The Complete Emissions Life Cycle Assessment of Electric Buses in the Australian Transport Sector

by Enoch Zhao

Thesis submitted in fulfilment of the requirements for the degree of

Doctor of Philosophy

under the supervision of

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CERTIFICATE OF ORIGINAL AUTHORSHIP

I, Enoch Zhao, hereby declare that this thesis is submitted in fulfilment of the requirements for the award of Doctor of Philosophy, in the School of Mechanical and Mechatronic Engineering, Faculty of Engineering and Technology at the University of Technology Sydney.

This thesis is wholly my own work unless otherwise referenced or acknowledged. In addition, I certify that all information sources and literature used are indicated in the thesis.

This document has not been submitted for qualifications at any other academic institution.

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I pour milk into my instant coffee mix before adding hot water because an Italian exchange student taught me that it prevents burning the coffee grounds. My default font is Times New Romans because my school used it back in 2001. I wanted to learn how to drive with three pedals, so for my first car I bought one with a manual transmission because James did the same several years before and "if he can do it, then so can I." He had to drive it home for me too. My wife once commented that I look good in black, therefore immediately after that comment I decided that I will be wearing the same colour for the next twenty-five years. And if I were to write down every one of the things my parents have taught me, from tying my own shoes to the philosophy of life, I suppose I could almost fill the world with books that would be written. Thus, I am a mosaic of the people I've ever met, even for the briefest moment.

In the same way, the completion of this thesis would not have been possible without the immense support, guidance, and assistance I have received. I had certainly enjoyed conducting research work and then writing this thesis. The lengthy page count is a good indicator that my love of writing could have launched me into a career as a novelist. But instead, by chance, I watched Iron Man 2 around the time I had to choose my year 11 subjects and I was so inspired (I blame you Robert Downey Jr.) that I decided to become an engineer instead. One thing led to another and now I have written my PhD thesis. So, as I compose each chapter of this thesis, I fondly reminisce the people who helped author this chapter of my life.

I would like to begin by thanking my principal supervisor, Dr Paul Walker, who from the start had opened doors and then mentored, encouraged, and trusted me as I completed my research. I am also immeasurably grateful to my co-supervisor, Dr Nic Surawski, for his invaluable insights and ongoing support. My special thanks to Scott Graham, Kevin Cook, Peter Bracken, Fatma Al-Widyan, Rami Haddad, and Peter Tawadros for their friendship and company. Special mention to Mighty Car Mods, FortNine, and Chris Fix.

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Finally, here's to Yamaha Motor Co. for building such beautiful motorcycles. For on these magnificent machines I have repeatedly experienced how two wheels truly do move the soul.

LIST OF PUBLICATIONS

Journal Articles

1. ***Zhao, E.**, May, E., Walker, P.D. & Surawski, N.C. 2021, 'Emissions Life Cycle Analysis of Charging Infrastructures for Electric Buses', *Sustainable Energy Technologies and Assessments*, vol. 48, pp. 1-14.

2. ***Zhao, E.**, Walker, P.D. & Surawski, N.C. 2021, 'Emissions Life Cycle Analysis of Diesel, Hybrid, and Electric Buses', *Journal of Automotive Engineering*, pp. 1-13.

3. ***Zhao, E.**, Walker, P.D., Surawski, N.C. & Bennett, N.S. 2021, 'Assessing the Life Cycle Cumulative Energy Demand and Greenhouse Gas Emissions of Lithium-Ion Batteries', *Journal of Energy Storage*, vol. 43, pp. 1-19.

Conference Papers

4. **^Zhao, E.**, Walker, P.D., Ong, A. & Al-Widyan, F. 2020, 'Measuring Road Conditions with an IMU and GPS Monitoring System', Asia-Pacific Vibration Conference (APVC) 2019, eds S. Oberst, B. Halkon, J. Ji & T. Brown, vol. 1, Springer, Sydney, Australia, pp. 95-101.

*****Articles related to this thesis.

^Publications made during the PhD candidature but do not relate to this thesis.

NOMENCLATURE

ABS	Australian Bureau of Statistics
ABPS	Australian Battery Performance Standard
ADB	Asian Development Bank
AGCER	Australian Government Clean Energy Regulator
APRAA	Auto Parts Recyclers Association of Australia
ARENA	Australian Renewable Energy Agency
BEV	Battery Electric Vehicle
BOM	Bill of Materials
BNEF	Bloomberg New Energy Finance
BSE	Battery Storage Equipment
C	Carbon (Graphite)
CDP	Center for Disaster Philanthropy
CED	Cumulative Energy Demand
CH_4	Methane
CO	Carbon Monoxide
CO_2	Carbon Dioxide
CO_2e	Carbon Dioxide Equivalent
DAWE	Department of Agriculture, Water, and the Environment
DEE	Department of the Environment and Energy
DoD	Depth of Discharge
DPIS	Department of Planning, Industry and Science
EIA	U.S. Energy Information Administration
EoL	End of Life
EPA	Environmental Protection Agency
EPRS	European Parliamentary Research Service
ESS	Energy Storage System
EV	Electric Vehicle
FCV	Fuel Cell Vehicle
GHG	Greenhouse Gas
GWP	Global Warming Potential
HEV	Hybrid Electric Vehicle
ICEV	Internal Combustion Engine Vehicle
IEA	International Energy Agency
IPCC	Intergovernmental Panel on Climate Change
L(R)	Lithium Rich
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCN	Lithium Cobalt Nickel
LCO	Lithium Cobalt Oxide
LCP	Lithium Cobalt Phosphate
LFP	Lithium Iron Phosphate
Li	Lithium
LIB	Lithium-Ion Battery
LMO	Lithium Manganese Oxide
LMR	Lithium Manganese Rich
LTO	Lithium Titanate Oxide
MftE	Ministry for the Environment
MoS_2	Molybdenum Disulphide
N_2O	Nitrous Oxide
NCA	Lithium Nickel Cobalt Aluminium Oxide
NMC	Nickel Manganese Cobalt
NOx	Nitrogen Oxides
OPC	Opportunity Pantograph Charger
OPR	Overhead Pantograph Rails
PEMS	Portable Emissions Measurement System
PHEV	Plug-In Hybrid Electric Vehicle
RDE	Real-World Driving Emissions
PM	Particulate Matter
S	Sulphur
SiC	Silicon Graphite
SiNT	Silicon Nanotube
SiNW	Silicon Nanowire
SoC	State of Charge
SOx	Sulphur Oxides
TfNSW	Transport for New South Wales
USyd	University of Sydney
WPT	Wireless Power Transfer

TABLE OF CONTENTS

TABLE OF FIGURES & TABLES

ABSTRACT

Australia is increasingly experiencing the environmental impact of global warming. In recent decades, society has gradually become increasingly aware of the harm caused to the global environment by excess fossil fuel consumption. Greenhouse Gas (GHG) emissions and their contribution to global warming are considered to be one of the most pressing environmental issues of the present day. Transportation is the third-largest contributor of GHG emissions in Australia, contributing to 18.9% of total GHG emissions. Therefore, there are strong and urgent incentives to reduce emissions from the transportation sector. This problem can be rectified through the electrification of the vehicle's powertrain; consequently improving energy efficiency, reducing GHG emissions, and yielding a number of additional benefits. Thus, transitioning the transport sector to electrified powertrains have been perceived as the optimal solution to decarbonise the transport sector. This thesis employs a technique known as Life Cycle Assessment (LCA) to properly quantify and assess the environmental impacts from the transport sector.

First, this research starts by introducing Australia's development in the transition to electrified heavy-vehicle powertrains, the LCA technique, research objectives, and the outline of this thesis. Next, this research conducted a study that evaluated and calculated the magnitude of GHG emissions produced from the implementation of electric bus charging stations. Results show that the operations phase is heavily dependent on the electricity grid-mixes carbon intensity and contributes the most greenhouse gas emissions (98.8%), followed by production (0.69%), recycling and disposal (0.48%), installation (0.01%), and transportation (0.01%). Then, an evaluation of the environmental impact of electricity generation and four different charging methods was conducted. The study finds that the optimal charging arrangement is to deploy electric buses with small battery capacity in urban and suburban settings, large battery capacity in highway settings, and recharge with opportunity pantograph chargers or stationary charging stations. Moving on, an LCA was conducted to investigate the production, assembly, transportation, maintenance, and decommissioning phases of diesel, hybrid, and electric bus production. The results show that the electric bus has a higher total environmental impact than the diesel and hybrid bus (18.2% and 14.7% higher, respectively). After that, this research assessed LCAs of Lithium-Ion Batteries (LIBs) from various literature sources and found that the average global warming potential and cumulative energy demand from LIB production were 187.26 kgCO_2e/kWh or 19.78 kgCO_2e/kg, and 42.49 kWh/kg, respectively. Finally, a summary and conclusion of this research as a complete entity concludes this dissertation.

1 CHAPTER 1: INTRODUCTION

1.1 TRANSITION TO ELECTRIFIED HEAVY-VEHICLE POWERTRAINS

Australia is increasingly experiencing the impact of global warming. Although Australia experiences bushfires annually, the recent 2019 – 2020 bushfires prompted the Australian government to declare a state of emergency in November 2019. From June 2019 to February 2020 more than 46 million acres of land have been burnt. At least 34 people, several firefighters, and more than one billion animals have tragically lost their lives (USyd 2020). Nearly 3,000 homes and several thousand buildings have been destroyed across the continent (CDP 2020). Climate change is lengthening the bushfire season and has worsened the current bushfire crisis. Without proper sustainable and substantial efforts to combat climate change fire risks will continue to escalate. From an economic point of view, the costs of fighting fires have increased. The Australian government has become increasingly constrained in its ability to share resources between countries, states, and territories (Climate Council 2019).

In recent decades, society has gradually become increasingly aware of the harm caused to the global environment by excess fossil fuel consumption. Greenhouse Gas (GHG) emissions and their contribution to global warming are considered to be one of the most pressing environmental issues of the present day. In 2015, Australia has committed to reducing GHG emissions by 26 ~ 28% on 2005 levels by the year 2030 (DAWE 2015). The New South Wales Government has set a goal of net-zero emissions by 2050 and has released a Net Zero Plan (Stage 1: 2020 – 2030) to fast-track emissions reduction over the following decade (DPIS 2020). The plan aims to reduce annual NSW emissions by 35.8 Mt by 2030 and reduce annual emissions by 35% on 2005 levels (from 160.7 Mt/year to 103.7 $MtCO_2e$/year).

Transportation is the third-largest contributor of GHG emissions in Australia. In the year to June 2019, transport accounted for 18.9% of total GHG emissions (DEE 2019e). These emissions and pollutions result in global warming through the greenhouse gas effect, consequently harming the environment and lives. Furthermore, fossil fuel and oil prices continue to increase. Therefore, there are strong and urgent incentives to reduce emissions from the transportation sector. Although public transportation by bus plays a key role in effectively mitigating traffic congestion in urban areas (Lin et al. 2019), a major contributor to fuel consumption and GHG emissions come from heavy commercial vehicle transportation.

In an effort to actively combat the increase of transportation GHG emissions, the European Union (EU) has implemented emission standards for most vehicle types. New standards are updated periodically, and new vehicle models introduced into the market must meet the current or planned standards (Williams & Minjares 2016). Transitioning the transport sector to electrified powertrains have been perceived as the optimal solution to decarbonise the transport sector (Hawkins et al. 2012; Lajunen & Lipman 2016; Hall & Lutsey 2018). With such strong and urgent incentives to reduce emissions from transportation, electrified powertrain technologies have been under rapid development.

Electrified powertrains offer other considerable advantages over the conventional Internal Combustion Engine Vehicle (ICEV) powertrains. Comparatively, BEVs, HEVs, and ICEVs share most of the same components, with the exception of different propulsion systems (internal combustion engine vs. electric motor) and energy storage (fuel tank vs battery). One need only look at the BEV advertisements to realise that the designation "Electric Vehicle (EV)" is synonymous with the keywords "green alternative", "environmentally sustainable", "zero emissions", etc. In reality, the tailpipe emissions are relocated from densely populated regions to the immediate vicinity of power plants, where electricity is generated that powers the BEVs. The traffic conditions in urban and suburban bus routes allow HEV and BEV buses to operate at low speeds and with frequent stops which assists with the recuperation of energy from regenerative braking. This is especially advantageous in high-congested areas with low air quality, such as city centres (Lajunen & Lipman 2016). Additionally, implementing electric recharging stations can be planned efficiently due to their key characteristic of servicing set routes.

There have been substantial development and impact evaluation results from the successful launch of electric powertrain systems in passenger vehicles, such as the Toyota Prius and Tesla Model 3. However, the extrapolation of the results into heavy commercial vehicles remains limited in terms of both research quantity and degree of success. To date, there are half a million BEV buses in operation globally, which represents 33% of the global bus fleet. China has over 400,000 BEV buses (approximately 99% of BEV buses in operation globally), Europe has 2,250, and the USA has 650 (Tigue, K. 2019; Henze, V. 2020). The Netherlands leads Europe in BEV bus adoption with 770 electric buses (15% of the entire Dutch fleet) at the end of 2019 and is expected to increase to nearly 1,400 by the end of 2020 (Maas, S. 2020).

At present, there are relatively few BEV buses in widespread use in the Australian transport sector. There exists a BEV bus adoption barrier due to their high purchasing cost, restricted performance, lack of supporting charging infrastructures, and extreme temperatures in some regions. The transition to BEV buses could take years to decades due to technological, operational, and infrastructural requirements. Most buses are driven 16 hours per day and would require much larger and more durable battery packs, thus increasing the mass of the vehicle. The BEV buses may suffer from volume or mass penalties, which would compromise the number of passenger/kilometres transported per day. Additionally, present battery technologies are comparatively low in energy density, meaning that the BEV buses require more charges to reach the same range as their diesel counterparts.

Understanding the environmental benefits of replacing conventional fossil fuel vehicles with EVs requires a life cycle perspective (Ellingsen et al. 2017). The initial rollout of infrastructure construction and equipment installation will produce significant GHG emissions and will be further exasperated if it goes without proper planning and coordination. It may not be cost-effective or time-efficient to construct new charging stations for BEVs only; indeed, depending on the size of the fleet or geographic coverage, transport operators have the ability to provide their own charging stations for their company-specific operations in their existing bus depots with the proper facilities for parking accommodation, servicing, and maintenance. There are also environmental concerns specific to the GHG emissions produced from BEV bus manufacturing and its associated processes, such as assembly, transportation, maintenance and disposal. The drive for electrified powertrains has raised the question of how BEV buses compare environmentally sustainable-wise with ICEV or HEV buses, especially with respect to LIB production.

The gradual transition to BEV buses would bring forth increases in electricity demands in order to satisfy operational requirements. Electricity generation around the globe varies considerably from country to country. On one hand, some countries represent an upper bound on fossil fuel reliance, such as South Africa, China, India, Indonesia, Australia, and Poland; where the majority of electricity generation is from combusting coal and natural gas. On the other hand, other countries such as France, Sweden, Switzerland, and New Zealand have higher shares of renewables and nuclear energy in their grid-mixes and are thus applying lower carbon-intense methods for electricity generation. The fossil fuels that are combusted to generate electricity comprises of black coal (bituminous coal), brown coal (lignite), natural gas, and oil. During the combustion process, GHG [Carbon Dioxide (CO_2), Methane (CH_4), and Nitrous Oxide (N_2O)] and other pollutants [Carbon Monoxide (CO), Nitrogen Oxides (NO_x), Sulphur Oxides (SO_x), and Particulate Matter (PM)] are emitted. The emission factors for CO_2 heavily depends on the carbon content of the fuel rather than the combustion conditions, thus CO_2 emissions can be accurately estimated based on the amount of fuel consumed during the combustion process (Jeon et al. 2010).

In 2018 global electricity generation reached approximately 26,614.8 ~ 26,700 TWh (IEA 2019; BP p.l.c 2019). Energy-related GHG emissions increased by 1.7% to 33,100 $MtCO_2e$ (IEA 2019). China, India, and the USA accounted for 85% of the net increase in GHG emissions, whereas there was a decline in emissions for France, Germany, the UK, and Mexico. From a global perspective, energy consumption from oil, natural gas, coal, and renewables increased by 1.3%, 4.6%, 0.7%, and 4%, respectively. The global energy demand was driven by the growing economy and severe weather conditions that led some parts of the world to increase energy demand for heating and cooling. Thus, the environmental sustainability of BEV buses has direct relations with the sustainability of the electric power production required to drive these vehicles (Hsu, T. 2013).

Understandably, there are concerns associated with electrifying the powertrain of the public transport fleet. To fully overcome the adoption barrier, it would be in Australia's best interest to look towards countries leading in transitioning to HEV and BEV buses and then adopt and tailor their methods to suit Australian conditions.

1.2 LIFE CYCLE ASSESSMENT

The increased awareness of the environmental impacts from transportation has increased interest in the development of methods to properly understand and address these impacts. Consequently, a technique known as Life Cycle Assessment (LCA) has been developed (ISO 14040:2006). LCA is one of several environmental management techniques and not all situations can be appropriately assessed via LCA. Typically, the economic or social aspects of a product are not addressed, although the life cycle approach and methodologies can still be applied to these aspects. The information developed by LCA studies can be used or assist with making comprehensive decisions. Thus, this thesis is guided by LCA principles in supporting the objectives of this research. The LCAs conducted in the corresponding chapters assist in three regards: to identify opportunities to optimise the environmental performance of electrified powertrains in heavy transportation at various phases in their life cycle, to inform and facilitate discussion among industry, governments, and policymakers, and to select relevant environmental performance indicators and measuring techniques.

The LCA conducted in the corresponding chapters addresses the current and potential environmental impacts from the study subject's entire life cycle, or Cradle-to-Grave (CTG). The investigated phases include but are not limited to: production, assembly, transportation, operations (use), maintenance, and EoL. The structure of an LCA study consists of four notable stages: goal and scope definition, Life Cycle Inventory Analysis (LCIA), impact assessment, and results interpretation.

The LCA scope includes the system boundary and level of detail, which are dependent on the study subject and the intended purpose of the study. The extent of the system boundary is bound by the objectives set by the study. The LCIA stage involves collecting and analysing the input/output data of a system regarding the study subject. The purpose of impact assessment is to supplement data on the environmental significance of the subject's Life Cycle Inventory (LCI) results. The last stage then interprets, summarises, and discusses the results obtained from the LCIA and impact assessment stages. This stage also provides the conclusions and any recommendations for future works.

1.3 RESEARCH OBJECTIVES

As described in the first two sections of this chapter, it is apparent that there is a major research gap for properly quantifying and assessing the life cycle environmental impacts of electric buses in the Australian transport sector. The novelty and innovation of this study are presented as the successful quantification and assessment of the environmental impact of transitioning the transport bus fleet to electrified powertrains by the application of a case study approach for Australia. Thus, this research aims to address this research gap through the application of four alternative case studies with the specific objectives to:

1. Calculate the life cycle GHG emissions from installing BEV bus charging stations in existing bus depots. This includes constructing a mathematical model of the Well-to-Wheel (WTW) energy consumption and GHG emissions during the operations (use) phase;

2. Construct mathematical models to (a) evaluate the GHG emissions produced from electricity generation, (b) define a break-even point for countries with high carbon-intense grid-mixes where the GHG emissions from electricity generation are equal to diesel fuel emissions, (c) analyse four different BEV bus charging methods with respect to their magnitude of GHG emissions, and (d) determine the optimal arrangement from the available charging strategies that contribute the least to climate change;

3. Calculate the life cycle GHG emissions from the production of a representative diesel, hybrid, and BEV bus. The calculation incorporates the GHG emissions released from the five phases of production, assembly, transportation, maintenance, and EoL. The operations phase is excluded from the calculation as it is not considered as specific to vehicle production; and

4. Critically review available life cycle studies that have assessed the environmental impact of Lithium-Ion Battery (LIB) production and have also provided detailed contribution analyses and transparent inventories. Determine the average life cycle values for Global Warming Potential (GWP) and Cumulative Energy Demand (CED). Investigate into LIB second-life applications.

To address the research gap and thus meet the research objectives, the four research objectives are consolidated into an entity termed the "Complete Life Cycle" model, as shown in *Figure 1*. The horizontal flow represents the equipment life cycle and incorporates the life cycles of the LIB, charging infrastructure, and the vehicle itself (research objectives 1, 3, and 4). The vertical flow represents the WTW life cycle (research objectives 1, 2 and 4), which is subdivided into Well-to-Tank (WTT) and Tank-to-Wheel (TTW) stages. The WTT stage focuses on the delivery of the energy carrier (fossil fuel for the diesel and hybrid bus and electricity for the BEV bus) from its source to the storage method of the vehicle. The TTW stage focuses on the conversion of energy to drive the wheels (chemical or electrical energy to mechanical energy). Thus, the primary outcomes of this research will be a comprehensive evaluation of the life cycle environmental sustainability and benefits of transitioning the Australian transport bus fleet to electrified powertrains, and demonstration of the most appropriate methodologies to effectively mitigate the carbon intensity of the electrical energy supply, which is the primary concern of the operations phase.

1.4 OUTLINE OF THIS THESIS

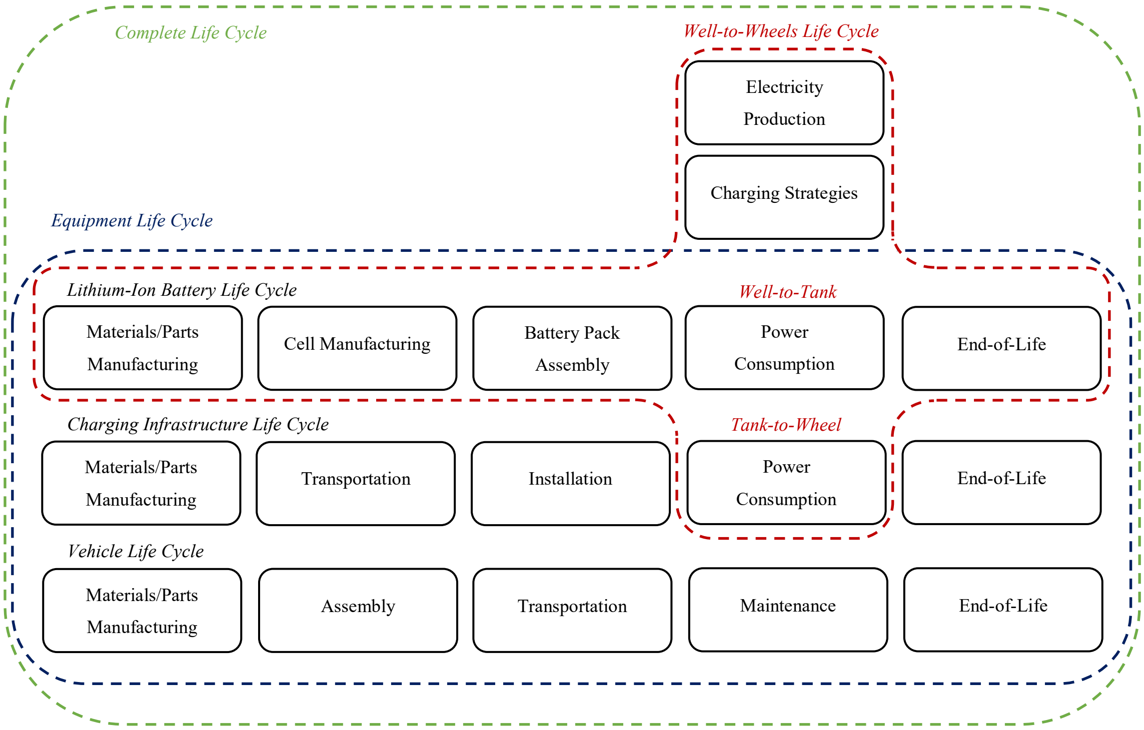


Figure 1 – Simplified view of the research scope, based on the WTW and equipment flows described by Nordelöf et al. (2014).

This dissertation is built on six chapters individually investigating a specific research area. To facilitate a proper understanding of the specific research area, each corresponding chapter incorporates a dedicated literature review section to provide the relevant background information. The results of each chapter are then synthesised into the conclusion of this thesis. The overview of the corresponding chapters is as follows. First, Chapter 1 introduces the background, statement, and objective of this research. Then, Chapter 2 presents a case study approach for Australia and evaluates the life cycle GWP when implementing BEV bus charging stations into existing bus depots in conjunction with the transitioning of the Australian transport bus fleet into electrified powertrains. Next, Chapter 3 evaluates the environmental impact of electricity generation and four different charging methods: stationary charging station, Wireless Power Transfer (WPT), Opportunity Pantograph Charger (OPC), and Overhead Pantograph Rails (OPR). This chapter further estimates the GHG emissions produced from different operation charging strategies to determine the optimal arrangement that benefits bus operators and also contribute the least to GWP. After that, an LCA of diesel, hybrid, and BEV bus production is conducted in Chapter 4. Chapter 5 then assesses LCAs of LIBs from various literature sources that provide detailed contributions and transparent inventories. In this chapter, the average life cycle emissions, the disparity in GWP and Cumulative Energy Demand (CED) values, and the significance of battery End-of-Life (EoL) are examined and documented. Finally, Chapter 6 summarises the results from all previous chapters with concluding remarks and recommendations.

2 CHAPTER 2: LCA OF CHARGING INFRASTRUCTURES[1]

2.1 INTRODUCTION

This chapter addresses the charging infrastructure life cycle segment of the Complete Life Cycle model. To address the first research objective on the central theme of BEV bus charging infrastructures and their respective environmental impacts, a case study approach is applied for Australia and calculates the magnitude of greenhouse gases produced from the implementation of electric bus charging stations into existing bus depots concurrent with the transitioning of the commuter bus fleets into electrified powertrains. First, the methodology section defines the scope, system boundary, and functional unit adopted for this study. In this section, the readers are introduced to the representative bus and charge station chosen for analysis. Next, a comprehensive and in-depth LCA of BEV bus charging stations is conducted and incorporates the GHG produced from the production, transportation, installation, operations, and decommissioning phases. Finally, the results from a detailed estimation of the amount of emissions produced throughout the life cycle of a charging station, followed by a discussion on the environmental sustainability of BEV bus charging infrastructures conclude this chapter.

2.1.1 ASSESSING LCA STUDIES ON CHARGING INFRASTRUCTURES

To provide the relevant background information relevant to the context of this chapter, the following are reviews of LCA studies conducted by researchers regarding charging infrastructures. Lucas et al. (2012) presented a methodology to evaluate the energy use and GHG emissions from the construction, maintenance, and decommissioning of supporting infrastructures for BEVs in Portugal. Three types of chargers were analysed: home, normal, and fast. Although the study accounted for the construction and installation phases of the charging points, the authors did not account for transportation. Their results showed that the energy supply infrastructure for BEVs is more carbon and energy-intensive than the conventional refuelling stations. However, the authors suggested that future upgrades to the Portuguese electricity grid-mix to incorporate higher shares of low emission sources could lower the carbon and energy intensity from a life cycle perspective. Similarly, Zhang et al. (2019) conducted a comprehensive environmental analysis of the four main types of BEV chargers in China. The study evaluated the energy consumption and GWP in their manufacturing, use, and end-of-life stages, but have omitted the transportation stage. The authors did however report the individual stage's life cycle GWP, thus the use stage produced the highest GWP, followed by manufacturing, and then recycling/disposal. Their results show that

[1] The contents of this chapter have been adapted from the publication: **Zhao, E.**, May, E., Walker, P.D. & Surawski, N.C. 2021, 'Emissions Life Cycle Analysis of Charging Infrastructures for Electric Buses', *Sustainable Energy Technologies and Assessments*, vol. 48, pp. 1-14.

home chargers contributed the least to energy consumption and GHG emissions, followed by public AC and DC charges, and finally public mix chargers. This was explained as due to the differences in construction materials and charging technologies. As these two studies differ in their functional units, it makes it difficult to adequately compare their numeric results.

A more relatable study to the aim of this chapter is one analysed by Nansai et al. (2001) on the life cycle of public charging stations in eligible sites in Japan. Their study included the production, transportation, and installation phases, but had excluded the decommissioning phase. The analysed charging station was equipped with a charger, battery storage unit, and a charger stand. The authors calculated the environmental impact of the cumulative value of 14,000 charging sites throughout the country. In summary, the production, transportation, and installation of a single charging station produced 3,857.1 kgCO_2e, 128.6 kgCO_2e, and 283.6 kgCO_2e, respectively. The summation of total GHG emissions amount to 4,269.3 kgCO_2e, however, if this study was to assume the same emissions value for the decommissioning phase as the production phase, then the additional decommissioning phase increases the total GHG emissions to 8,538.6 kgCO_2e.

In summary, all three reviewed studies focused on the LCA of charging infrastructures for light-duty BEVs. Although the studies are relevant and their methodologies and results can be used for comparison, a detailed analysis is required for the context of BEV buses.

2.2 METHODOLOGY

2.2.1 SCOPE DEFINITION

In general, life cycle assessment is an evaluation method used to assess the environmental impact of a process, product, or service throughout the entire service life of the subject (Nugent & Sovacool 2014). The study in this chapter, in particular, performs an emissions LCA of implementing electric bus charging stations in existing bus depots in Sydney Central Business District (CBD) and Inner West regional areas. The City of Sydney was chosen as it is the most populous city in Australia and suffers from significant traffic congestion. The LCA conducted in this study is prepared according to the ISO 14040:2006. The focus is on bus routes that consist of city, suburban, and freeway driving. The objective of this study is to calculate the magnitude of GHG emissions produced from infrastructure constructions and equipment installation for the electric bus fleets and to evaluate and discuss the environmental implications for the transition to electric buses.

Furthermore, this study intends to provide an update of current emissions caused by manufacturing, transportation, electricity production associated with charging infrastructures. However, it should be noted that this is only going to be current at the time it was authored and may be superseded at the time it is being read. Therefore, it is recommended to use this work as a guide and reflection.

