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Citation for the original published paper (version of record):
Nordelöf, A., Romare, M., Tivander, J. (2019). Life cycle assessment of city buses powered by electricity, hydrogenated vegetable oil or diesel. Transportation Research Part D: Transport and Environment, 75: 211-222. http://dx.doi.org/10.1016/j.trd.2019.08.019

Transportation Research Part D 75 (2019) 211–222

Contents lists available at ScienceDirect

Transportation Research Part D

journal homepage: www.elsevier.com/locate/trd

# Life cycle assessment of city buses powered by electricity, hydrogenated vegetable oil or diesel

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ARTICLE INFO

*Keywords:*
LCA
City
Bus
Battery
Electric
Vehicle
Plug-in
HVO
Diesel
Public transport

ABSTRACT

This study explores life cycle environmental impacts of city buses, depending on the: (1) degree of electrification; (2) electricity supply mix, for chargeable options; and (3) choice of diesel or hydrogenated vegetable oil (HVO), a biodiesel, for options with combustion engine. It is a case study, which uses industry data to investigate the impact on climate change, a key driver for electrification, and a wider set of impacts, for average operation in Sweden, the European Union and the United States of America. The results show that non-chargeable hybrid electric vehicles provide clear climate change mitigation potential compared to conventional buses, regardless of the available fuel being diesel or HVO. When fueling with HVO, plug-in hybrid and all-electric buses provide further benefits for grid intensities below 200 g $CO_2$ eq./kWh. For diesel, the all-electric option is preferable up to 750 g $CO_2$ eq./kWh. This is the case despite batteries and other electric powertrain parts causing an increase of $CO_2$ emissions from vehicle production. However, material processing to make common parts, i.e. chassis, frame and body, dominates the production load for all models. Consequently, city buses differ from passenger cars, where the battery packs play a larger role. In regard to other airborne pollutants, the all-electric bus has the best potential to reduce impacts overall, but the results depend on the amount of fossil fuels and combustion processes in the electricity production. For toxic emissions and resource use, the extraction of metals and fossil fuels calls for attention.

## 1. Introduction

### 1.1. Background

Electromobility is gradually becoming more and more important for road transportation, in the interest of climate change mitigation, reduced local air pollution and decreased dependence on foreign oil. This trend has created focus on the environmental impacts of electrifying vehicles, and triggered a large number of life cycle assessment (LCA) studies to investigate electrically propelled passenger cars (Nordelöf et al., 2014). In contrast, heavy road vehicles, for example public transportation buses operated in cities, have until recently drawn less attention in LCA literature (Nordelöf et al., 2014), despite being highly applicable for electrification and possessing a suitable business case (Sushandoyo and Magnusson, 2014). However, a growing number of bus studies focus on different fuel types, including hydrogen in fuel cells and various biofuels, as well as the production of electricity for charging, by assessing the energy carrier life cycle from well-to-wheels (WTW) (Ercan et al., 2016; Frey et al., 2007; Kliucininkas et al., 2012;

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https://doi.org/10.1016/j.trd.2019.08.019

Available online 10 September 2019

Lajunen and Lipman, 2016; Ma et al., 2017; Ou et al., 2010; Xu et al., 2015; Zhou et al., 2016). Several other studies apply more complete LCA, i.e. they cover the life cycle of the vehicle equipment and the WTW stage, e.g. comparing electrically chargeable buses with those driven by diesel or hydrogen fuel cells (Cooney et al., 2013; García Sánchez et al., 2013; Harris et al., 2018; Ribau et al., 2014).

One study, by Ercan and Tatari (2015), investigates how environmental impacts differ between a conventional bus running on diesel or biodiesel compared to a range of other fuel options, including a non-chargeable diesel hybrid and a chargeable battery electric bus. Their study combines national and sector economic input-output data with manufacturing process information specifically for biodiesel and traction batteries, using model data from the Argonne National Laboratory (2018), in a setup referred to as a hybrid-LCA. The results indicate clear climate impact benefits from both hybridization and full electrification of buses, but also point to a strong dependence on the underlying electricity grid mix (Ercan and Tatari, 2015), this being in line with earlier general results for chargeable vehicles (Nordelöf et al., 2014).

Xylia et al. (2019) also study the use of different types of biodiesel in bus fleets and compare these with the operation of chargeable all-electric buses, but exclude hybrid and plug-in hybrid vehicles. Scope covers the WTW life cycle of the energy carriers but only with the battery for the equipment life cycle (see for example Nordelöf et al. (2014) for explanation of terms), under the assumption that these two parts are the dominating contributors to the total environmental impacts of the bus life cycle. An optimization model is then used to locate suitable charging points and to estimate linked carbon dioxide emissions for several implementation scenarios (Xylia et al., 2019).

However, many relevant questions coupled to city bus electromobility remain to be explored. For example, until now no studies in the research literature examine the combination of using biofuel in vehicles with different degrees of electrification (including plug-in hybrid vehicles) for the same city bus application or take a full spectrum of impacts categories into account, for example toxic emissions and metal depletion caused by resource extraction. Several previous articles point out the general lack of data for bus design, operation and manufacturing (Cooney et al., 2013; Ercan and Tatari, 2015; García Sánchez et al., 2013). Likewise, Harris et al. (2018) concludes that there is a need for additional process-based LCA studies (i.e. not using the hybrid-LCA setup) covering the full vehicle equipment life cycle, while remarking these are missing partly due to difficulties in acquiring component composition and manufacturing data.

### 1.2. The aim and content of the article

In response to existing gaps in the research field, this paper's aim is to report a cohesive LCA case study that uses industry data to investigate the environmental impacts of different city buses of the same model series for public transportation in cities. These bus options differ depending on (1) the degree of electrification, (2) the electricity supply mix for charging, and (3) the choice of liquid fuel for the alternative drive mode – diesel or hydrogenated vegetable oil (HVO), a type of biodiesel attracting growing interest for public transportation applications and promising environmental performance (Arvidsson et al., 2011; Xylia and Silveira, 2017). The study is based on data from an existing bus line (no. 55) in Gothenburg, Sweden, with vehicle content, manufacturing, maintenance and operation data provided by Volvo Bus Corporation and the local bus operator, Keolis Sverige AB.

Following the introduction, this paper presents: the details of the study's goal and scope definition; a description of the inventory analysis; impact assessment results and a discussion on robustness; and conclusions phrased as answers to the research questions of the study. A supplementary report (SR) is coupled to this article, presenting more detailed technical descriptions, schematics and tables linked to the inventory analysis, as well as figures complementary to the presentation of the results and different types of sensitivity analysis. All figures or tables denoted with an "S" refer to the SR.

