.

	% secondary Li	% secondary Co	% secondary Ni	% secondary Mn
2020	0%	0%	0%	0%
2025	5%	6%	5%	6%
2030	5%	6%	5%	6%
2035	44%	69%	40%	69%
2040	100%	100%	100%	100%
2045	100%	100%	100%	100%
2050	100%	100%	100%	100%

Table 4. Shares of metals demand for battery manufacturing that are supplied by recycling in the "TaaS" scenario.

2.4. LIB recycling sub-model

Battery pack dismantling was excluded from the model due to the lack of information on the associated processes at scale, but all the non-cell parts of the battery pack were assumed to be treated as scrap, and their recycling was modelled under the same assumptions as for the other metal parts of the EoL vehicle (*cf*. section 2.2). Currently the most mature process for recycling LIBs is pyrometallurgical recycling. However, due to its high energy demand and low metal recovery efficiency (Arambarri et al., 2019), and the fact that a shift to hydrometallurgical recycling is already underway in Asia, all future LIB cells were assumed to undergo hydrometallurgical recycling in the UK, using inorganic acid leaching to recover the key battery materials: Li, Ni, Mg and Co. The assumptions on metal-specific recovery efficiencies, and the sources used to inform them, are reported in Table 5.

The foreground material and energy use inventory for the recycling process was informed by a recent Argonne Laboratory report (Dai and Winjobi, 2019). It was assumed that the recycling plants will be based in the UK; accordingly, the UK grid mix model (*cf*. section 2.5) was used to track the evolving impacts of electricity use over time, from 2020 to 2050. It was further assumed that electricity will be used for industrial furnaces and all machinery.

Finally, as discussed in Section 2.3, it was assumed that all the recovered cathode metals will be reused directly for the manufacturing of new LIB cells (i.e., closed-loop recycling). Accordingly, only the net surplus of recycled metals, after the annual demand for new LIBs to equip newly registered EVs has been satisfied, was accounted for towards the calculation of EoL impact credits. This was calculated based on the displacement of the respectively primary supply chains (up to chemical form in which the metals are fed to the LIB manufacturing industry).

Metal	Recovery Efficiency	Reference
Li	95%	(Greim et al., 2020)
Co	94%	(Cheret & Santen, 2007)
Ni	99%	(Cheret & Santen, 2007)
Mn	95%	(Chen & Zhou, 2014; Melin, 2019)

Table 5. Recovery efficiencies of metals in hydrometallurgical recycling process

2.5. Electricity Grid mix sub-model

The electricity grid mix model takes account of the evolution over time of the impacts related to the generation of electricity in the UK. This LCA sub-model is largely based on research previously developed and published (Raugei et al., 2020). However, whereas the previous model was based on the "2 degree" future energy scenario published by National Grid in 2019 (National Grid, 2019), this new model reflects the latest "leading the way" scenario (National Grid, 2020). The main technology changes between the "2 degree" and "leading the way" scenarios are that the latter assumes an uptake of biomass with carbon capture and storage (CCS) instead of natural gas combined cycles (NGCC) with CCS, increased hydrogen production, and reduced deployment of small modular nuclear reactors (SMRs). It is assumed that hydrogen production will be from excess onshore wind generation, whereby the latter is converted to hydrogen with an efficiency of 80% (i.e., "green" hydrogen). Table 6 reports the assumed percentages of grid generation contributed by each technology up to 2050, with 5-year resolution, based on the FES 2020 "leading the way" scenario (FES, 2020). Grid storage demand is assumed to be met by an evolving combination of pumped-hydro storage (PHS), lithium-ion batteries (LIBs), vehicle-to-grid (V2G), and abiotic compressed air energy storage (ACASE).

Generation technology	2020	2025	2030	2035	2040	2045	2050
Biomass	9.6%	8.8%	1.2%	0.5%	0.4%	0.3%	0.3%
Waste	3.4%	4.3%	3.4%	2.4%	1.6%	1.0%	0.5%
Coal	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
NGCC	31.4%	6.3%	4.9%	1.2%	0.1%	0.0%	0.0%
Biomass + CCS	0.0%	0.0%	5.3%	10.5%	10.4%	11.4%	10.5%
Nuclear (PWR - current)	21.9%	16.1%	9.2%	7.7%	7.0%	5.8%	5.3%
Nuclear (SMR - future)	0.0%	0.0%	0.0%	0.9%	0.7%	0.4%	0.3%
Hydro	2.1%	2.2%	1.8%	1.5%	1.4%	1.2%	1.1%
Marine (tidal)	2.3%	3.7%	3.5%	2.2%	1.5%	1.1%	1.0%
Wind (onshore) [incl. for H2 prod.]	11.7%	18.8%	17.2%	16.9%	17.9%	16.5%	16.4%
Wind (offshore)	13.2%	33.5%	45.8%	47.0%	49.0%	52.4%	54.3%
Photovoltaic	4.4%	6.3%	7.6%	9.2%	10.1%	9.9%	10.4%

