Journal of Cleaner Production 322 (2021) 129052

Contents lists available at ScienceDirect

Journal of Cleaner Production

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

Regionalized climate footprints of battery electric vehicles in Europe

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ARTICLE INFO

Handling editor: Zhifu Mi

Keywords:
Battery electric vehicle (BEV)
Life cycle assessment
Climate change
Electrification
Greenhouse gas emissions (GHG)
Software tool

ABSTRACT

The climate mitigation benefits of battery electric vehicles (BEVs) relative to internal combustion engine vehicles (ICEVs) are highly dependent on the carbon intensity of the electricity consumed during their production and use-phase. A consistent and dynamic approach to grid-mix regionalization of BEV life-cycle assessments in Europe is therefore necessary to offer accurate guidance to consumers and policy makers. To this end, we present ReDyFEV, a simple open-source software tool that can be used to calculate attributional, regionalized lifecycle climate impacts of BEVs in Europe for user-defined time periods, including near real-time. We determine the national lifecycle carbon footprints across all EU states for four BEV size segments and compare them to those of fossil-fuelled vehicles of similar sizes. Simplified sensitivity analyses investigate the effect of lifetime assumptions, electricity demand in battery production, and of relocating battery production to Europe on the carbon footprints of BEVs.

1. Introduction

The transport sector was responsible for 24% of total global greenhouse gas emissions arising from fuel combustion in 2017 (International Energy Agency, 2019). Of these emissions, 74% originates from road transport. Despite the urgent need to address anthropogenic greenhouse gas emissions, there is continued rapid growth in demand and emissions expected in the transport sector, making it a key target for climate mitigation efforts. Within the transport sector, passenger cars, or light duty vehicles (LDVs) used approximately 50% of the total oil demanded by the transport sector in 2017 (IEA, 2018a). In an attempt to curb greenhouse gas emissions from the transport sector, many cities and countries are pledging to phase out fossil-fuelled vehicles in favour of battery electric vehicles (BEVs) (IEA, 2018b). Since BEVs do not have the tailpipe emissions associated with internal combustion engine vehicles (ICEVs), the extent to which BEVs contribute to climate mitigation is sensitive to the carbon footprint of the electricity used in both the production and use-phases of the vehicle.

A complete lifecycle perspective is necessary in the environmental assessment of BEVs, as they generally have higher production emissions than equivalently-sized fossil-fuelled vehicles (Hawkins et al., 2013; Ellingsen et al., 2016; Kim et al., 2016). These higher production emissions are attributed to the manufacturing of the traction battery, which is electricity-intensive and is generally performed in countries with carbon-intensive electricity mixes (Ellingsen et al., 2014; Kim et al., 2016; Sun et al., 2020; Crenna et al., 2021). To compensate for these higher production emissions and offer significant climate advantages compared to ICEVs, BEVs require a local electricity mix with as low a carbon footprint as possible in the use-phase. Multiple studies demonstrate that the sources of production and use-phase electricity (hydropower, nuclear energy, natural gas, coal, etc.) are the most important contributors to variations among carbon footprint estimates (Marques et al., 2019; Ellingsen et al., 2016; Cusenza et al., 2019). In other words, the lifecycle impacts and benefits of BEVs are highly sensitive to the regions of production and use.

Unfortunately, a coherent, systematic regionalization of BEV LCAs is still lacking. This is particularly problematic in Europe, where a large polychotomy of electricity mixes exist between countries within a relatively small geographical area. Some BEV LCAs simply rely on "an average European mix," (Bicer and Dincer, 2017; Miotti et al., 2017;

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https://doi.org/10.1016/j.jclepro.2021.129052
Received 1 April 2021; Received in revised form 26 August 2021; Accepted 14 September 2021
Available online 17 September 2021

Cusenza et al., 2019; Hawkins et al., 2013) and Moro and Lonza (2018) note that this assumption is "commonly used for national regulatory purposes". However, using a continental average implicitly makes the assumption that the level of interconnection within the European grid is sufficient to lead to a homogeneous electricity mix. This leads to high uncertainties as to the magnitude of the climate mitigation benefits (if any) that consumers and policy makers can expect from BEV adoptions in each country. Conversely, single-region, local studies have been performed independently in individual countries and states (Qiao et al., 2019; Marques et al., 2019; Shi et al., 2019; Wolfram and Wiedmann, 2017; Lombardi et al., 2017), but differing system boundaries and assumptions complicate their comparison and validation. This work in isolation leads to research inefficiencies, as the same tasks are essentially repeated by each local LCA group. More importantly, a myriad of local LCAs can prove hard to keep consistently updated, which is crucial considering the rapid pace of progress in battery manufacture (Kwade et al., 2018; Duffner et al., 2021; Blomgren, 2017; Bresser et al., 2018) and in the decarbonization of electricity grids, both targeted (European Council, 2014) and realized (Eurostat, 2021). In short, there is a need for a broadly applicable LCA model with a high geographical granularity and a dynamically updated representation of the electricity grid and its greenhouse gas emissions.

An important regionalization effort was conducted for the United-States by Wu et al. (2019a), Yuksel et al. (2016), Tessum et al. (2012) and Archsmith and Kendall (2015) yielding state-specific vehicle footprints that show important levels of inter-state variations. These studies, however, provide a snapshot in time (e.g., fixed at the year 2010 for Tessum et al., and 2016 for D. Wu et al. (via United States Environmental Protection Agency (EPA) 2016)), requiring further work to keep updated. These regionalization studies also did not address the increasingly pressing question of the geographical location of battery production.

Similarly, Moro and Lonza (2018) provide a one-off European state-level regionalization centered around the year 2013. This study omits the significant contributions of BEV production emissions to the lifecycle emissions. Moreover, use-phase impacts in this study are based on the Joint Research Centre Well-to-Tank report (Edwards et al., 2014), which omits the construction and maintenance of the electricity generating infrastructure. Incomplete system boundaries and truncation issues can prove a significant source of uncertainty in LCA when comparing technologies that rely on widely different value chains (Lenzen, 2002). Hybrid LCAs, relying on complementary data from national economic accounts to fill data gaps, have been developed precisely to avoid such truncation errors (Gibon et al., 2015; Agez et al., 2020) but have scarcely been applied to the question of regionalized BEVs, with Wu et al. (2019b), Karaaslan et al. (2018) and Wolfram and Wiedmann (2017) being three exceptions. In a similar thread, while there exists regionalized well-to-wheel studies comparing ICEVs and BEVs, such studies omit the vehicle production cycle (Woo et al., 2017; Canals Casals et al., 2016). However, the vehicle cycle for BEVs, particularly the production phase, have been found to contribute significantly to the overall carbon footprint of these vehicles (Ellingsen et al., 2016; Kim et al., 2016; Yang et al., 2020).

The European Union (EU) is evaluating "the possibility of developing a common Union methodology for the assessment and the consistent data reporting of the full life-cycle CO_2 emissions of such vehicles placed on the Union market" (European Parliament and Council of the European Union, 2019). Given the context of previous studies, we now know that such a common approach should capture the entire lifecycle of the vehicle with an accurate and up-to-date representation of regional electricity mixes. Such a precise understanding of regional differences may prove particularly important if BEV deployment faces production and capacity constraints (Cohen, 2020; Valero et al., 2018), thus leading to a need to prioritize BEV uptake where they can yield maximum climate benefits. Furthermore, in a bid to assert energy independence, the European Commission is working to establish a battery manufacturing sector in Europe (European Commission, 2019a). Such evolving dynamics and local production should be reflected in vehicle footprints to better inform consumers and comply with emissions standards.