2.2.2 SYSTEM BOUNDARY

The topology of Sydney CBD and Inner West regional areas is beneficial in representing real-world driving under everyday driving conditions, route distances, operation logistics, and recharging times. Sydney is set to grow by 30% by 2031, which will result in an increase in population by 1.6 million people. More than 220 million bus trips consisting of over 600 bus routes are made each year in Sydney. During peak morning traffic, more than 1,000 (> 40%) bus services travel through the Sydney CBD (TfNSW 2013). It is likely that in the near term BEV buses will be operating in heavy trafficked and short-distanced routes. A BEV bus was selected as the representative electric vehicle, including all technical specifications, along with various representative duty cycles and routes (Hall & Lutsey 2019). The charging needs are assessed for buses performing each of these duty cycles in terms of hours of charging per day at both fast and normal charging stations.

This study performs a GHG emissions LCA of a BEV bus charging station specific to Australian conditions by examining the five phases of production, transportation, installation, operations, and decommissioning (see *Figure 2*). Production involves the investigation of the main GHG pollutants emitted during the manufacturing of components required for the charging station. Transportation accounts for the emissions based on the assumption of determining distances from the factory and accounting for the distance the device is required to be transported. Additionally, a standard truck is determined to use for transportation. To avoid unnecessarily complicated calculations with the limited data available, the study excludes any transportation emissions outside of Australia (e.g. sea transport) from the system boundary and only accounts for the transportation emissions produced within the country. The installation phase includes the processes required to install the charging station to an existing depot in Sydney. Operations examine the electricity required to charge a BEV bus. The charging cycles of the BEV buses are determined, and the data is then extrapolated to estimate the total amount of electricity consumed over the lifetime of the BEV bus. The emissions from this electricity consumption will be assessed via the carbon intensity evaluation of the average Australian grid-mix. Finally, the decommissioning of the device involves equipment disassembly, transportation, materials segregation, recycling, and disposal. The scope of recycling is limited to the process prior to implementing the recycled materials into new products. The estimation of the amount of emissions produced throughout the life cycle of a charging station allows for an accurate comparison of BEV buses against the current bus fleet in Sydney.

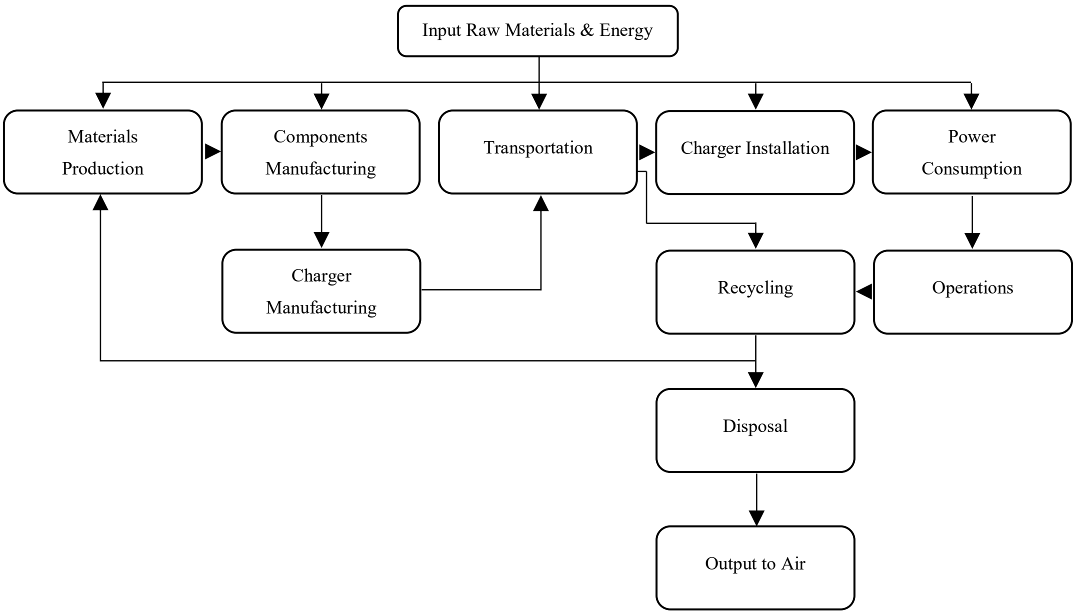


Figure 2 – System boundary of a charging station life cycle.

2.2.3 FUNCTIONAL UNIT

The functional unit of the environmental load is defined as a unit mass of GHG – or Carbon Dioxide equivalent (CO_2e) – per unit of energy/material production: **$kgCO_2e/kWh$** or **$kgCO_2e/kg$** (McIntyre et al. 2011). In the operations phase, the functional unit is defined as **$kgCO_2e/100km$**. The Carbon Dioxide Equivalent (**$kgCO_2e$**) of other greenhouse gases is calculated by multiplying the mass of the gas by the gas' Global Warming Potential (GWP). GWP is defined as the measure of energy the emissions of a unit mass of gas will absorb over a given period (usually 100 years), relative to the emissions of a unit mass of reference gas CO_2 (1): CH_4 (28) and N_2O (265) (IPCC 2014; AGCER 2016; EPA 2017). This study, therefore, uses this functional unit as a measure to provide a reference to the inputs and outputs defined and incorporated into the LCA study.

2.2.4 LIFE CYCLE INVENTORY ANALYSIS

This section introduces the chosen routes, representative bus, and charging station specifications used for analysis. It is assumed that all BEV buses are to be fully charged at 60 kW overnight (up to 6 hours) at the home bus depot. Fast charging provides the operating range needed for the assumed travel requirements. This study assumes charging stations in the home bus depot only, fast charging time is therefore optimised to maximise the available operating time during the day. Thus, the total amount of fast charging time required for each day is calculated. The BYD K9 BEV bus is capable of charging at 80 kW and has the potential of saving time. However fast charging expedites the optimum and useful life of the $LiFePO_4$ (lithium-ion) batteries. These batteries have limited charge-discharge life cycles and their internal integrity will noticeably degrade upon reaching the maximum limit of the life cycle, hence restricting the charging capacities (Ivankov, A. 2018).

2.2.4.1 ROUTE SPECIFICATIONS

Four different bus routes and track lengths are assessed via Google Maps (see *Figure A1 – 4*), as shown in *Table 1*. The routes chosen are a close representation of urban, suburban, and highway road and traffic conditions. The route settings are determined by driving distance, service frequency, and speed. Thus, routes 470 and 309 best represent urban conditions with their short driving distances, high service frequencies, and low operating speeds. Route 550 operates in mixed traffic on residential and commercial streets, and hence bests represent suburban conditions. Then, on the other end of the spectrum, route 607X best represent highway conditions with longer driving distance, low service frequency, and high speeds.

Table 1 – Specifications of selected bus routes.

	Bus Route Number			
	550	470	607X	309
Trip Distance (km)	~16.5 km	~8.5 km	~38 km	~11 km
Peak Service Frequency (min/bus)	10 – 20 min	3 – 15 min	20 min	10 min
Speed Range (km/h)	20 – 60 km/h	10 – 40 km/h	20 – 100 km/h	10 – 50 km/h

Realistically, the initial deployment of BEV buses will most likely be limited to short routes close to the operating bus depots. The gradual adoption of BEV bus use will then encourage and increase the demand for charging stations, thus progressively deploying BEV buses on a much broader variety of bus routes throughout the city.

2.2.4.2 REPRESENTATIVE BUS SPECIFICATIONS

This study has chosen the Chinese-made BYD K9 BEV bus as the representative bus (see *Figure A5*). The chosen bus specifications, such as dimensions and passenger capacity, closely matches the current bus fleet in Australia. Additionally, BYD dominates the global market in the BEV bus business (Hampel 2019) and can provide the necessary data needed to conduct an LCA. To simplify the assessment, it is assumed that the same type of BEV bus to be deployed on the chosen routes. Key specifications for the BEV bus in this paper are provided in *Table 2*.

Table 2 – Key specifications of a BYD K9 BEV bus.

Specifications	
Wheelbase (m)	6.2 m
Length (m)	12.0 m
Width (m)	2.6 m
Height (m)	3.2 m
Mass (kg)	14,400 kg (kerb mass), 19,700 kg (Gross Vehicle Mass)
Passenger Capacity	35 seated, 27 standing

Top Speed (km/h)	96 km/h
Max Gradeability (%)	≥17%
Acceleration (m/s^2)	2.5 m/s^2 (0-50 km/h in 20 seconds)
Range (km)	250 – 350 km
Motor Type	AC Synchronous (in-wheel motors)
Max Power (kW) | Torque (Nm)	2 x 150 kW | 2 x 550 Nm
Battery Capacity (kWh | Ah)	324 kWh | 600 Ah
Battery Composition	Lithium Iron Phosphate ($LiFePO_4$)
Power Consumption (kWh/km)	1 – 1.2 kWh/km (100 – 120 kWh per 100 km)
Charging Capacity (kW)	80 kW
Charge Time (h)	3 h @ 80 kW (fast), 6 h @ 60 kW (normal)

2.2.4.3 CHARGING STATION SPECIFICATIONS

A locally produced Tritium BEV charging station was chosen (Model Veefil^{pk}, dual cable)[2] as the representative charging station (see *Figure A6*). The key specifications of the charging station are shown in *Table 3*. It is a high energy efficient, minimal maintenance, scalable, and flexible High Power Charging (HPC) system for commercial operators. The system includes a user unit, a power unit, and a control unit that is capable of delivering up to 350 kW of power. The charging station can be installed outdoors with an International Protection Rating of IR65 and Impact Rating of IK10. The user unit is slim in design and requires very little space for installation. Two user units operate on one power unit. The system is scalable by adding more power units when there is more charging demand. The units are cooled internally with ethylene glycol ($C_2H_6O_2$). One power unit powers up to two user units and can be placed independently from each other. The control unit contains the communication unit and is the central control system for site power and load management. To simplify the assessment, it is assumed that the same type of charging station is installed in the bus depots of the four chosen routes. Furthermore, it is assumed that there is only one BEV bus per charge station and that both bus and charger have the same service life.

Table 3 – Key specifications of a Tritium BEV charging station.

Specifications	Value
User Unit	
Connectors	CCS Type 1, CS Type 2 & CHAdeMO Single or Dual Cable Option
Output Power (kW)	Up to 350 kW
Output Voltage (V)	Up to 920 Vdc
IP Rating	IP65
Efficiency (%)	98.5%
Operating Temperature (ºC)	-35ºC ~ 50ºC
Dimensions (mm)	1,998(H) x 980(W) x 525(D) mm
Weight (kg)	260 kg

[2] Tritium Pty Ltd. 48 Miller Street, Murarrie QLD 4172 Australia (Tritium 2018).

Cable Length (m)	4.3 m
Total Harmonic Distortion (THD) (%)	< 5%
IK Rating	IK10
Power Unit	
Input	2 x 480 V, 3-Phase, 50 Hz
Output	2 x 95 Vdc, 350 kW
IP Rating	IP54
Efficiency (%)	> 98%
Power Factor	0.99
Operating Temperature (ºC)	-35ºC ~ 50ºC
Network Connection	Ethernet to User Unit and Control Unit Site Power Control
Weight (kg)	700 kg
Dimensions (mm)	2,350(H) x 603(W) x 1,230(D) mm
IK Rating	IK10
Control Unit	
Weight (kg)	220 kg
Dimensions (mm)	2,350(H) x 603(W) x 1,230(D) mm
IK Rating	IK10
Power Supply	Battery-Backed UPS Functionality for Reliable Telemetry at All Times

2.2.5 MASS, RANGE, AND TIME LIMITATIONS

BEV buses may encounter foreseeable challenges regarding hauling additional weight and spending time charging. Assuming the $LiFePO_4$ battery energy density of 0.12 kWh/kg (Rydh & Sandén 2005a; Majeau-Bettez, Hawkins & Strømman 2011; Dunn et al. 2014; Dunn et al. 2015; Lu et al. 2016; Ambrose & Kendall 216; Hao et al. 2017a; Yu et al. 2018; Ioakimidis et al. 2019), the 324-kWh battery would weigh at least 2,700 kg. However, the official specifications for current city transit buses within the same category in Sydney (e.g. Volvo B8RLE and Scania K280UB) shows that the conventional diesel bus weighs approximately 19,000 kg and can carry the approximate number of passengers as the BEV bus (65 and 63 passengers, respectively) (Volvo 2019; Scania 2019). Therefore, there is next to no loss in passenger capacity, albeit the BEV bus weighs approximately 700 kg more.

There exists the challenge of operating range. The BEV bus has an approximate operating range of 250 km per charge (depletion to no less than 20% state-of-charge to avoid damaging the LIB). According to the Australian Bureau of Statistics (2019), the average diesel bus fuel consumption is 29.2 L/100km. The current city transit diesel buses (e.g. Volvo B8RLE and Scania K280UB) have 300-litre fuel tanks. The average fuel range of the conventional diesel bus is calculated to be approximately 950 km per tank. Since operation hours vary across bus operators, this study designates an operation time of 16 hours per day. *Table 4* shows that the conventional diesel bus full tank of fuel makes 3.8 times more trips than the BEV bus on a full charge. Only the BEV bus for route 470 is theoretically capable of covering the maximum number of trips in a 16-hour day on a single charge. However, this analysis does not consider auxiliary equipment use (e.g. air conditioning) or energy recovery (regenerative braking). BEV buses will be disadvantaged due to the long charging times (3 hours fast charging minimum), which can also be reflected in the fewer amount of passengers transported by the BEV bus compared to the diesel bus.

Table 4 – Bus route specifications.

Bus Route Number	550		470		607X		309	
	BEV	Diesel	BEV	Diesel	BEV	Diesel	BEV	Diesel
Trip Distance (km)	~16.5 km		~8.5 km		~38 km		~11 km	
Min. Number Trips per Full Charge/Tank	15	57	29	111	6	25	22	86
Number of Passengers Transported	764	1103	1482	2141	332	479	1145	1655

2.2.6 INFRASTRUCTURE ASSESSMENT AND ANALYSIS

This section calculates the total amount of emissions associated with the implementation of electric bus charging stations in existing bus depots, as well as the emissions produced when generating the electricity to charge the BEV bus. The GHG emissions (CO_2, CH_4, and N_2O) of the charging stations are calculated from the production, transportation, installation, recycling, and disposal phases.

2.2.6.1 GREET© MODEL

There is very limited data available in Australia in the public domain relating to materials processing emissions. Therefore, the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET® 2019) model is utilised to estimate the GHG emissions rate per unit of material weight. Sponsored by the U.S. Department of Energy's Office of Energy, Efficiency, and Renewable Energy, the GREET model is an analytical tool that simulates the emissions output and energy consumption of a diverse range of vehicles and fuel combinations. The model considers the complete life cycle (Cradle-to-Grave) of fuels production, raw material mining, material production (virgin and recycled), and the disposal of materials.

2.2.6.2 PRODUCTION PHASE

2.2.6.2.1 MATERIALS BREAKDOWN

The production phase incorporates large-scale resource extraction, raw material processing, and manufacturing of the final product. For this phase, an input-output approach is applied to estimate the emissions of a charging station. First, the weight of each constituent component of the charging equipment (input) is determined. The product specifications are used to track the materials used in constructing the charging station. The charging station unit body is assumed to be made of cold-rolled powder-coated aluminium with a thickness of 1 mm. Next, the weight of each component is estimated by multiplying the estimated volume (m^3) by its associated density (kg/m^3). The emissions produced from these materials are calculated using the cradle-to-grave emissions data extracted from the GREET® 2019 model. Here, an estimation of the proportional weight of each material type is made that contributes to the total weight of each unit given in the product specifications. Then, *Table 5* shows the GHG emissions intensity per unit of material weight (output) is determined; e.g. $kgCO_2e/kgAl$ for aluminium. Lastly, the total emissions of each component are calculated by multiplying the GHG emissions intensity with the weight of each component. *Table 6* shows the estimated weight of each material and the calculated emissions produced in this phase. The environmental load from the production phase is designated as $\boldsymbol{E_{production}}$ ($kgCO_2e$) hereinafter.

Table 5 – Emissions intensity per unit of material weight.

Emissions/Material (kg/ton)	CO_2	CO	NO_x	SO_x	CH_4	N_2O
Aluminium (Virgin Production)	7,085.3	2.7	5.9	29.3	12.6	0.1
Aluminium (Recycled Production)	1,545.2	1.0	1.6	2.8	4.1	0.0
Copper	2,570.6	2.3	6.1	131.8	5.1	0.1
Glass	1,065.9	0.6	1.6	1.1	2.2	0.0
Plastic (PVC)	1,959.9	3.0	2.9	12.0	14.9	0.1
Steel (Virgin Production)	2,623.9	22.0	2.6	10.7	4.4	0.0
Steel (Recycled Production)	1,160.0	3.7	1.0	2.0	2.6	0.0
Rubber	3,283.3	2.0	4.6	12.5	7.0	0.8
Ethylene Glycol	2,870.2	1.7	4.3	11.2	7.1	0.1

Source: GREET® model (2019).

Table 6 – Charging equipment materials breakdown.

Material	Estimated Weight (kg)	Emissions (kg)					
		CO_2	CO	NO_x	SO_x	CH_4	N_2O
User Unit							
Aluminium	39.1	277.0	0.1	0.2	1.1	0.5	4.30E-03
Copper	37.2	95.7	0.1	0.2	4.9	0.2	1.79E-03
Glass	0.2	0.2	0.0	0.0	0.0	0.0	3.31E-06
Plastic	20.0	39.2	0.1	0.1	0.2	0.3	1.37E-03
Steel	50.0	131.2	1.1	0.1	0.5	0.2	1.12E-03
Ethylene Glycol	109.7	314.9	0.2	0.5	1.2	0.8	6.59E-03
Rubber	3.8	12.5	0.0	0.0	0.0	0.0	3.09E-03
Total	260.0	870.6	1.5	1.1	8.1	2.0	1.83E-02
Power Unit							
Aluminium	80.0	566.8	0.2	0.5	2.3	1.0	8.80E-03
Copper	500.0	1,285.3	1.2	3.0	65.9	2.6	2.40E-02
Plastic	20.0	39.2	0.1	0.1	0.2	0.3	1.37E-03
Steel	100.0	262.4	2.2	0.3	1.1	0.4	2.25E-03
Total	700.0	2,153.7	3.6	3.8	69.6	4.3	3.64E-02

Control Unit							
Aluminium	80.0	566.8	0.2	0.5	2.3	1.0	8.80E-03
Copper	100.0	771.2	0.7	1.8	39.6	1.5	4.80E-03
Plastic	10.0	19.6	0.0	0.0	0.1	0.1	6.84E-04
Steel	30.0	78.7	0.7	0.1	0.3	0.1	6.74E-04
Total	220.0	922.2	1.1	1.2	16.0	1.8	1.50E-02
Cables							
Copper	200.0	514.1	0.5	1.2	26.4	1.0	9.59E-03

2.2.6.3 TRANSPORTATION PHASE

A second phase involves transporting the finished product from the loading site (Port Botany) to the bus depots. An Isuzu NLR 45-150 truck (4-cylinder 3.0-litre turbo diesel, Euro 5 emissions standard) is chosen as the representative delivery truck. At 4.5 tonnes Gross Vehicle Mass (GVM) the truck is considered as a Light Rigid vehicle (GVM up to 8 tonnes) in NSW. It should be noted that travel speeds and fuel efficiencies will differ due to varying traffic conditions. This study assumes the average diesel fuel consumption to be 29.1 L/100km (ABS 2019). Transport emissions from the truck travel prior to reaching the loading site (Port Botany) are not included in the calculations.

In the absence of recorded tailpipe emissions data, it is possible to roughly estimate the amount of emissions produced in transportation. The average Well-to-Wheel (WTW) diesel fuel emission conversion factor is 2.7 – 2.9 $kgCO_2/L$ (McKinnon & Piecyk 2011; ADB 2016; MftE 2019). The estimated CH_4 and N_2O emissions (in $kgCO_2e$) given from the data given by the Department of the Environment and Energy (2019a) are also incorporated into the calculations. *Table 7* shows the GHG emissions produced for this stage, calculated from driving distance and the amount of diesel fuel consumed. Thus, the environmental load from the transportation phase is designated as $\boldsymbol{E_{transportation}}$ ($kgCO_2e$) hereinafter.

Table 7 – Approximate emissions ($kgCO_2e$) produced from charging station transportation.

	Ryde Bus Depot (Route 550)	Leichhardt Bus Depot (Route 470)	Seven Hills Bus Depot (Route 607X)	Randwick Bus Depot (Route 309)
Distance from Port Botany (km)	25.1	17.3	44.4	9.7
Diesel Fuel Consumed (L)	7.3	5.0	12.9	2.8
GHG Emissions ($kgCO_2e$)	20.3	14.0	36.0	7.9

2.2.6.4 INSTALLATION PHASE

This next phase involves calculating the GHG emissions produced from the power consumption during the installation phase. The environmental load from the installation phase is designated as $\boldsymbol{E_{installation}}$ ($kgCO_2e$) hereinafter. The charging equipment setup involves installing a user unit, power unit, and control unit in an existing bus depot. The power unit is capable of supplying power to two user units at 350 kW each, from a maximum distance of 100 m from the user units. Tritium claims that the system requires minimal maintenance, therefore it is assumed that any emissions produced by maintenance, if any, would also be negligible.

2.2.6.4.1 SYSTEMS INSTALLATION

Installing the charging equipment involves a two-step process. The first step involves laying down cables to connect the control unit to the electricity grid. The second step involves installing the control and power units in the bus depot. With the entirety of the system weighing 220 kg, 260 kg, and 700 kg and contact surface areas of 0.74 m^2, 0.51 m^2, and 0.74 m^2, respectively, the largest normal force to the ground is 9.28 kPa. According to the Australian Concrete Structures Standard (AS 3600-2009), the minimum standard for concrete in Australia is 25 MPa. Since the forces applied here is much lower than the minimum standard, it can be assumed that the units will be attached to the ground without any additional procedures.

According to Electrical Installations – Selection of Cables (AS/NZS 3008:2017), the cross-section of the cables is required to be 70 mm^2. The size requirements are too large for the cables to be suspended above the ground, therefore the cables are buried in a trench with the use of an excavator. The activity of laying of the cables is identified as the main source of emissions, due to the operations of machinery. Emissions from other minor activities are considered negligible and are not considered in this assessment. The emissions produced from machinery is calculated based on the time in operation.

A Kubota KX41-3V excavator is chosen with a rate of work of 4.5 m^3/h (Methvin 2012) and an average diesel fuel consumption of 3.5 L/h. An operations time of 9 hours is assumed to dig a trench 80 m long, 0.25 m wide, and 0.9 m deep. To simplify the assessment the worksite is assumed to have a hard ground that requires a moderate work rate. Based on the above work rate, the total amount of fuel consumed is 31.5 L and produces 86.3 $kgCO_2e$. Thus, the environmental load from the installation phase is $\boldsymbol{E_{installation}}$ **= 86.3** $\mathbf{kgCO_2e}$.

2.2.6.5 OPERATIONS PHASE

This section addresses the vertical flow TTW segment shown in *Figure 1*. The TTW segment represents the operations phase and calculates the amount of GHG emissions produced when charging a BEV bus. First, the carbon intensity of the current Australian grid-mix is assessed to determine the electricity emissions factors. Next, the charging cycles of the BEV buses are determined, and the data is then extrapolated to estimate the total amount of electricity consumed over the lifetime of the BEV bus. Finally, the operations lifetime emissions of a BEV bus is then calculated from the emissions factors and the total electricity consumption.

The LCA results of BEVs are largely dependent on the environmental impact of the electricity grid-mix of a region, as each region has its unique resources and environmental characteristics (Woo, Choi & Ahn 2017). Additionally, calculating the operations GHG emissions allows for performance comparisons of different countries (Moro & Lonza 2018). Although it is a popular selling point for many automotive manufacturers, to proclaim that BEVs when in operations have absolute zero emissions is vastly inaccurate. Indeed, in the case of BEVs the vehicle tailpipe emissions are shifted to electricity generation emissions (Notter et al. 2010). Naturally, the BEVs operations phase will result in high GHG emissions whenever high shares of fossil fuels electricity generation dominate the grid-mix. In Australia, there are multiple electricity generation sources with each source producing GHGs in varying magnitudes. Estimation for electricity generation was published in March 2019 by the Department of the Environment and Energy (2019d) for the 2018 calendar year. Australia's electricity generation sources vary significantly across the country. In the calendar year 2018, the total electricity generation in Australia was estimated to be approximately 261,405 GWh and an increase of approximately 1% compared with the previous year. The electricity generation estimation for the calendar year 2018 and 2017 – 2018 was based on the data from the Australian Energy Market Operator (2020) and the Australian Government Clean Energy Regulator (2016). The statistics cover all on- and off-grid electricity generation across all states and territories in Australia, including electricity generated by power plants, businesses, and households (DEE 2019a,b,c). The environmental load from the operations phase is designated as $\boldsymbol{E_{operations}}$ (kgCO_2e) hereinafter.

Electricity generation is the largest source of GHG emissions in Australia's inventory, accounting for 33.1% of emissions in Quarter 1 of 2019 (up to March 2019) (DEE 2019d). The calendar year 2017 was the third consecutive year that Australia's emission has risen, reaching 556.4 MtCO_2e. The generation of electricity contributed to 33%, equating to 184.5 MtCO_2e (Bourne et al. 2018). Since 2005, emissions produced by electricity generation have decreased by 6.4%, where at that time the electricity sector reached 197 MtCO_2e. It is worth noting that the GHG emission factors depend heavily on the carbon content of the fuels, rather than the conditions during the combustion process. It is, therefore, accurate to estimate the GHG emissions based on the amount of fuel consumed (Jeon et al. 2010). Here, the operations functional unit is **kgCO_2e/100km**.

2.2.6.5.1 ASSESSING THE GHG EMISSIONS FROM RENEWABLES ENERGY

This section of the study assesses the GHG emissions produced by renewable energy technologies. Renewable electricity generation systems do not generate significant GHG emissions, nevertheless, processes comprising of raw material production, manufacture, construction, transportation, and decommissioning contributes to non-negligible life cycle emissions (Lund & Biswas 2008).

Nugent & Sovacool (2014) critically assessed 153 life cycle studies covering the life cycle GHG emissions for wind and solar PV technologies. The authors screened the studies and filtered out 41 studies that were deemed the most accurate, relevant, original, and recent. From these studies, their assessment showed that for both wind and solar PV technologies cultivation and fabrication (71.5% and 71.3%, respectively) contributed to the most to life cycle GHG emissions, followed by construction (24.0% and 19.0%, respectively) and operation (23.9% and 13.0%, respectively). Interestingly, their assessment revealed that the decommissioning phase often recycles materials back into future production processes, thus creating 'negative' emissions (-19.4% and -3.3%, respectively). For wind energy, the study reported a range of 0.0004 ~ 0.36 $kgCO_2e/kWh$ and a mean value of 0.03 $kgCO_2e/kWh$. For solar energy, the study reported a range of 0.001 ~ 0.22 $kgCO_2e/kWh$ and the mean value of 0.05 $kgCO_2e/kWh$. Therefore, the authors concluded that wind and solar PV technologies are not actually emissions-free.

In their study, Ocko & Hamburg (2019) analysed the environmental impacts of hydropower over time, including the GHG emissions produced from constructing and creating the reservoir, and compared the data with the environmental impacts of both fossil fuel and other renewable technologies. The study utilised a comprehensive database of net life cycle emissions of nearly 1,500 plants operating in over 100 countries. For hydropower, the study reported global median emissions of 0.07 $kgCO_2e/kWh$ and global weighted average emissions of 0.25 $kgCO_2e/kWh$. From the mounting evidence shown in the study, hydropower plants produce considerably higher emissions than wind and solar technologies. However, the authors articulated that the magnitude of GHG emissions is influenced by meteorological conditions, facility features, and environmental characteristics.

Similarly, Thornley et al. (2015) conducted a full life cycle assessment on biomass technologies to establish the total GWP of their entire life cycle. The study reveals that the electricity generation from biomass incorporates a long string of processes and results in an emissions factor of 55 ~ 60 $kgCO_2e/MWh$ (or approximately 0.06 $kgCO_2e/kWh$). In their study, Kadiyala, Kommalapati, & Huque (2016) assessed 22 studies that analysed the LCA of the two biomass electricity generation methods: biomass-only and co-firing in existing coal power plants (essentially replacing coal with biomass). The biomass feedstock categories include dedicated energy crops, forestry, wastes, agricultural residues, and industry. Their statistical evaluation of the two-generation methods results in a mean emissions factor of 0.04 ~ 1.73 $kgCO_2e/kWh$ and 0.93 ~ 1.07 $kgCO_2e/kWh$, respectively, where forestry and industry contribute the least amount of GHG emissions.

A summary of the lifecycle GHG emissions factors is illustrated in *Figure 3*. The published life cycle GHG emissions from the studies reviewed in this section and also by McIntyre et al. (2011) differ substantially depending on each study's magnitude of scope, life cycle stages definitions, and varying assumptions related to a multitude of influencing factors. Additionally, an important factor to consider is the selection of facilities included in the lifecycle GHG emissions studies, where the emission rates are site-specific, region-specific, and also power generation plant-specific (McIntyre et al. 2011). The reviewed literature indicates that there is a consistent reported emissions factor between different generation methods, although the differences in the scopes of the studies allow fluctuation of the absolute emission intensity. Hence, *Figure 3* graphically shows that a given generation method has a varying emissions factor range, which is dependent on the influencing factors assumed by the reviewed literature. Additionally, this section provides evidence to show that renewable technologies are not actually emissions-free, although the magnitude of emissions produced is significantly lower than fossil fuel technologies.