## 2. Methods, goal and scope

### 2.1. Goal definition

The goal of the presented study is to assess the life cycle environmental impacts of city buses used for public transportation. Three main questions were researched:

(1) How do the environmental impacts of city buses depend on their degree of electrification?
(2) How do the environmental impacts of electrically chargeable city buses depend on the production of the electricity used for charging?
(3) How do the environmental impacts of electrical operation compare with combustion of HVO instead of diesel?

The overall aim of the study is to create a knowledge base intended for use in the procurement of vehicles, electricity and fuels and strategic planning of public transportation, and for further development of bus powertrain technology. The main target group consists of local and regional public transportation agencies and companies.

### 2.2. Objects of study and functional unit

Four different vehicles were compared in the study:

(1) all electric powertrain; charging only; referred to as the “electric” bus
(2) plug-in hybrid electric (vehicle) powertrain; charging combined with refueling; “PHEV” bus
(3) hybrid electric (vehicle) powertrain; refueling only; “HEV” bus
(4) diesel engine powertrain; refueling only; “conventional” bus

Data for Volvo’s 7900 model series was used (Volvo Bus Corporation, 2011, 2016a,b,c). All refueling (buses 2–4) considers two fuel alternatives – diesel or HVO – with equivalent emissions and performance when used in the internal combustion engine (of Euro 6 emission standard) (Volvo Lastvagnar, 2013). The chargeable buses (1–2) are combined with different electricity production average grid mixes: for Sweden, the European Union (EU) and the United States of America (USA). All in all, the different bus versions, the fuel alternatives and the selected charging electricity mixes, combine into 13 examined options.

The functional unit of the study is person × km, corresponding to the city bus task of transporting people on a specific line. There are differences in the capacity for maximum number of passengers between the bus models due to deviating component packaging and total mass (see Section A.1 of the SR), but the main assumption throughout the study is that this variation makes no difference to the ability to fulfil the designated transportation work. Accordingly, e.g. fuel- and electricity consumption for each bus was gathered to calculate average operation on the line, expressed per passenger. Further, the effect of maximum ridership for each bus type was examined in a sensitivity analysis (see Section D.2.2 of the SR). Energy usage includes the ability to maintain an equivalent on-board temperature year-round, i.e. the electric and PHEV options use dedicated fuel heaters during winter time, while the conventional and HEV versions make use of excess heat from the engine. In the electric buses a small amount of HVO is used, and in the PHEV buses the same fuel as used for propulsion is used for heating.

### 2.3. Technical, temporal and geographical system boundaries

The LCA study is attributional and comparative, i.e. it models and evaluates the potential impacts of operating different technical options in an existing system. Accordingly, the study is based on average data, unless specified otherwise. Foreground bus data, including vehicle design and composition, component and vehicle manufacturing, driving, and maintenance, was provided by the Volvo Bus Corporation. Other stages in the life cycle of the different bus options include the extraction and production of materials, generation of electricity, production of fuel, and treatment of materials consequent to ”End-of-Life” (EoL). These stages constitute the background system for which general inventory data from Ecoinvent v.3.3 was used (Wernet et al., 2016), complemented with updated information about the production of HVO from different feedstock (Källmén et al., 2019). This version of the Ecoinvent database was released in 2016. In terms of geography, the background data generally represents global averages, or European averages where global averages were not available (see Table S3 in Section C.1 of the SR). Specific electricity grid mixes were linked to the sites of the foreground manufacturing data, as described briefly in the following paragraph, and with more details in Section 3.2 and chapter B of the SR.

For the manufacture of subsystems and components, and assembly of complete buses, the Volvo foreground data refers to specific production sites in Europe. Electric motor and battery manufacturing were assumed to take place at suppliers in Germany and the USA respectively. The bus operation stage was assessed more broadly for geographical representation. For climate change, the full range of carbon intensity in electricity production was examined. Average grid mixes for the EU and the USA were assessed for all selected impact categories. Additionally, the Swedish electricity grid mix provides an example of charging using a very low share of fossil energy.

Other operating data is based on conditions in Sweden. Data for the amount of electricity used for charging, fuel consumption and number of passengers is based on an existing high-traffic, inner city bus line. Furthermore, the feedstock mix used in the production of HVO was based on average Swedish data for fuels sold during 2016 (Swedish Energy Agency, 2017), see Table S12 in Section C.3 of the SR. In 2016, as a result of a rapid expansion in the use of HVO, Sweden consumed more than 20% of global production, this in turn primarily deriving from other European countries and the USA (Grenea, 2017; REN21, 2017; Swedish Energy Agency, 2017), making it the best available estimate of the global average in terms of HVO feedstock. For this reason, we argue that this operating data is representative regarding the use of HVO in public transportation in cities in both the EU and the USA.

Finally, the EoL was modeled with reference to the disposal of vehicles and recycling of materials in Sweden. The life span of buses in operation is 12 years for all bus options, according to Volvo. Transports of materials and subcomponents are included in the background Ecoinvent data and were added to the foreground system by means of specific modeling of the distances and transport modes corresponding to the routes between supplier sites and the bus assembly facility (see Table S9 in Section C.2 of the SR).

### 2.4. Selection of life cycle impact assessment methods

Nine impact categories of high relevance to environmental problems relating to traffic and vehicles were included in the life cycle impact assessment (LCIA): climate change (GWP 100), acidification, eutrophication (marine end compartment), human toxicity (carcinogenic and non-carcinogenic), particulate matter, photochemical ozone formation, and abiotic resource use (two methods for mineral and non-renewable elements). Eight of these categories were assessed with midpoint characterization methods recommended by ILCD (International Reference Life Cycle Data System) (JRC-IES, 2011), with characterization factors according to JRC-IES (2013), with the exception of abiotic resource depletion (ADP) for which updated factors by CML (CML, 2016; Guinée et al., 2002) were used. The analysis of resource use includes EPS (Environmental Priority Strategies) (Steen, 2015), this being a broader endpoint LCIA method, and methodologically complementary to ADP.

Land use was not included among the selected LCIA methods, despite HVO being a biofuel. The reason being that biological waste is the primary feedstock and, unlike cultivation the of crops, it is not directly linked to land use. Consequently, suitable inventory data for land use was lacking.