Table 6. Expected UK grid mix composition up to 2050, with 5-year resolution, in terms of total electricity generated. NGCC = natural gas combined cycles; Biomass + CCS = Biomass combined cycles plus carbon capture and storage; PWR = pressure water reactors; SMR = small modular reactors.

From a modelling perspective, the electricity grid mix model is fundamentally attributional. However, it too includes an element that can be characterised as consequential, since the way in which the total projected LIB storage requirement is fulfilled in the model is by preferentially sending as many EoL EV batteries to second life as possible (more detail on this aspect of the model is provided in Kamran et al. (2021)). This has only a small effect on the overall environmental impacts of each kWh of electricity supplied by the grid; however, as discussed in Section 2.3, it also has an effect on the calculated impacts for the manufacturing of new EV batteries, since it affects the associated input shares of primary/recycled metals.

2.6. Life Cycle Impact Assessment (LCIA)

In addition to global warming potential (GWP), a range of other sector-relevant impact categories were also considered, namely: photochemical ozone creation potential (POCP), abiotic depletion potential (ADP), and human toxicity potential (HTP). The purpose of including these additional impact categories was to move beyond simple “carbon accounting”, and to be able to identify and discuss any potential trade-offs and impact shifting arising from the planned transition to electrical mobility. GWP, POCP and ADP were calculated using the widely adopted CML method (University of Leiden, 2021).

The GWP indicator was calculated using IPCC-derived characterisation factors over a standard 100-year time horizon, and excluding the contribution of biogenic carbon emissions (thereby assuming that all CO_2 emissions arising from the combustion of woody and other biomass feedstocks are offset by the capture of the same amount of CO_2 during the growth phase of sustainably managed short-rotation plantations).

Photochemical ozone creation is a type of impact that takes place at the local/regional scale. Its effects on human health are more severe when the emissions take place in densely populated areas (such as in cities), as opposed to remote or primarily industrial locations (such as, for instance, at mineral mining sites or at metal processing and battery manufacturing facilities). Consequently, when calculating the POCP results, the decision was made to restrict the boundary of the analysis to the vehicle use phase only, in order to provide a clearer indication of the evolution of the LDV fleet’s impact in terms of local air pollution and potential for photo-smog formation in urban centres and along busy motorways in the UK.

ADP is based on estimates of ultimate reserves and current extraction rates. This indicator was calculated excluding the contributions of all energy inputs (such as fossil fuels and uranium), in order to focus on the depletion of non-energy resources, and specifically metals. It should be noted that abiotic depletion is an impact category that is still frequently the object of methodological discussion, and alternative approaches exist to the quantification of the associated impact, often reflecting differences in problem definition [Guinée et al., 2002; Schulze et al., 2020a,b; Sonderegger et al., 2020; Berger et al., 2020]. Additionally, ADP’s specific dependence on the estimate of ultimate reserves makes it susceptible to obsolescence, especially when using this indicator to assess a depletion-related impact taking place several decades into the future [van Oers et al., 2019].