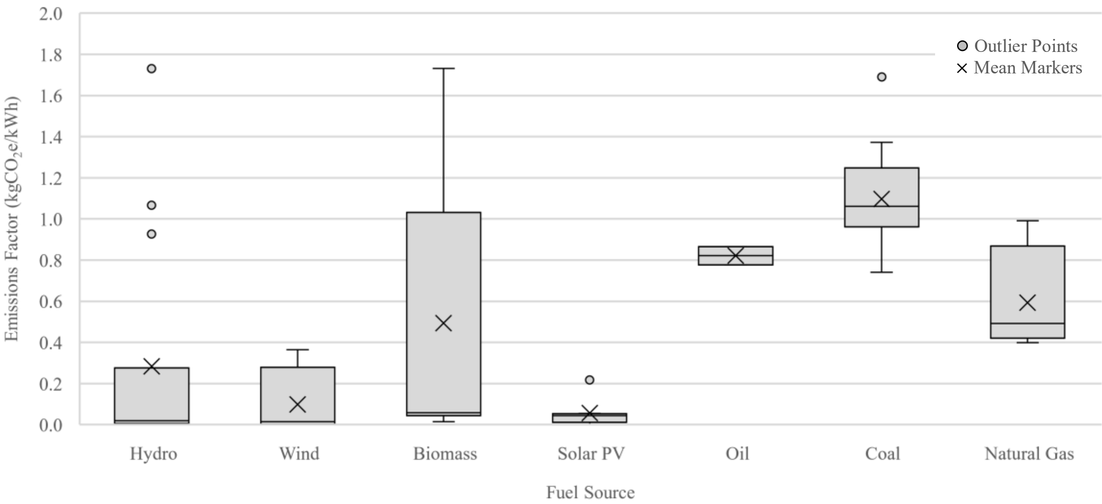


Figure 3 – Lifecycle GHG emissions factor summary.

Source: McIntyre et al. (2011); Nugent & Sovacool (2014); Thornley et al. (2015); Kadiyala, Kommalapati, & Huque (2016); Ocko & Hamburg (2019).

2.2.6.5.2 OPERATIONS EMISSIONS FACTOR CALCULATIONS

McIntyre et al. (2011) created a summary of life cycle GHG emissions factors for both renewable and fossil fuel technologies. The Department of the Environment and Energy (2019f) provided the emissions factors of each Australian state and territory. The two sets of data are illustrated in *Table 8*.

Table 8 – Electricity generation emissions factors, by state or territory, and fuel source.

State or Territory	Emissions Factor	Fuel Source	Emissions Factor
NSW & ACT	0.81	Black Coal	0.89
Victoria	1.02	Brown Coal	1.05
Queensland	0.81	Natural Gas	0.50
South Australia	0.44	Oil	0.73
SWIS	0.69	Hydro	0.03
NWIS	0.59	Wind	0.03
DKIS	0.55	Bioenergy	0.05

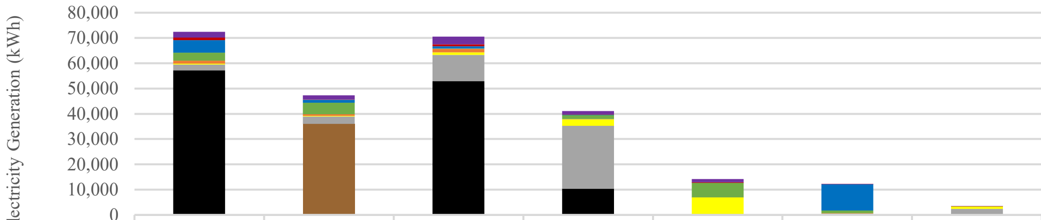
Tasmania	0.15	Solar PV (small)	0.09
Northern Territory	0.63	Solar PV (large)	0.09

Source: McIntyre et al. (2011); DEE (2019e). Emissions Factor: **$kgCO_2e/kWh$**; NSW: New South Wales; ACT: Australian Capital Territory; SWIS: South West Interconnected System in Western Australia; NWIS: North Western Interconnected System in Western Australia; DKIS: Darwin Katherine Interconnected System in the Northern Territory.

Figure 4 shows the scale of electricity generated by fuel type of each state and territory across Australia. It is clear that the grid-mixes of each region varies by fuel type. Much effort has been made to reduce the GHG emissions of different fossil-fuel electricity generation methods. For example, a conventional coal power plant can reduce GWP by incorporating CO_2 capture and sequestration. For this particular clean-coal technology, however, there is a penalty of increased fossil fuel consumption as the CO_2 capture and compression processes and additional maintenance requirements decrease the power plant efficiency (Lund & Biswas 2008).

The GHG emissions from operations can be calculated from the emissions factor per state, as reported by the Department of the Environment and Energy (2019e). From *Table 8*, the emissions factor for New South Wales is 0.81 $kgCO_2e/kWh$. Taking the predetermined 324 kWh battery pack and assuming the worst-case range of 250 km per charge, approximately 262.4 $kgCO_2e$ is produced when charging the BYD BEV bus battery from 0% to 100% capacity. Thus, the electricity emissions factor is calculated to be **105.0 $kgCO_2e/100km$**.

Alternatively, the operations emissions can also be calculated from the emissions factor by fuel source, as reported by McIntyre et al. (2011). Taking the same 324 kWh battery pack and assuming the worst-case range of 250 km, approximately **236.1 $kgCO_2e$** is produced per charge and an emissions factor of **94.4 $kgCO_2e/100km$**. Thus, the calculated electricity emissions factor is **94.4 ~ 105.0 $kgCO_2e/100km$**.



	New South Wales	Victoria	Queensland	Western Australia	South Australia	Tasmania	Northern Territory
Solar PV (small)	2,367.9	1,645.3	3,018.4	1,345.2	1,316.4	149.0	99.8
Solar PV (large)	967.9	177.3	767.3	47.5	142.4	1.2	35.5
Hydro	4,946.8	1,114.2	816.0	217.9	6.4	10,350.7	0.0
Wind	3,122.0	4,580.0	142.8	1,615.3	5,704.0	1,102.4	0.0
Bioenergy	1,166.1	666.8	1,437.8	147.9	89.1	24.6	7.1
Oil Products	411.5	164.8	999.5	2,540.9	6,916.6	33.0	969.0
Natural Gas	2,238.7	3,027.1	10,352.6	24,796.7	0.0	632.9	2,420.2
Brown Coal	0.0	35,962.1	0.0	0.0	0.0	0.0	0.0
Black Coal	57,251.2	0.0	52,947.4	10,402.1	0.0	0.0	0.0

States and Territories

Figure 4 – Electricity generation across Australia.

Source: Department of the Environment and Energy (2019b).

2.2.6.5.3 DIESEL FUEL COMBUSTION EMISSIONS

The combustion emission factors of fossil fuels differ slightly depending on the source of literature (Maennel & Kim 2018). From the studied literature, this study sets the average diesel emission factor to be 2.74 kgCO_2e/L (McKinnon & Piecyk 2011; ADB 2016; MftE 2019; DEE 2019e). According to the Australian Bureau of Statistics (2019), the average diesel bus and hybrid bus consumes 29.2 L/100km and 20.4 L/100km, respectively. Therefore both buses produces approximately 79.9 kgCO_2e/100km and 55.7 kgCO_2e/100km, respectively.

To consider diesel fuel combustion from another angle, another calculation is made under the assumption of an ideal world scenario, where the complete combustion process of diesel fuel (saturated hydrocarbons C_nH_{2n+2} and unsaturated hydrocarbons C_nH_{2n} and C_nH_{2n-2}) with oxygen (O_2) would produce only CO_2 and water vapour (H_2O). The range is between $C_{10}H_2O$ to $C_{15}H_{28}$; assuming the generally chemically formulated dominant molecule $C_{12}H_{23}$, the equation of the diesel fuel combustion process is as follows:

$$4C_{12}H_{23(l)} + 71O_{2(g)} \rightarrow 48CO_{2(g)} + 46H_2O_{(g)} \qquad (1)$$

Based on the combustion equation (Equation 1), the calculated result shows a diesel bus and hybrid bus emissions factors of 76.5 kgCO_2e/100km and 53.4 kgCO_2e/100km, respectively. Thus, from the two independent methods of calculations, this section finds the diesel bus and hybrid bus emissions factors to range from **76.5 ~ 79.9 kgCO_2e/100km** and **53.4 ~ 55.7 kgCO_2e/100km**, respectively. Comparing the GHG emissions when charging a battery and combusting diesel fuel, the generation of electricity with the current grid-mix produces approximately **1.2 ~ 1.3 times** more GHG emissions than when combusting diesel fuel. The calculations provide strong evidence that the high carbon-intensive grid-mixes in Australia has a significant impact on the environmental impact of BEV buses, which corresponds with the findings of Sharma et al. (2013).

2.2.6.5.4 BEV BUS OPERATIONS LIFETIME EMISSIONS

BYD claims that the K9 BEV bus life expectancy is over 388,000 km with a 12-year battery warranty and a battery life of above 4,000 cycles (roughly 11 years or 1,000,000 km). Multiple studies have set the service life expectancy of electric buses to 10 ~ 12 years and 500,000 ~ 800,000 km (Potkány et al. 2018; Franca 2018; Lajunen 2018; Borén 2019). Therefore, the service life expectancy of electric buses is set to **12 years** and **650,000 km** (or 2,600 cycles). Adopting the value of 105.0 kgCO_2e/100km from the previous section, the lifetime emissions amount to **682,344 kgCO_2e**.

Alternatively, the emissions factor is calculated by the fuel source. The GHG emissions produced in proportion to the contribution of each fuel source that makes up the NSW grid-mix results in 236.1 kgCO_2e per charge, which ultimately amounts to **613,778 kgCO_2e** in lifetime emissions. Consequently, the total operations environmental load ranges from $\boldsymbol{E_{operations}}$ **= 613,778 ~ 682,344 kgCO_2e** per BEV bus.

2.2.6.5.5 DIESEL AND HYBRID BUS OPERATIONS LIFETIME EMISSIONS

To provide a frame of reference, the operations lifetime emissions of diesel and hybrid buses are calculated. Over the lifetime of 650,000 km, the operations lifetime emissions amount to **497,250 ~ 512,785 kgCO_2e** and **347,100 ~ 362,050 kgCO_2e**.

2.2.6.5.6 ANALYSIS OF ALTERNATIVE SCENARIOS

The operations lifetime emissions above are estimated based on the current grid-mix and vehicle technology scenarios (hereinafter designated as the base case scenario). As technology progresses for BEV buses and shifts away from consuming fossil fuel, the burden on the environment will change. This section explores alternate scenarios and the environmental load is calculated according to the assumptions made.

2.2.6.5.6.1 SCENARIO 1: TIME VARIANCE

Time variance is considered in this scenario. Charging the BEV buses at different times (day or night) will have an impact on the operations environmental load, as the emissions factors of electricity generation will vary significantly. The environmental load is calculated with the assumption that no solar PVs are operating during the night. It is assumed that the amount of electricity generated by solar PVs during the day would be evenly distributed amongst the remaining fuel sources for generation during the night. The results show that when considering time variance, GHG emissions increase by 1.77% (at 96.1 $kgCO_2e/100km$ or lifetime emissions of 624,809 $kgCO_2e$).

2.2.6.5.6.2 SCENARIO 2: HIGH SHARES OF RENEWABLES

This scenario consists of low carbon electricity generation and improved BEV bus technology. The grid-mix is set to comprise of 95% renewables (Tasmanian grid-mix). Now, assuming a 324 kWh battery pack and the Tasmanian grid-mix emissions factor of 0.15 $kgCO_2e/kWh$, approximately **48.6 $kgCO_2e$** is produced when charging the BYD BEV bus battery from 0% to 100% capacity (or an emissions factor of **19.4 $kgCO_2e/100km$**). Assuming the same diesel emissions factor of 76.50 ~ 79.89 $kgCO_2e/100km$, the environmental load ratio in Tasmania from electricity generation and diesel combustion is approximately **0.25:1.**

2.2.6.5.6.3 SCENARIO 3: NET ZERO EMISSIONS BY 2050

The NSW Government has set a goal of net-zero emissions by 2050 and has released a Net Zero Plan (Stage 1: 2020 – 2030) to fast-track emissions reduction over the following decade (Department of Planning, Industry and Science 2020). The plan aims to reduce annual NSW emissions by 35.8 Mt by 2030 and reduce annual emissions by 35% on 2005 levels (from 160.7 Mt/year to 103.7 $MtCO_2e/year$). In this scenario, it is assumed that the amount of electricity generated in NSW remains fixed at 72,472.1 GWh/year (DEE 2019a). In the first year (2020) the fossil fuel share is set to 82.5% (DEE 2019b) and intermittently decreases each year until the fossil fuel share is at 0% by 2050. Assuming the same BEV bus service life, the average annual operations environmental load is calculated. The total operations environmental load is then calculated from the summation of the annual operations environmental loads for the duration of the assumed service life from the year (inclusive) 2020 – 2031 (Scenario 3.1), 2032 – 2043 (Scenario 3.2), and 2044 – 2055 (Scenario 3.3), respectively. Since the Net Zero Plan has a set goal of net zero emissions by 2050, the annual operations environmental loads for the duration of 2051 – 2055 are assumed to be equivalent to 2050.

2.2.6.6 RECYCLING AND DISPOSAL PHASE

The final phase involves the decommissioning of the charging station at the end of its service life. A critical analysis shows that recycling materials, such as cobalt and nickel in the cathode materials of Li-ion batteries, instead of production from virgin resource supplies result in a 51% natural resource savings (Dewulf et al. 2010). The charging station is reverted back into its original state of raw materials, where the waste materials are separated into individual components, to the point where they have their lowest value. Here, the recyclable materials are determined to include but are not limited to: electronics, glass, metals, plastics, and rubber. The recyclable materials are then transported to respective recycling plants to be sorted, cleaned, and then reprocessed into fresh materials for manufacturing new products. The implementation process of the fresh materials into new products is not included in the analysis. Finally, the remaining non-recyclable materials are then transported to landfills and disposed of.

The challenge with recycling modern equipment lies in that the modules are intricately integrated with plastics, electronics, metals, and other materials. While most of the metals are stripped and recovered, the remaining materials ultimately end up in landfills. Unfortunately, Australia is one of the only countries part of the Organisation for Economic Co-operation and Development (OECD) without a deliberate plan for dealing with automotive component recycling (APRAA 2017). Currently, the Commonwealth Government of Australia is working with the Motor Trades Association of Australia (MTAA) and the Victoria Automotive Chamber of Commerce (VACC) to develop a vehicle End-of-Life (EoL) recycling scheme.

The ultimate objective of the waste management sector is to maximise resource efficiency and reduce GHG emissions simultaneously (Turner, Williams & Kemp 2011). Nugent & Sovacool (2014) and Turner, Williams & Kemp (2015) have also reported numerous international studies to have unanimously shown that waste material recycling can result in the reduction of GHG emissions. *Table 9* articulates the estimated weight of each material and the calculated emissions produced in this phase. Similar to the virgin materials GHG emissions production, the calculation of waste material GHG emissions observe the same functional unit of a unit mass of GHG (or equivalent) per unit of energy/material production: **$kgCO_2e/kg$** (kg of CO_2 equivalent per kg of material reprocessed/recycled). With the limited information available on the emissions produced from recycling and reusing waste materials, this section assumes the emissions of virgin material production, with the exception of aluminium and steel. Thus, the environmental load from the recycling phase is designated as $\boldsymbol{E_{recycling}}$ ($kgCO_2e$) hereinafter.

Table 9 – Charging equipment recycled materials breakdown.

Material	Estimated Weight (kg)	Emissions (kg)					
		CO_2	CO	NO_x	SO_x	CH_4	N_2O
User Unit							
Aluminium	39.1	60.4	0.0	0.1	0.1	0.2	1.5E-03
Copper	37.2	95.7	0.1	0.2	4.9	0.2	1.8E-03
Glass	0.2	0.2	0.0	0.0	0.0	0.0	3.3E-06
Plastic	20.0	39.2	0.1	0.1	0.2	0.3	1.4E-03
Steel	50.0	58.0	0.2	0.0	0.1	0.1	1.1E-03
Ethylene Glycol	109.7	314.9	0.2	0.5	1.2	0.8	6.6E-03
Rubber	3.8	12.5	0.0	0.0	0.0	0.0	3.1E-03
Total	260.0	580.8	0.6	0.9	6.6	1.6	1.5E-02
Power Unit							
Aluminium	80.0	123.6	0.1	0.1	0.2	0.3	3.1E-03
Copper	500.0	1,285.3	1.2	3.0	65.9	2.6	2.4E-02

Plastic	20.0	39.2	0.1	0.1	0.2	0.3	1.4E-03
Steel	100.0	116.0	0.4	0.1	0.2	0.3	2.2E-03
Total	700.0	1,564.1	1.7	3.3	66.6	3.4	3.1E-02
Control Unit							
Aluminium	80.0	123.6	0.1	0.1	0.2	0.3	3.1E-03
Copper	100.0	257.1	0.2	0.6	13.2	0.5	4.8E-03
Plastic	10.0	19.6	0.0	0.0	0.1	0.1	6.8E-04
Steel	30.0	34.8	0.1	0.0	0.1	0.1	6.5E-04
Total	220.0	435.1	0.4	0.8	13.6	1.1	9.2E-03
Cables							
Copper	200.0	514.1	0.5	1.2	26.4	1.0	0 9.6E-03

2.2.6.6.1 TRANSPORTATION

The emissions produced by the transportation of waste materials from each bus depot to respective recycling facilities and landfill sites are included. *Table 10* shows the GHG emissions produced transportation for the recycling phase, calculated from driving distance and the amount of diesel fuel consumed. This study excludes the transport emissions from the truck travel prior to reaching the loading site (bus depots). The same Isuzu NLR 45-150 truck is assumed to be the representative delivery truck to deliver the decommissioned charging equipment from the bus depots to our selected resource recovery facility Lucas Heights NSW. This facility process wastes to be either recycled, reused, or disposed of in their landfill site.

Table 10 – Approximate emissions (kgCO_2e) produced by charging station transportation to the resource recovery facility.

	Ryde Bus Depot (Route 550)	Leichhardt Bus Depot (Route 470)	Seven Hills Bus Depot (607X)	Randwick Bus Depot (309)
Distance from Recycling Facility (km)	34.0 km	31.9 km	43.2 km	37.5 km
Diesel Fuel Consumed (L)	9.9 L	9.3 L	12.6 L	10.9 L
GHG Emissions (kgCO_2e)	27.6 kg	25.9 kg	35.0 kg	30.4 kg

2.3 RESULTS

Table 11 shows the total life cycle emissions of a charging station for BEV buses. It is evident that with the current grid-mix, the operations phase contributes the most to GHG emissions. It is then followed by production, recycling and disposal, installation, and transportation. The results reveal that charging the 324 kWh battery in the operations phase yielded a carbon intensity of 94.4 ~ 105.0 kgCO_2e/100km, whereas the diesel and hybrid bus operations carbon intensity amounted to 76.5 ~ 79.9 kgCO_2e/100km and 53.4 ~ 55.7 kgCO_2e/100km, respectively. This indicates that the generation of electricity with the current grid-mix produces approximately **1.2 ~ 1.3 times** more GHG emissions than when combusting diesel fuel.

Table 11 – Life cycle GHG emissions results.

	$kgCO_2$	$kgCH_4$	kgN_2O	$kgCO_2e$	% of Total Emissions
Production	4,460.6	9.1	0.1	4,737.6	0.7%
Transportation	71.0	0.0	0.0	71.0	0.0%
Installation	86.3	0.0	0.0	86.3	0.0%
Operations	682,344.0	0.0	0.0	682,344.0	98.8%
Recycling & Disposal	3,094.1	7.1	0.1	3,310.7	0.5%
Total Emissions (kg)	690,056.0	16.3	0.1	690,549.6	100%

The amount of transportation emissions are calculated from the transportation between the shipping port to the bus depots, and from the bus depots to the resource recovery facility. Since transportation emissions vary significantly depending on the distance travelled, the highest values are used as the upper limit of emissions produced (Seven Hills Bus Depot). Here, the transportation emissions are found to be of limited significance to the calculated net GHG emissions, on average contributing to just 0.01% of total GHG emissions. Similarly, emissions from the installation are found to be of limited significance to the calculated net GHG emissions, on average contributing to just 0.01% of total GHG emissions.

For the base case scenario, the total environmental load is designated as E_{total} ($kgCO_2e$) for a single charging station. Here, E_{total} is obtained by the sum of $E_{production}$, $E_{transportation}$, $E_{installation}$, $E_{operations}$, and $E_{recycling}$.

$$E_{total} = \sum(E_{production} + E_{transportation} + E_{installation} + E_{operations} + E_{recycling}) = \mathbf{690,549}\ \boldsymbol{kgCO_2e} \quad (2)$$

In summary, the total GHG emissions produced from one charging station in Sydney, Australia is approximately 690.5 $tonCO_2e$. The production, transportation, installation, and recycling and disposal phases contributed to 0.69%, 0.01%, 0.01%, and 0.48% of the total emissions, respectively. This study's best estimate of a BEV bus charging station's total environmental load, excluding the operations and decommissioning emissions, is approximately 47.4% to 52.2% higher than reported by Nansai et al. (2001), due in part to inclusions of assumptions, such as electronic components and decommissioning phases not previously considered.

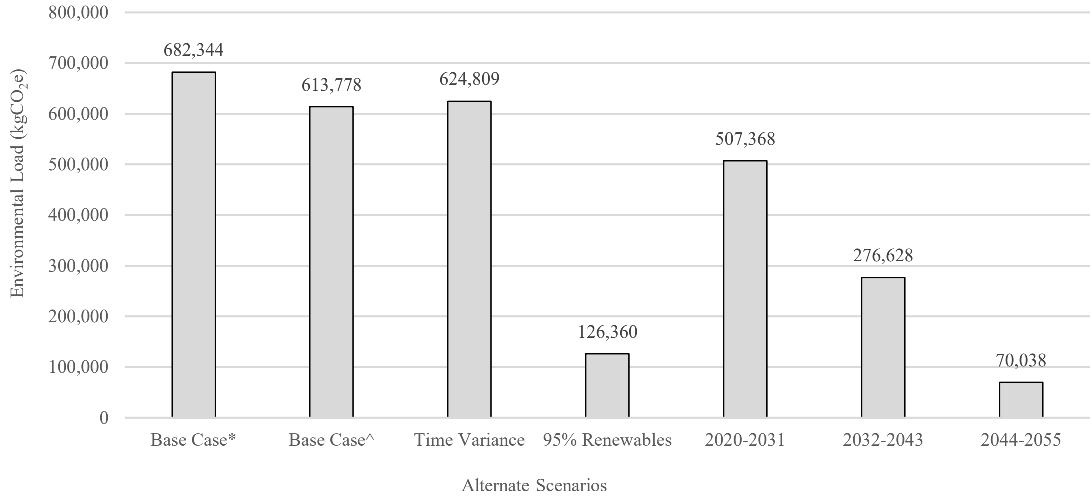


Figure 5 – GHG emissions of alternate scenarios.

*Based on the NSW emissions factor provided by the Department of the Environment and Energy (2019a).

^Based on the emissions factor of each fuel source provided by McIntyre et al. (2011).

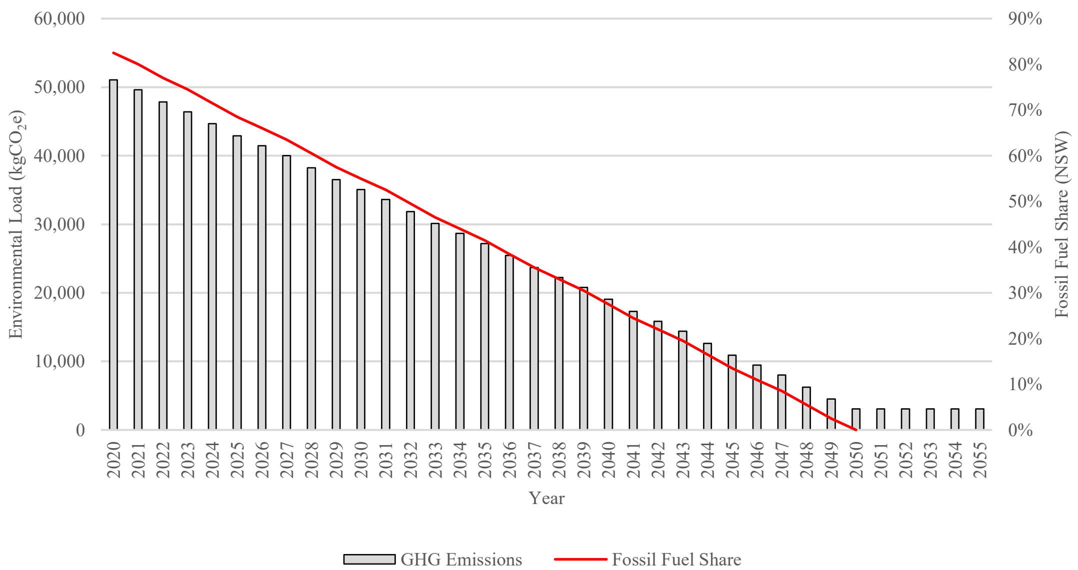


Figure 6 – Projection of the environmental load from the net-zero emissions by 2050 plan.

Figure 5 shows the comparison between the operations GHG emissions of alternate scenarios. First, operations emissions of the base case scenario were calculated based off NSW emissions factor (0.81 $kgCO_2e/kWh$; 105.0 $kgCO_2e/100km$) and individual fuel source emissions factors (94.4 $kgCO_2e/100km$). Next, the environment load due to Scenario 1 (time variance) was calculated with the assumption of no solar PV was used during the night. The results show that at a carbon intensity of 96.1 $kgCO_2e/100km$, charging at night would increase the overall lifetime emissions by approximately 1.77%. Then, in Scenario 2 (high shares of renewables) the results show an emissions reduction of 81.5% (19.4 $kgCO_2e/100km$). Finally, the last three columns in

Figure 5 and *Figure 6* graphically shows the projected annual operations emissions of Scenario 3 (Net Zero Emissions by 2050). The scenario illustrates the steady linear decrease in environmental load following the decrease in fossil fuel share of the NSW grid-mix. Assuming the same service life and comparing with the base case emissions, a BEV bus operating during 2020 – 2031, 2032 – 2043, and 2044 – 2055 is projected to have an operations lifetime emissions reduction of 17.3%, 54.9%, and 88.6%, respectively. The small amounts of emissions shown from 2050 onwards are owing to the non-negligible emissions produced from the life cycle of the respective renewable power plants. Therefore, a comprehensive switchover to renewable technologies does not equate to complete emissions-free electricity generation, although the magnitude of emissions produced is significantly lower than fossil fuel technologies.

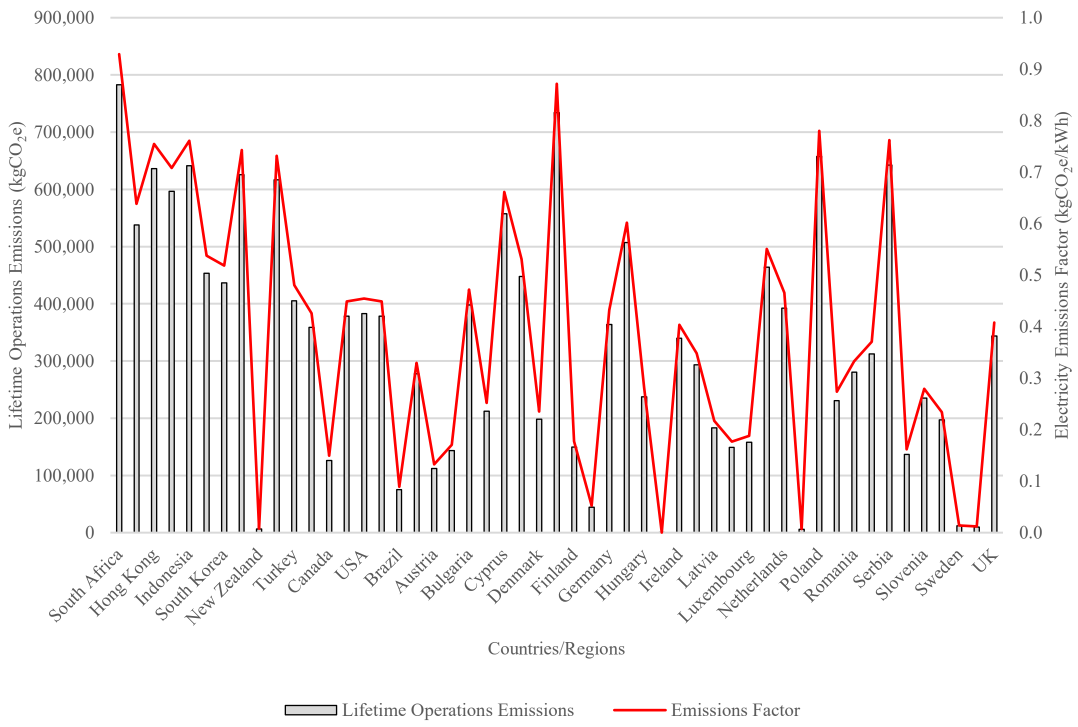


Figure 7 – Carbon intensity of electricity for selected countries and regions.

Source: Moro & Lonza (2018); Department of the Environment and Energy (2019c); GREET© (2019); Carbon Footprint (2020).