## 3. Inventory analysis

### 3.1. Vehicle specifications and operation data

The data for material composition of the different bus options derives from existing bus models in Volvo's 7900 series. All bus options studied were modeled on the same chassis, frame and body, using data from the fully electric bus version (which has the largest passenger capacity), whereas data for powertrains (including all subsystems) and batteries was gathered for each specific variant. Mass data was available for all parts, although some component material compositions were lacking. This included printed circuit boards (PCBs) in the various electronics and permanent magnets of the electric motors, where generic data from Nordelöf et al. (2017, 2018) was used. All internal combustion engines meet Euro 6 emission standards and all high voltage batteries are of lithium iron phosphate (LFP) type. For more details, see Section A.1 and chapters B and C of the SR. Composition and manufacturing data for the batteries was provided by Volvo but originate from Volvo suppliers. These primary datasets were found to be in line with other datasets for non-giga-scale battery production (Ellingsen et al., 2017).

More thorough technical information about the buses can be found in Section A.1 of the SR. Fig. S1 illustrates mass distribution between major vehicle sub-segments for all model versions and Fig. S2 shows the distribution of different materials in the vehicles. Table S1 specifies power and electrical storage capacities, and total masses of all bus options.

Minor shifts of the interior design between options, linked to the variation in maximum passenger capacity (see Section 2.2), were disregarded. The average passenger count per kilometer was calculated to be 16 based on bus line statistics (Keolis, 2017; Lund, 2017). The line is 7.6 km long and has 11 stops in each direction, with charging stations at both ends. The buses run at an average speed of 18 km/h, for 65,000 km per year. Thus, the total transport work over a total bus life equals 12.5 million person-km. Further details about the bus line is provided in Section A.2 of the SR.

The amount of energy required by each bus option when operating on the line, i.e. the average amount of charging electricity and the fuel consumption, was provided by Volvo, based on a combination of direct measurements and calculations (for the options not currently running on the line). This includes the energy necessary to maintain the same temperature in the passenger compartment in all weather conditions. The diesel fuel includes a low blend of 7% biodiesel (rape methyl ester). The study also takes urea for exhaust after-treatment and energy losses in the charger stations into account, but not the chargers in terms of equipment. Thus, the data for the tank-to-wheels (TTW) stage, i.e. "charger-to-wheels" for the chargeable options, is case-study specific. For upstream production and distribution of electricity, diesel and HVO – referred to as the well-to-tank (WTT) part of the WTW life cycle for these energy carriers – generic background data was used, as described in the previous methodology chapter.

For regulated tailpipe exhaust emissions, the limits of EU emission standards (Euro 6) were used for both HVO and diesel. This implies a worst-case approach in both cases, taken to avoid underestimating the exhaust emissions by using certification data that typically is very low as it is collected under ideal laboratory conditions. In fact, and similarly for both fuels, the difference between Volvo certification data and the Euro 6 emissions limits is specifically large for carbon monoxide (CO), hydrocarbons and particles, but also clearly different for nitrogen oxides ($NO_x$). This is an indication that real-world tailpipe emissions might be lower than the study reports in these cases. The same certification data also specifies that HVO may have lower emissions of sulfur dioxide and hydrocarbons than diesel, but higher emissions of CO.

Section B.2 of the SR describes WTT and TTW data in more detail, for example the underlying composition of the HVO. Table S2 reports fuel and electricity use for all options.

### 3.2. Life cycle inventory for vehicle equipment from raw materials to End-of-Life

In analogy with the WTW life cycle data for the energy carriers, the inventory for the life cycle of the vehicle equipment can largely be divided into case-study specific data from Volvo for the foreground system, and more general database records for the background system. The latter includes raw material extraction, material processing, EoL treatment procedures and energy supply for various processes throughout the life cycles, e.g. from electricity or natural gas.

The foreground system covers component and bus manufacturing, maintenance and use of the vehicle (see Section 3.1). The data for material inputs, energy use and waste from Volvo's production facilities is from 2016. However, for production steps where primary data was not judged to provide sufficient detail, complementary sources were used: for casting, forging, curing and forming procedures (Nordelöf and Tillman, 2018; Wernet et al., 2016); for the manufacturing of electronics (Nordelöf, 2018; Nordelöf and Alatalo, 2017); and for the production of magnets and electric motors (Nordelöf et al., 2019; Nordelöf et al., 2017; Nordelöf and Tillman, 2018). Likewise, data for lithium-ion battery production was obtained from Volvo's supplier and reported as stemming from 2015. Recycling of production waste was taken into account for steel, aluminum and copper.

All maintenance procedures are based on Volvo's documentation for service and preventive repairs, including replacements of the high-voltage batteries. For the electric bus, all battery packs are replaced once, and for HEV and PHEV options they are replaced twice during the 12-year life.

EoL modeling was conducted in accordance with the LCA cut-off approach (Atherton, 2007; Frischknecht, 2010), and covers all waste handling procedures required to recover three main material fractions: aluminum, copper, and ferrous metals (iron and steel alloys). Using this method, there is no allocation of burden to the recycled materials (i.e. downstream products). Instead, raw material input to upstream component and vehicle production has a recycled content (carrying no load from its previous use, except for upgrading) (Wernet et al., 2016). However, an exception from this approach was made for energy recovery from incineration of a fraction of miscellaneous materials (including electronics, plastics, rubber, wood and textiles). Some of these resulting emissions were allocated to waste incineration by-products, i.e. by crediting for emissions otherwise generated by electricity from municipal waste incineration and heat for district heating, using average data from Sweden (see Table S13 in Section C.4 of the SR for more details). This modeling choice was made to avoid double counting of emissions from energy recovery in municipal waste incineration, which otherwise would be included in both the upstream electricity production and the EOL stage.

Other specific recovery processes were modeled for certain parts, such as catalysts and batteries. For the lithium-ion batteries, disassembled cells were modeled to go through a pyrometallurgical recovery process (Dunn et al., 2014). All residual pack materials are treated and sorted in fractions in the same way as all other bus parts. Section B.4 of the SR provides complete details of EoL modeling.

As a last note on inventory modeling, and presentation of results, it is important to mention that the model is set up in such a way that when material losses occur in a specific life cycle stage, then that same life cycle stage carries the load of all upstream processes required to balance the waste flows. For example, the maintenance stage includes the load caused by extraction of both the materials and production required to make new components when older components are replaced and scrapped. Similarly, the “component and full vehicle manufacturing” stage is ascribed the full upstream burden of wasted materials. As a consequence, the “material extraction and processing” step covers only the materials ending up in the bus at the factory gate.