The objective of this study is therefore to offer a coherent and agile solution to the regionalization and the dynamic updating of ICEV's and BEV's carbon footprints, across all European countries, and account for their entire life cycle with consistent system boundaries. To this end, we provide a simplified, open-source software tool for the REgionalized DYnamic Footprinting of Electric Vehicles (ReDyFEV) in Europe. This tool extracts grid data and calculates regionalized, country-specific electricity mixes for any arbitrary time period, which it then translates into attributional lifecycle carbon intensities using hybridized lifecycle factors for electricity generation technologies. With these regionalized electricity mixes, we determine lifecycle BEV carbon intensities in these countries for four vehicle size segments. The combination of using hybridized, spatially explicit emission factors for the electricity mix combined with the use of a full LCA approach including the vehicle cycle in a multi-country European context has not yet, to our knowledge, been performed. The outcomes from this work contribute to a better understanding of the heterogeneity of BEV climate footprints across Europe under different assumptions and guide the potential framework under development for communicating LDV footprints across Europe. We also perform simplified scenario analyses to investigate the potential climate savings in moving battery production to Europe, and the robustness of results against different assumptions for vehicle lifetime and electricity use in manufacturing.

2. Methods

2.1. Modelling the European electricity grid

To test the implicit "average" European mix assumption, we calculated and compared the carbon intensity of the production and consumption electricity mixes for European countries. We used the ENTSO-E Transparency Platform (Hirth et al., 2018) via the bentso Python package (Mutel et al., 2019) to retrieve the national production mixes ("Actual Generation per Production Type") and trade relationships ("Cross-Border Physical Flow") for the 2020 calendar year. Production data represent net generation and were aggregated to annual production values. National electricity consumption was calculated as in equation (1), where net generation is the generation after electricity for the producing plants' own use and hydro pumping is accounted for.

$$\text{consumption} = \text{net generation} + \text{import} - \text{export} \tag{1}$$

To calculate the lifecycle carbon intensity of the energy technologies at the country level, we use a hybridized LCA-environmentally extended input-output (EEIO) database (Agez et al. 2019, 2020). These hybridized factors are built from *ecoinvent 3.5* and *EXIOBASE 3* datasets (Stadler et al., 2018). The ENTSO-E electricity technology categories were matched to electricity production activities in the hybridized database, which follows the ecoinvent 3.5 naming convention. For many of the ENTSO-E categories, there are several relevant *ecoinvent* activities (see Table 11 in Supplementary Information); in these cases, a weighted average approach was used to determine the emissions intensity of each ENTSO-E category and country. This weighted average calculation uses the production shares of each relevant *ecoinvent* technology for each country's electricity mix as represented in *ecoinvent*. The correspondences of ecoinvent 3.5 to the ENTSO-E technology categories assumed in this study are described in Table 10 in the Supplementary Information. As a simplifying assumption, the emission intensities for the ENTSO-E categories 'Other' and 'Other renewable' were the continental average of the emissions from non-renewable (fossil, and nuclear) and renewable (hydro, solar, wind, and geothermal) generation technologies, respectively. Electricity generated by waste incineration, representing a maximum of 4.6% of the national production mix, follows the convention of allocating emissions to the main function, i.e., waste

Table 1
Key parameters for the vehicles used in the base case scenario. The assumed lifetime for both vehicle technologies in the baseline scenario is 180 000 km.

			Vehicle segment			
			A mini	C medium	JC compact SUV	JE mid-size SUV
BEV	Battery size	kWh	36.8	62	82	95
	Battery production emissions, excluding electricity	t CO_2	2.8	4.7	6.2	7.1
	Electricity requirements, battery production	kWh	1582	2666	3526	4085
	Vehicle production emissions, excluding battery	t CO_2	4.8	6.5	8.5	9.8
	Use-phase energy demand	Wh km^{-1}	145	180	184	261
	End-of-life treatment	t CO_2	0.5	0.7	0.9	1.1
ICEV	Production emissions	t CO_2	3.6	5.3	9.3	11.3
	Use-phase energy demand	kJ km^{-1}	1672	1893	2714	2935
		(Wh km^{-1})	(464)	(526)	(754)	(815)
	Use-phase emissions (lifetime, including fuel chain)	t CO_2	26.7	30.2	43.2	47.1
	End-of-life treatment	t CO_2	0.3	0.5	0.6	0.7

treatment, used in ecoinvent and described in Majeau-Bettez et al. (2014, 2016) and Heijungs and Suh (2002). Electricity from waste incineration therefore has an emission factor of 0 g CO_2-eq kWh^{-1} for all countries.

Some of the technology-country combinations reported by ENTSO-E did not have emission factors available in ecoinvent. These missing emission intensity factors were estimated as the average of the generation technology over the countries studied. For example, emission factors for hard coal power in Hungary and Romania were not found in ecoinvent. Proxy values for these two missing emissions factors were therefore calculated using the average of the hard coal emission factors for hard coal over the other countries in the system, ranging between 984 and 1481 CO_2-eq kWh^{-1}. These values are highlighted in Table S2 in the Supplementary Information Spreadsheet. For countries with trade relations but no ENTSO-E production data (Albania, Belarus, Croatia, Luxembourg, Malta, Russia, Turkey and Ukraine), the assumed footprint is taken from the high-voltage electricity mixes in ecoinvent (Table S1, Supplementary Information (shaded cells)) in order to calculate the consumption mix footprint for the countries they trade with. These countries with no production data from ENTSO-E are otherwise not included in this study. Lifecycle emissions for electricity from shale oil does not have representation in ecoinvent and is therefore assumed to have the same emission factor as fossil oil for a given country.

The carbon footprints of country-level consumption electricity mixes are calculated using the trades retrieved from ENTSO-E. While many approaches exist to calculate consumption electricity mixes (Ryan et al., 2016; Weber et al., 2010), we implemented two methods in ReDyFEV. The first is a grid-average approach including trade, as in Moro and Lonza (2018). In this method, it is assumed that the net generation mix of the exporting country is used for trade. The second is the flow tracing method, used in Tranberg et al. (2019), where it is assumed that each node is a perfectly homogeneous market, and that therefore each country's consumption mix is also representative of the mix it exports. The flow tracing method is analogous to the Leontief approach used in LCA calculations and therefore used for the results presented here. Results obtained using the grid-average approach are available in Supplementary Information Figs. 9–14.

Country-specific transmission and distribution losses from 2014 (IEA Statistics, 2018) as a percentage of output are applied to each country's electricity footprints to obtain the carbon intensity of low-voltage electricity.

2.2. Lifecycle emissions of light duty vehicles

Table 1 summarizes key characteristics and emissions profiles of the vehicles considered in the base case scenario of this work. In the demonstration of the tool and results, we calculate footprints for four vehicle segments: A (mini), C (medium), JC (compact SUV) and JE (mid-size SUV). These SUV segments were selected to address the increasing market shares of SUVs (IEA, 2021), the medium segments for a mid-range "average" vehicle, and the mini class as a counterpoint to the large SUVs. The following text describes the calculation of the default values for lifecycle emissions of light duty vehicles, however, within the tool, users are free to modify these values.

Battery production emissions are based on Sun et al. (2020), scaled to the different battery capacities. In order to calculate the emissions for domestic production of BEVs, the electricity intensity (Table 1) for battery production processes was separated out from the total emissions intensity from battery materials and production provided in Sun et al. (2020), and is assumed to be 43 kWh electricity for each kWh of battery storage capacity. This value represents the cell manufacture, active material production (including NCM precursors), dry room operation, electrode drying, mixing and formation processes, as well as cell and pack manufacturing for lithium-ion batteries of NCM 622 chemistry. The emissions attributed to this electricity use are subtracted from the total emissions intensity for production to provide the "Rest of battery production" emissions intensity of 75 kg kWh^{-1}. The Chinese electricity mix for medium voltage from ecoinvent 3.0 is used to calculate these emissions from electricity use by Sun et al. (2020), as this is the database they use. Further details for both powertrain types are available in the Supplementary Information.