This study analysed the recent carbon intensity of the grid-mixes of 49 countries, illustrated graphically in *Figure 7*. Assuming the same BEV bus service life of 12 years and 650,000 km, the electricity carbon intensity of each country is used to calculate their respective lifetime operations emissions. The results reveal that 39 out of 49 countries have low carbon-intense grid-mixes, indicating that it would be intuitive to implement BEV buses in those countries, as the electricity generated to charge the BEV bus would yield less GHG emissions than its diesel counterpart. It is worth noting that the low emissions factor of Europe (0.33 $kgCO_2e/kWh$) should be interpreted as an average value only. A few selected European countries such as Estonia, Poland, and Serbia have relatively high shares of fossil fuel in their grid-mixes.

2.4 SENSITIVITY ANALYSIS AND UNCERTAINTY

The total life cycle emissions of a charging station for BEV buses calculated in this study are influenced by several factors, assumptions, and uncertainties. The specific values to some parameters that influence the life cycle emissions of the studied charging infrastructure vary from region to region. This study applies a case study approach to calculate the life cycle emissions, hence the results would be specific to Australia, but can be readily adapted to other regions with knowledge of specific factors. In terms of manufacturing and production, the GHG emissions vary with the degree of virgin and recycled materials used. In addition, impact factors such as energy consumption are heavily influenced by the carbon intensity of a region's grid-mix and induce uncertainty in the raw material manufacturing emissions. For example, if this study was to conduct an emissions LCA exclusively for France, then the phases that are influenced by energy consumption (e.g. production, installation, operations, and disposal and recycling) would result in much lower lifetime emissions. Realistically, it should be assumed that numerous sub-components and raw materials may originate from various parts of the world. The manufacturing and assembly processes may vary in their degrees of carbon intensity, making it difficult to determine an accurate environmental load for imported sub-components and raw materials. As there is limited data available in Australia in the public domain relating to materials processing emissions, this study utilised the GREET© model was utilised to estimate the emissions rate per unit of material weight for the production, recycling, and disposal phases. The GREET© model reported higher levels of GHG emissions for virgin materials production than for recycled materials, which corresponds with the numerous international studies that have unanimously shown that waste material recycling can result in a reduction of GHG emissions (Carbon Footprint 2020).

The results show that BEV charging infrastructure environmental impacts are highly influenced by the GHG emissions from the operations phase. The analysis showed that the operations phase produced the most GHG emissions, is highly dependent on the carbon intensity of the electricity grid-mix and will differ depending on the electricity generation method of a country or region. For the current Australian grid-mix, the generation of electricity produces approximately 1.2 ~ 1.3 times more GHG emissions than when combusting diesel fuel. Additionally, emissions from production, transportation, installation, recycling, and disposal phases were found to be of limited significance to the calculated net GHG emissions. The results from the analysis of alternative scenarios suggest a very high potential for Australia to reduce the environmental impact gap between diesel and electric powertrain technologies. Since the operations phase contributed to

98.8% of the total environmental load, the environmental performance of a BEV bus is heavily influenced by energy production. As NSW executes the Net Zero Emissions Plan, from 2050 onwards the annual BEV bus operations emissions would eventually be reduced to approximately 3,048 $kgCO_2e$. The foreseeable improvements to the NSW electricity grid-mix (Net Zero Emissions Plan) would potentially allow for a significant reduction of environmental impacts from transportation, namely urban air pollution and climate change.

2.5 POLICY IMPLICATIONS

The results of this study establish that transitioning the transport bus fleet into BEVs has the potential to significantly reduce the impact of climate change compared to diesel buses. The environmental benefits of BEV technologies are substantial and a rich area for research; however, immediately transitioning the current bus fleets exclusively into BEV buses for all countries will be counterproductive. This is only possible if the electricity used to charge the BEV buses is generated from low carbon-intense sources, such as renewable energy. This study's analysis has shown that with the current electric grid-mixes of 49 observed countries, the use of BEV buses in 10 countries (including Australia) actually leads to an increase in GHG emissions as the electricity grid-mix of these countries are comprised predominately of fossil fuels. This study also emphasised the importance of assessing the environmental impact of renewable energy technologies, as electricity generation from renewables does generate minor amounts of GHG emissions. Each renewable technology offers significant GHG emissions reduction compared to fossil fuel technologies, but will not be completely emissions-free. Ultimately, governments and policymakers would have to consider the trade-offs between various renewable and fossil fuel technologies that would best benefit the environment of a given region.

Transitioning to BEV buses may be a motivator for committing to decarbonising the electricity grid-mixes, however, their introduction into the bus fleets should only serve to complement such commitments. With the absence of tailpipe emissions, the mass introduction of BEV buses may reduce urban air pollution. In this case, the emissions are relocated from high-traffic roads to specific regions, this relocation does not effectively reduce emissions. Consequently, the operations GHG emissions will be shifted away from densely populated regions and instead be relocated to the vicinity around electricity power plants, which are mostly out of city limits. This may bring forth new challenges, opportunities, and risks for policymakers and government bodies, as the operations emissions would be theoretically easier to control at a selected few locations rather than thousands of mobile sources.

2.6 CONCLUSION

The study in this chapter applied a case study approach for Australia and calculated the magnitude of greenhouse gases produced from the implementation of electric bus charging stations into existing bus depots concurrent with the transitioning of the commuter bus fleets into electrified powertrains. Utilising the Australian-based fleets as a case study and baseline scenario, the GHG emissions from production, transportation, installation, operations, recycling, and disposal phases were estimated to establish a comprehensive and in-depth emissions life cycle assessment.

The best estimate of a BEV bus charging station's total environmental load, excluding the operations and decommissioning emissions, is approximately 47.4% to 52.2% higher than the reviewed literature, due in part to inclusions of assumptions, such as electronic components and decommissioning phases not previously considered. The operations phase contributed the most GHG emissions (98.8%), followed by production (0.7%), recycling and disposal (0.5%), installation (0.01%), and transportation (0.01%). Thus, the contributions from infrastructure development and the transition to electrified buses are substantially outweighed by operation emissions. Additionally, three alternate scenarios were explored: time variance, high shares of renewables, and net-zero emissions by 2050. Achieving a net-zero emissions outcome still requires a substantial change to the electricity generation methods.

This study's analysis has shown that with the current electric grid-mixes of 49 observed countries, the use of BEV buses in 10 countries (including Australia) actually leads to an increase in GHG emissions as the electricity grid-mix of these countries are comprised predominately of fossil fuels. The generation of electricity with the current Australian grid-mix produces approximately **1.2 ~ 1.3 times** more GHG emissions than when combusting diesel fuel. Consequently, it would be counterproductive to introduce BEV buses in countries and regions that generate electricity from high carbon-intense sources.

The environmental benefits of BEV technologies are substantial and a rich area for research. Transitioning the transport bus fleet into BEVs has the potential to significantly reduce the impact of climate change compared to diesel buses. This progression may be a motivator for committing to decarbonising the electricity grid-mixes, however, their introduction into the bus fleets should only serve to complement such commitments. The absence of tailpipe emissions will relocate emissions from densely populated regions to the vicinity of electric power plants. This may bring forth new challenges, opportunities, and risks for policymakers and government bodies, as the operations emissions would be theoretically easier to control at a selected few locations rather than thousands of mobile sources.

3 CHAPTER 3: OPERATIONS EMISSIONS ASSESSMENT & EVALUATION

3.1 INTRODUCTION

In the year to June 2019, electricity generation and transportation is the first and third largest contributor of GHG emissions in Australia, accounting for 33.8% (179.9 $MtCO_2e$) and 18.9% (100.4 $MtCO_2e$) of total GHG emissions, respectively (DEE 2019e). GHG emissions from electricity generation across the country have been significantly reducing since 2016 as the grid mixes continue to decarbonise. The Australian Department of Environment and Energy (2019e) projected emissions decline to reach 170 $MtCO_2e$ by 2020, 149 $MtCO_2e$ by 2025, and 131 $MtCO_2e$ by 2030.

Specific to Australia, the electricity generation sources varies considerably from state to state. The state of Victoria represents an upper bound on fossil fuel reliance, where virtually all brown coal that Australia produces is consumed to generate electricity. On one hand, in 2018 the electricity in New South Wales (NSW) and the Australian Capital Territory (ACT), Victoria (VIC), Queensland (QLD), Western Australia (WA), and the Northern Territory (NT) were generated from more than 80% fossil fuels. The grid mixes of these states are predominately black and brown coal, with the exception of WA and the NT (predominately natural gas). On the other hand, renewables generation dominated the energy's share significantly in Tasmania (TAS) (94.6%) and marginally so in South Australia (SA) (51.2%). There are no nuclear power plants in Australia. There is increasing popularity with wind power and small-scale solar power (residential rooftop solar).

Over the past decade, there has been a dramatic increase in interest in developing and producing BEV buses. The transition to and utilisation of BEV buses have been perceived as environmentally friendly as they emit no tailpipe greenhouse or toxic gases into the atmosphere. Nevertheless, the sustainability regarding the electricity generation that powers the BEV buses have seldom been questioned. It is tempting to envision a futuristic world where public road transportation is driven solely by BEV buses. After all, we currently process the technology to make electrified transportation a reality (Hall & Lutsey 2019; Miller & Jin 2019).

Several studies show that the environmental performance of BEVs is influenced by the electricity grid-mix used for charging these vehicles. Nordelöf et al. (2014) demonstrated that WTW assessments generally use the average emissions of the electricity grid-mix. In reality, however, the emissions from the grid-mix vary depending on the time the vehicles are charged. Since BEVs are flexible with their charging, the optimal charge time should be whenever the share of low carbon emissions sources is high (renewables or nuclear). Foley et al. (2013) analysed the Irish electricity grid-mix and highlighted that off-peak charging is more beneficial to CO_2 emissions reduction than peak charging. This study only assessed the CO_2 emissions savings of replacing a fleet of conventional ICEVs with BEVs, thus only the CO_2 emissions on the energy conversion phases of ICEVs and electricity generation systems. Faria et al. (2013) conducted a study on the impact of BEV charging characteristics on the overall GHG emissions for three electricity grid-mixes: 1) mainly fossil fuel, 2) large contribution from nuclear, and 3) large contribution from renewables. Their analysis shows that the time of charging is only influential to GHG emissions reduction when the share of renewables is high in the grid-mix. The authors further concluded that the operation phase (and also driving behaviour) contributed the most to GHG emissions over the life cycle of any given vehicle, regardless of its propulsion technology. Rangaraju et al. (2015) compared the life cycle environmental emissions of BEVs with diesel and petrol vehicles. Their assessment was based on real-world energy consumption data. Due to the large share of nuclear energy in the Belgian electricity grid-mix, their results show that the BEV had lower GHG emissions than the two ICEVs. Additionally, the authors suggested that WTW emissions can be further reduced by charging the BEVs at the right time, mainly when the share of high emissions sources (non-renewables) is lower. Thus, the unanimous agreement across the reviewed studies to reduce the life cycle environmental impact of BEVs was to charge the vehicles when the shares of low emissions sources in the electricity grid-mix are high.

This chapter addresses the WTW life cycle segment (vertical flow in *Figure 1*) within the Complete Life Cycle model. A series of studies are conducted to evaluate the environmental impact of electricity generation. The structure of this study is as follows. First, the scope definition and functional unit adopted in this study are defined in the methodology section. Second, the study applies a case study approach and analyses the GHG emissions produced from electricity generation in Australia and then compares the results against the electricity generation of 48 other countries. In this section, six countries with high carbon-intense grid-mixes have been selected and a model is created to determine a break-even point where the environmental load for electricity generation is equal to diesel fuel combustion. Third, in the next section, the life cycle emissions of four different charging methods are compared: stationary charging station, wireless power transfer, opportunity pantograph charger, and overhead pantograph rails. Fourth, the GHG emissions produced from different operation charging strategies are analysed to determine the optimal arrangement from the available charging strategies that will not only benefit the bus operators but also contributes the least to climate change. In this section, three representative BEV buses are introduced that are specifically chosen for this study's analysis. Finally, the results from the break-even analysis and a discussion on the operation charging strategies, followed by a sensitivity analysis to conclude the study of this chapter.

3.2 METHODOLOGY

3.2.1 SCOPE DEFINITION

To begin, this study evaluates the GHG emissions produced from the electricity grid-mixes of the 49 countries, determines a break-even point for countries with high carbon-intense grid-mixes, analyse four different BEV bus charging methods in their magnitude of GHG emissions, and determines the optimal arrangement from the available charging strategies that will contribute the least to climate change. This chapter also evaluates the operations phase of a diesel and hybrid bus and compare the environmental impact of diesel combustion with electricity generation. The results of this study contribute to the evaluation of GHG emissions from the BEV bus operations phase, which in turn contributes to the emissions LCA of BEV buses. To assist with the evaluation of operation scenarios, the study adopts bus routes that consist of urban, suburban, and highway driving. It is worth mentioning that operational emissions have significant environmental impacts on other lifecycle phases, such as human toxicity, acidification, eutrophication, etc. Although these impacts cannot be ignored, they are deemed as out of the scope of this study and therefore not included in this analysis.

3.2.2 FUNCTIONAL UNIT

In this study, three Functional Units (FU) are used. First, the FU for electric energy analysis is defined as a unit mass of GHG equivalent (CO_2e) per unit of energy production: **kgCO_2e/kWh** (McIntyre et. al. 2011). Then, the FU for operations GHG emissions calculations is defined as a unit mass of GHG equivalent per 100 km travelled: **kgCO_2e/100km**. Lastly, the FU for the LCA of recharging methods is defined as a unit mass of GHG equivalent per unit mass/length of material production: **kgCO_2e/kg** or **kgCO_2e/km**.

3.2.3 OPERATIONS GHG EMISSIONS CALCULATION

This section calculates the GHG emissions produced when charging a BEV bus. This study has chosen the Chinese-made BYD K9 BEV bus as the representative bus. The BYD K9 BEV bus has a 324 kWh battery pack ($LiFePO_4$) and a claimed range of 250 km ~ 350 km per charge (BYD 2017). The amount of emissions produced to fully charge the battery pack is calculated by multiplying a given region's emissions factor (kgCO_2e/kWh) by the battery capacity (kWh). Applying a case study approach for Australia and assuming the grid-mix carbon intensity of 0.63 ~ 0.74 kgCO_2e/kWh (see *Section 2.2.7.5.2*), charging the BEV bus battery from 0% to 100% amounts to approximately 204.1 ~ 240.7 kgCO_2e. The following formula is then used to calculate the operations GHG emissions factor:

$$E_i = \frac{Q \times EF_i}{D_r} \times 100\ km \tag{3}$$

Where,
E_i = Emissions produced from electricity generation in jurisdiction i ($kgCO_2e/100km$)
Q = Battery capacity (kWh)
EF_i = Emissions factor from electricity generation in jurisdiction i ($kgCO_2e/kWh$)
D_r = Driving range (km)

This section assumes the worst-case driving range of 250 km per charge. Therefore, by applying the values to Equation 3, the resulting operations emissions factor of a BEV bus in Australia ranges from **81.6 ~ 95.9 $kgCO_2e/100km$**.

3.2.4 DIESEL FUEL COMBUSTION EMISSIONS

Similarly, the operations GHG emissions produced by a diesel bus can be calculated from the following formula:

$$E_d = EF \times FC \tag{4}$$

Where,
E_d = Emissions produced from combusting diesel fuel ($kgCO_2e/100km$)
EF = Emissions factor of diesel ($kgCO_2e/L$)
FC = Fuel consumption (L/100km)

The combustion emission factors of fossil fuels differ slightly depending on the source of literature (Maennel & Kim 2018). The reported Well-to-Wheel (WTW) diesel fuel emission conversion factor ranges from 2.66 ~ 2.90 $kgCO_2e/L$ (McKinnon & Piecyk 2011; ADB 2016; EPA 2018; MftE 2019; DEE 2019e). According to the Australian Bureau of Statistics (2019), the average diesel bus consumes 29.2 L/100km. Therefore, by applying these values into Equation 4, the resulting operations emissions factor of a diesel bus in Australia amounts to approximately **77.7 ~ 84.7 $kgCO_2e/100km$**.

3.2.4.1 ELECTRICITY GENERATION DISPARITY FOR VARIOUS COUNTRIES

To provide a performance comparison with this case study's grid-mix carbon intensity, this section analyses the grid-mixes of 49 countries. From the previous section, the emissions factor for a diesel bus is calculated to be 77.7 ~ 84.7 $kgCO_2e/100km$. Consequently, the electricity generation GHG emissions factor of the 49 analysed countries can be calculated with Equation 3 and the results are graphically illustrated in *Figure 8*. The red bar in the legend indicates the emissions factor for diesel. To simplify the assessment, this study does not analyse electricity imports from other countries, therefore it is assumed that electricity produced satisfies the demands of electricity consumption. A summary of life cycle GHG emissions factors for both fossil fuels and renewable technologies is presented in *Table 12*.

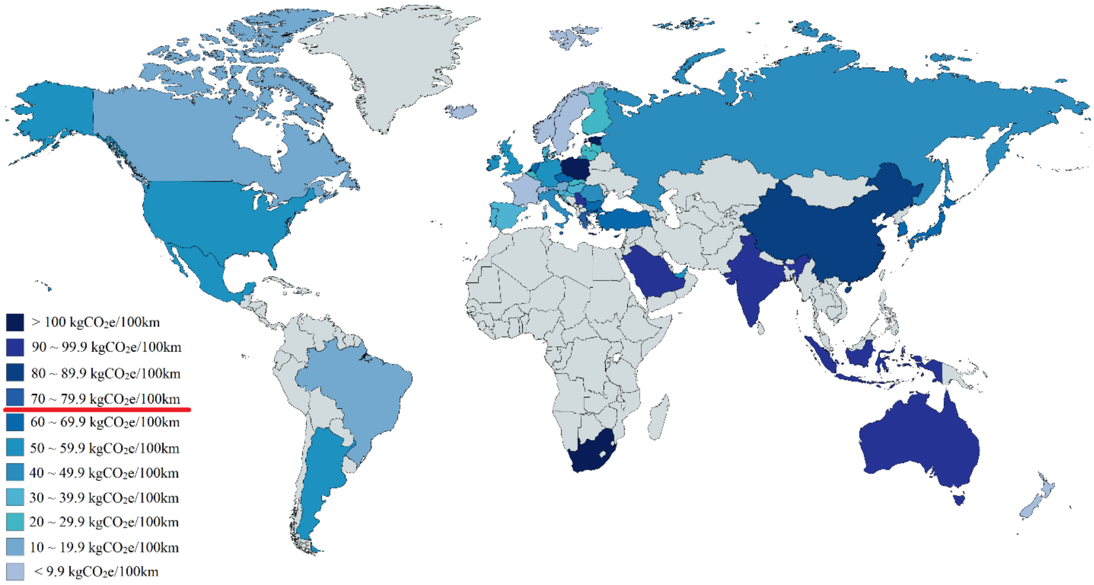


Figure 8 – Electricity generation carbon intensity of selected countries and regions.

Source: Moro & Lonza (2018); GREET© (2019); Department of the Environment and Energy (2019d); Carbon Footprint (2020).

Table 12 – Emission factor of fuel sources for electricity generation.

Fuel Source	Emission Factor (kgCO_2e/kWh)		
	Low	High	Mean
Black Coal	0.76	1.31	0.89
Brown Coal	1.06	1.69	1.30
Natural Gas	0.36	0.89	0.50
Oil	0.55	0.94	0.73
Hydro	2×10^{-3}	0.24	0.03 ~ 0.25
Wind	4×10^{-4}	0.36	0.03
Nuclear	2×10^{-3}	0.13	0.03
Biomass	5.5×10^{-3}	0.06	0.04 ~ 0.05
Solar Photovoltaics (PV)	1×10^{-3}	0.22	0.05 ~ 0.09

Source: Lund & Biwass (2008); McIntyre et al. (2011); Nugent & Sovacool (2014); Thornley et al. (2015); Kadiyala, Kommalapati, & Huque (2016); Ocko & Hamburg (2019).

3.2.5 BREAK-EVEN ANALYSIS

In the previous section, the carbon intensity of 49 countries' grid-mixes was analysed. However, an unanswered question remains: at what point would it be intuitive to promote and implement BEV buses in a country? To answer this question, three critical pieces of information would need to be determined: the GHG emissions factor from diesel fuel combustion, the carbon intensity of a given country's grid-mix, and the grid-mix fossil fuel to renewables energy ratio. The BP Statistical Review of World Energy (2019) report provides data on a country's grid-mix contributing fuel types, and from the data, the country's fossil fuel to renewables energy ratio was determined with respect to the total amount of electricity generated in the year 2019. The value for the emissions factor of a diesel bus is designated to be 77.7 ~ 84.7 $kgCO_2e/100km$ (provided in the previous section) and the BEV bus's driving range is set to 250 km per charge. Using the reported grid-mix carbon intensities, the values were applied to Equations 3 & 4. The calculated results show that 10 out of the 49 countries yielded higher GHG emissions from electricity generation than diesel fuel combustion.

With the available data, a break-even analysis is therefore conducted for the following outstanding countries: South Africa, China, India, Indonesia, Australia, and Poland. To achieve this analysis, a model was created to determine a break-even point where the environmental load for electricity generation is equal to diesel fuel combustion (see *Figure 9*). The model applies values into Equations 3 & 4 and calculates the GHG emissions from electricity generation with respect to the percentage of fossil fuels that comprises a country's grid-mix. Starting from 100% fossil fuels, the model incrementally reduces the grid-mix fossil fuel share to the break-even point and then continues until the grid-mix reaches 100% renewables.

BREAK-EVEN ANALYSIS MODEL					User Input
Conversion Table	Value	Units			
Bus Battery Capacity	324.00	kWh			
Bus Battery Range	250.00	km			
Per 100km	100.00	-			
GWh to kWh	1,000,000.00	-	Whole	Split	Australia
Black Coal	156,562.90	GWh	59.89%	73.83%	81.13%
Natural Gas	50,244.90	GWh	19.22%	23.69%	
Oil	5,258.60	GWh	2.01%	2.48%	
Hydro	17,451.90	GWh	6.68%	35.37%	18.87%
Wind	16,266.50	GWh	6.22%	32.97%	
Nuclear	0.00	GWh	0.00%	0.00%	
Biomass	3,539.30	GWh	1.35%	7.17%	
Solar PV	12,081.10	GWh	4.62%	24.49%	

Environmental Load		Fuel Source	Emission Factor ($kgCO_2e/kWh$)	Electricity Generation (GWh)	Electricity Generation (kWh)	Ratio Split (%)	$kgCO_2e$	User Input Share (%)
0.6495	$kgCO_2e/kWh$	Black Coal	0.89	156,320.61	156,320,607,656.54	73.83%	138,812,699,599.01	81.00%
84.17	$kgCO_2e/100km$	Natural Gas	0.50	50,167.14	50,167,142,405.01	23.69%	25,033,404,060.10	
		Oil	0.73	5,250.46	5,250,461,938.45	2.48%	3,848,588,600.88	
			Total	211,738.21	211,738,212,000.00	Total	167,694,692,259.99	
		Hydro	0.03	17,567.99	17,567,985,193.75	35.37%	456,767,615.04	19.00%
		Wind	0.03	16,374.70	16,374,700,242.04	32.97%	425,742,206.29	
		Nuclear	0.03	0.00	0.00	0.00%	0.00	
		Biomass	0.05	3,562.84	3,562,842,441.01	7.17%	160,327,909.85	
		Solar PV	0.09	12,161.46	12,161,460,123.21	24.49%	1,033,724,110.47	
			Total	49,666.99	49,666,988,000.00	Total	2,076,561,841.65	

Figure 9 – Break-even analysis model used to calculate the break-even point where the environmental load for electricity generation is equal to diesel fuel combustion for the six analysed countries.

3.2.6 ASSESSING LCA STUDIES ON RECHARGING METHODS

As the global transport sector gradually pushes for the adoption of BEV buses, bus operators would be presented with the many charging methods currently available, each with its associated advantages and disadvantages. The majority of transit buses operate continuously throughout the day and night along planned routes, therefore bus operators must maintain a consistent schedule. Depending on the daily operational distance and battery capacity, a BEV bus may require recharging if the battery's State of Charge (SoC) falls below the minimum level of energy required to complete the route. A long charging duration may result in dispatching additional buses to continue servicing the route (Gao et al. 2017). In this section, the life cycle emissions of the four different charging methods are compared: stationary charging station, wireless power transfer, opportunity pantograph charging, and overhead pantograph rails.

3.2.6.1 STATIONARY CHARGING STATION

Suppose a bus operator chooses a BEV bus with a large battery capacity (e.g. BYD K9 or Yutong E12). The BEV bus will not require frequent charging as it has a sufficient range per charge to complete the daily scheduled route partially or fully. The charging time ranges from two to five hours, therefore charging may only be done at the depot or in a dedicated charging complex. Hence, bus operators favouring this solution will be required to install and utilise stationary charging stations.

The calculated GHG emissions values from the life cycle of charging infrastructures are obtained from the previous chapter. The following is a summary of the results. The production and decommissioning phases contributed similar amounts of GHG emissions, at 4,460.6 kgCO_2e and 3,094.1 kgCO_2e, respectively. The transportation and installation phases were found to be of negligible contribution to the calculated net GHG emissions. The operations phase contributed to 682,344 kgCO_2e (or 98.8% of total emissions) due to high shares of fossil fuels in the Australian electricity grid-mix. Thus, the result shows that one charging station in Australia produces approximately 690,549.6 kgCO_2e over a service life of 12 years.

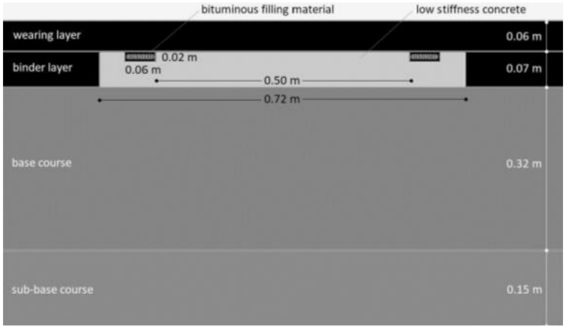
Similarly and also mentioned in the previous chapter, Nansai et al. (2001) analysed the life cycle of public charging stations in eligible sites in Japan. The production, transportation, and installation GHG emissions of a single charging station amounted to 3,857.1 kgCO_2e, 128.6 kgCO_2e, and 283.6 kgCO_2e, respectively (or a cumulative of 4,269.6 kgCO_2e). If the decommissioning phase was to be included, then the total GHG emissions would amount to 8,538.6 kgCO_2e (assuming the same decommissioning emissions as the production emissions).

3.2.6.2 WIRELESS POWER TRANSFER

Static WPT charging involves charging a parked or temporarily stopped vehicle with wireless charging coil pads embedded in the road. Dynamic WPT charging involves equipping a road with WPT which recharges a vehicle when in motion. Both charging methods are achieved via inductive charging. The application of inductive charging is especially advantageous for urban buses; some benefits include readily available information regarding vehicle specifications, fixed routes, and municipalities' involvement. The fixed routes have been set to provide enough charge to cover the entire bus route (thus acting as a range extender), therefore the BEV buses can be equipped with smaller batteries that are instantaneously recharged by the WPT systems whilst the bus is in motion (Lee, Ji, & Cho 2019).

Marmiroli, Dotelli, & Spessa (2019) conducted a detailed LCA of an electrified road that incorporated the environmental impacts resulting from construction and maintenance. The study analysed the impact of a stretch of high-density motorway that has been upgraded to include dynamic WPT technology. Unlike the emissions LCA of stationary charging stations, WPT technology merges charging infrastructure and the roads together into a single system. Their study has declared a functional unit of 1 km of the lane on the far right of an average European motorway, upgraded to an electrified road capable of supplying 100 kW to heavy-duty vehicles. In real-world applications, the charging coils are embedded approximately 60 ~ 70 mm in the road surface, and the power electronics and control equipment are installed in plastic manholes off to the side of the road (see *Figures 10 & 11*). The study considered construction, transportation, and maintenance phases into the LCA. The GHG emissions reported in the study are as follows. First, the construction phase includes road and charging coils material production, transportation, and construction of the electrified road, and produces a total of 168,337 kgCO_2e/km. Next, the maintenance phase (single wear layer rehabilitation) produces a total of 31,069 kgCO_2e/km and the expected maintenance frequency is once every 2 ~ 5 years. The wear and binder layer rehabilitation produces a total of 57,014 kgCO_2e/km and the expected maintenance frequency of at least once in the lifetime of the road. Finally, in summary, the final results show a varying cumulative energy demand of 287,469 ~ 473,882 kgCO_2e/km, which is dependent on the wear layer rehabilitation frequencies of every 2 to 5 years.

Balieu, Chen, & Kringos (2019) conducted a life cycle study to compare the environmental impacts of conductive rail, pantograph, and inductive power transfer technologies for Sweden. The required energy consumption in the analysis was set to be produced from the Swedish grid-mix. Similarly, the considered functional unit is 1 km of a highway lane, with a lifetime of 20 years. The authors consider GHG emissions produced road construction, maintenance, and rehabilitation phases into the final LCA results. In the context of inductive power transfer, the GHG emissions reported in the study are as follows. First, the construction phase includes raw material production, transportation, and paving operations, amounting to approximately 115.0 tonCO_2e/km. Then, the maintenance phase includes winter maintenance and rehabilitation operations and amounts to 149.5 tonCO_2e/km. The rehabilitation frequency was set to three top layer replacements and one complete pavement replacement for the lifetime duration of 20 years. Lastly, in summary, the final results from construction, maintenance, and rehabilitation produced a life cycle GHG emissions of 281,610 ~ 590,320 kgCO_2e/km. The frequency of rehabilitation operations heavily influences the final value of the life cycle GHG emissions.



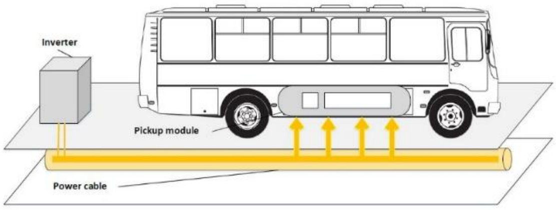


Figure 10 – (Left) WPT road geometry for motorway applications.