## 4. Results and discussion

### 4.1. Climate change

The total results for greenhouse gas emissions, expressed as $CO_2$ equivalents per person × km, are presented in Fig. 1 for all bus options, as a function of greenhouse gas intensity in the electricity supplied. Next, Fig. 2 shows bus equipment life cycle only (i.e. the four bus types with the operating phase excluded). For the total results (Fig. 1), all electrified and HVO options show clear benefits compared to the conventional diesel bus. A significant impact reduction is already achieved with the HEV diesel. And it can be noted that the conventional HVO bus causes significantly lower emissions than the HEV and PHEV diesel options. More specifically, the WTT stage, i.e. the production of HVO, causes higher emissions than the production of diesel, but in turn emissions are much lower from the vehicle TTW stage, because $CO_2$ emitted from combustion of HVO is biogenic. This is illustrated in Fig. S4 (of the SR), and can be read in Table 1 (which reports the results for all impact categories), showing the contribution to climate impact per life cycle stage for all options when charging takes place by means of the average electricity grid mixes in Sweden, the EU and the USA.

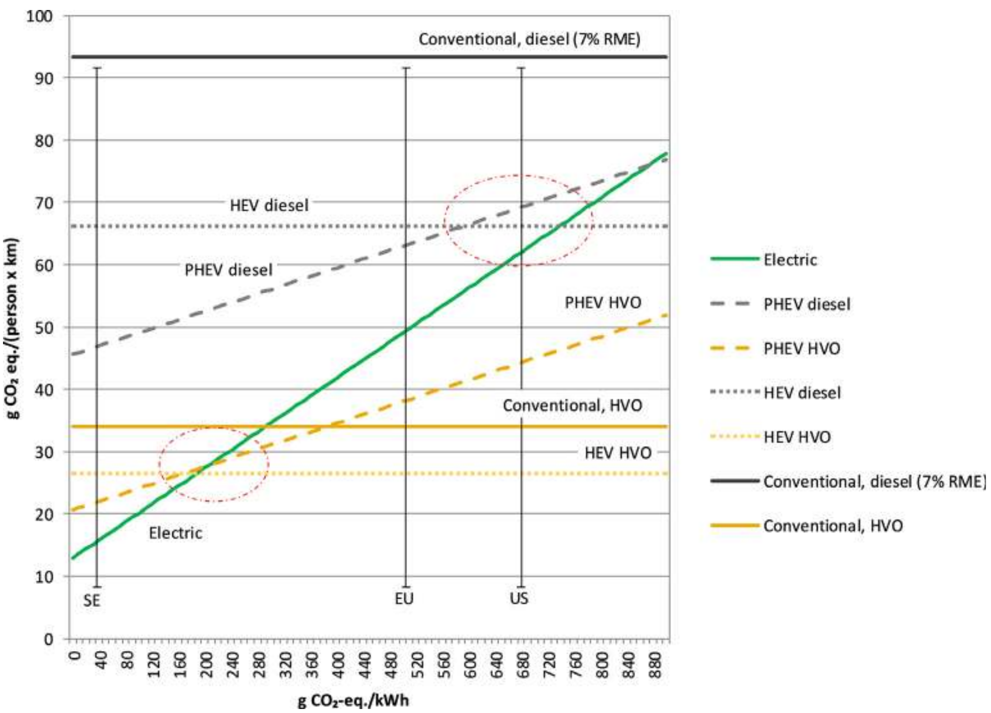


**Fig. 1.** Total results per vehicle option for the climate change impact category, presented as a function of grid mix greenhouse gas intensity (measured in $CO_2$ equivalents) of the electricity supplied for charging.

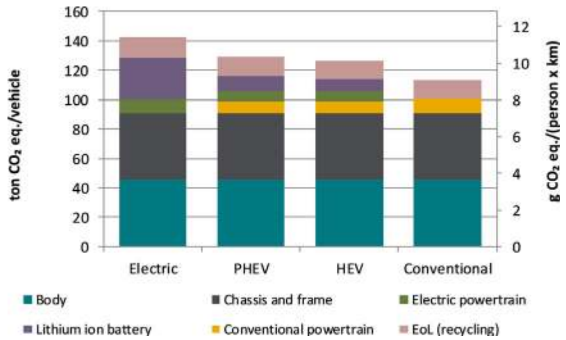


**Fig. 2.** Vehicle life cycle results and contribution to climate change impacts from different parts of the bus (equipment only, i.e. bus operation and WTW life cycle of fuels and electricity for charging are excluded).

Parenthetically, TTW results for HVO illustrate that fuel combustion causes $N_2O$ and $CH_4$ emissions, which translate into noticeable amounts of $CO_2$ equivalents.

In Fig. 1, there are two zones marked in red indicating intersections of special interest. The first is positioned at a grid intensity of around 200 g $CO_2$ eq./kWh. Below this intersection, where the Swedish mix can be found, electric buses come out best among all options. Up to this point there is also a benefit of operating chargeable PHEVs compared to HEVs, even when fueled with HVO. On the other hand, above this value, the HEV HVO option comes out as the lowest $CO_2$ emitting alternative. As an analogy, for grid intensities above 750 g $CO_2$ eq./kWh, i.e. to the right of the upper zone marked in red and higher than the average mix both in the EU and the USA, both chargeable options emit more than the HEV diesel bus. Similarly, ranging up to around 600 g $CO_2$ eq./kWh, total emissions decline with an increasing level of electrification, from the HEV option to the PHEV, and decline further still to the fully electric bus.

Fig. 2 focuses on a different trend, namely that an increasing level of electrification implies higher $CO_2$ emissions embedded in the vehicle life cycle (i.e. when the WTW life cycle of the energy carrier is excluded). Especially the batteries, but also the (other) electric powertrain parts contribute to this increase of emissions compared to the conventional vehicle. However, the chassis, frame and body dominate the vehicle life cycle load for all models. Fig. S4 also adds that emissions relating to these common subparts derive mainly from material processing, particularly steelmaking. Li-ion batteries contribute most in the vehicle maintenance stage owing to scheduled replacements.

Clearly, with an increasing level of electrification, the combined effects of higher emissions from the vehicle life cycle and lowered impact during operation, implies an increased relative importance of the production stages. For example, for a conventional diesel bus the usage phase accounts for about 90% of climate impact. For an all-electric bus charged from the Swedish grid mix, the corresponding figure is 30%, albeit out of a much lower total.

Finally, the EoL stage causes a roughly equal climate load for all vehicle options in terms of absolute numbers. This load relates to energy required for disassembly and separation of all bus parts, and the preparation of materials for recycling. All the energy benefits of recycling are accounted for upstream, in material processing, since secondary materials were modeled to be included in the raw material supply.