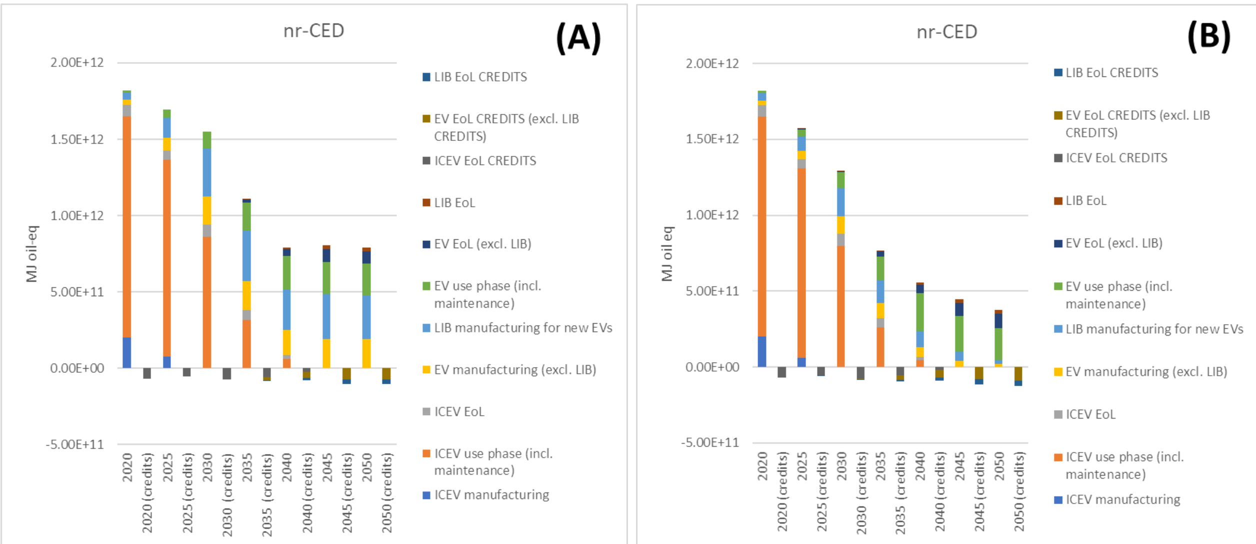
HTP was calculated using the USETox method (Hauschild et al., 2008; Rosenbaum et al., 2008), which is widely reputed to be the most sophisticated, up-to-date and accurate method to estimate potential toxicity impacts in LCA (UNEP-SETAC, 2021). Separate impact indicators are calculated for potential cancer and non-cancer effects on human health, and the results are expressed in terms of dimensionless “comparative toxic units” (CTUs). Even so, HTP results are still inevitably affected by a larger margin of uncertainty than those for all other impact categories, due to the intrinsic methodological difficulty of comparing and combining into a single indicator the individual toxicity potentials of a wide and diverse range of organic and inorganic emissions. The uncertainty is especially large in the case of metal emissions.

Finally, although not an LCIA indicator in the strict methodological sense, the non-renewable cumulative energy demand (nr-CED) is also reported, thereby providing an indication of the total non-renewable primary energy directly and indirectly harvested from the environment, expressed in units of crude oil equivalent [Frischknecht et al., 2015].

3. Results and Discussion

Figures 2 and 3 respectively illustrate the projected UK LDV fleet’s overall demand for non-renewable primary energy and GHG emissions. As expected, the trends for these two indicators are very similar, with a clear overall reduction of impact as EVs gradually displace ICEVs over time. The ICEV use phase initially represents the major contribution to CED, mainly due to the consumption of petrol and diesel; this then gradually decreases over time, as the number of ICEVs dwindles. In both scenarios, the total number of EVs surpasses the number of ICEVs on the roads after the year 2030. With the rise in EVs, there is a clear steady increase in nr-CED and GWP due to the EV manufacturing phase and use phase for the baseline scenario regardless of the UK grid mix evolution. This is mainly due to the increase in demand for number of vehicles. The same trends can be seen for LIB manufacturing (it is worth reminding that in the model, it is assumed that the batteries are manufactured outside the UK, where the grid mix is taken as a background process and is assumed to remain static over time). On a per-unit basis, the energy demand and carbon emissions associated with the manufacturing of new EVs (and specifically the battery packs) are significantly higher than those for ICEVs. However, this is overcompensated by the positive effect of low-carbon electricity replacing petrol and diesel as the energy carrier used to power the vehicles during their use phase, coupled with the intrinsically higher tank-to-wheel efficiency of electric *vs*. internal combustion power trains (typically, approximately 85% vs. 25%).

The improvement in nr-CED and GWP is even larger in the "TaaS" scenario because of an overall reduction in the total fleet size, which leads to an overall net impact reduction in nr-CED of 36% and 74% for years 2035 and 2050 respectively, and an overall net impact reduction in GWP of 39% and 84% for years 2035 and 2050 respectively. Relative to the "baseline" scenario, there is thus a clear decrease in the impacts of the EV and LIB manufacturing stages, despite the fact, that the shared mobility vehicles are replaced every 3 years. A further differentiator between the two scenarios is that in the "TaaS" scenario the batteries become available for recycling much earlier on.