Figure 11 – (Right) Simplified layout of a WPT system.

Source: Marmiroli, Dotelli, & Spessa (2019); Lee, Ji, & Cho (2019).

3.2.6.3 PANTOGRAPH CHARGING

Another factor to consider is choosing the proper battery capacity of the BEV bus that best suits the needs of the bus operator. The battery is the most expensive component of a BEV bus; indeed, a BEV bus equipped with a small battery capacity will have a significantly lower initial cost. The following is a summary of the two types of pantograph charging: opportunity pantograph charging (*Figure 12*) and overhead pantograph rails (*Figure 13*).

Suppose a BEV bus equipped with a smaller battery capacity satisfies the needs of a particular bus operator. Depending on the daily operational distance, the BEV bus may require recharging to complete the daily scheduled routes. Volvo and ABB are pioneering this opportunity pantograph charging technology by taking advantage of the resting time at the bus terminal to recharge the BEV bus. After the bus has stopped at the designated area, the charging interface located at the top of the pylon lowers down automatically and makes contact with the bus's pantograph located on the roof. Implementing opportunity pantograph charging infrastructures minimises the impact on route schedules, in that the chargers can be installed at terminals, intermediate stops, endpoints, and depots. The charging power ratings are currently at 150 kW, 300 kW, 450 kW, and 600 kW. ABB (2020) claims to have a typical charge time of three to six minutes. At the time when this study is authored no studies have been conducted on the GHG emissions LCA for this technology. The technical specifications of the opportunity pantograph charger are similar to the stationary charging station, therefore this study assumes the same life cycle GHG emissions values for the opportunity charging infrastructure as the values reported in the previous chapter.

In the same study mentioned earlier, Balieu, Chen, & Kringos (2019) calculated the life cycle GHG emissions from implementing Overhead Pantograph Rails (OPR) onto the roads. For this technology, the study has set the requirement of utilising steel pillars spaced 50 m apart with copper cabling. The GHG emissions reported in the study are as follows. First, the construction phase includes raw material production, transportation, and installation processes. The authors stated that significant amounts of copper and steel used produced high amounts of GHG emissions and resulted in approximately 140.6 tonCO_2e/km. Then, the maintenance phase observes the same winter maintenance and rehabilitation operations and amounts to 164.6 tonCO_2e/km. Lastly, in summary, the final results from construction, maintenance, and rehabilitation produced a life cycle GHG emissions of 319,960 ~ 484,340 kgCO_2e/km, dependent on the frequency of rehabilitation operations. To simplify the assessment, this study assumes that the overhead pantograph rails are capable of supplying 100 kW to heavy-duty vehicles.

Figure 12 – (Left) Volvo bus with a rooftop pantograph for opportunity pantograph charging.

Figure 13 – (Right) BEV buses with overhead pantograph rails in Vienna.

Source: Rail.cc (2017); Volvo (2018).

3.3 OPERATION CHARGING STRATEGIES

In the previous section, the life cycle GHG emissions from the four charging methods were analysed. *Table 13* summarises the life cycle GHG emissions for each charging method. Then, the next question is: which charging strategy to choose from that will not only benefit the bus operators but also contributes the least to climate change? In this section, the GHG emissions produced from the different operation charging strategies are analysed. To create a comparison of the environmental impacts of different BEV buses, *Table 14* shows the technical specifications of three BEV buses selected for this study.

Table 13 – Summary of the GHG emissions produced by different charging methods.

Recharging Method	Stationary Charging Station	Wireless Power Transfer	Opportunity Pantograph Charger	Overhead Pantograph Rails
Life Cycle GHG Emissions	8,182.3 ~ 8,538.6 kgCO_2e	287,469 ~ 473,882 kgCO_2e/km	8,182.3 ~ 8,538.6 kgCO_2e	319,960 ~ 484,340 kgCO_2e/km

Table 14 – Technical specifications of selected BEV buses.

Vehicle Specifications	BYD K9	Yutong E12	Volvo 7900
Kerb Weight (kg)	14,400	12,500	12,060
Length (m)	12	12.2	12
Battery Capacity (kWh)	324	374	150, 200, 250

Battery Composition	Lithium Iron Phosphate ($LiFePO_4$)	$LiPePO_4$	Lithium-Ion
Operational Range (km)	250 – 350	225 – 280	≥ 200
Energy Consumption (kWh/km)	0.93 ~ 1.30	1.34 ~ 1.66	0.75 ~ 1.25
Charging Options	AC 80 kW	DC 150 kW	OppCharge[3] 150 kW, 300 kW, 450 kW CCS[4] DC 150 kW CCS AC 11 kW
Service Life (km)	650,000	650,000	650,000

Energy consumption and charging schedules are taken from the literature. Gao et al. (2017) evaluated the energy consumption and battery performance of BEV city transit buses operating in real-world day-to-day routes and standardised bus drive cycles. To reflect real-world driving conditions, the study utilised the Oak Ridge National Laboratory (ORNL) MD conventional bus database and conducted a case study for the city of Knoxville, USA. The bus simulation included a 324 kWh battery, an EV charger power of 90 kW (minimum two hours per charge), energy consumption of 1.35 kWh/km, average regenerative braking of approximately 0.39 kWh/km, and an overall simulated distance of 66,461 km. The simulated results showed that 14.2% (87 out of 610 days) of the overall drive days are occupied by proactive charging. The simulated 324 kWh battery offered 10 hours of operational driving per day and a range of 242 km without any on-route charging. Their simulation also included decreasing the battery size to 150 kWh, in which the results showed an increase to 208 days of proactive charging (approximately 34.1%). The operation flexibility is dependent on the bus operator's BEV bus selection, such as buses with large battery capacities to satisfy any route lengths, or small battery capacities to operate on shorter routes but with frequent opportunity charging.

To address the next focus for this study, an operations model was created to analyse the GHG emissions produced from the different operation charging strategies. Consider a particular bus route with the total route length L (km) and a daily service time of 18 hours. The total average travel time T_t (min.) of a single BEV bus along route L can be calculated from the following formula:

$$T_t = \frac{L}{S} + \sum_{s=1}^{N} T_{i,s} + T_r \tag{5}$$

Where,
L = Total route length (km)
S = Average speed (km/h)
T_i = Idling time at bus stop s (s)
T_r = Time spent in traffic (min.)

[3] Opportunity Charging System by ABB (2020), Volvo Bus Corporation, and several other stakeholders
[4] Combined Charging System (CCS)

The following section explores the alternative scenarios of various charging strategies. The environmental load (e.g. GHG emissions) is calculated according to the following variables inputs. First, with every route, the service frequency starts at five min/bus and increases in increments of five up to 60 min/bus. Next, in the absence of real-world driving data, the following average speeds are therefore set for each route type: urban 40 km/h, suburban 50 km/h, and highway 80 km/h. Then, the number of stops ranges from 1 ~ 50 (e.g. N = 50) and the idling time at stops ranges from 15 ~ 60 seconds. Finally, to account for traffic or other unforeseeable disruptions to the bus service, slack times are therefore introduced into the bus schedules and ranges from 5 ~ 60 minutes, increasing in increments of five. Hence, these variables affect the overall travel time per route and consequently the fleet size of a bus operator. The required buses to service a given route can be calculated from the following formula:

$$B_N = \frac{f_h}{f_r} = \frac{T_t}{f_m} \tag{6}$$

Where,
B_N = Number of buses (buses/day)
f_h = Service frequency (buses/hour)
f_r = Route frequency (trips/hour/bus)
f_m = Service frequency (min./bus)

The charging frequency can be calculated from the following formula:

$$f_c = \frac{B_d}{R_c} = \frac{f_r \times T_o \times L}{R_d} \tag{7}$$

Where,
f_c = Charging frequency (charges/day/bus)
B_d = Number of buses (trips/day/bus)
R_c = Service operational range (routes/charge)
R_d = Bus operational range (km/charge)
T_o = Operation hours (hours/day, for this study the operation hours is set to 18 hours per day)

Finally, the GHG emissions produced by charging a single BEV bus to service a particular route can be calculated from the following formula:

$$M_{GHG,d} = E_c \times EF \times f_c \tag{8}$$

Where,
$M_{GHG,d}$ = Mass of GHG emissions ($kgCO_2e$/bus/day)
E_c = Battery capacity (kWh)
EF = Electricity generation emissions factor ($kgCO_2e$/kWh, this study observes the Australian emissions factor of 0.74 $kgCO_2e$/kWh)

3.3.1 OPERATIONS SCENARIOS

The combination of the multiple influencing variables indicates that the operations model will produce approximately four million GHG emissions results. To simplify the assessment, this section has set the following predetermined influencing variables:

Service frequency:	5, 10, and 15 min./bus
Number of stops:	30
Idling time at stops:	30 s
Traffic:	15 min.
Operation Hours:	18 hours/day

First, consider a series of bus routes that operate predominately in urban city traffic conditions. The average speed in this urban setting is 40 km/h. For this urban setting, the operations model (see *Figure B1*) analyses five route lengths for every service frequency: 5, 7.5, 10, 12.5, and 15 km. Next, consider a series of bus routes operating on suburban streets and traffic conditions. The average speed is increased to 50 km/h, and the operations model analyses 13 route lengths for every service frequency: 15 ~ 45 km. Last, consider a series of highways with an average speed of 80 km/h. The operations model analyses 11 route lengths for every service frequency: 25 ~ 50 km. The series of results are graphically illustrated in *Figures B4 – B24*.

3.4 RESULTS

The South African electricity grid-mix is the most carbon-intensive amongst the analysed countries, followed by Poland, Indonesia, Australia, India, and then China. On the other hand, China generated the most electricity and consequently produced the most GHG emissions. However, China also has the highest shares of low emission and renewables technologies in their grid-mix, with the majority of the low emission electricity generated by hydro and nuclear power. Indonesia, Australia, and Poland do not have nuclear power plants. The total amount of electricity generated by the corresponding countries in 2019 are shown in *Table 15*, and *Table 16* shows the carbon intensities of the analysed countries. Here, the results show that the emissions factors are significantly higher than that of a diesel bus. The break-even points for each country can be observed in *Figure B2*. This is the result of the break-even points' sensitivity to three factors: individual fuel emissions factor (*Table 12*), the total amount of electricity generated per year (*Table 15*), and the country's grid-mix. All six countries rely heavily on black coal as the predominant fossil fuel combusted to generate electricity. From *Table 12* it is evident that black coal has a high emissions factor, meaning that compared to other fossil fuels, black coal produces more GHG emissions per kWh generated. Thus answers the question raised in *Section 3.2.5*: when the fossil fuel percentage of an analysed country's grid-mix reaches or falls below the break-even point, the operations environmental load of a BEV bus will be equivalent to that of a diesel bus, therefore it would then be intuitive to promote and implement BEV buses in that country.

The key findings of the break-even analysis specific to Australia are as follows. For the calendar year 2018:

1. The Australian grid-mix comprises of approximately 81.1% of fossil fuels and 18.9% of renewables. The environmental load from electricity generation is approximately 0.65 ~ 0.67 $kgCO_2e/kWh$. BEV buses that operate in Australia will have an average operations environmental load of approximately 84.7 ~ 87.6 $kgCO_2e/100km$. When considering Australia's grid-mix as a combination of all states, the break-even analysis shows that when fossil fuel electricity generation is reduced to approximately 72.0% ~ 81.5% the BEV bus operations environmental load will be equivalent to that of a diesel bus. This indicates that from below the break-even range onwards, the operations environmental load of electricity will be lower than diesel. The analysis also reveals that if the grid-mix is completely independent of fossil fuels, the BEV bus operations environmental load will be approximately 5.42 $kgCO_2e/100km$ and the grid-mix emission factor will be approximately 0.04 $kgCO_2e/kWh$.

2. The New South Wales grid-mix comprises of 82.7% fossil fuels. New South Wales generated the highest amount of electricity than any other state. That being said, significant amounts of electricity is generated from renewables such as hydro, wind, and small-scale solar PVs. The NSW renewables generated more than twice of Queensland's renewables, the next highest generating state. The break-even point is at approximately 67.5% for NSW.

3. The Victorian grid-mix comprises of 82.7% fossil fuels. The state has the highest environmental load in the country at 0.84 $kgCO_2e/kWh$ and 108.9 $kgCO_2e/100km$. Victoria is the only state that combusts brown coal to generate electricity. Brown coal has the highest emissions factor and contributes to approximately 76.0% of all electricity generation. Here, the break-even point is at approximately 58.0%, the lowest of all states.

4. The Queensland grid-mix has a large share of fossil fuels at 91.2% and generated the most electricity from fossil fuels than other states. The break-even point is at approximately 71.0%.

5. Western Australia's grid-mix has a large share of fossil fuels at 91.8%. However, the overall electricity generation environmental load is lower than diesel, as natural gas is the predominant share of the fossil fuels combusted. The break-even point is at approximately 96.5%, which is already higher than the current fossil fuels share.

6. South Australia's grid-mix has an approximate even share of fossil fuels and renewables. Oil is the only fossil fuel combusted to generate electricity; wind and small-scale solar PV has large shares in renewables. The break-even point is at approximately 81.5%, which is

significantly higher than the current fossil fuels share of 48.8%.

7. Tasmania grid-mix comprises of 94.6% renewables. The state has the highest share of electricity generation from hydroelectricity. Indeed, the electricity generated by Tasmanian hydro power stations was higher than any other state's hydro power stations. Even in the scenario of 100% fossil fuels (66.2 $kgCO_2e$/100km; natural gas and oil), the environmental load from operating a BEV bus is still significantly lower than operating a diesel bus.

8. The Northern Territory's grid-mix has the largest share of fossil fuels at 96.0%. Similar to Western Australia, the overall electricity generation environmental load is lower than diesel, as natural gas has a predominant share of the fossil fuels combusted. Even in the scenario of 100% fossil fuels (73.3 $kgCO_2e$/100km; natural gas and oil), it is less environmentally taxing to operate BEV buses than diesel buses.

Applying the scenario of 100% renewables does not nullify GHG emissions. Indeed, the emissions factors of renewable technologies are derived from their respective life cycle emissions. Each renewable technology offers substantial GHG emissions reduction compared to fossil fuel combustion, but will never be completely emissions-free. Implementing renewable technologies will vary across each country and are dependent on geography, climate, government incentives, and energy generation efficiency. For example, in the scenario of 100% renewables, Queensland, Western Australia, and the Northern Territory will have the highest operations environmental load of 8.62 $kgCO_2e$/100km, 6.63 $kgCO_2e$/100km, and 10.76 $kgCO_2e$/100km, respectively. These states have an abundance of sunshine throughout the year, therefore small-scale solar PV's contributed to large shares of renewables, which also has higher emissions factors than other renewables (see *Table 12*). On the other hand, Tasmania receives large amounts of rainfall annually, and therefore have built 30 hydro power stations in high-rainfall catchments across the state. Thus, it would be under the discretion of governments and policymakers to consider and determine the compromises of the available fossil fuel and renewables technologies so that the combination of the grid-mixes would be the most beneficial to the environment of a given region. This will most certainly conceive new opportunities, risks, and challenges.

Table 15 – Electricity generation by country.

GWh	South Africa	China	India	Indonesia	Australia	Poland
Black Coal	225,000.0	4,732,400.0	1,176,300.0	156,400.0	156,562.9	134,700.0
Natural Gas	1,900.0	223,600.0	74,300.0	59,600.0	50,244.9	12,400.0
Oil Products	100.0	10,700.0	10,100.0	20,200.0	5,258.6	1,200.0
Hydro	900.0	1,202,400.0	139,700.0	16,400.0	17,451.9	2,000.0
Wind	8,160.1	99,822.8	24,388.7	3.0	16,266.5	10,765.9
Nuclear	11,100.0	294,400.0	39,100.0	0.0	0.0	0.0
Biomass	572.9	32,021.1	23,719.3	840.6	3,539.3	7,162.0
Solar PV	6,836.6	31,697.0	7,679.3	10.4	12,081.1	109.5

Total	254,569.6	6,627,040.9	1,495,287.3	253,454.0	261,405.2	168,337.4

Table 16 – Emissions factors, operations emissions factors, and break-even point of the analysed countries.

	South Africa	China	India	Indonesia	Australia	Poland
Total GHG Emissions ($ktCO_2e$)	201,985.8	4,368,321.3	1,096,153.7	183,895.4	170,017.5	127,344.3
Emissions Factor ($kgCO_2e/kWh$)	0.79 ~ 0.93	0.66 ~ 0.72	0.71 ~ 0.73	0.73 ~ 0.76	0.79 ~ 0.81	0.77 ~ 0.78
Operations Emissions Factor ($kgCO_2e/100km$)	102.8 ~ 120.4	82.7 ~ 85.4	91.8 ~ 95.0	94.0 ~ 98.6	84.3 ~ 96.3	98.0 ~ 101.2
Fossil Fuel (%)	89.2%	75.0%	84.3%	93.2%	81.1%	88.1%
Break-Even Point	66.0% ~ 72.5%	68.0% ~ 74.5%	68.5% ~ 75.0%	76.5% ~ 83.5%	74.5% ~ 81.5%	69.0% ~ 75.5%

Source: Moro & Lonza (2018); BP Statistical Review of World Energy (2019); GREET© (2019); Department of Environment and Energy (2019e); Carbon Footprint (2020).

The results of the operations scenarios are as follows. First, *Figures B4 – B6* demonstrates that the WPT and OPR charging methods have high life cycle GHG emissions that increase linearly with route length. Suppose a bus operator chooses to charge the buses using the WPT and/or OPR technologies. Taking an urban route length of 5 km, for example, WPT and OPR would produce approximately 1,437 ~ 2,369 tonCO_2e and 1,600 ~ 2,422 tonCO_2e, respectively. Here, the electricity generation GHG emissions must then be additionally included in the calculations. On the other end of the spectrum, for a highway route length of 50 km, WPT and OPR would produce approximately 14,373 ~ 23,694 tonCO_2e and 15,998 ~ 24,217 tonCO_2e, respectively and once again does not include electricity generation GHG emissions. On the other hand, suppose a bus operator chooses to charge the buses when stationary with two units. In this instance, the stationary charging stations would be installed at the bus depot, and for the opportunity pantograph charger, one would be installed at the bus terminal and the other one at the bus depot. For both options, the life cycle GHG emissions would be reduced to 16.4 ~ 17.1 tonCO_2e, excluding electricity generation GHG emissions.

Then, *Figures B7 – B9* provides insight on how battery capacity and bus route range influences the proactive charging requirements. In terms of charging requirements, the small battery capacity of the Volvo 7900 requires up to 2.5 and 2.25 times more frequent charging compared to the BYD K9 and Yutong E12, respectively. If the model was to assume the best case range of 200 km per charge, the Volvo 7900 has the potential to operate the full 18 hours/day for routes up to 7.5 km. In comparison, the BYD K9 and Yutong E12 have the potential to operate the full 18 hours/day for routes up to 15 km and 12.5 km, respectively. The charging requirements are further analysed from another perspective. Suppose a bus operator chooses to charge the buses using the stationary charging stations. *Table 17* provides the minimum recharging time required for each bus. It is worth mentioning that the WPT and OPR technologies recharge the batteries whilst the bus is in motion, therefore in this study, these two technologies require no charging time.

Table 17 – Minimum proactive charging time for the two stationary charging stations.

	BYD K9	Yutong E12	Volvo 7900
Stationary Charging Station	4 h 3 min.	4 h 41 min.	1 h 53 min.
Opportunity Pantograph Charger	1 h 5 min.	1 h 15 minutes	30 min.

Last, in the absence of real-world driving data, *Figures B10 – B21* indicates that the route length is a major contributing factor of energy consumption and consequently influences the amount of GHG emissions produced by an individual BEV bus per day. *Figures B22 – B24* shows that the service frequency impacts the required number of buses required per day on a particular route and that in turn impacts the amount of GHG emissions produced by the bus fleet as a whole.

3.5 DISCUSSION

It is vastly inaccurate to claim BEV buses as absolute zero-emissions vehicles. In the case of BEV buses, their tailpipe emissions are shifted to electricity generation emissions. The carbon intensity of a grid-mix is determined by a given region's unique resources, environmental characteristics, and generation methods. There are multiple electricity generation methods available with each source producing GHG emissions in varying magnitudes. To achieve net-zero emissions the supply power must have no emissions and emissions generated as a result of production and transportation of the BEV bus should be offset.

When applying a case study approach for Australia, the operations emissions factor of a BEV bus amounts to approximately **81.6 ~ 95.9 kgCO$_2$e/100km**. In comparison, the resulting operations emissions factor of a diesel bus amounts to approximately **77.7 ~ 84.7 kgCO$_2$e/100km**. Therefore, utilising this value and comparing it to the reported grid-mix carbon intensities of 49 countries, the calculated results show that South Africa, China, India, Indonesia, Australia, and Poland yielded higher GHG emissions from electricity generation than diesel fuel combustion. The results from the break-even analysis indicate that when the fossil fuel percentage of an analysed country's grid-mix reaches or falls below the break-even point, the operations environmental load of a BEV bus will be equivalent to that of a diesel bus, therefore it would then be intuitive to promote and implement BEV buses in that country. It is worth noting that replacing all fossil fuels with renewables does not nullify GHG emissions, as renewables technologies produce their respective life cycle emissions, albeit significantly lower in comparison to fossil fuel technologies. Implementing renewable technologies will vary from country to country and are dependent on geography, climate, government incentives, and energy generation efficiency. It would be under the discretion of governments and policymakers to consider and determine the compromises of the available fossil fuel and renewables technologies so that the amalgamation of the grid-mixes would be the most beneficial to the environment of a given region. This will most certainly conceive new opportunities, risks, and challenges.

Thus, the results obtained answer the question raised in *Section 3.3*: which charging strategy to choose from that will not only benefit the bus operators but also contributes the least to climate change? From a logistical point of view, the optimal arrangement for BEV buses in the urban and suburban settings is to deploy a BEV bus (Volvo 7900) with a small battery capacity and charge with the opportunity pantograph charger (150 ~ 300 kW) at the terminals during operational hours and the depots at night. Although the smaller battery capacity requires more frequent charging, the Volvo 7900 can be recharged in approximately 30 minutes, depending on the charging capabilities chosen. The optimal arrangement in highway settings is to deploy a BEV bus with a larger battery capacity (BYD K9 and Yutong E12) that do not require frequent charging as it has sufficient range per charge to complete partial or total of the daily scheduled route. The larger battery requires longer charging times that range from approximately one to five hours (dependent on charging capacity), therefore charging may only be done at the depot or in a dedicated charging complex. It should be noted that bus operators may consider using stationary charging stations if the charging capacity matches the opportunity pantograph charger. However at the time when this study is authored the stationary charging station is capable of up to 80 kW of charging power only, as determined by the manufacturer. Additionally, the opportunity pantograph charger has the additional benefit of charging the bus autonomously, which can contribute to increased safety and time reduction.

Alternatively, deploying WPT or OPR technologies may be considered in urban settings due to the low traffic velocity and omits the need for scheduled charging. However, from an environmental point of view upgrading a 1 km stretch of road with WPT or OPR technologies produces the equivalent life cycle GHG emissions of installing 34.4 ~ 69.1 stationary charging stations or opportunity pantograph chargers. Therefore the WPT and OPR technologies will have an adverse effect on the environment. The equations from the operations charging strategies section demonstrate that BEV bus charging emissions are influenced by battery capacity and energy efficiency (consequently the operational range). Thus, when assuming the best-case range, a BEV bus with a smaller battery capacity contributes the least to the charging emissions (Volvo 7900, BYD K9, and then Yutong E12). On the contrary, when assuming the worse-cast range the BYD K9 contributes the least to the charging emissions, followed by the Volvo 7900 and then the Yutong E12.

This study emphasises that the fixed bus routes are extremely beneficial to the recharging schedules of the BEV buses. While the WPT technology allows BEV buses to instantaneously recharge whilst the buses are in motion, it also requires bus operators to analyse the dynamic traffic environment of each of their bus routes. The velocity profile of the route is dependent on the operation time. To clarify this further, take an arterial road as an example. During heavy traffic the transit bus's mean operational acceleration and velocity are lower, implicating that it will be optimal to deploy BEV buses with smaller battery capacities and recharge with WPT technology because the low velocity increases charging time. As the operational velocity increases, the charging time decreases. Hence, on roads where the traffic flows freely and at higher velocities, it would be more beneficial to deploy BEV buses with larger battery capacities. Additionally, from an efficiency point of view, the improvements are clear: smaller batteries improve energy efficiency due to their lighter weight, and significantly less time is spent charging the BEV buses in the bus depot (Lee, Ji, & Cho 2019).

The OPR technology has the same charging advantages as the WPT technology, however logistically it may disadvantage other road users. BEV buses recharging with the OPR technology are confined to the roads where the rails are installed. In the event of a bus breakdown in the middle of the route, the entire service route will be disrupted as the buses from the following services would not be able to overtake the broken-down bus. This is also true for high traffic congestions, where it would not be possible to reroute the BEV buses. Additionally, the rails are exposed to the elements and also subject to wear-and-tear, consequently increasing maintenance requirements and reducing operations efficiency.

On the other hand, the arrangement will be different for BEV buses in highway settings. The larger battery capacities on the BYD K9 and Yutong E12 will have the advantage of operating over longer bus routes. The BEV buses will not require frequent charging as it has sufficient range per charge to complete partial or total of the daily scheduled route. The larger battery requires longer charging times that range from approximately one to five hours (dependent on charging capacity), therefore charging may only be done at the depot or in a dedicated charging complex. Alternatively, deploying WPT or OPR technologies may be costly and impractical, especially for OPR technology. Furthermore, the charging time decreases at higher vehicle velocities, thus reducing the benefits of WPT charging.

3.5.1 INFLUENCES OF ASSUMPTIONS AND UNCERTAINTIES

The break-even analysis and operation charging strategies calculations demonstrated in this study are influenced by a number of assumptions and uncertainties. First, in the absence of real-world driving data, this study assumes the BEV buses' technical specifications provided by their respective manufacturers. In reality, the claimed performances are significantly different from real-world performances. BEV buses' energy consumptions and consequently their operational ranges are highly influenced by driving behaviour, traffic condition, vehicle speed, road conditions, road topology, and weather conditions. The multiple factors constitute to varying vehicle performances that cannot be replicated under laboratory conducted type-approval tests. Additionally, auxiliary equipment use (such as air conditioning for vehicle cabin heating or cooling), or regenerative braking are weather and traffic dependent and therefore not considered in this study.

The specific values for charging infrastructure life cycle GHG emissions vary depending on the parameters set by the reviewed literature. The construction phase, which also includes materials production and installation, induces uncertainty into the analysis with values from raw material manufacturing emissions. The values for the WPT and pantograph chargers are taken from reviewed literature based on European conditions, whereas the stationary charging station was analysed under Australian conditions. The energy consumption throughout the life cycle phases of each technology is subjected to a region's grid-mix carbon intensity. For example, *Figure B3* graphically shows the grid-mix emissions factor variation across Australia. The electricity generation GHG emissions vary with the grid-mixes fuel share. Victoria is the only state that combusts brown coal to generate electricity and thus has the highest environmental load in the country. From this example, it is evident that the life cycle GHG emissions would be reported higher if the reviewed literature of the WPT and pantograph chargers were conducted in countries and regions with high carbon-intense grid-mixes instead. Therefore it is also necessary to assume that numerous raw materials and sub-components in the construction phase may originate from other regions and countries, which further induces complications and uncertainty to the analysis. The maintenance phase contributes to the degree of uncertainty by the frequency set by the reviewed literature. In reality, the degree of wear and tear on the equipment or road commissions unpredictable maintenance requirements. At the time when this work is authored no information can be found in the literature regarding the life cycle GHG emissions of opportunity pantograph chargers. Therefore, this study had assumed the same life cycle GHG emissions values as the stationary charging infrastructure reported in the previous chapter.

3.6 CONCLUSION

This chapter investigated the production of GHG emissions from implementing heavy vehicle charging infrastructures into the current road network. The study analysed the GHG emissions produced from electricity generation of 49 different countries from available data in the public domain. As the analytical process is dependent on localised emissions outputs, Australia was used as a case study to demonstrate the process and compared against 48 other countries. This led to the analysis of six countries with high carbon-intense electricity grid-mixes, to which a break-even point was determined where the environmental load for electricity generation is equal to diesel fuel combustion. The study then moved on to comparing the life cycle GHG emissions of a stationary charging station, wireless power transfer, opportunity pantograph charging, and overhead pantograph rails. Three representative BEV buses were then chosen for this study's analysis, and together with the analyses of the GHG emissions produced from the different operation charging strategies the optimal arrangement from the available charging strategies was determined. Thus, based on the results from the break-even analysis and operation charging strategies, this study makes the following conclusion.

Considering the environment, it would be intuitive to promote and implement BEV buses in countries with low carbon-intense grid-mixes, as the operations environmental load of a BEV bus will be equivalent to or less than that of a diesel bus. Implementing renewable technologies to decarbonise the grid-mix will vary across each country and are dependent on geography, climate, government incentives, and energy generation efficiency. New challenges, opportunities, and risks would be conceived when governments and policymakers consider and determine the compromises of the available fossil fuel and renewables technologies so that the amalgamation of the grid-mixes would be the most beneficial to the environment of a given region.