As a sensitivity analysis, the analysis is also conducted using the energy use for battery production coarsely estimated in Kurland (2019) for the Tesla Gigafactory 1, a value corresponding to 65 kWhel/kWhbattery. In the base case, the electricity associated with battery manufacture is assumed to be the Korean medium-voltage electricity mix, with a lifecycle CO_2 intensity of 679 g CO2-eq kWh^{-1} (Wernet et al., 2016). For reference, corresponding carbon intensity of electricity mixes in China and Japan, the other current dominant LIB manufacturing regions, range between 660 and 1020 g CO_2-eq kWh^{-1} (Wernet et al., 2016). The use-phase energy demand for the BEVs is based on the Worldwide Harmonised Light Vehicles Test Procedure (WLTP) rated energy consumption values, as reported by the manufacturers, and which include charging losses. Rest-of-vehicle production emissions are calculated from the inventory in Hawkins et al., (2013), and scaled as in Ellingsen et al. (2016). Battery end-of-life is calculated using mass-based intensity factors and battery weights, and is according to Dewulf et al. (2010). The remainder of the vehicle end-of-life is based on vehicle curb weights and corresponding mass-based intensity factors from Hawkins et al. (2013).

The ICEs used in the comparison are selected to match the BEV curb weight without the battery to make as equal a comparison as possible (e.

g., avoiding comparing small and large vehicles from the same segment). The default values for production inventories of internal combustion engine vehicles (ICEVs) are also from Hawkins (Hawkins et al., 2013), with scaling as performed in Ellingsen et al. (2016). The lifetime use-phase emissions are based on the manufacturer reported WLTP values. These WLTP values, presented in g CO_2 km^{-1}, represent the direct tailpipe emissions. For the indirect use phase emissions, that is, fuel chain emissions, the use-phase energy demand and upstream fuel chain emissions are calculated from the WLTP fuel efficiency values, assuming an energy density of 31.6 MJ L^{-1}, and upstream emissions intensity of 17 g CO_2-e MJ^{-1} for petrol (Prussi et al., 2020). The use-phase WTW emissions intensity values are multiplied by the assumed lifetime for total use phase emissions. The vehicle specifications used in calculations and mentioned above and the representative models used can be found in Table 1 in the Supplementary Information.

All vehicle segments for both vehicle powertrain technologies in the base case scenario are assumed to have a lifetime of 180 000 km. This value was selected as an approximate average of the commonly assumed lifetimes of 150 000 (Bekel and Pauliuk 2019; Wu et al. 2019b; Yang et al. 2021) and 200 000 km (Cox et al. 2020), and which was used in Ellingsen et al. (2016). However, traction batteries lifetimes are often measured in terms of equivalent full cycles (EFC), which means that larger batteries inherently have a longer lifetime in terms of km driven than smaller batteries. Consequently, on a km-driven basis, the lifetime of the larger segment batteries is longer than that of the smaller batteries; these lifetimes may possibly be longer than that of the vehicle itself and thus these batteries may lend themselves to second-life applications such as grid integration of renewables or other stationary energy storage uses (Kamath et al. 2020a, 2020b; Martinez-Laserna et al., 2018). Thus, to account for the greater remaining cycle-life of the battery beyond the vehicle km-lifetime in the case of larger batteries and thereby allow a fair comparison between vehicle segments, we proportionately allocate a share of the total battery production emissions to the use in vehicle, and a share to the stationary use (Equation (2)) in the base case scenario. In technical terms, the second life application constitutes a by-product of BEV production and use, as grid operators or end consumers would incur a cost to obtain these (still highly functional) EOL batteries. This approach of allocating production emissions to by-products is addressed in Weidema (2000), Nakamura and Kondo (2002), Schrijvers et al. (2016) and Schulz et al. (2020). The other impacts of these second life applications, however, are outside of the scope of this study. The production emissions allocated to the BEVs for each segment in the base case scenario are shown in Table 1 in the Supplementary Information. This assumption effectively assumes that the vehicle itself, and not the battery, is the lifetime bottleneck for BEVs. We make the simplifying assumptions that a battery reaches end-of-life in the vehicle at a 20% capacity fade (Ruiz and Moretto 2018; Pelletier et al., 2017), and that this level of capacity fade occurs at 1500 EFC (Equation (3)). Equation (3) also assumes a linear degradation of battery capacity from 100% to 80%. We also provide a full set of results without the base case scenario's allocation assumption in the Supplementary Information.

$$\text{allocated battery emissions} = \frac{(\text{vehicle lifetime, km})}{(\text{battery lifetime, km})}(\text{total battery manufacturing emissions}) \tag{2}$$

where

$$\begin{aligned}\text{battery lifetime, km} &\approx \frac{\text{total energy stored, kWh}}{\text{fuel efficiency, kWh km}^{-1}}\\ &= \frac{\int_{x=0}^{x_{max}} \text{capacity}(x)dx}{\text{fuel efficiency, kWh km}^{-1}}\\ &\approx \frac{(\text{initial capacity})(x_{max@80\%})(0.9)}{\text{fuel efficiency, kWh km}^{-1}}\\ &\approx \frac{(\text{initial capacity})(1500)(0.9)}{\text{fuel efficiency, kWh km}^{-1}}\end{aligned} \tag{3}$$

As a sensitivity analysis, we also adjust the lifetimes of both powertrain technologies; for BEVs, we examine lifetimes of 150 000 to 250 000 km and for ICEVs, 200 000 to 250 000 km. No battery replacement in the lifetime of the BEVs is assumed. While BEV use-phase energy use and battery lifetime is highly dependent on a number of factors such as consumer charging patterns, including use of fast chargers, and climate (i.e., temperature) (Yang et al. 2018; Barré et al. 2013; Lacey et al. 2017), these effects are not explicitly accounted for in this study. The absolute lifecycle climate mitigation effects of BEVs presented in *Fig. 5* are calculated by subtracting the absolute lifecycle climate emissions of ICEVs from similarly sized BEVs. These absolute mitigation effects are also normalized by the ICEV lifecycle climate emissions to obtain the relative mitigation effects in *Fig. 5*.

2.3. European production of electric vehicle batteries

Currently, the majority of BEV batteries are manufactured in China, Japan and South Korea, where fossil fuels power dominate the electricity mix, resulting in relatively high carbon intensity (Wernet et al. 2016). As a result, battery manufacturing, particularly electricity use in cell manufacture, is partially responsible for the relatively high production emissions for batteries and BEVs (Crenna et al. 2021; Sun et al. 2020; Yuan et al. 2017; Ellingsen et al. 2014, 2017; Kim et al. 2016). As a hypothetical scenario, we explore the effects of European battery manufacturing on the BEV footprint. We substitute the country-level electricity mix for the electricity used in the battery manufacturing.

2.4. ReDyFEV: a spatially explicit footprinting tool

As part of this work, we present REgionalized DYnamic Footprinting of Electric Vehicles (ReDyFEV), an open-source software tool for calculating regionalized climate footprints for electric vehicles in Europe (Hung, 2021). ReDyFEV can receive parameter values from users including the carbon intensity of the electricity used in the manufacturing of the traction batteries, the ICEV and BEV lifetimes in km, the calculation approach for national consumption mixes (flow or grid average approaches), the period from which to fetch electricity data from the ENTSO-E Transparency Portal, whether to apply the second-life allocation assumption, and the electricity use assumptions in battery production. The vehicle specifications may also be modified from those presented in the base case here by editing the input spreadsheet. This

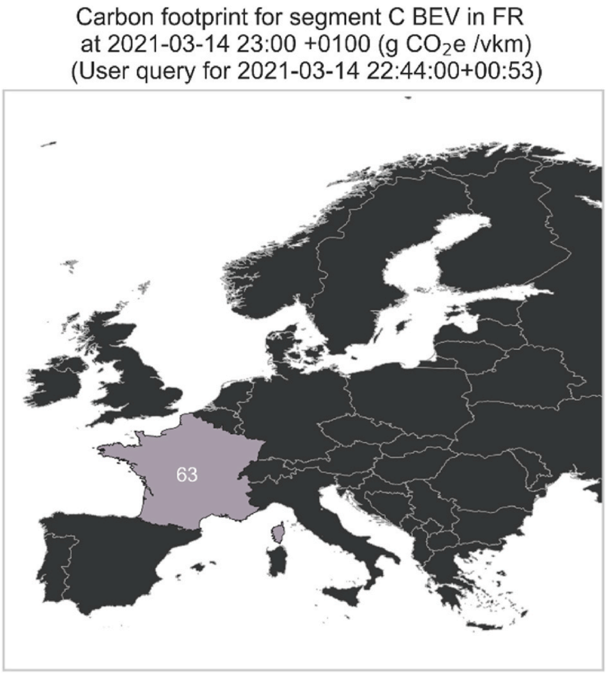

Fig. 1. Example screenshot of vehicle reporting functionality from ReDyFEV.

capability allows users to choose alternative representative vehicles (e.g., changing vehicle sizes), and emissions intensities (e.g., from other sources in literature). Users can also choose to calculate the lifecycle carbon footprint of a desired vehicle segment in a specific country for a specified time period. As the data in the Transparency Portal is updated with an approximate 2-h delay, these footprints can be calculated in near real-time (see example screenshot in Fig. 1). ReDyFEV also generates the figures presented in this article.