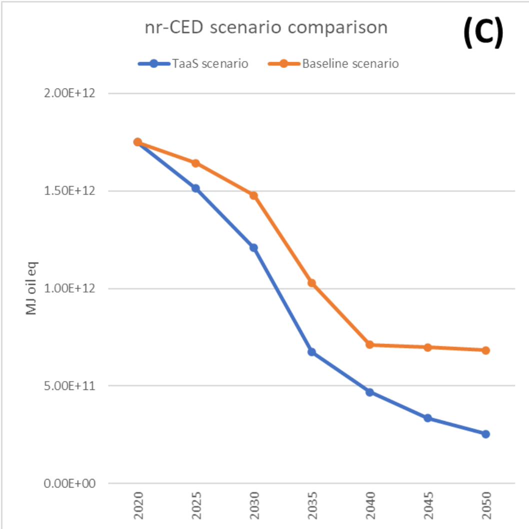
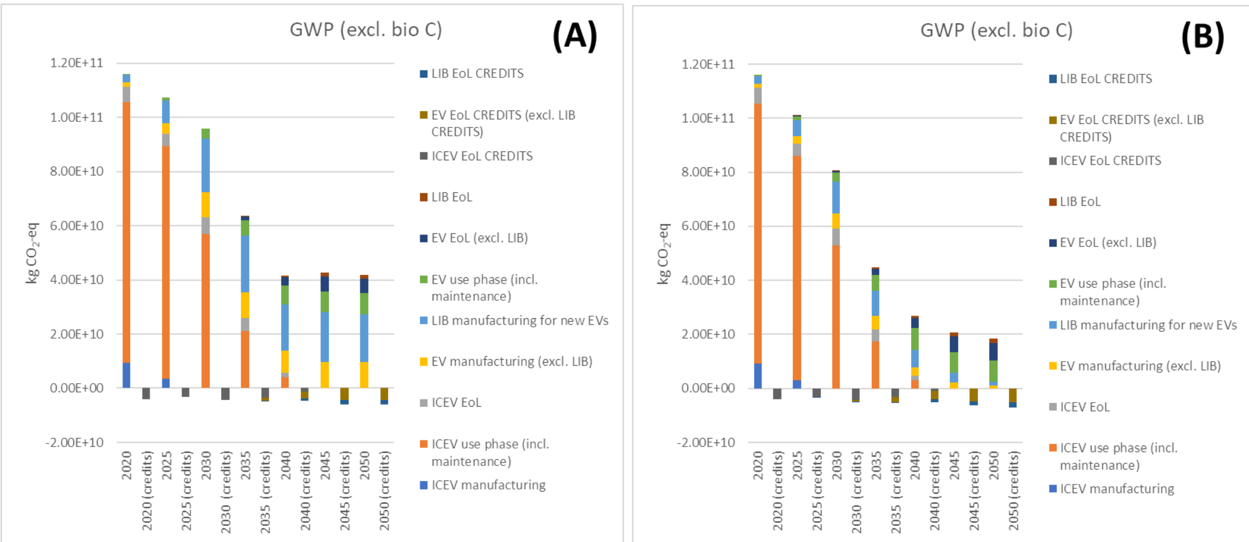


Figure 2 - Non-renewable Cumulative Energy Demand (nr-CED) of the UK light-duty vehicle fleet. (A) = "Baseline" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (B) = "TaaS" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (C) comparison of overall net impact (= total impact – credit) for "Baseline" vs. "Taas" scenarios.



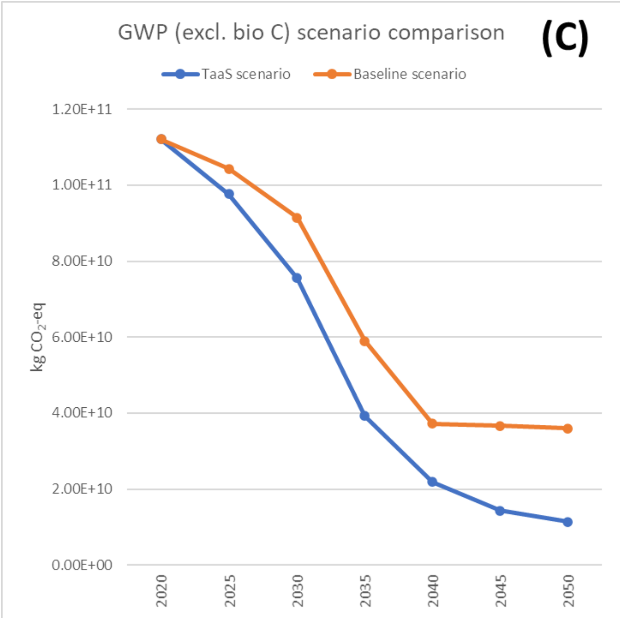


Figure 3 – Global warming potential (GWP, excluding biogenic C) of the UK light-duty vehicle fleet. (A) = “Baseline” scenario, with total impact broken down by system component, and end-of-life credits reported separately; (B) = “TaaS” scenario, with total impact broken down by system component, and end-of-life credits reported separately; (C) comparison of overall net impact (= total impact – credit) for “Baseline” vs. “Taas” scenarios.