Considering the logistics of the operation, the optimal arrangement for BEV buses in the urban and suburban settings is to deploy a BEV bus with a small battery capacity and charge with the opportunity pantograph charger at the terminals during operational hours and the depots at night. The BEV will require more frequent charging, however depending on the charging capabilities chosen the BEV bus can be recharged in approximately 30 minutes. The optimal arrangement in highway settings is to deploy a BEV bus with a larger battery capacity that does not require frequent charging as it has sufficient range per charge to complete partial or total of the daily scheduled route. Charging may only be done at the depot or in a dedicated charging complex as the larger battery requires up to five hours of charging. It would be up to the bus operator's discretion to use stationary charging stations if the charging capacity matches the opportunity pantograph charger. However at the time when this study is authored the stationary charging station is capable of up to 80 kW of charging power only, as determined by the manufacturer. The opportunity pantograph charger has the additional benefit of charging the bus autonomously, which can contribute to increased safety and time reduction.

4 CHAPTER 4: LCA OF BUS PRODUCTION[5]

4.1 INTRODUCTION

The environmental burden from the life cycle of EV buses is non-negligible and complex to analyse. To fulfil this need, the present chapter applies a case study approach and builds upon existing literature to investigate the life cycle GHG emissions from the production of diesel, hybrid, and BEV buses. Thus, the vehicle life cycle segment of the Complete Life Cycle model is addressed here. Within this chapter, first, the methodology section defines the scope, system boundary, and functional unit adopted for this study. Next, an in-depth and comprehensive process-based LCA of diesel, hybrid, and electric buses is conducted which includes the environmental impact resulting from the production, assembly, transportation, maintenance, and disposal phases. Then, the results present the detailed estimation of GHG emissions produced throughout the life cycle of transit buses and discuss the environmental sustainability of the three bus variants. Finally, a sensitivity analysis is conducted to address the technological developments uncertainties and assumptions made in this case study.

4.1.1 ASSESSING LCA STUDIES ON THE EMERGING BEV TECHNOLOGIES

There are many LCA studies conducted by researchers on the environmental impact of emerging BEV technologies. Pero, Delogu & Pierini (2018) conducted a comparative LCA of ICEVs and BEVs. The production, operations, and disposal stages were considered in both vehicles' entire life cycle. It was reported that BEVs has a lower operations impact, contributing to a 36% reduction of total life cycle impact with respect to ICEVs. The BEV's lower operations impact is due to the absence of tailpipe emissions and lower environmental loads involving electricity generation from different fuel sources. Additionally, the average European electricity grid-mix is relatively low in GHG intensity. The break-even point for the operations stage of a BEV compared to an ICEV occurs at about 45,000 km (when assuming a service life of 150,000). The break-even point is further decreased to approximately 30,000 km when assuming the Norwegian electricity grid-mix. However, when assuming the Polish electricity grid-mix, the study found no break-even point occurs even when the considered service life is extended to 250,000 km. Additionally, the study excluded the production transportation and vehicle maintenance from the system boundaries, as it was stated that their influence on total life cycle impact is negligible and no specific information was available.

[5] The contents of this chapter have been adapted from the publication: **Zhao, E.**, Walker, P.D. & Surawski, N.C. 2021, 'Emissions Life Cycle Analysis of Diesel, Hybrid, and Electric Buses', *Journal of Automotive Engineering*, pp. 1-13.

Similarly, Tagliaferri et al. (2016) presented a life cycle assessment of a BEV based on the lithium-ion technology in Europe and compared the results with an ICEV. The study applied a cradle-to-grave approach and included the manufacturing, operations, and disposal phases into the assessment. The results show that the GHG emissions produced from the ICEV operations phase are higher than BEVs by almost 50%, whereas the manufacturing phase of BEVs is almost double that of ICEVs. The manufacturing phase of BEVs from complex electrified powertrain systems and batteries productions contributed to higher GWP. The higher environmental burden from BEV manufacturing is associated with the toxicity from producing the materials required to manufacture specific electronic components and batteries. It was also concluded that the disposal phase was not a significant contributor of GWP to the total environmental impact.

The GHG emissions of various types of transit buses were assessed Lajunen & Lipman (2016). The study was based on the operating environment case scenarios for Finland and California (USA) WTT and operations phases. Simulated results showed that BEV buses consumed very low energy due to higher powertrain efficiency and regenerative braking capabilities. Regarding climate change, BEV buses have reported lower CO_2 emissions for both Finland and California due to both regions having high shares of nuclear and renewables in their respective grid-mixes.

Mierlo, Messagie & Rangaraju (2017) performed a comparative environmental assessment of alternative fuelled vehicles using LCA in Belgium. The results of the study align with the results reported by Tagliaferri et al. (2016). The study found that BEVs have next to no local pollution (no tailpipe emissions) and the operations GHG emissions have been shifted away from areas of dense population and instead relocated to electric power plants, mostly out of city limits. It was also found that a major fraction of toxic emissions stem from batteries and power electronic equipment manufacturing. The mining process of these materials releases toxic substances into the environment.

Hawkins et al. (2012) developed a transparent life cycle inventory of ICEVs and BEVs and applied the inventory to assess the vehicles over a range of impact categories. The studies showed that BEVs powered by the European electricity grid-mix offered a 10% to 24% reduction in GWP relative to ICEVs assuming a service life of 150,000 km. However, it was reported that the production of BEVs is significant in polluting the environment, such as increasing human toxicity, freshwater eutrophication and eco-toxicity, and resource depletion impacts. The results were sensitive to the assumptions made regarding electricity source, operations energy consumption, vehicle service life, and battery replacement frequency. The study concluded that improving the environmental profile of BEVs requires heavy reduction of vehicle production supply chain impacts and reducing the carbon intensity of electricity generation.

Likewise, Ellingsen et al. (2013) provided a transparent inventory for a lithium-ion nickel-cobalt-manganese (NCM) traction battery based on primary data and to report the battery's cradle-to-grave environmental impacts. The results showed that the environmental burden was mainly caused by the production chains of battery cell manufacture, positive electrode paste, and a negative current collector. It was also shown that producing the battery cells with electricity from low carbon intensity grid-mixes, recycling spent materials, and developing longer battery lifetimes are the most effective approaches to reduce GWP.

In summary, the unanimous agreement across the reviewed studies shows that the production of BEVs significantly impacts the environment, mainly caused by the production materials required to manufacture LIB specific components. Furthermore, most studies focused on the LCA of light-duty BEVs. Although the studies are relevant to the LCA of vehicle production, a detailed analysis is required for the context of BEV buses.

4.2 METHODOLOGY

4.2.1 SCOPE DEFINITION

Applying a case study approach for Australia, this study performs an emissions LCA of diesel, hybrid, and BEV buses in the Australian bus fleet. Several bus operators have begun trailing BEV buses on bus routes in the City of Sydney. The objective of this study is to calculate the amount of life cycle GHG emissions produced during production to investigate the environmental implications for when the Australian bus fleet eventually transitions to BEV buses.

4.2.2 SYSTEM BOUNDARY

A system boundary is set within which process data are collected to meet the objective of the study. Processes found to contribute negligibly to the end results are excluded. There is a complex interaction between vehicles and larger systems, such as infrastructure, emerging technologies, power generation, and transportation options specific to a region. It is difficult to determine which processes can be excluded, however, this study presents what is believed to be a complete LCA for diesel, hybrid, and electric (hereinafter referred to as ICEV, HEV, and BEV buses) specific to vehicle production.

A complete LCA can be divided into two studies: the Well-to-Wheel (WTW) life cycle and the equipment life cycle (or Cradle-to-Grave) (Nordelöf et al. 2014). The former focuses on the life cycle of the energy carrier (fossil fuel or electricity) that propels the vehicle. The latter, and also the focus of this study, consists of processes specific to vehicle production. Regarding the WTW life cycle, it is beyond the scope of this article and rigorously addressed in another study.

Thus, this study performs an equipment LCA of diesel, hybrid, and electric buses through the investigation of the main GHG pollutants released during the five phases of production, assembly, transportation, maintenance, and disposal (see *Figure 14*). Materials production is the first process that involves the extraction of raw materials and the manufacturing of vehicle components. It is followed by the assembly phase, which examines the energy consumed to assemble vehicle components together and build a functioning bus. Then, the transportation phase involves the GHG emissions produced from the process of shipping the fully built buses from their respective manufacturing plants to Sydney, Australia. The next phase relates to the aspects of the operations phase that relates solely to the equipment life cycle, namely the periodic replacement of components and servicing, which this study refers to as the maintenance phase. The remaining aspects of the operations phase relate to the WTW life cycle and are excluded from the study. Finally, the disposal phase involves vehicle components disassembly, materials segregation, recycling, and disposal. The scope of recycling is limited to the process prior to implementing the recycled materials into new products.

An ICEV, HEV, and BEV bus are selected that best represent the current bus fleet operating in Sydney NSW, Australia. It is sufficient to assume that many vehicle components do not differ significantly from each other, nonetheless, the technical specifications of the buses are standardised to provide a frame of reference. To clarify this further, this study standardises most of the components, such as vehicle chassis, interior, exterior, wheels, tyres, etc. The emissions produced from manufacturing components are specific to the bus variant, namely the powertrain and associated components.

Unless stated otherwise, the functional unit employed in this study is a unit mass of GHG per unit of material production or assembly process: **$kgCO_2e/kg$**.

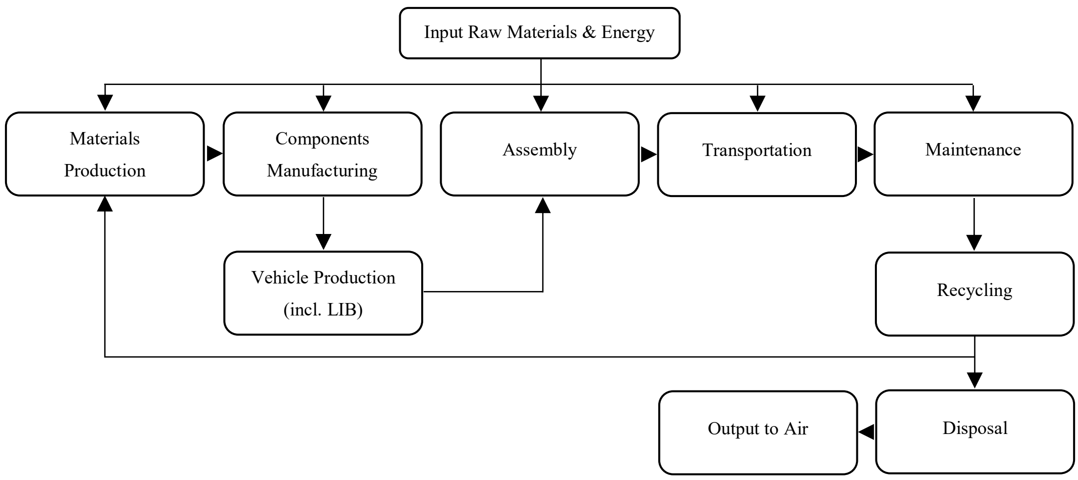


Figure 14 – System boundary for bus production life cycle.

4.2.3 LIFE CYCLE INVENTORY ANALYSIS

Manufacturing a diesel, hybrid, and BEV bus requires components, materials, and processes unique to bus variant, insinuating that the life cycle emissions from constructing the buses differ at each life cycle phase (Nealer, Reichmuth, & Anair 2015). Thus, this section calculates the total amount of emissions associated with the production, assembly, maintenance, and disposal of diesel, hybrid, and BEV buses in the Australian bus fleet. The focus, in particular, is on the GHG that contributes to global warming: Carbon Dioxide (CO_2), Methane (CH_4), and Nitrous Oxide (N_2O), with the functional unit **kgCO_2e**. An appropriate comparison of the LCA requires the inclusion of all relevant differences and similarities across the three bus variants (Hawkins et. al 2012).

4.2.3.1 REFERENCE BUS SPECIFICATIONS

An ICEV, HEV, and BEV bus [Volvo B8R Low Entry (Volvo 2019), Volvo B5L Hybrid (Volvo 2019), and BYD K9 (BYD 2017), respectively] are chosen as a baseline model for comparison. The specifications of the chosen buses, such as passenger capacity and dimensions, are currently in service in the Australian bus fleet. Furthermore, the manufacturers have readily provided the necessary data the authors need to conduct an LCA. To ensure the comparability of the three bus variants, a common generic glider (a vehicle absent of its powertrain components) is established, hereinafter referred to as Reference Bus (see *Figure 15*). Here, the bus specifications are standardised wherever possible to provide a frame of reference. *Table 18* provides key specifications for the Reference Bus used in this study.

Table 18 – Key specifications of diesel, hybrid, BEV, and representative bus.

Specifications	Volvo B8RLE	Volvo B5L Hybrid	BYD K9	Reference Bus
Dimensions				
Wheelbase (m)	6.80 m	6.30 m	6.20 m	6.50 m
Length (m)	12.5 m	12.5 m	12 m	12.5 m
Width (m)	2.5 m	2.6 m	2.6 m	2.5 m
Height (m)	2.3 m	2.3 m	3.2 m	2.5 m
Kerb Weight (kg)	12,700 kg	12,400 kg	14,400 kg	-
Gross Weight (kg)	19,000 kg (GVM)	18,600 kg (GVM)	19,700 kg (GVM)	-
Passenger Capacity	35 seated, 27 standing			
Chassis				
Suspension	Air Suspension			
Brakes	Front & Rear Disc, Anti-Lock Braking System (ABS)			
Tyres	275/70R 22.5”			
Frame	Carbon Steel			
Powertrain				

Engine/Motor Type	8 L Inline 6-Cylinder	5.1 L Inline 4-Cylinder AC Permanent Magnet	AC Synchronous (in-wheel motors)	-
Max Power (kW)	246 kW	177 kW; 110 kW	2 x 150 kW	-
Torque (Nm)	1,200 Nm	918 Nm; 800 Nm	2 x 550 Nm	
Gearbox	6-Speed Automatic	12-Speed Automatic	-	-
Fuel Tank & AdBlue (L)	300 L; 50 L	220 L; 30 L	-	
Battery Capacity (kWh | Ah)	-	19 kWh[6]	324 kWh | 600 Ah	-
Fuel Type	Diesel	Diesel	-	-
Battery Composition	-	$LiFePO_4$	$LiFePO_4$	
Fuel/Power Consumption (L/100km, kWh/km)	29.2 L/100km[7]	20.4 L/100km[5]	120 kWh/100 km[8]	-
Charging Capacity (kW)	-	-	80 kW	-
Charge Time (h)	-	-	3 h @ 80 kW (fast), 6 h @ 60 kW (normal)	-

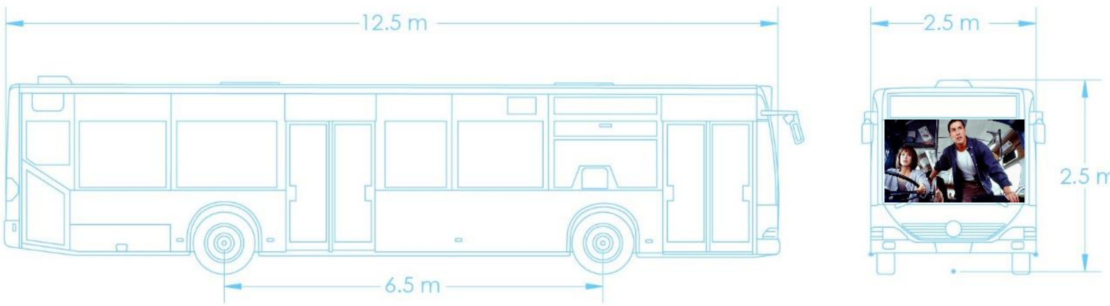


Figure 15 – Reference bus technical drawing.

Source: Speed (1994); CAD-Block (2020).

4.2.3.2 PRODUCTION PHASE

The production phase covers the entire manufacturing process of the buses. This includes the stages from raw materials extraction to the manufacturing of various bus components. For this phase, the data collection involves determining the components' typology, weight, and quantity of materials, as well as the components' manufacturing processes. Before analysing the emissions produced in this phase, it is necessary to apply a breakdown approach and divide the reference bus into assemblies, components, and structures. Industry inventories and reports regarding component masses, manufacturing processes, and materials were utilised whenever the information and data were available in the public domain. Similar to the previous chapters, this study utilises the GREET® (2019) model to estimate the GHG emissions rate per unit of material

[6] Lajunen & Lipman (2016)
[7] Australian Bureau of Statistics (2019)
[8] BYD Auto (2017)

weight. The total production emissions can be found in *Table C4*. There are uncertainties regarding materials production, as the emissions vary depending on the assumptions made, such as the degree of virgin and recycled materials used. Additionally, the carbon intensity of a region's grid-mix influences energy consumption, which in turn induces uncertainty in the raw material manufacturing emissions. For example, if a bus is produced in a certain region, it is still necessary to assume that numerous sub-components and raw materials may originate from various parts of the world, where the manufacturing and assembly processes will then vary in their degrees of carbon intensity.

4.2.3.2.1 MATERIALS BREAKDOWN

An input-output approach is applied to estimate the emissions of the three buses in the production phase. First, the weight of each essential component (input) is determined. Then, the GHG emissions rate per unit of material weight (output) (e.g. $kgCO_2e/kgAl$ for aluminium) is estimated. Last, the total emissions of each component are calculated by multiplying the GHG emissions rate with the weight of each component. The environmental load from the production phase is designated as $\boldsymbol{E_{production}}$ **($kgCO_2e$)** hereinafter. The GHG emissions produced from these materials are calculated using the cradle-to-grave emissions data extracted from the GREET® (2019) model. This section estimates the proportional weight of each material type that contributes to the total weight of each unit given in the product specifications.

Consumable components such as various fluids (transmission fluids, brake fluids, engine oil, and coolant), brake pads, and tyres have been included in the materials analysis. Furthermore, these consumable components require periodic replacement during the lifetime of the buses. It is, therefore, necessary to incorporate the emissions produced from the consumable components' initial installations and replacements. The GHG emissions intensity per unit of material weight produced can be found in *Table C1*.

4.2.3.2.2 BATTERY MANUFACTURING

The battery pack is an essential component to the HEV and BEV bus, comprising of a cooling system, battery cells, packaging, and Battery Management System (BMS). The modelled battery pack is split into three smaller packs, installed on the back of the bus (see *Figure 16*) (Tagliaferri et al. 2016). The battery thermal management is done by the cooling system and is made up of the radiator, manifolds, clamps, pipe fittings, thermal gap pads, and coolant. Battery performance is achieved by the BMS, which includes the Battery Management Board (BMB), Integrated Battery Interface System (IBIS), fixings, and high and low voltage systems. The battery cells are made up of five subcomponents: anode, cathode, electrolyte, separator, and cell container. The packaging is divided into three subcomponents: battery retention, battery tray, and module packaging. In their study, Ellingsen et al. (2013) report that the battery assembly process requires little energy, as it is mainly performed using manual labour. The only direct energy requirement is for the welding process, which itself only amounts to 3.89×10^{-3} kWh per kWh of battery capacity. There is a lack of access to industry data for the GHG emission of battery packaging and BMS production. The production of lithium-ion batteries requires extracting and refining rare earth metals. It is a carbon-intense process involving high heat and sterile conditions during manufacturing. The GHG emissions from energy use are highly sensitive to a region's electricity grid-mix.

Literature has suggested that most early BEV battery LCAs relied on only a few primary sources for emissions inventories, rendering high degrees of uncertainty and may not accurately represent the multiple BEV battery production facilities operating around the globe (see *Table C2*). Many of these studies have indicated that a large share of GHG emissions is produced from the electricity used in manufacturing. Different battery types also influence the final LCA results, as some battery chemistries hold higher concentrations of energy-intense metals. Furthermore, these studies also typically do not incorporate the disposal (including recycling) phase into the end results, therefore there is significant uncertainty regarding a battery's end-of-life environmental load (Hall & Lutsey 2018). The studies listed in *Table C2* reported battery production emissions from the combination of several different types of lithium-ion batteries. As the specific emissions values for a $LiFePO_4$ battery is needed, it is therefore decided to determine the battery production emissions from the GREET® (2019) model. The $LiFePO_4$ battery energy density is assumed to be 0.12 kWh/kg (Rydh & Sandén 2005a; Majeau-Bettez, Hawkins & Strømman 2011; Dunn et al. 2014; Dunn et al. 2015; Lu et al. 2016; Ambrose & Kendall 216; Hao et al. 2017a; Yu et al. 2018; Ioakimidis et al. 2019).

Further detailed information on the environmental impact of lithium-ion batteries will be discussed in the next chapter.

4.2.3.2.3 BILL OF MATERIALS

This section establishes the Bill of Materials (BoM), including any components and powertrains relating specifically to ICEVs, HEVs, or BEVs. An investigation is conducted on the additional electronics components (the $LiFePO_4$ battery, BMS, and controller) specific to the HEV and BEV bus. *Table 19* and *Table 20* provide estimated BoMs of vehicle components and LIB batteries, respectively.

Table 19 – ICEV, HEV, and BEV bus bill of materials.

Material	Material Weight (kg)		
	ICEV	HEV	BEV
Aluminium	635	565	650
Battery Management System	-	50	50
Cast Iron	1,540	1,050	125
Fiberglass Composites	965	965	965
Copper	65	565	975
Nylon 66	45	45	45
Fluids & Lubricants	385	420	415
Glass	475	475	475
Polyurethane Flexible Foam	75	75	75
Lead	25	-	-
Lithium Battery	-	115 (7.7 kWh)	2,700 (324 kWh)
Magnesium	75	75	75
Paint	45	45	45
Plastics	445	445	445
Rare Earth	15	35	90
Rubber	645	645	645
Stainless Steel	545	475	520
Steel	6,655	6,290	6,040
Zinc	65	65	65
Total	12,700	12,400	14,400

Source: $(S\&T)^2$ Consultants (2014); Lajunen & Lipman (2016); BYD (2017); Scania (2019); Volvo (2019).

Table 20 – Lithium iron phosphate battery bill of materials.

Components (%)	HEV	BEV
Active Material	15.5%	23.8%
Graphite/Carbon	9.1%	13.8%
Binder	1.3%	2.0%
Copper	24.3%	10.4%
Wrought Aluminium	20.1%	23.1%
Electrolyte ($LiPF_6$)	1.9%	2.5%
Electrolyte (Ethylene Carbonate)	5.4%	6.8%
Electrolyte (Dimethyl Carbonate)	5.4%	6.8%
Plastic (Polypropylene)	2.0%	1.0%
Plastic (Polyethylene)	0.5%	0.3%
Plastic (Polyethylene Terephthalate)	0.3%	0.2%
Steel	1.4%	0.7%
Thermal Insulation	0.7%	0.5%
Coolant (Glycol)	5.7%	5.1%
Electronic Parts	6.4%	3.0%

Source: Dai et al. (2018); GREET® model (2019).

Figure 16 – BYD K9 battery modules.

Source: Halvorson (2014).

4.2.3.3 ASSEMBLY PHASE

Accordingly, this section calculates the GHG emissions produced during the assembly of the buses. The environmental load from the assembly phase is designated as $E_{assembly}$ **(kgCO$_2$e)** hereinafter. There is very limited data available in Australia in the public domain relating to automotive assembly emissions. Therefore the GREET® (2019) model is utilised to estimate the GHG emissions rate per unit of material weight. The assembly emissions are subject to the assumptions incorporated by the GREET (2019) model. Here, an input-output approach is applied to simulate the vehicle assembly line. The per-ton vehicle output is used to connect the vehicle assembly processes together. *Table 21* provides the emissions intensity per ton of vehicle assembled. The overall emissions from assembly can be found in *Table C5*.

Table 21 – Emissions intensity per ton of vehicle assembled.

Process	Emissions (kg/kg)					
	CO_2	CO	NO_x	SO_x	CH_4	N_2O
Painting	211.5	0.2	0.3	0.1	0.6	5.94E-03
HVAC & Lighting	131.8	0.1	0.1	0.2	0.3	2.06E-03
Heating	195.1	0.2	0.2	0.04	0.7	6.46E-03
Material Handling	27.3	0.01	0.02	0.04	0.05	4.30E-04
Welding	36.3	0.02	0.02	0.07	0.07	5.70E-04
Compressed Air	54.4	0.02	0.04	0.09	0.1	8.50E-04

4.2.3.4 TRANSPORTATION PHASE

This section estimates the amount of emissions produced from transporting the buses from their respective manufacturing plants into Sydney, Australia. The ICEV and HEV bus (Volvo B8RLE and Volvo B5L Hybrid) are produced in Borås, Sweden, and the BEV bus (BYD K9) is produced in Shenzhen, China. To simplify the assessment, it is assumed that the buses are fully built before being loaded onto a cargo ship (Port of Gothenburg and Port of Shenzhen, respectively) bound for Australia. The environmental impact of transportation is estimated with an activity-based calculation method (McKinnon & Piecyk 2011):

Environmental Impact (kgCO_2e) = Transport Mass (kg) by Transport Mode x Transport Distance (km) x CO_2e Emissions Factor per kg/km.

The environmental load from the transportation phase is designated as $\boldsymbol{E_{transportation}}$ **(kgCO_2e)** hereinafter. The overall emissions from transportation can be found in *Table C6*.

4.2.3.5 MAINTENANCE PHASE

Moving on to the maintenance phase, this section accounts for the regular preventive maintenance for the studied buses. The GHG emissions produced during the maintenance phase buses is calculated by determining the emissions produced from manufacturing the replacement components. The environmental load from the maintenance phase is designated as $\boldsymbol{E_{maintenance}}$ **(kgCO_2e)** hereinafter. There is very limited data available in the Australian public domain relating to automotive maintenance emissions, therefore the power consumption emissions are excluded from this study. The overall emissions from maintenance can be found in *Table C7*.

Multiple studies have also set the service life expectancy of electric buses to 10 ~ 12 years and 500,000 ~ 800,000 km (Potkány et al. 2018; Franca A., 2018; Lajunen, A. 2018; Borén, S. 2019). To simplify the assessment, the service life expectancy of the studied buses is set to 12 years and 650,000 km. The lead-acid batteries in the ICEV bus and the HEV lithium-ion batteries are set to be replaced every 5 years (Brecher, A. 2012; Lajunen & Lipman 2016). According to literature, the lithium-ion batteries for heavy-duty vehicles such as the HEV and BEV bus are assumed to have an average life expectancy of approximately six to eight years (Zackrisson, Avellán & Orlenius 2010; Kushnir & Sandén 2011; Majeau-Bettez, Hawkins & Strømman 2011; Brecher, A. 2012; Amarakoon, Smith & Segal 2013; Hawkins et al. 2013; Dunn et al. 2014; Grütter, J. 2014; Dunn et al. 2015; Lu et al. 2016; Ambrose & Kendall 2016; Zackrisson, M 2017; Vandepaer, Cloutier & Amor 2017). Thus, with the lack of industrial data on the real-world battery performance of heavy-duty vehicles, this study assumes that the HEV and BEV bus requires approximately one battery replacement for the set service life expectancy.

Consumable components have been accounted for in this study. The periodic replacement of the components is based on the first-hand data available in the international public domain (Volvo 1999; Gillig 2007; Mitsubishi 2014). To simplify the assessment, the replacement components are set to be produced from virgin materials. The tyre service life is set to 50,000 km, thus a total of 13 full sets of new tyres are replaced per bus. Brake pads are set to be replaced every 50,000 km. Components specific to the ICEV, HEV, and BEV bus powertrains have also been accounted for. Notably, since damage caused by accidents often occur unexpectedly, replacement components caused by accidents are therefore excluded from this study.

4.2.3.6 RECYCLING AND DISPOSAL PHASE

Lastly, this section decommissions the buses at their end-of-life. A critical analysis shows that recycling lithium-ion battery materials, such as cobalt and nickel in the cathode, will result in a 51% reduction in natural resource consumption (Dewulf et. al 2010). A scenario analysis was performed, where the three buses are assumed to be recycled and disposed of within the Australian border. The scenario analysis also applies a high recycling approach, where the buses are reverted back into their original state of raw materials. The spent materials are separated into individual modules, to the point where they have their lowest value. The recyclable materials are set to include but are not limited to: electronics, glass, metals, plastics, and rubber. Recyclable materials are sorted, cleaned, and then reprocessed into fresh materials in their respective recycling plants. The implementation process of the fresh materials into new products is excluded. The remaining non-recyclable materials are then disposed of in landfills. In addition, the high recycling approach includes recycling waste oil from the BEV and HEV bus.