3. Results

3.1. *Carbon footprints of electricity mixes in European states*

Fig. 2 compares the lifecycle carbon footprint of country-level production and consumption electricity mixes. The larger the deviation from the 45-degree line, the more electricity trading influences the carbon footprint of the consumption mix. Most countries lie relatively close to this line, demonstrating that electricity trade generally has little effect on carbon intensity in European countries. These results contradict the approach taken in much of the traditional LCA literature, where the assumption of an 'average European electricity mix' is often considered sufficient (Kawamoto et al. 2019; Peng et al. 2018; Miotti et al. 2017). The use of an average European mix suggests that trade occurs at such an extent that differences in production mixes between countries are "averaged out" to a homogeneous value. Thus, for such an assumption to be valid, the points in Fig. 2 would approximate a horizontal line, where despite differing production mix intensities, the consumption mix intensities would align near the same values. Despite the relatively close geographical proximity of European states, electricity trading does not occur to an extent that justifies the assumption of a homogeneous European electricity mix. We can thus conclude that the average European mix assumption is particularly inadequate for the evaluation of electricity-intensive technologies, such as operation or production of BEVs, where this assumption would hold greater weight over the results. Indeed, the carbon intensity of European state

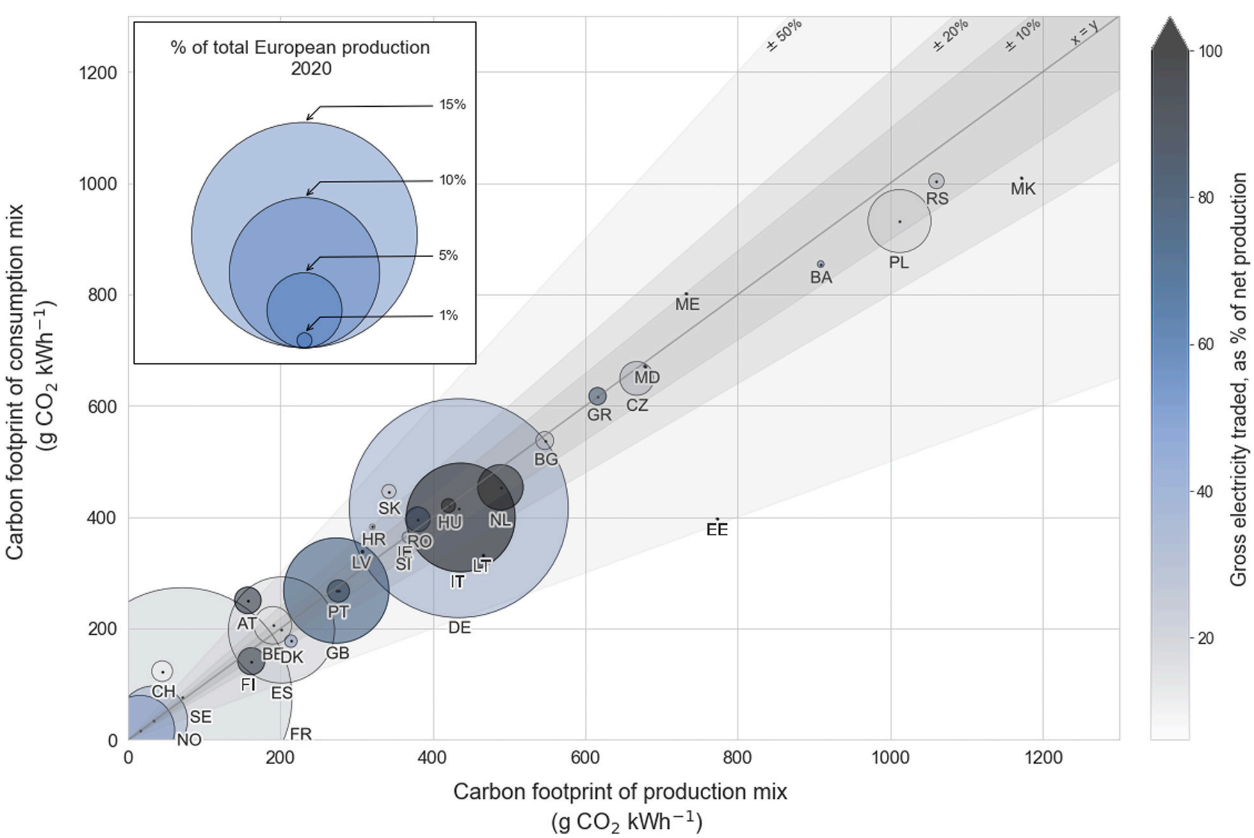

Fig. 2. *Lifecycle carbon footprints of European electricity mixes, 2020. Bubble colour indicates total (gross) electricity traded (export and imports) as a percentage of net domestic production. Bubble area indicates each country's total electricity traded as a share of its total production. Shaded areas indicate ± 10%, 20% and 50% of 45-degree line.*

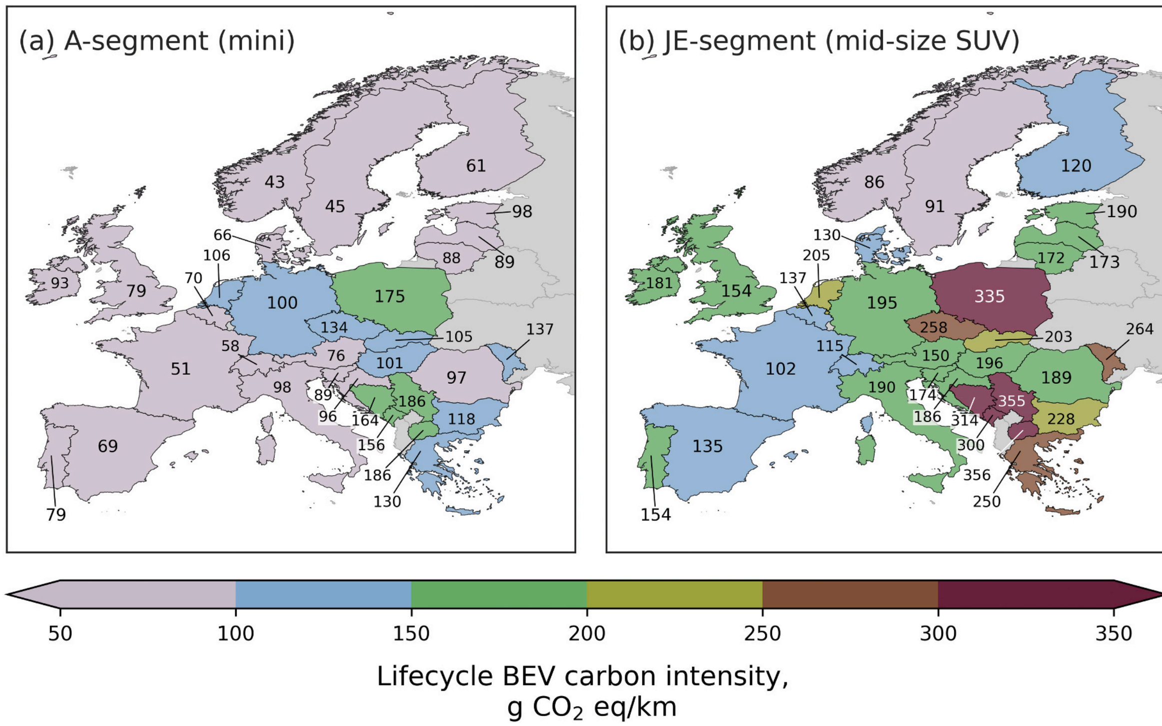

Fig. 3. *Absolute BEV lifecycle carbon intensities (g CO_2-eq km^{-1}) for mini (A-segment, left) and mid-size SUV (JE-segment, right) vehicles using domestic consumption electricity mix in the operation phase and Korean-produced batteries in the base case scenario.*