### 4.2. Acidification, eutrophication and air pollution

At first glance in Table 1 results for acidification (Fig. S5 of the SR), eutrophication (Fig. S6), particle emissions (Fig. S7) and photochemical smog (Fig. S8) send a mixed message about benefits of electrification and HVO utilization. Firstly, when charging with the Swedish electricity mix, there is a trend of decreasing total impacts with an increasing level of electrification, while there are small increases in vehicle life cycle load for all the LCIA methods, in line with results of global warming impact. But in contrast to climate change category, impacts and reductions with increasing electrification are now roughly the same for the HVO and diesel options. In fact, in the eutrophication category, using HVO has a worse effect than diesel. Secondly, for options charging with electricity mixes from the EU and the USA, results diverge from the global warming pattern in the acidification and particles impact categories. Here, the HEV options have lowest emissions and the trend then points to increasing impact according to the level of electrification in both diesel and HVO use. The eutrophication category also shows a break of the pattern for diesel use in combination with the EU and USA electricity mixes, but in this case the hybrid options are preferable compared to the all-electric and conventional ones.

Looking closer, it is noticeable that the extent to which various life cycle stages depend on combustion processes is a main driver of emissions influencing all these impact categories, regardless if this involves large scale firing of coal in power plants to produce electricity or the burning of diesel or HVO in an onboard engine. $NO_x$ is the key emission type for all but the particles impact category, where it nevertheless remains important. $NO_x$ is generated in all types of combustion involving air. Further contributions to $NO_x$ derive from nitrogen contained in the fuels. Similarly, the presence of sulfur in fossil fuels leads to the generation of acidic sulfur dioxide during combustion. As pointed out in the discussion around certification data and Euro-6 emissions limits (see Section 3.1), real driving emissions of $NO_x$ can be expected to be lower than the emission data used in this study, implying that TTW impact may be

**Table 1**
Results for all impact categories – total contribution and per life cycle stage for all options. Electric and PHEV options are reported for charging with the average electricity grid mixes (including losses in the charger) in Sweden (SE), the EU and the USA. Abbreviated life cycle stages: Matr. = material extraction and processing; Manuf. = Component and full vehicle manufacturing; Maint. = Maintenance; WTT = Well-to-tank (charging electricity and fuels for operation); TTW = Tank-to-wheels (energy conversion in vehicles); and EoL = End-of-Life.

| Impact category | Life cycle stage | Electric SE el | Electric EU el | Electric US el | PHEV Diesel SE el | PHEV Diesel EU el | PHEV Diesel US el | PHEV HVO SE el | PHEV HVO EU el | PHEV HVO US el | HEV Diesel | HEV HVO | Conv. Diesel | Conv. HVO |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Climate change | Matr. | 6.4 | 6.4 | 6.4 | 6.2 | 6.2 | 6.2 | 6.2 | 6.2 | 6.2 | 6.2 | 6.2 | 5.6 | 5.6 |
| g $CO_2$ eq./(person × km) | Manuf. | 1.9 | 1.9 | 1.9 | 1.5 | 1.5 | 1.5 | 1.5 | 1.5 | 1.5 | 1.4 | 1.4 | 1.2 | 1.2 |
| | Maint. | 2.0 | 2.0 | 2.0 | 1.7 | 1.7 | 1.7 | 1.7 | 1.7 | 1.7 | 1.6 | 1.6 | 1.3 | 1.3 |
| | WTT | 4.5 | 37 | 49 | 7.8 | 23 | 29 | 11 | 27 | 32 | 10 | 15 | 16 | 23 |
| | TTW | 0.1 | 0.1 | 0.1 | 29 | 29 | 29 | 0.7 | 0.7 | 0.7 | 46 | 1.1 | 69 | 1.7 |
| | EoL | 1.1 | 1.1 | 1.1 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 |
| | **All** | **16** | **48** | **60** | **47** | **63** | **68** | **22** | **38** | **43** | **66** | **26** | **93** | **34** |
| Acidification | Matr. | 62 | 62 | 62 | 57 | 57 | 57 | 57 | 57 | 57 | 57 | 57 | 51 | 51 |
| µmol H+ eq./(person × km) | Manuf. | 15 | 15 | 15 | 13 | 13 | 13 | 13 | 13 | 13 | 12 | 12 | 11 | 11 |
| | Maint. | 20 | 20 | 20 | 15 | 15 | 15 | 15 | 15 | 15 | 14 | 14 | 9.8 | 9.8 |
| | WTT | 26 | 201 | 220 | 77 | 162 | 171 | 56 | 140 | 149 | 108 | 74 | 167 | 116 |
| | TTW | 2.6 | 2.6 | 2.6 | 18 | 18 | 18 | 18 | 18 | 18 | 28 | 28 | 42 | 42 |
| | EoL | 1.3 | 1.3 | 1.3 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 1.2 | 0.9 | 0.9 |
| | **All** | **126** | **301** | **320** | **181** | **266** | **275** | **160** | **244** | **253** | **220** | **186** | **282** | **231** |
| Eutrophication | Matr. | 8.2 | 8.2 | 8.2 | 7.6 | 7.6 | 7.6 | 7.6 | 7.6 | 7.6 | 7.5 | 7.5 | 6.8 | 6.8 |
| mg N eq./(person × km) | Manuf. | 2.1 | 2.1 | 2.1 | 1.7 | 1.7 | 1.7 | 1.7 | 1.7 | 1.7 | 1.7 | 1.7 | 1.5 | 1.5 |
| | Maint. | 2.6 | 2.6 | 2.6 | 2.0 | 2.0 | 2.0 | 2.0 | 2.0 | 2.0 | 1.8 | 1.8 | 1.3 | 1.3 |
| | WTT | 8.5 | 32 | 33 | 11 | 22 | 23 | 23 | 34 | 34 | 13 | 32 | 20 | 48 |
| | TTW | 1.4 | 1.4 | 1.4 | 9.2 | 9.2 | 9.2 | 9.2 | 9.2 | 9.2 | 15 | 15 | 22 | 22 |
| | EoL | 0.3 | 0.3 | 0.3 | 0.3 | 0.3 | 0.3 | 0.3 | 0.3 | 0.3 | 0.3 | 0.3 | 0.2 | 0.2 |
| | **All** | **23** | **46** | **48** | **32** | **43** | **44** | **43** | **54** | **55** | **39** | **58** | **52** | **80** |
| PCOF | Matr. | 25 | 25 | 25 | 24 | 24 | 24 | 24 | 24 | 24 | 24 | 24 | 22 | 22 |
| mg NMVOC eq./(person × km) | Manuf. | 5.4 | 5.4 | 5.4 | 4.6 | 4.6 | 4.6 | 4.6 | 4.6 | 4.6 | 4.5 | 4.5 | 4.0 | 4.0 |
| | Maint. | 8.8 | 8.8 | 8.8 | 6.6 | 6.6 | 6.6 | 6.6 | 6.6 | 6.6 | 6.1 | 6.1 | 5.0 | 5.0 |
| | WTT | 17 | 75 | 71 | 41 | 69 | 67 | 33 | 61 | 59 | 55 | 42 | 84 | 65 |
| | TTW | 6.0 | 6.0 | 6.0 | 40 | 40 | 40 | 40 | 40 | 40 | 65 | 65 | 96 | 96 |
| | EoL | 0.79 | 0.79 | 0.79 | 0.72 | 0.72 | 0.72 | 0.72 | 0.72 | 0.72 | 0.71 | 0.71 | 0.63 | 0.63 |
| | **All** | **64** | **121** | **117** | **118** | **146** | **144** | **110** | **137** | **136** | **155** | **142** | **212** | **193** |
| PM | Matr. | 14 | 14 | 14 | 13 | 13 | 13 | 13 | 13 | 13 | 13 | 13 | 12 | 12 |
| mg PM 2.5 eq./(person × km) | Manuf. | 3.4 | 3.4 | 3.4 | 2.1 | 2.1 | 2.1 | 2.1 | 2.1 | 2.1 | 1.9 | 1.9 | 1.3 | 1.3 |
| | Maint. | 3.5 | 3.5 | 3.5 | 2.9 | 2.9 | 2.9 | 2.9 | 2.9 | 2.9 | 2.7 | 2.7 | 2.2 | 2.2 |
| | WTT | 2.8 | 18 | 140 | 6.5 | 14 | 75 | 4.4 | 12 | 72 | 8.5 | 5.1 | 13 | 8.3 |
| | TTW | 0.11 | 0.11 | 0.11 | 0.76 | 0.76 | 0.76 | 0.76 | 0.76 | 0.76 | 1.2 | 1.2 | 1.8 | 1.8 |
| | EoL | 0.19 | 0.19 | 0.19 | 0.17 | 0.17 | 0.17 | 0.17 | 0.17 | 0.17 | 0.17 | 0.17 | 0.14 | 0.14 |
| | **All** | **24** | **39** | **160** | **26** | **33** | **94** | **24** | **31** | **92** | **28** | **24** | **31** | **26** |
| Humtox cancer | Matr. | 3.1 | 3.1 | 3.1 | 3.1 | 3.1 | 3.1 | 3.1 | 3.1 | 3.1 | 3.1 | 3.1 | 3.0 | 3.0 |
| $10^{-9}$ $CTU_h$/(person × km) | Manuf. | 0.31 | 0.31 | 0.31 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.28 | 0.27 | 0.27 | 0.24 | 0.24 |
| | Maint. | 0.65 | 0.65 | 0.65 | 0.54 | 0.54 | 0.54 | 0.54 | 0.54 | 0.54 | 0.50 | 0.50 | 0.42 | 0.42 |
| | WTT | 0.32 | 2.6 | 4.1 | 0.32 | 1.4 | 2.1 | 0.32 | 1.4 | 2.1 | 0.28 | 0.29 | 0.44 | 0.45 |
| | TTW | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| | EoL | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.07 | 0.06 | 0.06 |
| | **All** | **4.4** | **6.7** | **8.2** | **4.3** | **5.4** | **6.1** | **4.3** | **5.4** | **6.1** | **4.2** | **4.2** | **4.2** | **4.2** |