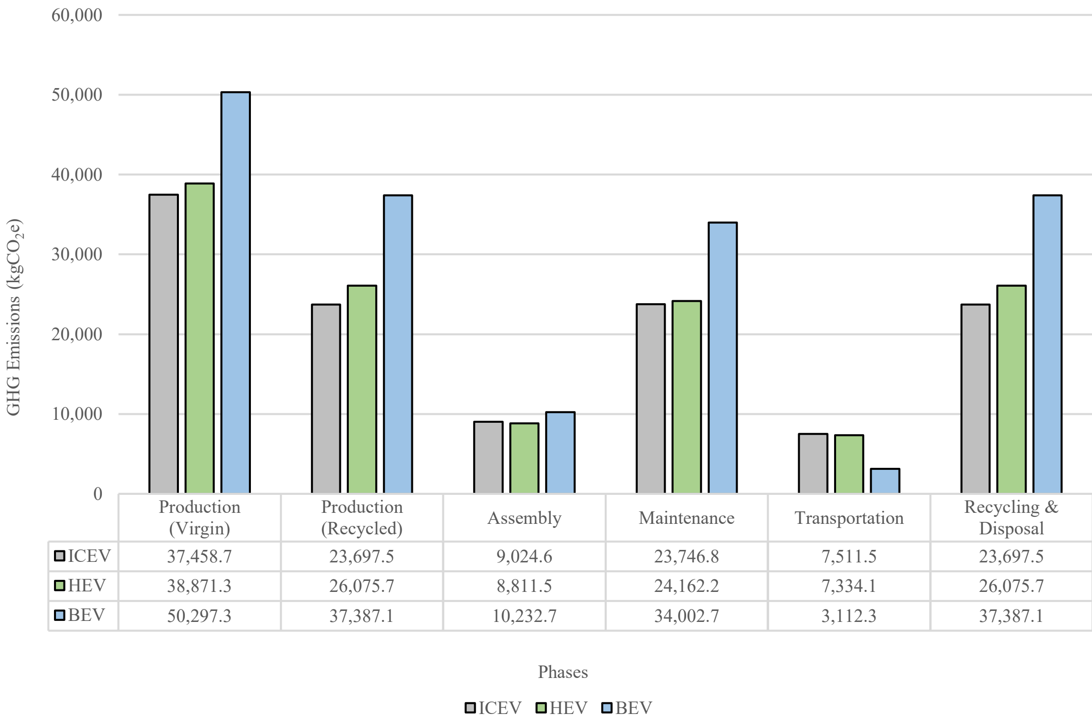
Most modern equipment is intricately integrated with plastics, electronics, metals, and other materials, making the recycling process challenging but not impossible. In the high recycling approach, most of the metals are stripped and recovered, the remaining waste materials will ultimately end up in landfills. The primary objective of the disposal phase is to maximise resource efficiency and reduce GHG emissions simultaneously (Turner et al. 2011). Numerous international studies have unanimously shown that waste material recycling can result in a reduction of GHG emissions (Manfredi, Tonini, & Christensen 2011; Franchetti & Kilaru 2012; Nordelöf et al. 2014; Turner et al. 2015). Similar to the virgin materials GHG emissions production, the calculation of waste material GHG emissions observe the same functional unit of a unit mass of GHG (or equivalent) per unit of energy/material production: **$kgCO_2e/kg$** (kg of CO_2 equivalent per kg of material reprocessed/recycled). The environmental load from the disposal phase is designated as $\boldsymbol{E_{disposal}}$ **($kgCO_2e$)** hereinafter. With the limited literature investigating the GHG emissions produced from recycling and reusing materials, the GHG emissions from disposal are assumed to be the same as virgin material production emissions, with the exception of aluminium and steel. According to the UE 2000/53/EC directive, requirements have been introduced to obtain the minimum recovery and recycling rates of EoL vehicles. Thus, in harmony with the directive, the reuse and recycling rate of a minimum of 85% by an average weight per vehicle was assumed.

4.2.3.6.1 BATTERY DISPOSAL OPTIONS

It is assumed that most components of the HEV and BEV bus are recycled similarly to the ICEV bus. The major difference then lies in the disposal of lithium-ion batteries. At present, there are three disposal options available: repurpose, recycle, or landfill. Repurposing is a relatively new concept, however, there are opportunities for reusing these batteries in stationary storage applications at the vehicle's end-of-life. This allows for a more thorough use of the batteries, as they are likely to retain approximately 75% to 80% of their original capacity at their vehicle end-of-life (Hall & Lutsey 2018). The repurposed batteries can then be applied in other applications, for example, stationary electricity storage from renewable energy sources (such as solar PV) used in households. The advantage of this application could potentially displace fossil-fuel electricity generation to some extent and offset the GHG emissions produced (Nealer, Reichmuth, & Anair 2015). This study has deemed repurposing BEV bus batteries as out of the study's scope and therefore excluded from this analysis, however, it is a rich area to be considered for future work. In terms of recycling, the majority of lithium-ion battery components can be reverted back into raw materials and then recycled for use in producing new batteries. The degree of how much of the battery can be recycled depends on battery design and a given region's economic and technical abilities. This study considers battery recycling, therefore the GHG emissions produced from the battery recycling process is supplemented into the disposal phase environmental load calculations. Further detailed information on battery disposal options will be discussed in the next chapter.

4.3 RESULTS AND DISCUSSION

Figure 17 shows the life cycle GHG emissions, with the total environmental impact separated into production, assembly, maintenance, transportation, and disposal phases. The results from the production, assembly, maintenance, and disposal phases are based on the functional unit reported in the GREET© (2019) model. Detailed emissions intensity per unit of material weight produced can be found in *Tables C1 – C7*. The total life cycle environmental loads are calculated from the sum of GHG emissions produced by production, assembly, maintenance, and recycling & disposal phases.



	Production (Virgin)	Production (Recycled)	Assembly	Maintenance	Transportation	Recycling & Disposal
ICEV	37,458.7	23,697.5	9,024.6	23,746.8	7,511.5	23,697.5
HEV	38,871.3	26,075.7	8,811.5	24,162.2	7,334.1	26,075.7
BEV	50,297.3	37,387.1	10,232.7	34,002.7	3,112.3	37,387.1

Figure 17 – Life cycle GHG emissions results.

Given the results obtained by this study, the main findings are as follows:

Producing a BEV bus has a higher environmental load than its ICEV and HEV counterparts, much of which is due to the manufacturing of lithium-ion batteries. Indeed, manufacturing the 324 kWh (weighing approximately 2,700 kg) $LiFePO_4$ battery contributes 11,038.8 $kgCO_2e$ of GHG emissions in its production phase. However, several studies have shown that the electricity used in the battery manufacturing process accounts for approximately 50% of GHG emissions (Ellingsen et al. 2013; Tagliaferri et al. 2016; Hall & Lutsey 2018). Therefore, an effort in grid decarbonisation, such as increasing the use of renewable energy and more efficient power plants will lead to reduced emissions in battery manufacturing. Furthermore, the results show that producing the buses from recycled materials yielded significant GHG emissions savings of 13.6, 12.2%, and 9.6%, respectively.

Assembling the three bus variants produces similar amounts of GHG emissions and contributed very little to the total life cycle environmental loads. It is still worth mentioning that the BEV bus has higher emissions from assembly due to its additional mass. The environmental impact of transportation is heavily dependent on the transport distance. For this case study where the buses are shipped to Sydney, Australia, the results show that both the ICEV and HEV buses reported higher GHG emissions, as both buses have to be transported for more than double the distance of the BEV bus. Here, the transportation phase contributed the least to the total life cycle environmental loads.

The emissions from maintenance are highest for the BEV bus, followed by the HEV, and then the ICEV bus. The maintenance needs for the ICEV and HEV buses are similar given that both buses have an ICEV powertrain. As per the assumption made previously, over their service lives the HEV and BEV bus requires one battery replacement every six to eight years. Thus, an additional 940.3 kgCO_2e and 11,038.8 kgCO_2e are included. The largest contribution to maintenance emissions comes from the replacement of tyres. A pessimistic approach is applied and has assumed that tyres are to be replaced every 50,000 km over the buses' 650,000 km service life. The amount of GHG emissions from producing tyres amount to 16,808.5 kgCO_2e. Replacement components caused by accidents have also been excluded, as such incidences often occur unexpectedly and are impractical to predict.

Very little literature investigated the GHG emissions produced from recycling and reusing materials, and there is limited data available in the public domain relating to materials processing emissions. The GHG emissions from virgin material productions for the materials processing in the disposal phase are therefore assumed, with the exception of some recycled materials data reported by GREET© (2019).

4.3.1 CONTRIBUTION ANALYSIS

The total life cycle environmental load is designated as $\boldsymbol{E_{total}}$ **(kgCO_2e)**. Here, E_{total} is obtained by the summation of environmental loads from all life cycle phases.

$$E_{total} = \sum(E_{production} + E_{assembly} + E_{maintenance} + E_{transportation} + E_{disposal}) \qquad (9)$$

Therefore, when using virgin materials in the production phase, the total life cycle environmental loads of an ICEV, HEV, and BEV bus are $\boldsymbol{E_{total}}$ **= 101,439.1 kgCO_2e**, **105,254.8 kgCO_2e**, and **135,032.1 kgCO_2e**, respectively. Alternatively, when accounting for the scenario of using recycling materials in the production phase, the total life cycle environmental loads are $\boldsymbol{E_{total}}$ = **87,677.9 kgCO_2e**, **92,459.1 kgCO_2e**, and **122,122.0 kgCO_2e**, respectively.

4.3.2 SENSITIVITY ANALYSIS AND UNCERTAINTY

The ICEV, HEV and BEV bus life cycle emissions calculated in this study are influenced by several factors, assumptions, and uncertainties. This is especially the case for the BEV bus, as the chosen BYD K9 BEV bus only entered mass production in 2010. It is rather difficult fixing specific values to some parameters that influence the life cycle environmental loading of bus production. The production phase emissions vary with the degree of virgin and recycled materials used. In addition, factors such as energy consumption in all phases are heavily influenced by the carbon intensity of a region's grid-mix and induce uncertainty in the raw material manufacturing emissions. If this study was to set the bus production to a certain region, it is still necessary to assume that numerous sub-components and raw materials may originate from various parts of the world. The manufacturing and assembly processes may vary in their degrees of carbon intensity, making it difficult to determine an accurate environmental load for imported sub-components and raw materials.

Figure C1 demonstrates that variations in key parameters regarding materials production methods, maintenance frequency, service lifetime, and transportation distances can all influence the life cycle phases and consequently the total life cycle GHG emissions. A series of 19 scenarios (A-R) for each bus variant is illustrated through the vertical bars, which in turn demonstrate the potential range of life cycle GHG emissions under the influence of parameter changes. First, the variation of emissions in the production phase is due to the mixed manufacturing with virgin and recycled materials. Next, transportation emissions vary by the travel distance of transporting the buses from their respective manufacturing plants into Sydney, Australia. Then, GHG emissions from the maintenance phase are heavily influenced by unpredictable factors, such as traffic conditions, drive patterns, drive style, weather conditions, and road conditions. For instance, this study had applied a pessimistic approach and have set the replacement of brake pads to every 50,000 km. However, the HEV and BEV buses have regenerative braking abilities that will greatly extend the intervals of replacing brake pads. Last, at the discretion of bus operators, the predetermined service life (ranging from 500,000 ~ 800,000 km) contributes to additional maintenance requirements, which then exacerbates the GHG emissions from the maintenance phase. For example, the assumed battery service life of this study is six to eight years. The real-world driving conditions may impact battery performance for the HEV and BEV buses, thus increasing the service life will require an additional battery replacement.

From the analysis of the grid-mix carbon intensities of 49 countries from the previous chapters, Australia is among the rest of the countries with significant carbon-intense grid-mixes. This signifies that any raw material and sub-component manufacturing and production processes in these countries that consume high electric power (for example, electric arc furnaces and machining processes) will yield high environmental loads. This corresponds with the study of Cooney, Hawkins & Marriott (2013), where the authors stated that there will be strong preferences for BEVs over ICEVs in regions where the grid is powered predominately by renewable energy or nuclear.

Regarding vehicle emissions, Hall & Lutsey (2018) reported that incorporating BEV life cycle manufacturing emissions into vehicle regulations would be misguided. Many governments pioneering the decarbonisation of the transport sector have been investigating the environmental impact of BEVs, especially the manufacturing emissions of BEV batteries. However, the regulations on vehicle emissions and energy efficiencies should also incorporate manufacturing emissions for all conventional vehicle components, in addition to vehicle batteries, so that BEVs would not be unfairly penalised.

The GHG emissions from the maintenance phase are heavily influenced by unpredictable factors, such as traffic conditions, drive patterns, drive style, weather conditions, and road conditions. Furthermore, the maintenance phase is greatly influenced by bus service life and scheduled maintenance frequency, both at the discretion of bus operators. If the service life was set to 800,000 km for all buses, the HEV and BEV bus would both require a battery replacement.

The slow deterioration of the $LiFePO_4$ battery may influence the final environmental load of the BEV bus. Currently, there are opportunities to repurpose the batteries after the BEV bus's end-of-life, thus allowing a more thorough and efficient operations phase. This study finds significant variety in environmental load reported across the literature studied based on life cycle methodologies and battery chemistries. Earlier literature reported higher production emissions whereas the emissions gradually reduce in more recent literature. The improvement of battery technology allows for longer vehicle service life, which offers fewer replacements in the vehicles and an increase in secondary use for stationary storage applications. With the increase in BEV bus implementation into the transport sector, battery manufacturers will also scale their production to suit the demand. The energy intensity of manufacturing batteries relies heavily on the composition of battery chemistries. Eventually, batteries may be manufactured with less carbon-intense materials. As promising as the proposed technological improvements may be, these technologies are still undergoing development and the time to commercialisation is still unknown. Consequently, this study does not attempt to quantify the GHG savings from future battery technology improvements and breakthroughs.

At the buses' end-of-life, there will be opportunities for reusing the BEV batteries, such as repurposing them for stationary storage applications which allows for a more thorough use of the batteries. This study has assumed the recycling of the batteries, and with the lack of available data, it is assumed that the recycling process will have the same emissions as virgin material production. However, in the scenario where the batteries are repurposed, there will be considerable disposal emissions savings, consequently reducing the total environmental impact.

4.4 CONCLUSION

The innovation and contribution of this study have been presented through an appropriate comparison of diesel, hybrid-diesel, and electric buses by applying a case study approach for Australia and addressing the research gap on the environmental impact of transitioning the transport bus fleet to electrified powertrains. This study has targeted a Volvo B8R Low Entry diesel bus, a Volvo B5L Hybrid bus, and a BYD K9 electric bus as baseline models for comparison. The buses were chosen as they are currently in service in the Australian bus fleet and manufacturers have readily provided the necessary data with the authors needed to conduct an LCA. Then, an in-depth and comprehensive LCA of the three bus variants was conducted which included the environmental impact resulting from the production, assembly, maintenance, and disposal phases. The detailed estimation of GHG emissions produced throughout the life cycle of transit buses assists in the accurate evaluation of the environmental sustainability between ICEV, HEV, and BEV buses. The uncertainty and assumptions made from the technological developments were addressed with a sensitivity analysis with respect to the parameters presented.

Based on the results obtained by this study, the following remarks concludes this chapter:

On average, in Australia, the BEV bus has a higher environmental impact in its product life cycle than the ICEV and HEV bus, much of which is due to the manufacturing of lithium-ion batteries. The manufacturing of the 324 kWh (weighing approximately 2,700 kg) $LiFePO_4$ battery contributes 11,038.8 $kgCO_2e$ of GHG emissions. Furthermore, the results show that producing the buses with recycled steel and aluminium instead yielded significant GHG emissions savings of 13.6%, 12.2%, and 9.6%, respectively. Additionally, these values will vary depending on the buses' country of origin. This study's sensitivity analysis shows that countries with high carbon-intense grid-mixes will yield higher environmental loads from raw material and sub-component manufacturing and production processes.

Lithium-ion battery, copper, and rare earth metals production accounts for the most significant difference between the buses but represents only a small percentage of the BEV bus's total equipment life cycle GHG emissions. Although the BEV bus's 2,700 kg lithium-ion battery is significant in weight, it represents only 21.9% of the production phase emissions and 8.2% of the total GHG emissions. Similarly, copper and rare earth metals account for 5.3% and 2.8% of the production phase emissions and only 2.1% and 1.2% of the total GHG emissions. In contrast, the sheer size of the buses dictate the large amounts of steel are used (6.1 ~ 6.7 tons), such as the chassis frame and suspension system, contributes the highest to the GHG emissions and represents 12.3% ~ 18.1% (16.6 ~ 18.3 $tonCO_2e$) of the total GHG emissions. In comparison, an average passenger vehicle only has approximately 900 ~ 1,000 kg worth of steel. Thus, it would not be accurate to simply extrapolate the known emissions data from BEV passenger vehicles and extend the application to BEV buses by assuming that the existing results will continue to be applicable.

Recycling BEV batteries will reduce product life cycle GHG emissions. Currently, there are opportunities to recycle the batteries after the BEV bus's end-of-life, thus allowing a more thorough and efficient operations phase. Additionally, as the energy intensity of manufacturing batteries rely heavily on the composition of battery chemistries, the improvement of battery technology may eventually lead to manufacturing batteries with less carbon-intense materials. This study's sensitivity analysis shows that the slow deterioration of the $LiFePO_4$ battery may influence the final environmental load of the BEV bus. There is significant variation in the environmental load reported across the literature studied based on life cycle methodologies and battery chemistries. The authors found that earlier literature reported higher production emissions and gradually reduces with recent literature.

Thus, the environmental burden from the life cycle of ICEV, HEV, and BEV buses is non-negligible and complex to analyse. Some studies have performed LCA evaluations on the environmental impact of the electrified powertrain technology at varying levels of detail, accuracy, and transparency. There are many opportunities to reduce product life cycle emissions, such as improvement in manufacturing efficiency, developing new battery technology, and production in regions with fewer carbon-intense grid-mixes. It is a rich area to be considered for future work.

This study has clearly compared the life cycle emissions of the three bus variants, and the results show that the BEV bus has a higher environmental impact than the ICEV and HEV bus. Yet it is strongly recommended for life cycle studies to be conducted and re-conducted in correspondence with the ever-innovating and developing BEV technologies. Although the data assumptions of materials and manufacturing emissions specific to the electric powertrain and BEV battery are current at the time this study was authored and may be superseded at the time it is being read, savings from these emissions are likely to increase in the future.

5 CHAPTER 5: LCA OF LITHIUM-ION BATTERIES[9]

5.1 INTRODUCTION

As Lithium-Ion Batteries (LIBs) have emerged as strong candidates among the battery of choice for EVs, multitudes of studies have conducted LCAs to assess their production environmental impact. The sustainability of mass global EV deployment is subject to the global regionalism of LIB supply chains and its effect on battery life cycle emissions (Kelly, Dai & Wang 2020).

This chapter evaluates the LCAs of LIBs from various literature sources, including the LCAs of BEVs, HEVs, and Plug-in Hybrid Vehicles (PHEVs) that provide detailed contribution analysis and transparent inventory of LIBs. Thus, the last item of the equipment life cycle segment is addressed here. The data and inconsistent results on automotive LIBs in terms of their life cycle environmental and energy impacts were examined. The usefulness of LCA studies available to date was scrutinised to facilitate the discussion among industry, governments, and policy makers seeking advice and guidance from LCA studies on the environmental impact of LIBs. Consequently, the study in this chapter recognises that each LCA study has its own goals and scope, and complements the work of Peters et al. (2016), Ellingsen, Hung & Strømman (2017), and Aichberger & Jungmeier (2020). Evidence was provided that contradicts the popular marketing propaganda and reveals that such "zero-emission" vehicles do indeed produce GHG emissions, and also mitigation opportunities were recommended to reduce their impact on the environment. This chapter has categorised the LIB product life cycle into four notable phases: 1) materials and parts production, 2) cell manufacturing, 3) battery pack assembly, and 4) End-of-Life (EoL) decommissioning. The study acknowledges that the operations phase is important and a rich area for research, however, it is deemed out of scope and therefore is not included.

To carry out the assessment, the evaluation process is as follows. First, this chapter starts by introducing readers to the present state of research on LIBs. Next, it explains the research methods utilised to filtrate studies that were deemed the most complete and relevant assessments. Then, the study critically evaluates the current literature and industrial data regarding the life cycle GHG emissions produced and the Cumulative Energy Demand (CED) from LIB technologies. After that, the average life cycle emissions, the disparity in Global Warming Potential (GWP) and CED values in terms of life cycle phases, and the factors that causes the disparities are determined. Last, the significance of battery recycling and repurposing is established, thus offering noteworthy insights and implications that conclude this chapter.

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[9] The contents of this chapter have been adapted from the publication: **Zhao, E.**, Walker, P.D., Surawski, N.C. & Bennett, N.S. 2021, 'Assessing the Life Cycle Cumulative Energy Demand and Greenhouse Gas Emissions of Lithium-Ion Batteries', *Journal of Energy Storage*, vol. 43, pp. 1-19.

5.1.1 PRESENT STATE OF RESEARCH

This section summarises the literature reviews conducted by the following three studies. Thus by summarising the literature reviews, this study has also taken into account all the LIB LCA studies that have been referenced. These studies were then meticulously examined and passed through the selection criteria described in the next section to be assessed based on relevance.

In their study, Peters et al. (2016) reviewed an overall of 79 studies that assessed the environmental impact of LIB production. They found that only 36 studies provided sufficient information to extract the environmental impact of LIB production. They also found that the majority of the reviewed studies relied on secondary data for their Life Cycle Inventory (LCI) and only a few studies published results with original data. Nevertheless, the results were found to vary significantly, and the authors attributed this outcome to the different assumptions studies have made regarding LIB key parameters. Their reported average CED and GHG emissions across all chemistries are 328 kWh and 110 kgCO_2e/kWh of storage capacity. The recycling of batteries was not considered as their review focuses primarily on battery production impact.

Ellingsen, Hung & Strømman (2017) examined the inventory data, key assumptions, differences, and results from nine LCA studies assessing the life cycle GHG emissions of LIBs based mainly on primary data and estimates. They reported that the main contributor to life cycle GHG emissions is the production phase (ranging from 38 ~ 356 kgCO_2e/kWh of storage capacity), and the use phase and EoL treatment phase held much smaller contributions. They also found that there was some disagreement concerning the quantity and sources of production-related emissions due to the widely different results for LIB production. Consequently, it is not possible to provide a unified answer to production-related emissions, especially since few studies considered the EoL phase in their LCA. The authors concluded that LIB technology is still under progression, thus continuous LCA studies in this area will be fundamental in updating information concerning prospects of improvement and environmental sustainability.

Aichberger & Jungmeier (2020) compiled 50 LCA publications between the years 2005 - 2020 and assessed the reported environmental impacts from production, use, and EoL for LIBs in automotive applications. Their investigation showed that the CED and GWP was 280 kWh/kWh and 120 kgCO_2e/kWh, respectively. Furthermore, when considering the recycling processes the GWP can be reduced by 20 kgCO_2e/kWh. Lastly, the authors conclude that many LCA results overestimated the cell manufacturing environmental impact due to disparity in assumptions and primary data sources.

5.2 RESEARCH METHODS AND SELECTION CRITERIA

In this literature review, the focus is on the two major indicators of GWP and CED, because of their great relevance and that these two indicators are the most represented in the reviewed literature, and thus allows for accurate comparison between results in the literature. Key assumptions and obtained results were extracted from all studies and recalculated into functional units set by this study. This allows for uniform comparison across studies with different functional units and for extracting the average value as the corresponding result, therefore the mean value is used for any study that provided a value range. The GWP is measured in kilograms of CO_2 equivalent (kgCO_2e) with a time horizon of 100 years relative to the emissions of a unit of the reference gas CO_2 (IPCC 2014, AGCER 2016, EPA 2017). The CED is the energy consumption from the entire manufacturing process of the LIB battery pack.

Therefore, the functional units by which the studies are evaluated are: 1) a unit mass of carbon dioxide equivalent per unit of battery energy capacity (**kgCO_2e/kWh)**, or per unit of battery weight (**kgCO_2e/kg)**, and 2) a unit of power consumption per unit of battery mass (**kWh/kg)**. Less emphasis was placed on studies that employed other functional units, as this implies that the methodology would be different and thus be incomparable. In addition, the LCI data reported by each study was traced to identify interdependencies and sourced data, analogous to the work done by Peters et al. (2016), Ellingsen, Hung & Strømman (2017), and Aichberger & Jungmeier (2020). A spreadsheet was utilised to quantify the collection of data from the published articles.

To ensure the selection of the "best" scientific literature, LCA studies were filtered through with the following terms within the titles, abstracts, and keywords: "life cycle assessment (LCA)", "greenhouse gas (GHG)", "carbon dioxide equivalent", "cumulative energy demand (CED)", "environment", "impact", "lithium-ion batteries (LIB)", "battery electric vehicles (BEV)", "hybrid electric vehicles (HEV)", and "plug-in hybrid vehicles (PHEV)". Variations of these terms were also accepted to cover a broader network of searches. Only studies that have been formally peer-reviewed and published in reputable academic journals, conference proceedings, and official government publications were assessed.

Using these search criteria, a total of 76 publications were identified. A total of 21 life cycle studies were excluded based upon relevance, as listed in *Table D3,* leaving a total of 55 studies suitable for this research. The excluded studies did not provide the necessary information regarding total energy consumed or total GHG emissions. Some studies have functional units other than those set above, many of which were normalised in their final results via mathematical unit conversions to suit the functional units described above. Within these studies, several were difficult if not impossible to have their results converted, therefore they were excluded from this assessment. Among those excluded, there were many complete, comprehensive, and detailed studies, however, their focus was primarily on other impact categories: life cycle cost, human toxicity, resource depletion, acidification, ecotoxicity, eutrophication, ozone depletion, air pollutants, and photochemical ozone. Some studies have included GWP and CED along with other impact categories, in which case these were included in this assessment. Continuing with this search, nine studies that have investigated the environmental impact of LIB decommissioning and provided invaluable insight were identified and included in this study's discussion.

5.3 LITERATURE ANALYSIS RESULTS AND DISCUSSION

From the 55 included studies, the most relevant information reported is compiled and presented in *Table D1*. In accordance with ISO 14040:2006, particular attention was given to the review of the 55 studies to four reporting criteria stated by the individual studies: the purpose and goal statements, the intended application, the intended audience, and the time frame. Eight studies focused on CED only, whereas the remaining studies focused on GWP only, or GWP and CED concurrently. A total of 32 studies were identified that relied exclusively or partially on their primary LCI data, thus the remaining LCA studies did not provide their own LCI and have based their assessments completely on the LCI of previous studies. An interesting discovery was that most of the LCA studies that were reviewed based their LCI data from eight comprehensive and transparent studies: Gaines & Cuenca (2000), Rydh & Sandén (2005), Hischier et al. (2007), Notter et al. (2010), Zackrisson, Avellán & Orlenius (2010), Majeau-Bettez et al. (2011), Dunn et al. (2012), and Ellingsen et al. (2014). The data provided should be considered accurate at the time when the studies were authored, but with the rapid development of LIBs, the LCI data may not accurately represent the current practices. Additionally, industry data was difficult to obtain and included large uncertainties for studies that were carried out when the LIBs were at their early commercialisation stage (Dai et al. 2019).

The majority of studies presented in *Table D1* investigated more than one battery chemistry, hence the total investigation amounted to 142 case studies. The results range (inclusive of emissions of materials/parts manufacturing, cell manufacturing, battery pack assembly, and decommissioning) reported by all LCA studies assessed by this study is presented in *Table D2*. *Figures 16 – 18* graphically shows the GWP impacts and CED reported from the reviewed studies. The battery chemistries with only one case study conducted are located to the right of the black bar in *Figures 16 – 18*. *Figure D1* illustrates the number of case studies that have been conducted with respect to battery chemistry reflected in the reviewed LCA studies. Not surprisingly, much more LCA case studies have investigated mature technologies, such as NMC-C, LFP-C, LMO-C, NCA-C, and LCO-C (27, 25, 15, 12, and 10, respectively; see *Table 22*). At initial glace it appears that LCO-Li, LFP-LTO, and LMO-NMC produced significant GHG emissions, however, it should be clarified that these battery chemistries exist in an early state of Research and Development (R&D),

thus the resulting GHG emissions and energy consumption are comparatively higher than those of mature technologies, as the production of emerging LIB technologies is of small laboratory-scales only. Consequently, it is to be expected that LCAs on LIBs in an early R&D stage can portray a new technology as less environmental sustaining (Troy et al. 2016). Additionally, fewer than two LCA case studies have been done for each of these three battery chemistries (Troy et al. 2016; Ioakimidis et al. 2019; Cusenza et al. 2019). Ioakimidis et al. (2019) and Kushnir & Sandén (2011) investigated the LFP-LTO battery, with the former evaluating solely on GWP, and the latter evaluating exclusively on CED.

When considering all LIB chemistries, the following results were observed: 1) the average reported GHG emissions are **187.26 kgCO_2e/kWh** (5.40 ~ 1,730.77 kgCO_2e/kWh) and **19.78 kgCO_2e/kg** (0.21 ~ 96.96 kgCO_2e/kg), and 2) the average reported CED is **42.49 kWh/kg** (5.44 ~ 393.33 kWh/kg). This study's results are comparable with the findings of Ellingsen et al. (2017): 38 ~ 356 kgCO_2e/kWh, Peters et al. (2017): average 110 kgCO_2e/kWh and 328 kWh/kg, and Aichberger & Jungmeier (2020): 70 ~ 175 kgCO_2e/kWh and 200 ~ 500 kWh/kg. The average values were calculated from the sum of all values (GHG and CED) divided by the total number of values reported by the assessed studies.

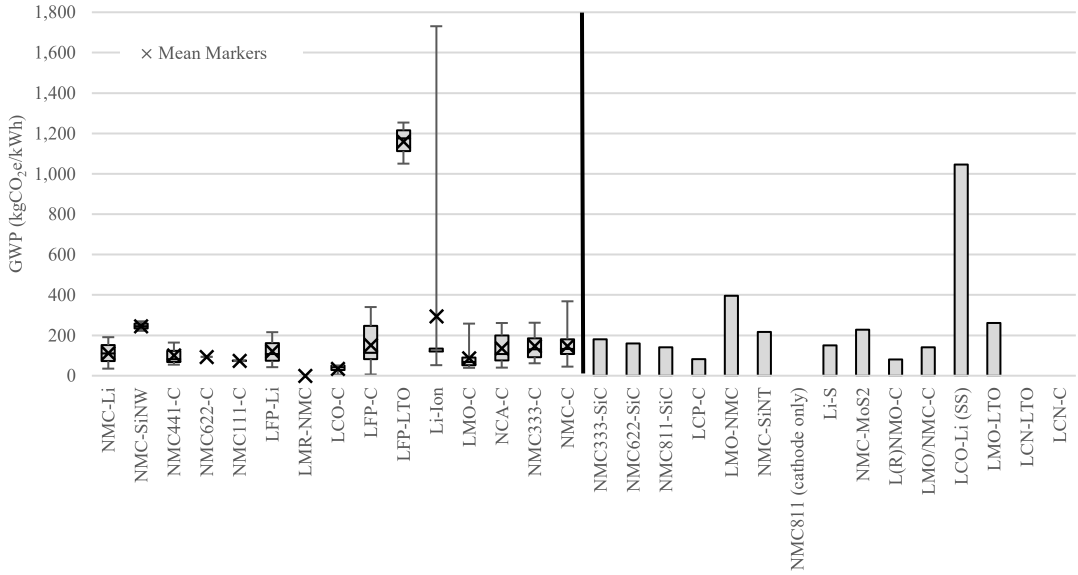


Figure 18 – GWP impacts in kgCO_2e/kWh, by battery chemistry.