consumption mixes are decidedly heterogeneous, as illustrated by the large spread in consumption mix intensities, and the use of a single, average European electricity mix would be inappropriate for most state-level considerations.

For simplification, only the results using the flow tracing method for calculating consumption mix are presented here. Results using the grid average approach used in Moro and Lonza (2018) are provided in the Supplementary Information. Note that while the approach used to calculate the consumption mix makes a difference in the consumption mix footprint for a small number of countries, the overall conclusions of

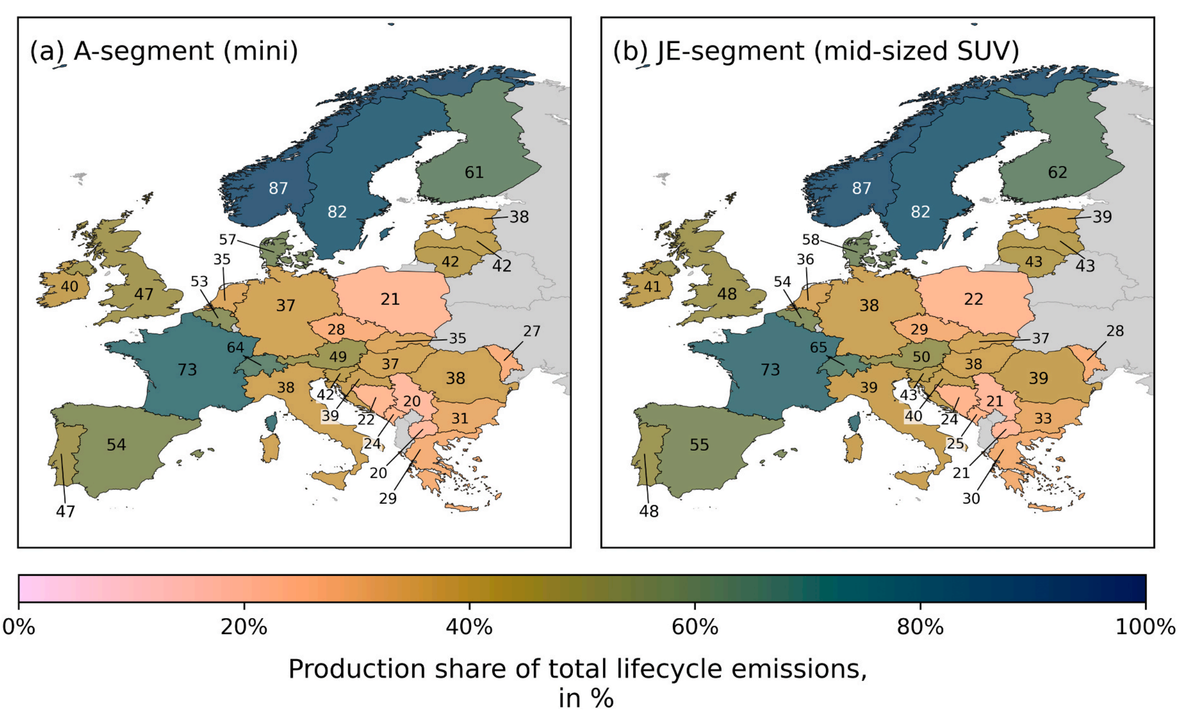

Fig. 4. *BEV vehicle manufacturing emissions, as % of total lifecycle CO_2 footprint for (a) mini (A-segment, 36.8 kWh), (b) mid-sized SUV (JE-segment, 95 kWh) vehicles in the base case scenario.*

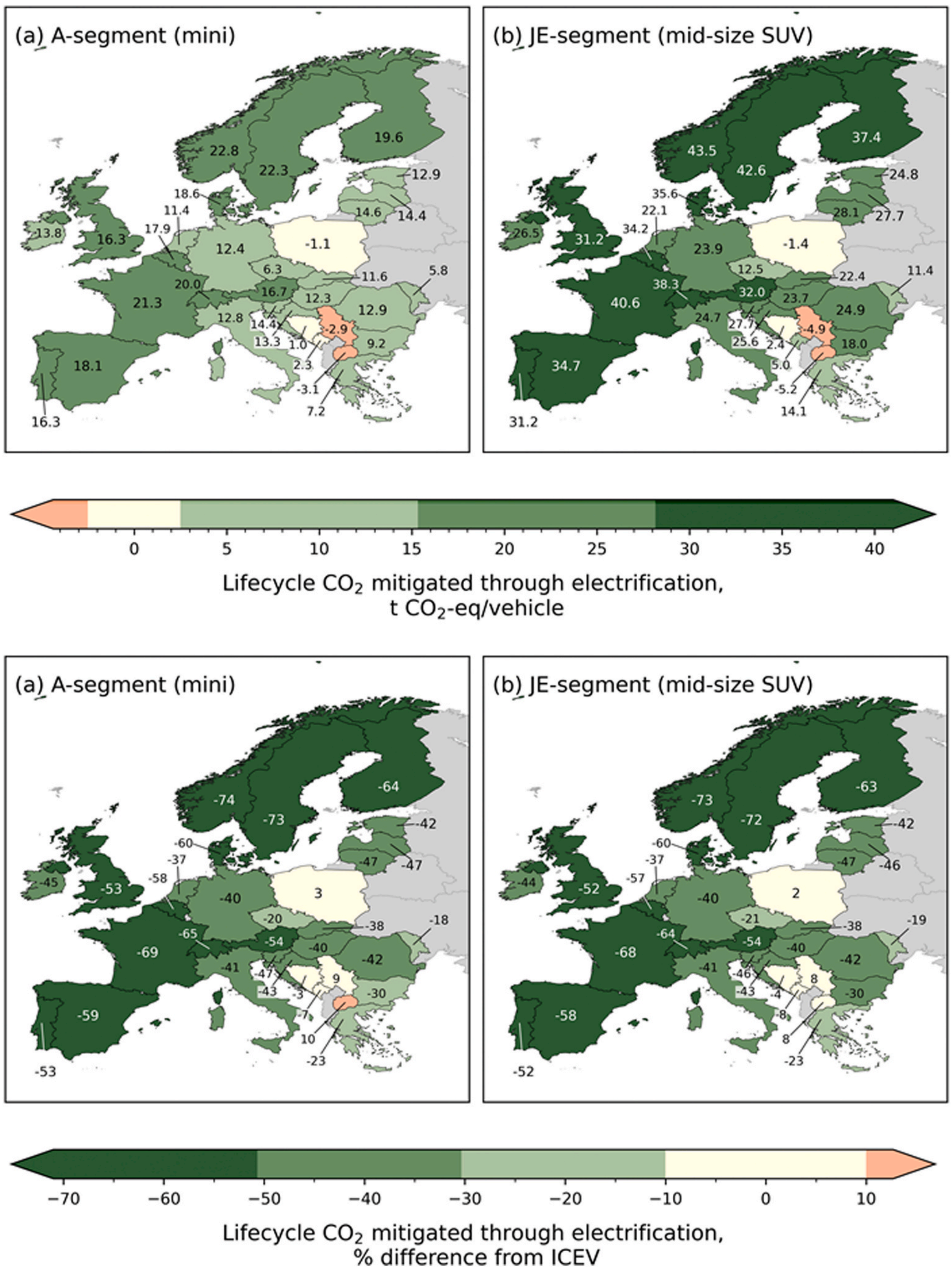

Fig. 5. *Absolute [top panels] and relative [bottom panels] lifecycle climate mitigation effects of electrification (t CO_2-eq/vehicle electrified) for (a) mini (A-segment), and (b) mid-sized SUV (JE-segment) vehicles in the base case scenario.*

this study remain the same.