(*continued on next page*)

**Table 1** (*continued*)

| Impact category | Life cycle stage | Electric SE el | Electric EU el | Electric US el | PHEV Diesel SE el | PHEV Diesel EU el | PHEV Diesel US el | PHEV HVO SE el | PHEV HVO EU el | PHEV HVO US el | HEV Diesel | HEV HVO | Conv. Diesel | Conv. HVO |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Humtox non-cancer | Matr. | 12 | 12 | 12 | 10 | 10 | 10 | 10 | 10 | 10 | 9.8 | 9.8 | 8.5 | 8.5 |
| $10^{-9}$ $CTU_h$/(person × km) | Manuf. | 2.7 | 2.7 | 2.7 | 2.5 | 2.5 | 2.5 | 2.5 | 2.5 | 2.5 | 2.5 | 2.5 | 2.4 | 2.4 |
| | Maint. | 3.7 | 3.7 | 3.7 | 2.4 | 2.4 | 2.4 | 2.4 | 2.4 | 2.4 | 2.0 | 2.0 | 1.2 | 1.2 |
| | WTT | 3.4 | 10 | 14 | 2.3 | 5.5 | 7.5 | 2.9 | 6.1 | 8.1 | 1.2 | 2.1 | 1.9 | 3.4 |
| | TTW | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| | EoL | 1.9 | 1.9 | 1.9 | 1.8 | 1.8 | 1.8 | 1.8 | 1.8 | 1.8 | 1.8 | 1.8 | 1.7 | 1.7 |
| | **All** | **23** | **30** | **34** | **19** | **22** | **24** | **20** | **23** | **25** | **17** | **18** | **16** | **17** |
| Abiotic depletion CML | Matr. | 300 | 300 | 300 | 280 | 280 | 280 | 280 | 280 | 280 | 280 | 280 | 220 | 220 |
| g Sb eq./(person × km) | Manuf. | 33 | 33 | 33 | 33 | 33 | 33 | 33 | 33 | 33 | 33 | 33 | 32 | 32 |
| | Maint. | 98 | 98 | 98 | 79 | 79 | 79 | 79 | 79 | 79 | 74 | 74 | 61 | 61 |
| | WTT | 12 | 14 | 14 | 13 | 14 | 14 | 8.6 | 9.6 | 9.6 | 11 | 4.4 | 20 | 9.8 |
| | TTW | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| | EoL | 38 | 38 | 38 | 34 | 34 | 34 | 34 | 34 | 34 | 34 | 34 | 26 | 26 |
| | **All** | **480** | **480** | **480** | **440** | **440** | **440** | **430** | **430** | **430** | **430** | **420** | **360** | **350** |
| Non-renew. depletion EPS | Matr. | 43 | 43 | 43 | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 50 | 46 | 46 |
| $10^{-3}$ ELU/(person × km) | Manuf. | 4.7 | 4.7 | 4.7 | 4.8 | 4.8 | 4.8 | 4.8 | 4.8 | 4.8 | 4.7 | 4.7 | 4.6 | 4.6 |
| | Maint. | 12 | 12 | 12 | 14 | 14 | 14 | 14 | 14 | 14 | 13 | 13 | 12 | 12 |
| | WTT | 1.7 | 2.1 | 2.0 | 2.2 | 2.4 | 2.4 | 1.2 | 1.4 | 1.4 | 2.2 | 0.6 | 3.7 | 1.3 |
| | TTW | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| | EoL | 9.2 | 9.2 | 9.2 | 8.3 | 8.3 | 8.3 | 8.3 | 8.3 | 8.3 | 8.3 | 8.3 | 6.3 | 6.3 |
| | **All** | **71** | **71** | **71** | **79** | **80** | **79** | **78** | **79** | **78** | **79** | **77** | **73** | 71 |

overestimated in terms of impacts caused by air pollution. A change in TTW results, i.e. if certification data for the tailpipe emissions was used instead, would not alter the overall results for most impact categories, except in the case of photochemical smog. In this category, lower emissions of $NO_x$ and CO could result in a shift, owing to PHEV or HEV being more favorable options.