Moving on to the POCP results for the use phase of the LVD fleet, a greater than 85% reduction in impact is found in both scenarios. This is a strong indication that the phasing out of internal combustion engines is a clear benefit in terms of local air pollution, and completely dominates over all other impacts resulting from non-tailpipe emissions, including those due to the provision of electricity to EVs. It should be noted that the POCP indicator only captures the impacts arising from the photochemical oxidation of gaseous emissions (leading to secondary respiratory irritants such as ozone and peroxy-acyl nitrates). Thus, it may fail to highlight the contribution to local air pollution caused by particulate matter (PM) emissions, both from vehicle tailpipes and from tyre, brake-pad and tarmac wear (Emissions Analytics, 2020). However, diesel engines are known to be a major source of PMs, and therefore phasing them out may be expected to be beneficial in this regard, too. Also, the regenerative braking systems on EVs suggest that brake-pad emissions are going to be reduced (Society of Motor Manufacturers and Traders, 2020).

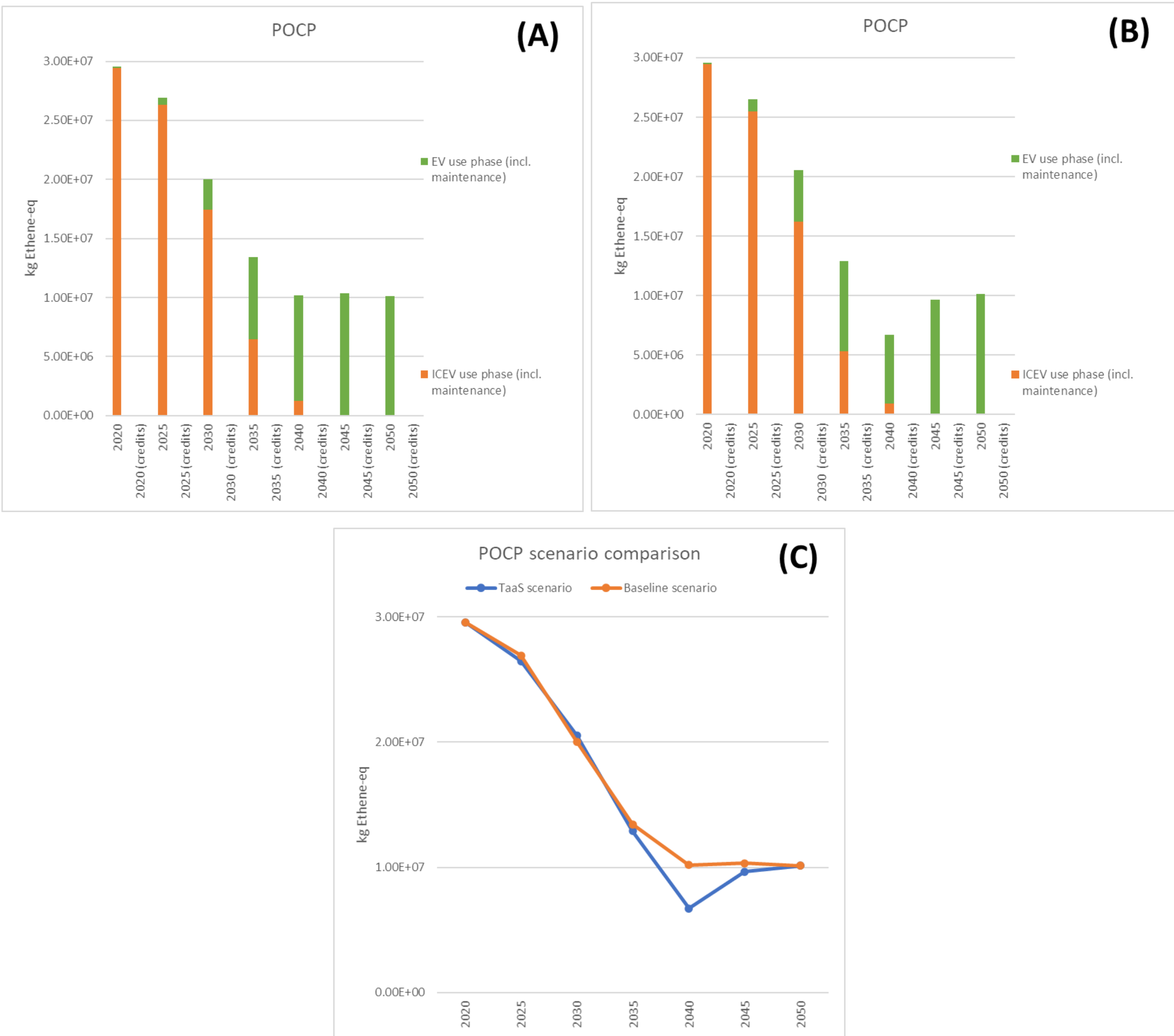


Figure 4 – Photochemical ozone creation potential (POCP) of the UK light-duty vehicle fleet, use phase only. (A) = "Baseline" scenario, with total impact broken down by vehicle type (ICEV and EV); (B) = "TaaS" scenario, with total impact broken down by vehicle type (ICEV and EV); (C) comparison of overall impact (= ICEVs + EVs) for "Baseline" vs. "Taas" scenarios.

Figure 5 shows that in the "Baseline" scenario, the growing demand for metals for new EV power trains (including for the motor as well as for the LIBs) causes a progressive increase in the net ADP by 72% in 2050 compared to year 2020. Notably, this impact indicator remains high even after the positive role of the EoL recycling of LIBs fully kicks in after 2035, with significantly reduced net demand for primary Li, Co, Ni, and Mn. This is explained by the fact that the demand for Cu was found to be a primary driver of abiotic resource depletion. These findings point to a potentially critical trade-off between reduced energy and climate impacts on one side, and increased resource depletion and toxicity impacts on the other side. Conversely, in the "TaaS" scenario there is an overall net decrease in ADP, by 52% and 101% for years 2035 and 2050 respectively, when compared to the "basline" scenario. Hence, results for the "TaaS" scenario show that the impact trade-off could be resolved by the widespread adoption of shared mobility, whereby a smaller overall EV fleet could be sufficient to satisfy the growing demand for personal mobility. In so doing, the total net demand for Cu and other critical metals, and the associated depletion impacts, could be kept in check, leading to an overall reduction, rather than increase, in total ADP and HTP over the next three decades.