3.2. Carbon footprints of battery electric vehicles in Europe

Previous studies have found that the use-phase electricity used to charge BEVs heavily influences their overall climate performance relative to ICEVs. Charging BEVs with carbon intensive electricity from coal-fired power plants, for example, gives BEVs a higher carbon footprint than similarly sized ICEVs, primarily due to the high battery production impacts (Ellingsen et al. 2016).

Fig. 3 illustrates the effect of vehicle size on the absolute lifecycle carbon footprint of BEVs. As one might expect, Fig. 3 shows that within each size class, the BEV footprints vary widely across the European states, thus reflecting the variation in electricity footprints shown in Fig. 2. These differences in BEV footprints confirm the sensitivity of the environmental impact of BEVs to the charging electricity.

Furthermore, Fig. 3 illustrates that the carbon intensity of the different BEV segments increases with increasing vehicle size, i.e., from the A-segment (mini) to the JE-segment (mid-sized SUV). This is attributable to the increased production emissions associated with the larger batteries typically installed in larger vehicles as well as the increased energy required for driving heavier vehicles. The regional differences in BEV footprint between countries also becomes more pronounced for larger vehicles due to the increased energy required for driving resulting from the increased mass of the larger vehicles.

Previous studies generally consist of well-to-wheel studies that exclude the vehicle cycle and therefore do not communicate full life-cycle impacts and particularly, the non-negligible influence of vehicle production. Fig. 4 illustrates the share of total lifecycle emissions attributed to vehicle production. Shares of production emissions in total lifecycle footprint vary between 20 and 87%. As one might expect, in countries with less carbon-intensive electricity, the vehicle cycle is more important in the entire lifecycle footprint because the overall operation emissions from charging are low. Battery size also plays a role in how much the vehicle production emissions contribute to overall CO_2 emissions intensity; larger batteries generally result in slightly higher contribution of production emissions to overall footprint. We can infer from this that the trend of increasing battery size within a given segment will increase the share of production emissions in the total footprint. These results highlight the importance of including vehicle production in lifecycle footprints, rather than simply well-to-wheel values. These results also show that the larger batteries being used in BEVs to alleviate range anxiety counteract to some extent the potential climate benefits of these vehicles. While our overall conclusions align with Moro and Lonza in that there is significant heterogeneity in BEV footprints between European states, the actual values of the footprints vary significantly due to the inclusion of vehicle production cycle.

Fig. 5 compares the potential climate mitigation effects of BEVs over ICEVs for the smallest and largest size segments. One can observe that a transition to BEVs represents a climate mitigation strategy for many European states. On the other hand, the relatively carbon-intensive electricity mix in countries such as Poland, Serbia, and North Macedonia lead to increased climate impacts from the adoption of BEVs relative to ICEVs in the smaller size segments, and approximately break-even mitigation effect for the largest segments. In these countries, BEVs have higher lifecycle emissions than ICEVs for some size segments. Although the electric powertrain in BEVs is far more energy efficient than internal combustion engines, the carbon intensity of the electricity used to power the BEVs results in high use phase emissions. In combination with the higher production emissions of BEVs compared to ICEVs, this results in a net climate disadvantage for BEVs in these countries.

In comparing the A- and JE-segments, it is evident that the magnitude of BEV climate benefit increases with larger segments. This follows from the use-phase efficiency gains of BEVs relative to their ICEV counterparts increasing with vehicle size, with the more powerful engines in JE-segment ICEVs being particularly fuel inefficient. In addition, the countries where BEVs offer the greatest climate benefits, the magnitude of the benefit of replacing ICEVs with BEVs increases for larger vehicles.

A side effect of these gains is the range of CO_2 mitigation effects between countries for each vehicle size class; the difference between BEV climate emissions in the countries with the least and most carbon intensive electricity mix is smaller for the A-segment than the JE-segment. Note that for some countries, the difference in lifecycle impacts between BEVs and ICEVs lies close to 0; it is difficult to draw a concrete conclusion between the two powertrain types for these states due to uncertainties in the lifecycle emission factors and the influence consumer behaviour, e.g., driving patterns.

In the countries with the most carbon intensive electricity mixes, such as Poland, Serbia and North Macedonia, current BEVs in different segments present either negligible advantages or even increases in lifecycle emissions when compared to their ICEV counterparts. In such countries, electrification represents a climate disadvantage. However, the extent of this disadvantage decreases (or even becomes a mitigation effect) with increasingly large vehicle segments; in other words, with larger vehicles, the difference between BEV and ICEV climate impacts becomes smaller.

In an electrification scenario where the assumption is that BEVs replace equally sized ICEVs, the implication would then be that in low-carbon grids, replacing larger (e.g., JE-segment) vehicles before smaller vehicles would maximize the climate mitigation benefits from electrification. In countries with carbon intensive electricity mixes, electrification is not necessarily a climate mitigation measure, however, larger segments provide the most climate benefit, if any, from electrification. These countries, however, might benefit from postponing the transition to electric vehicles in favour of decarbonizing the electricity mix.

3.3. Sensitivity analyses

While the knowledge regarding electricity and energy use in battery production has improved over the years, there are still considerable differences between the values found in literature (Sun et al., 2020; Crenna et al., 2021; Cox et al., 2020). This becomes especially evident when narrowing energy use to electricity use; in addition to differences in electricity estimates for certain processes, some manufacturers appear to rely more heavily on steam from natural gas, while others appear to use more electricity and less steam, e.g., Sun et al. (2020), in contrast to Dai et al. (2019). The electricity intensity of battery production we use in the baseline scenario, corresponding to 43 kWh of electricity for each kWh of battery capacity (Sun et al., 2020), represents a middle-of-the road value within current estimates, which range between 20.7 and 65 kWh/kWh, including both cell manufacture and precursor production (Dai et al., 2019; Davidsson Kurland 2019; Crenna et al., 2021). In the sensitivity analysis, we use the 65 kWh/kWh estimated by Kurland (2019), which is the estimated energy demand for a gigawatt-hour scale facility. This value for the electricity demand in battery production slightly increases the overall BEV lifecycle footprint; an increase in the electricity demand of approximately 50% results in a footprint increase ranging between 1 and 4% over the baseline scenario (see Table S12 in the Supplementary Information spreadsheet). As the electricity demand assumption only affects the production emissions, countries where production emissions constitute a large share of the total lifecycle emissions, i.e., those with relatively clean electricity mixes, are those most heavily influenced. Despite the higher overall emissions arising from this assumption, the increase is not sufficient to change the overall conclusions in the comparison between BEVs and ICEVs. In other words, this assumption does not lead to BEVs in countries with carbon intensive electricity mixes having lower carbon footprints better than similarly sized ICEVs.