Combustion in vehicles and power plants, along with extraction and processing of metals and fuels, cause particulate emissions. Fuel combustion leads to indirect formation of particles by reactions in the atmosphere involving sulfur dioxide, $NO_x$, ammonia and non-methane volatile organic compounds (often referred to as VOCs) (Johansson et al., 2003). However, for all chargeable vehicles relying on the US electricity mix, direct emissions of dust particles from fossil fuel use in electricity production dominate results (see Fig. S7).

The occurrence of photochemical smog is another version of secondary pollutant creation (Fig. S8). At times of little wind, sunlight causes $NO_x$ and VOCs to react to form ozone, which has adverse health effects at ground level. However, air pollution, including particles and smog along with the direct effects of $NO_x$, carbon monoxide etc., is most critical when taking place during vehicle operation in densely populated areas with large exposure to many people. This implies benefits for the electric options not depicted in the results, typically in cases when LCIA results are high or increasing, compared to diesel or HVO use. The reason being that emissions are shifted from the operation stage to other life cycle stages, with less human exposure.

Finally, one clear exception from the otherwise large dependence on airborne emissions in the results, is the high potential for eutrophication caused by HVO production. This is mainly a result of nitrate emissions in water from the use of crops.

### 4.3. Toxicity

Results for carcinogenic human toxicity, an LCIA category for potential health effects in terms of cancer, are reported in Table 1 and shown in Fig. S9. Other, non-carcinogenic, aspects of toxic emissions were also investigated in the study (Fig. S10). In both cases, differing from the previously reported LCIA categories, the operation of the vehicle (i.e. the TTW stage of the use phase) contributes negligibly to all options. Instead, toxic emissions, into both air and water, typically occur during the extraction of resources and during different steps of material production and fossil fuel processing. For example, the handling and treatment of mining waste can lead to leakage of toxic substances into ground water and rivers, and eventually the sources of fresh water used by humans.

Accordingly, results in Fig. S9 link largely to emissions of heavy metals such as chromium, cadmium and arsenic, mainly into water. The same heavy metals are also significant in the non-carcinogenic category. But while chromium influence is less, for example lead is more important. Cast iron and various steel grade production, especially stainless steel, for chassis, frames and bus bodies, contribute to a substantial part of life cycle impact in toxicity categories, especially in material extraction and processing phases. Similarly, mining and production of copper generate notable heavy metal emissions.

Clearly, result patterns reveal that toxic impacts are strongly dependent on the electricity supply for the chargeable options, and the underlying amount of fossil fuel use. The highest impacts are linked to the US and the EU grid mixes. Unsurprisingly, extraction, processing and combustion of fossil fuels are major sources of heavy metal emissions.

An important note on toxicity indicators is that they are coupled to a large degree of uncertainty, and the precision of characterization factors varies between two to three orders of magnitude (Finnveden et al., 2009; Rosenbaum et al., 2008). These impact categories relate to both the inherent toxicity of the emitted substances and the potential for humans and ecosystems being exposed to harmful effects. Consequently, and more than for other LCIA methods, it is useful to interpret these results as an aggregated reporting of relevant emissions, rather than a clear contribution to human health challenges. For example, there is a distinct difference between the carcinogenic (Fig. S9) and the non-carcinogenic (Fig. S10) impact categories of the EoL stage. This is explained by emissions of zinc into water during the treatment of used tires. A high level of zinc intake is toxic for humans, although an essential nutrient at lower intake (Linderholm, 2017). The high ranking of zinc illustrates the one-sided outcome of the LCIA method, while real-world effects can be much more complex.

### 4.4. Resource use

As mentioned, the analysis of resource use was conducted using two different methods: at midpoint for potential abiotic resource depletion (CML, 2016; Guinée et al., 2002), and at endpoint with the EPS indicator, but only for use of non-renewable elements (Steen, 2015). In the ADP result (Table 1 and Fig. S11 of the SR), conventional buses operating on diesel and HVO had lower impact than electrified options, and there is a small trend towards larger resource depletion with an increasing degree of electrification. Gold, silver and copper are available in all electronics and provide contributions for all options. In particular, the PCBs of power electronics, in battery management systems and in junction boxes contain gold, with clear effect in results on both material production and maintenance for options pertaining to electric powertrains. Relatively large amounts of gold are used in the PCB for the air purification units in all vehicles, influencing results for the maintenance stage. Molybdenum in steel alloys is another important contributor to all options. Copper use, for example in the electric traction motor, wires and PCBs, provides additional contributions to all electrically powered alternatives. It is worth mentioning that the lithium in the battery cells makes no significant contribution to the results of this LCIA method. Lastly, regarding ADP results for conventional buses, the lead-acid battery plays an important role due to the use of silver and lead and the extraction of cadmium, which is co-mined with lead.

In the EPS method (Fig. S12) the pattern differs compared to ADP results. Instead of indicating an increasing depletion of resources with an increasing level of electrification, EPS allots the highest load to hybrid alternatives and lowest load to the all-electric. Largely, many emissions caused by material production and maintenance mentioned previously in connection to ADP, explain the EPS results, even though the valuation of specific resources is different. The main explanation for divergence of patterns is that the EPS method clearly values the use and loss of expensive and scarce metals in the catalytic converters of the exhaust gas treatment systems, i.e. platinum, rhodium and palladium, in all vehicles with internal combustion engines, more highly than does the CML method. Although these metals are subject to dedicated recycling back to use in the same type of application (Andersson et al., 2017), because of their economic value, high shares of their upstream raw material input still originate from virgin resources.

## 5. Conclusions

### 5.1. Main findings

The study shows that although there is no direct correlation between the degree of electrification and the environmental impact of city buses, there is a link. Another way to express this is that questions 1, 2 and 3 stated in the goal definition of the article, are cross-dependent. One answer to question 1, i.e. if the environmental impact decreases or increases according to the degree of electrification, is that there is variation between the LCIA categories and that the electricity for charging (question 2) and the selection of fuel (question 3) to complement and compare with electricity, are decisive parameters. One clear conclusion is, however, that when the electrified bus is powered by low carbon electricity, like the Swedish mix, impacts related to air emissions decrease according to the degree of electrification.