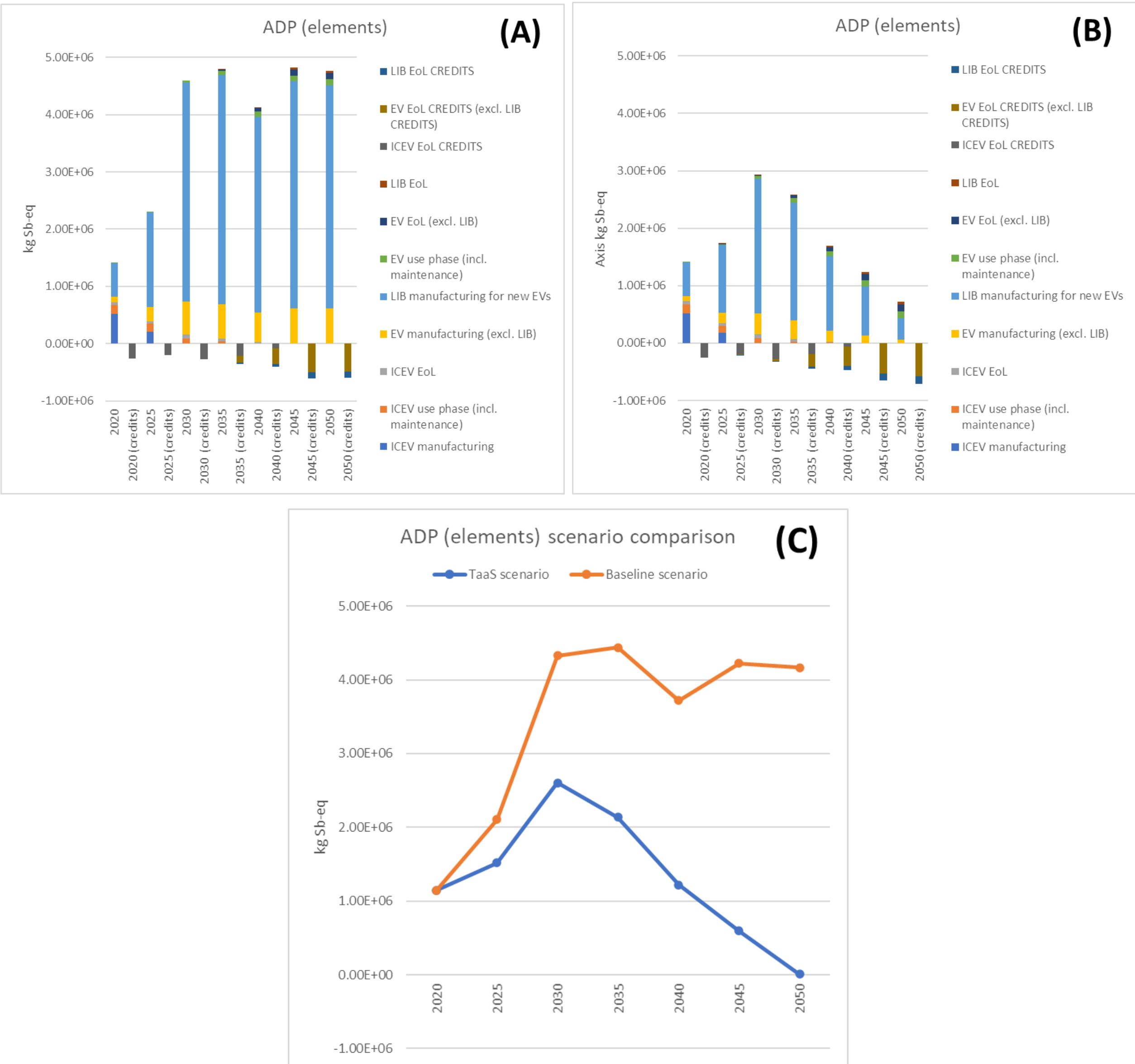


Figure 5 – Abiotic depletion potential (ADP, elements) of the UK light-duty vehicle fleet. (A) = "Baseline" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (B) = "TaaS" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (C) comparison of overall net impact (= total impact – credit) for "Baseline" vs. "Taas" scenarios.

Figures 6 and 7 paint a qualitatively similar story, but with a few significant differences. The human toxicity impacts are initially largely due to the manufacturing and decommissioning of ICEVs. Then, after 2030, in the "baseline" scenario, as the number of EVs start to increase significantly, the main share of the toxicity impacts is shifted to the manufacturing of EVs and LIBs, which leads to an overall net increase in HTP (cancer) and HTP (non-cancer) from year 2020 to year 2050 by 31% and 53% respectively. In particular, the associated metal supply chains (among which once again Cu plays a prominent role) are responsible for greater toxicological impacts than the emissions from the generation of the electricity required during the vehicles' use phase.

In the "TaaS" scenario, as the demand for new EVs is reduced thanks to the increase in shared mobility services, the major contributor to HTP becomes the EoL phase of EVs.Also, since in this scenario there is an oversupply of recycled LIB metals (Li, Co, Ni and Mn) after the year 2035, this results in not only a reduction in the impact associated with the extraction of raw materials for new battery manufacturing, but also in additional net HTP credits, which leads to an overall net reduction in HTP (cancer) and HTP (non-cancer) when compared to the "baseline scenario" by 93% and 122% in years 2035 and 2050 respectively.