Fig. 6 shows the results of the lifetime sensitivity analysis for each segment of BEVs and ICEVs studied. As we are examining lifecycle CO_2 intensity, the results illustrate the effect of distributing production and end-of-life emissions over a shorter or longer lifetime, as the operation emissions intensity is assumed to remain constant over the vehicle lifetime. Note that the lifecycle intensity for ICEVs is less sensitive to lifetime than for BEVs due to their lower production emissions relative to the use-phase emissions. The spread of lifecycle carbon intensity between the short and long lifetime assumptions increases with BEV segment. The upper values of the range, representative of a BEV lifetime 17% shorter than the baseline assumption, correspond to a 3–15% increase in lifecycle CO_2 intensity across all segments. In contrast, a BEV lifetime that is 39% longer than the baseline assumption, represents a 4–21% decrease in lifecycle intensity compared to baseline results. Furthermore, we can see that in most cases, the overall conclusions drawn regarding the performance of a BEV relative to an ICEV of the same segment are fairly robust across the spectrum of lifetime assumptions for both powertrain types, i.e., few countries cross the ICEV lines.

3.4. Domestic production of traction batteries

Previous studies have found that battery manufacture contributes 31–46% of the total production emissions for BEVs (Ellingsen et al. 2016; Kim et al. 2016). A large portion of these battery manufacturing emissions is attributable to electricity use in producing the battery cells (Ellingsen et al. 2017); from our calculations, electricity for precursor production, cell manufacture and pack assembly as reported in Sun et al. (2020) account for approximately 40% of the total climate change emissions of battery manufacturing. Today, most traction batteries are produced in China, Japan and South Korea (Lebedeva et al. 2017; Gifford 2015; Bernhart, 2014), all of which have relatively carbon intensive

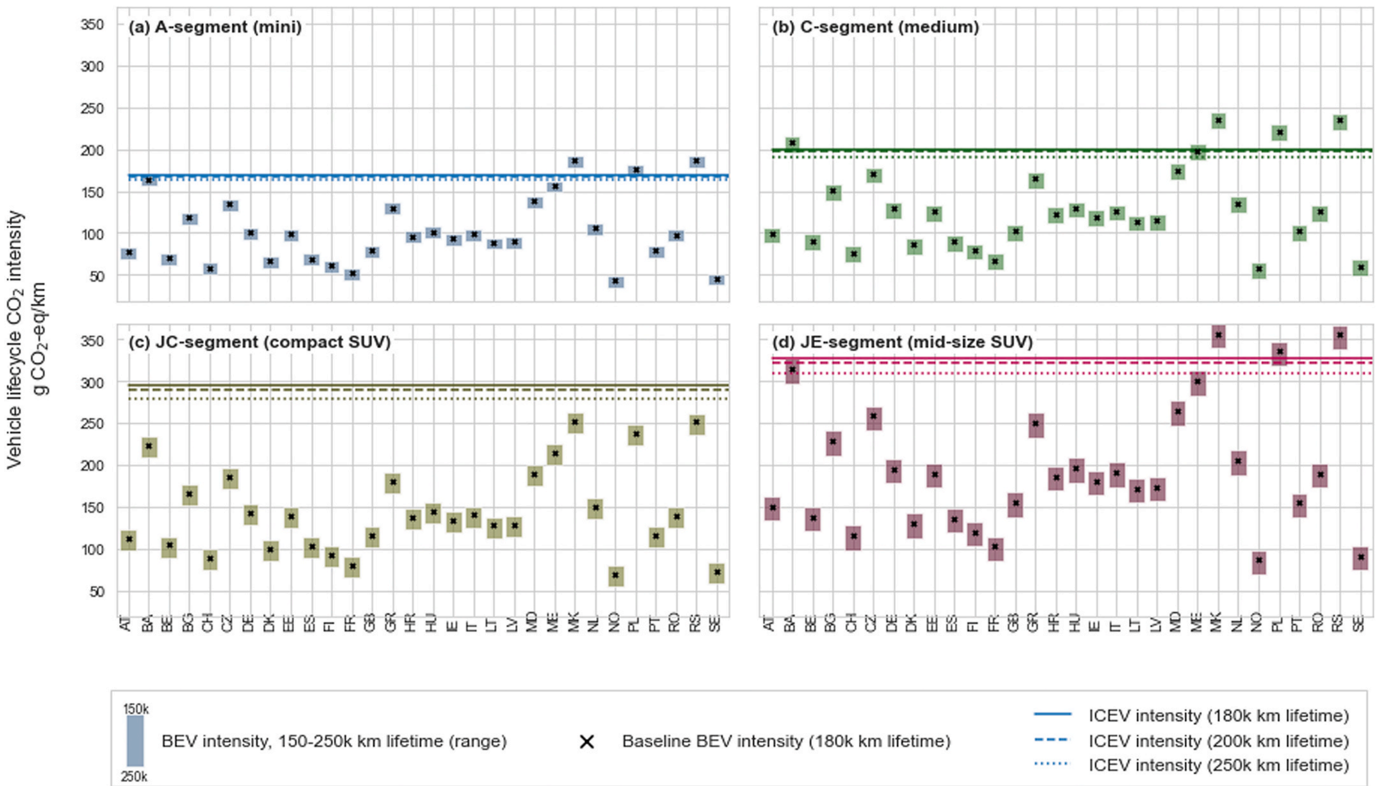

Fig. 6. Sensitivity of lifecycle intensity to BEV lifetime assumptions. Bars indicate range of BEV intensities for lifetime assumptions of 150 000 km (upper limit) and 250 000 km (lower limit). Markers indicate BEV under baseline assumption of 180 000 km. Horizontal lines indicate ICEV intensities for 180 000 (baseline assumption), 200 000 and 250 000 km.

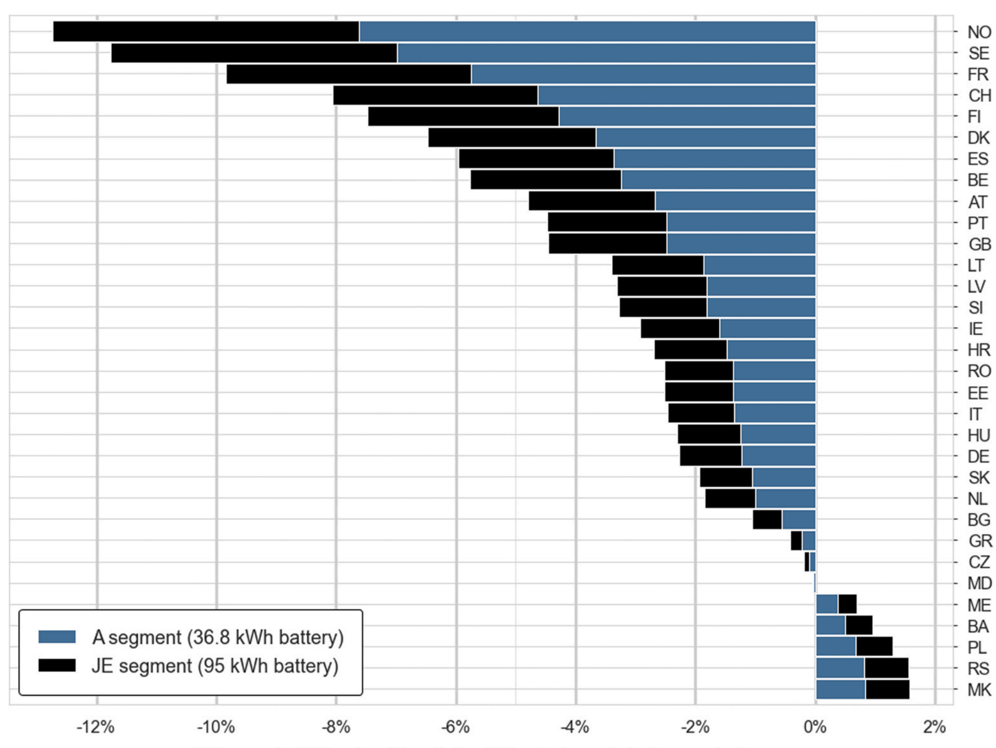

Fig. 7. Effect of domestic battery production on BEV lifecycle carbon intensity. Relative change from BEV of the same segment with Asian production of batteries.

electricity mixes (Wernet et al. 2016). As BEVs gain traction in the global market, however, the battery manufacturing sector is likely to geographically diversify.