When comparing options for the results of the vehicle equipment life cycle, most LCIA categories show that a higher degree of electrification is coupled to a growing environmental load. This is in line with LCA studies for other vehicle types (Nordelöf et al., 2014), although the overall shift upwards is relatively small for buses. In addition, the shift is small compared to variations in impact caused by charging and fueling.

In the climate change impact category – which is especially interesting as climate change mitigation is a key driving force for electrification – it can be noticed that the use of HEVs is beneficial, regardless of whether the complementary available fuel is diesel or HVO. Benefits of additional electrification with charging depend on the relationship between the emission profile of the available electricity mix and accessible fuel options. When diesel is the fuel option for a fleet of hybrid vehicles, both PHEVs and all-electric buses will reduce emissions further for operation in the majority of countries in the EU, and on average in the USA. Then again, if a fuel such as HVO is available for the bus fleet, i.e. a low emission biofuel based on waste products, then the conditions for achieving benefits by using chargeable vehicles become much more stringent, and from a climate mitigation perspective is motivated only in places with a relatively low grid mix carbon intensity (below 200 g $CO_2$ eq./kWh). The two final important conclusions from the assessment of the climate change category are: all options based on electric propulsion or HVO present clear savings compared with the conventional diesel bus, and it is the all-electric vehicle which has the largest potential to reduce emissions among all options, despite it having the heaviest vehicle equipment life cycle burden.

For regional environmental impacts caused by airborne emissions, i.e. acidification and eutrophication, as well as local air pollution, it is again the all-electric options which have the best potential to reduce impacts of emissions overall, but the link to the degree of electrification of the buses, the charging electricity and the choice of (reference) fuel is more complicated to summarize. Instead, the amount of energy conversion based on combustion overall throughout the life cycle, and the share of fossil fuels used to feed those processes, determine the outcome for these impact categories. However, a clear health benefit provided by electrification, even when a relatively large share of the of the life cycle energy derives from the burning of fossil fuels, is that people are less exposed to high concentrations of air pollutants in city centers.

As regards both toxicity and the use of resources, there is a shift in the division of burden between different life cycle stages. The actual operation is of little interest to these impact categories, and instead the extraction of metals and fossil fuels now comes into focus. Toxic emissions of heavy metals to ground and fresh water expand when a larger and broader spectrum of materials and metal alloys are in demand for electrified vehicles, but the decisive parameter when ranking between options is the matter of fossil fuel use in electricity production. Two common solutions to reduce toxic emissions in all the options investigated would be to (1) enforce stricter regulation of emissions from the excavation, refinement and waste handling in mining operations and (2) increase the use of recycled materials.

Results for the assessment of resources diverge for LCIA methods used, but both approaches agree that hybrid vehicles suffer from the fact that they demand advanced materials with high value elements from virgin sources, both in the electric powertrain and in internal combustion engine exhaust gas treatment components.

Another answer to the question of how city buses impact the environment according to their degree of electrification (question 1), is that material and component production required for electric powertrains and batteries contributes clearly to the difference between technical options in most impact categories, but that this difference is small compared to the production of common components. Instead, chassis, frame and body are the dominating factors behind the material extraction and processing life cycle stage. This was further verified in a sensitivity analysis, where the number of battery replacements made during the maintenance stage were varied for the all-electric, PHEV and HEV buses (see Tables S15–S17 in Section D.2 of the SR), causing only minor variation in results for all impact categories. In this aspect, buses in public transport clearly differ from passenger cars, where the production of electric powertrain parts, and in particular the battery pack, adds a much greater contribution to overall environmental impact of the vehicle (Hawkins et al., 2012; Nordelöf et al., 2014). Even when equipment life cycles for buses outweigh the use phase as a whole, the environmental impact of the use phase (the WTW stage) usually determines which option offers the lowest impact.

Finally, it can be observed that additional sensitivity analyses carried out to analyze the effect on the result when using the maximum number of passengers for each bus option, instead of the same average amount (by comparing Table 1 with Table S18 of the SR), further improves the results of Electric (largest capacity) and HEV options (second largest capacity). In contrast, the case for

PHEV options is weakened in several impact categories, e.g. for climate change, where PHEV diesel options charged with US electricity now only reduce impacts by 13%, compared to 27% in the original setup. Most importantly, apart from these shifts for the PHEVs, the main findings of the study remain the same also when accounting for maximum ridership.

### *5.2. Concluding remarks on strategies for reduced environmental impact of bus fleets*

Combined with other relevant groundwork, the LCA study presented could serve as a source of information for strategic planning of public transportation and in the procurement of bus fleets. However, for such use, it can be valuable to provide an additional discussion, in order to set results and conclusions in a broader context. Obviously, any effort to minimize the total environmental burden over the life cycle of a city bus implies prioritizing between different types of impact. Each case will depend on the specific vehicle design available, the electricity possible to procure and the feedstock and production route for available biofuels. In this context, it is important to emphasize that the data for HVO and electricity supply presented in this study indicates different trends.

Looking at HVO, in addition to containing biogenic carbon, which is key for climate change category, the relatively low impact across several categories is due to waste currently constituting most of the feedstock for its production. However, as an example, data from the Swedish Energy Agency (2017) indicates that when demand for HVO surmounts available waste resources, the use of food crops will increase. In turn, this entails a risk of a general increase of emissions caused by HVO production, e.g. if more palm oil is included as feedstock than the amount this study allows for. Generally, it can be argued this is true for most biofuels made from recovered waste flows, biogas being one such example.

On the other hand, regarding the supply of electricity for charging in both the EU and the USA, trends point towards a less emission-intensive electricity production, due to the ongoing shift from coal to natural gas, and the continued expansion of renewable energy generation such as solar and wind power (IEA, 2017). Assuming this trend will continue implies that the options contained in this study, which are propelled by grid electricity in the EU or USA, will lower their impact over time in most LCIA categories.

In a situation where electricity available for charging offers a load corresponding to or above the average electricity in both EU and USA, and where biofuels such as HVO are manufactured mainly from residues and waste, a hybrid vehicle using this biofuel can be the preferable option. As concluded above, ongoing trends point towards a continued reduction of emissions over time for chargeable electric buses on these markets, whereas an increase in biofuel demand might shift the environmental load upwards, strengthening the case for PHEVs and all-electric buses.

## Acknowledgement

The authors would like to express their gratitude to Region Västra Götaland and the Area of Advance Energy at Chalmers University of Technology for financing in different steps of the project.

## Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.trd.2019.08.019.

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