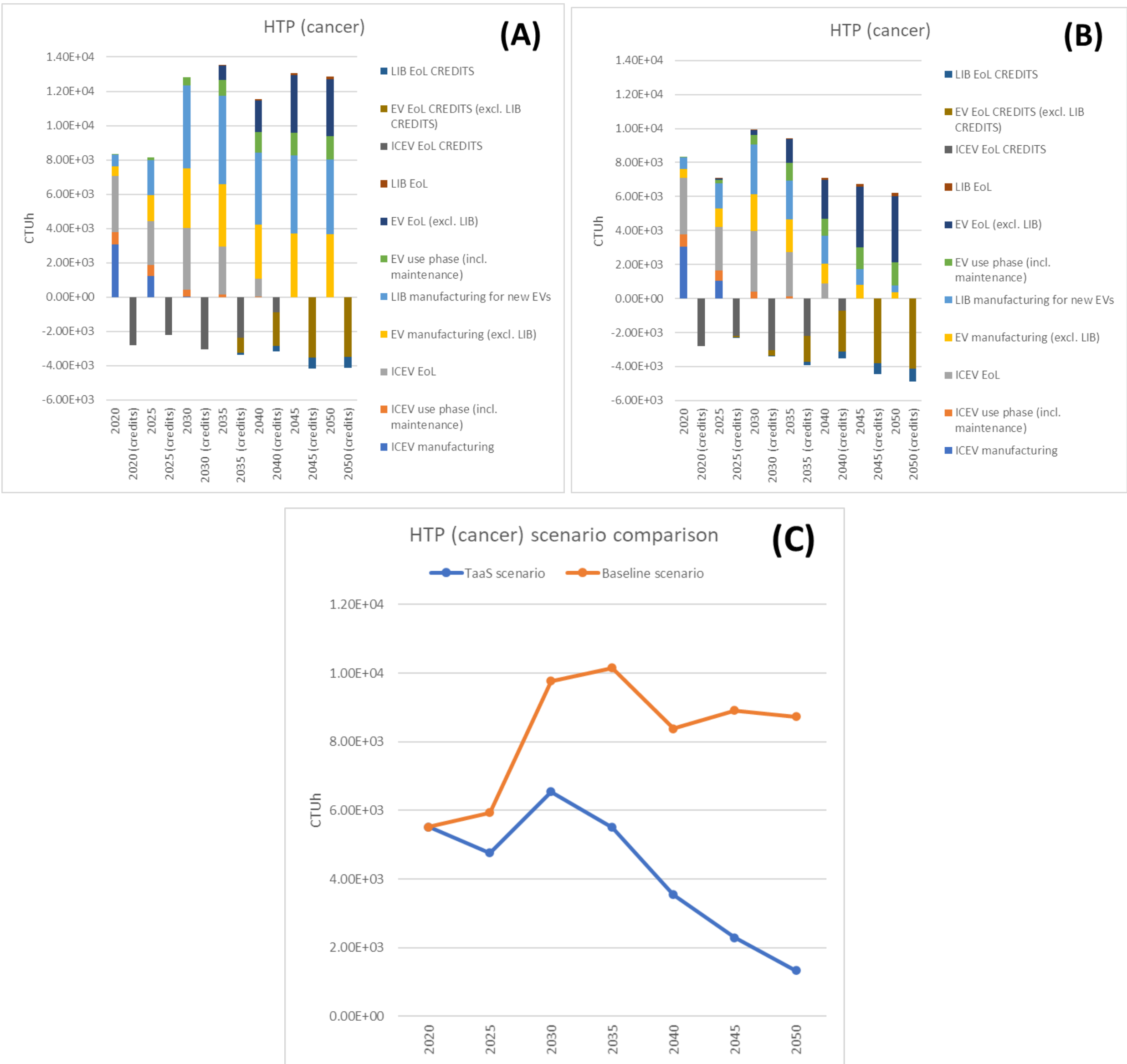


Figure 6 – Human toxicity potential, cancer (HTP, cancer) of the UK light-duty vehicle fleet. (A) = "Baseline" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (B) = "TaaS" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (C) comparison of overall net impact (= total impact – credit) for "Baseline" vs. "Taas" scenarios.

Figure 7 – Human toxicity potential, non-cancer (HTP, non-cancer) of the UK light-duty vehicle fleet. (A) = "Baseline" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (B) = "TaaS" scenario, with total impact broken down by system component, and end-of-life credits reported separately; (C) comparison of overall net impact (= total impact – credit) for "Baseline" vs. "Taas" scenarios.

4. Conclusions

The holistic prospective life cycle assessment of the future of the whole light duty vehicle fleet in the UK has shown beyond reasonable doubt that the ongoing shift from internal combustion engines to more efficient battery electric power trains will be markedly beneficial. This is in terms of: (i) reducing the overall demand for non-renewable primary energy sources (with welcome implications for the UK in terms of energy sovereignty), (ii) curbing greenhouse gas emissions (in large part thanks to a parallel effort to aggressively decarbonize the grid mix), and (iii) slashing local air pollution in cities and along busy roads.

The immediate consequence of introducing new electric power trains at scale is the sharp rise in demand for metals such as Cu, Li, Co, Ni and Mn. Even with large-scale implementation of end-of-life battery take-back and recycling, this risks counterbalancing the aforementioned positive effects with significant worsening rates of material resource depletion and human toxicity (the latter primarily occurring overseas along the metal supply chains, which raises issues of environmental justice). However, a parallel scenario analysis considered the possible widespread adoption of shared mobility schemes, whereby individual cars are used more efficiently, and ultimately a smaller number of vehicles are required to meet the same overall demand for personal mobility. Results for this latter scenario indicate that shared mobility would be an effective strategy to reverse these worrisome trends in resource depletion and human toxicity.

Acknowledgements

The financial support for this work provided by the Faraday Institution under project 'ReLIB' [grant number FIRG005] is gratefully acknowledged.

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