The European Commission (EC) has an objective that the EU will have "strategic autonomy" in the battery sector (European Commission 2019b). To this end, the EC aims to have 7–25% of global battery cells manufactured in Europe by 2028 of the estimated 250–1100 GWh of global battery sales in the same year (European Commission 2019b). As of 2018, the European share of cell manufacturing for traction batteries amounts to only 1% (Eddy et al. 2019). Currently, existing and planned European battery manufacturing capacity combined amount to an estimated 200–290 GWh $year^{-1}$ by 2025 (Scott 2020; Eddy et al. 2019). Of these facilities, the Tesla Gigafactory near Berlin, the combined ACC facilities in Germany and France, the LG Chem facility in Poland, the Northvolt facilities in Sweden and Germany and the Freyr facilities in Norway are the largest at approximately 100, 48, 35, 40 and 24 and 34 GWh (combined) annual manufacturing capacity, respectively (Amelang 2021; European Investment Bank, 2020; Hampel 2020; Freyr Battery AS). The remaining battery cell factories, however, are comparatively small, ranging between MWh-scale facilities to 16 GWh $year^{-1}$ (Eddy et al. 2019; Lebedeva et al. 2017). The demand for traction batteries in Europe is estimated to reach 400 GWh $year^{-1}$ by 2028 (European Commission 2019b), and 1 200 GWh $year^{-1}$ by 2040 (Eddy et al. 2019), which correspond to approximately 20 and 80 facilities with capacities of 15 GWh $year^{-1}$, respectively.

To investigate the potential ramifications of establishing a strong European battery manufacturing sector, we model a hypothetical scenario under the assumption that traction batteries are produced domestically (i.e., in the same country in which the vehicle is driven). Fig. 7 illustrates the changes in BEV footprints incurred by producing traction batteries domestically. Although most European countries would decrease BEV footprints by producing their own batteries, the magnitude of the net benefit arising from domestic battery production varies widely. As one might expect from the results, the domestic battery manufacturing scenario amplifies the heterogeneity between countries observed in the BEV footprint analysis. Minimizing the emissions in a BEV deployment scenario therefore requires the optimal placement of European battery manufacturing plants. Two of the largest planned battery cell manufacturing facilities in Europe will be in bottom half of the countries in Fig. 7 (Poland and Germany), thus providing either only a small climate advantage or even disadvantage over the current production situation. Indeed, a battery with Li-ion cells produced with the current Polish consumption mix would have a higher carbon footprint than the current batteries produced in Asia. Due to the higher production emissions associated with the larger batteries, the magnitude of the net benefit or disadvantage of domestic battery production increases with the larger battery capacities associated with larger-segment BEVs. The assumption of the electricity demand required for battery production, as explored in the sensitivity analysis, would also amplify the magnitude of the bars in Fig. 7.

Although it is unlikely that all the studied countries would produce traction batteries, this hypothetical scenario enforces the previous conclusions that the heterogeneity of electricity mixes in Europe have considerable implications for the climate performance of BEVs not only in the use phase, but also for production. These findings illustrate the climate effects of strengthening Europe's role in the global traction battery market.

4. Discussion

This study provides insight on the importance of adopting a consistent approach to regionalized life-cycle footprints of BEVs across Europe. Current LCAs often use an average European mix to determine the climate impacts of electrification or are one-off studies with differing system boundaries that make comparisons between studies and countries challenging. We calculate country-specific factors to determine the carbon footprint of electricity mixes and BEVs in Europe. From our results, we determine that, despite the coordinated electrification policy in the EU, heterogeneity is an important consideration for communicating BEV footprints in Europe and for BEV adoption in the European region.

Additionally, this work serves as a useful confirmation of the results by Moro and Lonza (2018), who also determined strong heterogeneity in BEV footprints across Europe using a different approach. Although some of the BEV intensities from this study are drastically different than those in this previous study, these are most likely attributable to the temporal difference in the electricity system data between the two studies and the inclusion of the vehicle cycle in this study; results from this study found a significant share of lifecycle CO_2 intensity (20–87% of total intensity) could be attributable to vehicle manufacturing. Given the trend of increasing vehicle and battery sizes, this finding suggests that curbing this trend as much as possible, e.g., encouraging uptake of smaller vehicles, will maximize the potential climate mitigation effects of electrification.

The relevance of regionalized footprints is particularly crucial as the EU evaluates the possibility of developing a common framework for calculating lifecycle emissions for current vehicles and potentially adopting such methods for communicating vehicle emissions to consumers. ReDyFEV presents a twofold advantage: the calculation of near real-time BEV footprints, and regional heterogeneity of these footprints. Relying on geographically generic and "snapshot" footprints for BEVs in the defining and enforcement of fleet emissions targets ignores an opportunity where Europe could encourage the mitigation-optimal pathway towards electrification of its LDV fleet. This accounting for geographical heterogeneity in the European electricity market is especially important in the short-to medium term, when there is a bottleneck in BEV manufacturing process that restricts global BEV supply (Cohen, 2020). This limited supply of BEVs should therefore be distributed in the regions where they will have the greatest mitigating effects. Over the longer term, the global production capacity for BEVs will likely be better equipped to address demand. At that point, the heterogeneities in the European electricity mix are expected to have decreased due to the Renewable Energy Directive 2018/2001/EU (European Commission), which stipulates national energy and transport decarbonization targets towards achieving significant shares of electricity from renewable sources across the EU. Thus, in the coming decade(s), the national difference in BEV climate impacts is also expected to decrease while BEV supply increases, facilitating a general Europe-wide electrification for reducing climate emissions.

We derived emission factors for average national electricity supply mixes to determine the lifecycle climate footprints for BEVs in Europe. Future work expanding on this study could be performed using a power dispatching model such as EMPS (Wolfgang et al., 2009) in conjunction with integrated assessment models, or adopting a consequential LCA approach to evaluate the effects of electrification policies. Furthermore, other factors that affect the lifecycle carbon intensity of LDVs, such as fuel or energy efficiency, may be considered in future work; many region-specific parameters, such as local climate, which affects battery life and range; driving patterns; and shares of diesel and gasoline ICEVs in the national fleet may affect the comparison between the two powertrain technologies. On a wider scope, given that resource and manufacturing bottlenecks may potentially slow BEV deployment, this study may be extended to account for these bottlenecks and determine the optimal electrification pathways.

Our work presents ReDyFEV, an open-source software tool that efficiently calculates the carbon footprints of BEVs across Europe using a systematic and consistent analysis framework. Such a parametrized regionalization is key to attain greater geographical and temporal granularity and coverage in LCA databases, while simultaneously increasing research efficiency and comparability. This study's findings should prove useful for guiding the current work of the European Commission's mandate to evaluate a potential LCA-based framework for assessing and reporting vehicle CO_2 emissions and for the short-term planning in both electric vehicle deployment and battery manufacturing localization for the most advantageous climate mitigation actions in the LDV sector.

CRediT authorship contribution statement

Christine Roxanne Hung: Conceptualization, Investigation, Formal analysis, Visualization, Software, Writing – original draft. **Steve Völler:** Conceptualization, Investigation, Writing – review & editing. **Maxime Agez:** Investigation, Writing – review & editing. **Guillaume Majeau-Bettez:** Conceptualization, Investigation, Writing – original draft. **Anders Hammer Strømman:** Conceptualization, Investigation, Project administration, Funding acquisition, Supervision, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

The authors gratefully thank Linda Ager-Wick Ellingsen for her invaluable input and discussion.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2021.129052.

Funding information

This research was made possible by financing from the Research Council of Norway (grant 255199) and the International Life Cycle Chair (CIRAIG).

Data availability

The electric power transmission losses data that support the findings of this study are available from the International Energy Agency, but restrictions apply to the availability of these data, which were used under license for the current study, and so are not publicly available. All other data generated or used during this study are included in this published article (and its Supplementary Information files) or are available free of charge from the relevant sources as cited.

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