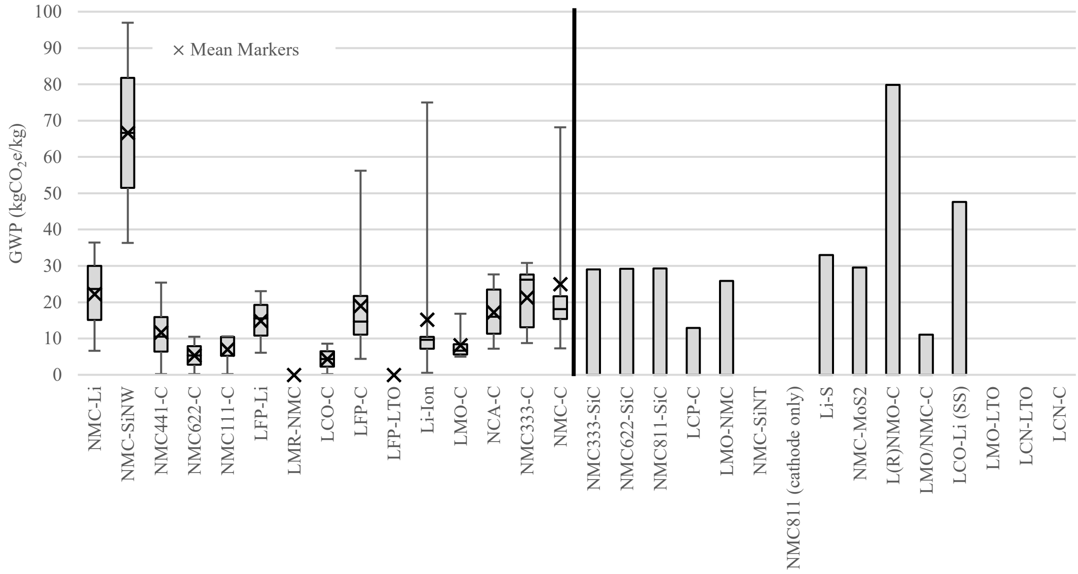


Figure 19 – GWP impacts in kgCO₂e/kg, by battery chemistry.

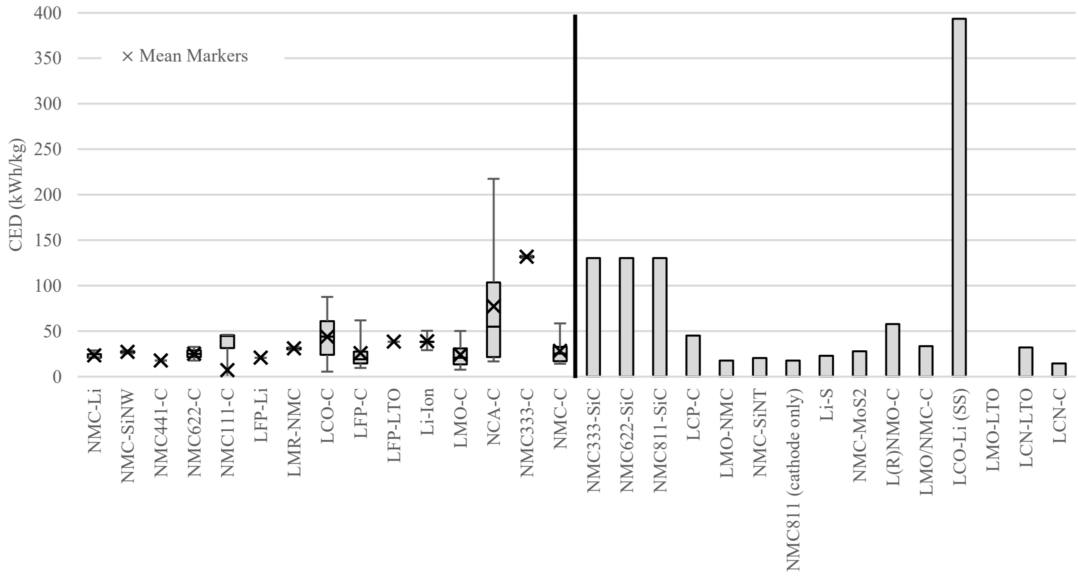


Figure 20 – CED results, by battery chemistry.

The performance and specifications of LIBs outlined by battery manufacturers typically include statements that define factors within which the claimed LIB performance can be delivered. Many such factors indicate LIB performance, such as energy efficiency, charge cycle, Depth of Discharge (DoD), temperature characteristics and effects, battery capacity, self-discharge characteristics, recharge voltage and rate, etc. Currently, the standards for LIB life cycle environmental impacts and performance are undergoing major investigation by government legislations and researchers. Domestically, a case study was conducted by the Australian Renewable Energy Agency (ARENA, 2020) that aimed to develop and propose an Australian Battery Performance Standard (ABPS) by reviewing existing local and international Battery

Storage Equipment (BSE) standards, guidelines, codes, and best practice documents. The project concluded that it is rather difficult to fairly compare the performance of battery storage equipment in the commercial market. This is due to the absence of a common standard on the measurement and reporting of battery performance characteristics. In an international example, a proposal concerning batteries and waste batteries for a regulation of the European Parliament and the Council that repeals Directive 2006/66/EC and amends Regulation (EU) No 2019/1020 was presented by the European Commission to modernise the EU's regulatory framework for batteries [European Parliamentary Research Service (EPRS, 2021)]. This proposal secures the sustainability and competitiveness of battery value chains and also introduces mandatory requirements for battery EoL management.

Table 22 – Results range for mature LIB technologies.

BattChem: Battery Chemistry; M/P: Materials/Parts Manufacturing ($kgCO_2e/kWh$); CM: Cell Manufacturing ($kgCO_2e/kWh$); PA: Battery Pack Assembly ($kgCO_2e/kWh$); DE: Decommissioning ($kgCO_2e/kWh$); GWP_1: $kgCO_2e/kWh$; GWP_2: $kgCO_2e/kg$; CED: kWh/kg

BattChem	M/P	CM	PA	DE	Other	GWP_1	GWP_2	CED
NMC-C	43.27	4.10	0.87	2.38	-9.08	45.00	7.29	14.15
	263.11	203.20	53.98	11.65	20.00	368.56	68.17	58.52
NCA-C	50.50	45.00	1.00	-	1.70	40.00	7.20	16.70
	81.00	208.80	-	-	-	261.00	27.65	217.50
LCO-C	0.20	-	2.17	-	2.80	5.40	0.20	5.44
	48.51	-	-	-	-	50.68	8.59	87.64
LFP-C	40.17	5.50	2.18	18.64	-7.67	6.16	4.40	9.52
	319.65	201.60	49.78	-	25.00	340.00	56.22	61.85
LMO-C	37.00	0.90	0.06	16.21	4.13	39.00	5.00	7.62
	91.50	206.40	19.60	-	5.69	258.00	16.83	50.17

5.3.1 DISPARITY IN GWP AND CED ESTIMATES

The inconsistencies of results found across the LCA studies were influenced by a variety of factors. In the absence of available primary data, studies relied on using secondary data, such as values from the literature, online databases, engineering modelling, or experimental data based on processes similar to the study's objectives, and extrapolate or modify them to estimate the actual operational data (Kim et al. 2016). It is rather difficult to obtain real battery inventory and production data from the industry sector, therefore LCA studies often labour with hypothetical designs acquired from combining available industry and literature data, or from modelling tools tailored specifically for the study. Thus, the diversity of battery specifications from literature, reports, and industrial data often induce errors in GWP and CED calculations.

The functional unit of most studies is $kgCO_2e/kWh$ or $kgCO_2e/kg$ – per storage capacity or mass, respectively – of the corresponding battery. Each study assumes different mass shares for each component, therefore this becomes a source of discrepancy (Peters & Weil 2018). Considering studies with vehicle lifetime as the functional unit, in either mileage or years, the resulting emissions per distance travelled (kgCO_2e/100km) are influenced by the assumptions made (Nealer & Hendrickson 2015). For example, a common vehicle lifetime the authors have found in literature is 150,000 ~ 200,000 km. A study assuming a lower-bound lifetime of 150,000 km will result in higher emissions per distance travelled, even if the emissions themselves remain the same. Thus, the emissions results can be skewed to construct either a positive or negative portrayal of BEVs compared to conventional ICEVs (Nealer & Hendrickson 2015).

An LCA's modelling approach – top-down or process-level – for the energy demand and GHG emissions of LIB production is the first aspect to influence the study's results. A top-down modelling approach starts with a given set of data of the energy consumption of an entire manufacturing facility, and then various processes and products are allocated with a percentage of the data value. Process-level modelling approaches on the other hand allocate values to specific individual processes along the production line, and the total value is calculated from the sum of all processes (Peters et al. 2017). In this research, three studies were found to have not specifically iterated any LCA approaches. From the remaining studies, it was shown that LCAs with top-down modelling approaches reported significantly higher values for energy demands due to including demands not linked to the examined LIB. LCAs with process-level modelling approaches may overlook certain aspects of the total energy demand. Thus, this research finds that top-down LCAs highlight cell manufacturing as the main contributor to environmental impacts, whereas process-level LCAs highlight materials production as the main contributor (see *Table D1*).

5.3.1.1 INFLUENCE OF LIB CHEMISTRY AND MATERIALS

This investigation found that large differences in GWP and CED are due to the variance in reported energy demands for cell manufacturing and battery pack assembly, as supported by Ellingsen et al. (2013). The lack of primary data makes it the most difficult aspect of battery production to analyse the GHG emissions (Kim et al. 2016). Considering the 55 studies, this study finds that while 47 studies reported GWP results, within these studies only six segregated the GHG emissions into the four respective life cycle phases. The remaining studies either omitted reporting values for some life cycle phases or only reported final emissions values.

Several LCA studies have stated that the processing of active materials for the LIB cells is responsible for the majority of environmental impacts in battery production (Notter et al. 2010; Majeau-Bettez et al. 2011; Dunn et al. 2012a; Ellingsen et al. 2014; Nordelöf et al. 2014; Kim et al. 2016; Deng et al. 2017a; Cusenza et al. 2019; Raugei & Winfield 2019; Philippot et al. 2019). When considering the five mature technologies, the following results were identified: whilst 10 case studies investigated the LCO-C battery, only five case studies evaluated the CED of LCO-C production. The remaining case studies reported only partial GWP impact values for the life cycle phases or reported only the total GWP impact value for the production process. The NMC-C, NCA-C, and LFP-C batteries have similar average total GHG emissions and are much higher than LCO-C and LMO-C batteries. Breaking down the production GHG emissions into their respective phases, it was found that the materials/parts manufacturing to be the most significant contributor to GHG emissions, with an average contribution of 45.35% of total GWP impact. For this phase, the GWP impact is very similar for the NMC-C and LFP-C batteries, and the NCA-C and LMO-C batteries. The next major contributor is cell manufacturing, with an average contribution of 20.16% of the total GWP impact. Comparatively, pack assembly (1.09%) and decommissioning (2.07%) contributed very little to the total GWP impact value. In terms of CED, it was found the NCA-C and LCO-C batteries have the highest energy consumption. When considering all LIB chemistries, it was found that the contributions of the four life cycle phases to the total GHG emissions are 29.72%, 12.76%, 1.99%, and 0.61%, respectively.

The lithium content ($LiPF_6$) in the electrolyte accounts for a very small percentage of the total battery mass, ranging from 5.4% ~ 6.8% per kg of conventional LIB (Dai et al. 2018). Additionally, Dai et al. (2018) reported that the extraction process of lithium from brines are simple and have low energy demands. There is a variation in the composition of LIBs and is primarily dependent on the cathode composition, as stated by Sullivan & Gaines (2012). Studies have unanimously identified that the production and processing of active cathode materials, together with high energy consumption from metals production are the key contributors to life cycle energy and environmental impacts (Hao et al. 2017a; Deng et al. 2017b; Kelly, Dai & Wang 2020). From these investigations, it was found that most studies used graphite as the target anode. Graphite only has low levels of CO_2 emissions as the carbon remains in the product in the processes.

However, analysis of different studies on potential next-generation LIBs has revealed varying results. The following text summarises the most comprehensive and transparent LCA studies on these emerging technologies. Deng et al. (2017a) investigated a Li-S battery and reported that the cell manufacturing process on a laboratory scale demands intensive electricity consumption. The authors also stated that cell manufacturing energy intensity is expected to reduce significantly when the Li-S battery is manufactured on an industrial scale. When compared to the conventional NMC-C battery, however, the Li-S battery was seemingly more environmentally friendly, where the life cycle environmental impacts were reported to be 9% ~ 90% lower (the Li-S battery was assessed on a mixed laboratory-scale and pilot production scale, whereas the NMC-C battery was evaluated on an industrial-scale production capacity). Deng et al. (2017b) examined a LIB with an NMC cathode and a MoS_2 anode. They found that the material synthesis of the MoS_2 anode was found to be extremely energy-intensive. Using the environmental impact of the conventional

NMC-C battery as the benchmark, the NMC-MoS_2 battery presented approximately 6% ~ 7% higher GWP impacts. Deng et al. (2019) conducted an LCA on the environmental impact of the NMC-SiNT battery. Similarly, the authors compared the LCA results with the conventional NMC-C battery. The results show that the NMC-SiNT battery produced 10% ~ 17% higher impacts in GWP, once again exasperated by the high energy intensity of cell manufacturing. Wu & Kong (2018) presented a prospective LCA of two new LIBs with a lithium metal anode (NMC-Li and NMC-SiNW), and compared the life cycle environmental impact with the traditional NMC-C battery. The authors found that within the same battery, the life cycle environmental impact of the NMC-SiNW battery production was higher than the other two batteries. During production, the NMC-Li battery was the most environmentally friendly, however, at present this particular battery is still under development and has not yet been practically used as a preferred energy carrier in BEVs. The authors hypothesize that the higher theoretical cycle lives of the NMC-SiNW and NMC-Li batteries would outperform the traditional NMC-C battery. Similarly, Li et al. (2014) performed an LCA on an NMC-SiNW battery. Their results show that the overall life cycle impacts of this new battery pack are moderately higher than conventional LIBs. The LCA was for pilot-scale laboratory designs only, and thus under the consideration of uncertainties and the eventual scaling up to industrial production of the technology, the increase in materials production efficiencies and decrease in energy consumption may lead to comparable life cycle impacts with conventional LIBs. Kim et al. (2016) reported the first LCA emissions assessment of an LMO/NMC-C battery pack for a Ford Focus BEV. The authors stated that cell manufacturing contributed the most to GWP and CED impact and was closely followed by materials production. In comparison with the GWP impact of other LCA studies, this new battery is positioned between the LMO and NMC batteries (NMC being the highest of all three). Raugei & Winfield (2019) presented a prospective LCA of the production and EoL of a new LCP-C battery. Not surprisingly, the cathode was the largest contributor to both GWP and CED impacts, which was stated as the result of comparatively high GHG emissions and energy consumption from the cathode input materials production process. The authors then performed a comparison of their results to the published literature that have also conducted LCAs on LMO, NMC, and LFP batteries. It was evident that the new LCP-C battery has lower GWP impacts than the NMC (Majeau-Bettez et al. 2011; Ellingsen et al. 2013; EPA 2013; Hao et al. 2017a) and LFP (Majeau-Bettez et al. 2011; EPA 2013; Hao et al. 2017a) batteries, and only slightly higher than the LMO (Notter et al. 2010; Dunn et al. 2012a; EPA 2013; Hao et al. 2017b) batteries.

Dunn et al. (2012a), supported by Notter et al. (2010), reported that certain battery components such as the binder, plastics, and graphite anode do not contribute much to battery life cycle environmental impacts. Majeau-Bettez et al. (2011) reported that BMS attributed to 15% of battery life cycle fossil fuel consumption, whereas Dunn et al. (2012b) calculated the maximised

BMS mass environmental impact to be less than 12% of the life cycle energy consumption. Notter et al. (2010) integrated the BMS with the battery steel box and cables as part of the battery pack and analysed it as over 20% of the overall impact; thus the BMS contributed to less than 20%. The values reported by these studies were accurate at the time they were authored, and since BMS designs are constantly under development and improvement with time, battery chemistry, and technological advancement, its contribution to battery life cycle environmental impacts will also evolve.

5.3.1.2 INFLUENCE OF LIB PRODUCTION VOLUME

Moving on to the actual assembly process of LIBs. Most studies have assumed manual assembly of the battery packs, therefore there are little energy and environmental impact associated with the process. Kim et al. (2016) and Ellingsen et al. (2013) have found that the assembly process that is not completely manual contributed very little to energy consumption and environmental impact. Indeed, Ellingsen et al. (2013) reported that the welding process is the only direct energy requirement and only amounts to 3.89×10^{-3} kWh per kWh of battery capacity.

The facility capacity is an important influencing factor of battery assembly energy intensity. That is, facilities that operate at or near the capacity for which they were designed would increase production efficiency, thus minimising energy intensity (Wu & Kong 2018). It should be noted that certain equipment is likely to consume the same amount of energy regardless of the operation capacity. If it were possible to produce recycled cathode materials at a lower energy intensity than to produce virgin cathode materials, the energy saved on a whole-battery level would be insignificant compared to the consumed energy from assembling the battery. If the battery assembly facilities operate at or near capacity, then the pack assembly process contributes to no more than 10% of total energy consumed, thus recycling the batteries will yield the benefits mentioned. In this case, the life cycle energy consumption is driven by the energy intensity of the active materials or of the wrought aluminium used for the structure material (Dunn et al. 2015).

5.3.1.3 INFLUENCE OF REGIONAL MANUFACTURING

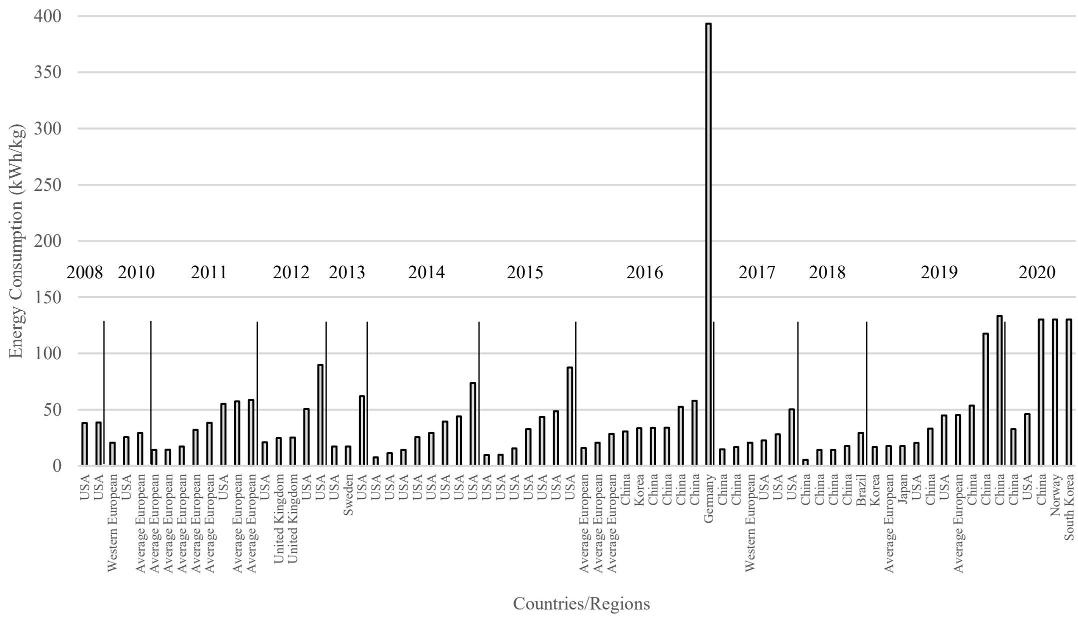


Figure 21 – Overview of CED from LIB production.

To visualise regional influence on the environmental impact of LIB production, the CED data reported by LCA studies with respect to study year and its country of reference were graphed in *Figure 21*. The time trend shows that from 2008 to 2015 the energy reference of LCA studies was predominately from the United States and Europe. From 2016 onwards, there is a gradual increase in LCA studies conducted in China. Additionally, it was also found that a substantial increase in LCA studies investigating the life cycle GWP and CED impacts of emerging LIB technologies. For example, the one-off high CED value (Germany, 2016) is due to the production of a new LCO-Li battery and on a small laboratory scale only.

LIB manufacturing countries with carbon-intense grid-mixes impact the GWP and CED of battery pack production directly. As of 2020, China continues to dominate the LIB supply chain (BNEF 2020), where their grid-mix is highly carbon-intense as a result of high shares of black coal used. Alternatively, countries with low carbon-intense grid-mixes (such as Sweden) shift the GWP and CED impact onto materials production instead (Philippot et al. 2019). Therefore, it is evident that the results vary significantly amongst different countries and regions due to the variable production techniques and specific manufacturing processes. In addition to varying grid-mixes, regional industrial practices can also differ significantly (Dai et al. 2019). Our investigation shows that the BOM of LIBs varies substantially with design, configuration, and materials, consequently the geographic locations influence the supply chains of LIBs.

5.3.2 INSIGHTS AND IMPLICATIONS

5.3.2.1 SIGNIFICANCE OF BATTERY RECYCLING AND REPURPOSING

This section discusses two major practices for LIB decommissioning: recycling and repurposing. Of the 55 studies, only ten included the decommissioning phase in the research. The majority of studies had omitted this phase, claiming that the magnitude of EoL contributions to total environmental impact is relatively small, and in addition, there is a lack of inventory data for LIB recycling (Aichberger & Jungmeier 2020). Most studies have defined their system boundary such that the environmental impacts of battery recycling or reuse were disregarded, and the decisions were made in part because of the uncertainty regarding the technology and scale of LIB EoL decommissioning in the future.

The LIB service life can be described as the function of battery degradation, characterised by progressive capacity reduction and impedance increase, consequently requiring either a battery replacement or retiring the vehicle. Battery degradation is caused by the Depth-of-Discharge (DoD), age, charge cycle frequency, thermal conditions, State-of-Charge (SoC), and voltage conditions (Ambrose & Kendall 2016). When the capacity loss reaches the stage where the travelling distance per charge is affected, the battery pack should be replaced (Faria et al. 2014). High ambient temperatures significantly impact battery performance degradation and ultimately affects the automotive life cycle (Song et al. 2013; Eddahech, Briat & Vinassa 2015; Kiel et al. 2016). Additionally, Bauer et al. (2018) estimated that calendar ageing is the major source of battery ageing, which is independent of charge-discharge cycling. The capacity degradation is related to the contribution of the driving profile and the number of charge cycles required to travel a given distance. For example, intensive use of the battery will require a higher number of cycles to travel the same distance due to higher energy consumption, losses, and lower energy extraction from the battery. Thus, one of the main contributors to capacity degradation is cycle aging from intensive uses and calendar aging from light uses (Faria et al. 2014; Casals, García & Canal 2019). Most of the reviewed literature has assumed the LIB first-life limit as 70% ~ 80% of a LIB's initial capacity. Researchers have also assumed that when a LIB's capacity drops below 50% of its initial capacity, it will no longer be suitable for any further use (Cready et al. 2003; Marano et al. 2009; Neubauer & Pesaran 2011; Ramoni & Zhang 2013; Faria et al. 2014).

A major concern relating to the environmental sustainability of LIBs is the disposal strategy of wastes generated when the BEVs reach their EoL (Richa et al. 2017; Wang et al. 2018). Considering that in 2020 a cumulative of 3.24 million BEVs and PHEVs were sold globally (Irle, R. 2020), it is imperative to develop an explicit and proactive LIB EoL management strategy. Ramoni and Zhang (2013) proposed that this strategy can be demonstrated via the Circular Economy (CE) principles (see *Figure 22 & 23*). The aim of a circular economy (or closed-loop economy) is to achieve high profitability and resource and energy efficiency via the elimination of waste by cycling the products and materials within the system (Richa, Babbitt & Gaustad 2017). This concept then allows for the reduction of virgin materials production and disposal of toxic and hazardous materials.

The first decommissioning practice is to recycle the retired LIB materials, although there is limited information on battery recycling available in the public domain, and automotive battery recycling has only begun gaining traction. Benefits include air emissions reduction from cathode metals mining, landfills, and management of metal resources (Dunn et al. 2015), extending to a reduction in landfill waste and overcoming materials shortage (Dunn et al. 2012b). The production of cathodes is the main driver of LIB life cycle environmental impacts under the assumption of assembly energy consumed in at-capacity facilities. Battery recycling may recover cathodes at lower energy intensity and emissions than producing virgin cathode materials. Recycling materials such as aluminium, nickel, steel, and copper the secondary production process reduces energy consumption by 24.6% ~ 74.1% (Gaines et al. 2011). On average, producing LIBs from recycled materials is 75% less energy-intensive than producing LIBs from virgin materials (Hammond & Hazeldine 2015). LIBs produced with recycled cathodes, aluminium, and copper reduce GHG emissions by up to 50% compared to batteries produced entirely from virgin materials (Dunn et al. 2015). The GHG reductions for producing the cathode material with the commercial pyrometallurgical process for LCO batteries was reported to be approximately 60% ~ 75%. If the recycled LCO were reused into BEV batteries that would have otherwise been produced with virgin LCO hydrothermally, the cathode material contribution to the overall battery GHG intensity was reported to decline approximately 57% ~ 25%, and consequently the overall battery GHG intensity was reported to decline by 43%. The GHG emissions from battery production with different cathode materials reduce between 11% ~ 91% when the cathode material is produced from recycled Co and Li_2CO_3. LMO is the least energy and GHG intensive cathode material to produce, thus it was reported that using recycling cathode material from intermediate and direct processes may have an overall battery GHG reduction of 2% and 16%, respectively. The overall battery GHG emissions can be further reduced when aluminium and copper are recycled and reused into new battery production. The recovery process of used battery materials can be accelerated when battery manufacturers consider recycling in the design, thus allowing materials disassembly and separation processing simplicity at the battery's EoL. To initiate autonomous recycling, it may be beneficial to standardise battery materials, configurations, and specifications (Dunn et al. 2015).

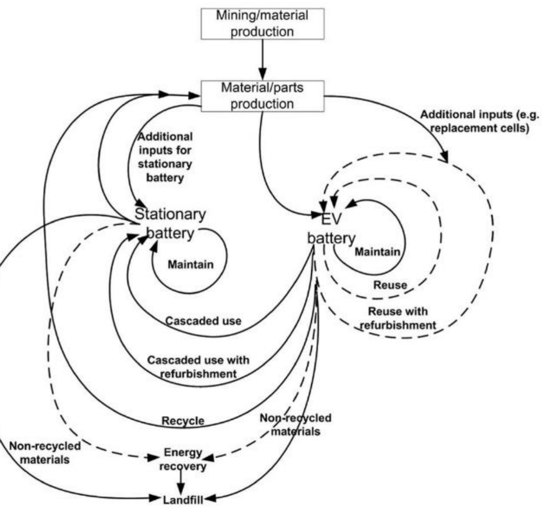
It is worth mentioning that some LCA studies performed estimates based on basic engineering calculations (Dunn et al. 2015). This signifies that battery cathode recycling additional processing may be required to reproduce a matching level of performance as cathodes produced from virgin materials. Nevertheless, these estimates serve to endorse the possibility of energy savings and emissions reduction from battery recycling. Thus, the estimates for GHG emissions should be regarded as an indication of environmental impact reduction potential, and not simply as true results.

The second decommissioning practice is to repurpose LIBs into mobility (as refurbished batteries in EVs) or utility (as stationary Energy Storage Systems (ESS)) applications. Repurposed batteries have four utility applications: supporting EV fast charging, stationary ESS integrated with solar photovoltaics (PVs) for self-consumption, supporting grid stability for area regulation, and providing power support to neighbourhood grid transformer during high power demands (Casals, García & Canal 2019). Considering the factors of capacity degradation mentioned previously and assuming a second-life EoL of 40 ~ 60% initial capacity, the estimated lifespan varies from six to 30 years. Consequently, LIB repurposing yields environmental, social, and economical benefits, since there is still some capacity remaining and thus extending the service life of the battery pack. For stationary ESS, this option shifts power demands to off-peak demand times, subsequently reducing the energy supply pressures of the electric grid and the price of energy purchased during peak times. From this literature review search, a considerable amount of research studies investigating the topic of repurposing automotive LIB batteries for supporting electricity grid operations were studied. Researchers have pointed out that proper policies and economic incentives are required to successfully promote LIB repurposing before recycling (Ahmadi et al. 2014; Faria et al. 2014; Heymans et al. 2014; Gohla-Neudecker, Bowler & Mohr 2015; Richa, Babbitt & Gaustad 2017; Richa et al. 2017; Bauer et al. 2018; Bobba et al. 2018; Casals, García & Canal 2019).

From an economic point of view, the application of such implementation strategies will depend on the resale value of the LIBs and the consumer needs. Reusing LIBs in stationary ESS contributes to a more constant loading by storing energy generated from periods of lower environmental impacts and then using the stored energy to complement the generated energy from periods of higher environmental impacts, i.e. charging the LIBs from the grid during off-peak periods or when the contribution of renewable energy sources is high. Homes, businesses, and utilities can therefore manage their own power consumption based on energy pricing and time needed (Heymans et al. 2014). BEV and PHEV battery packs with capacities of 8 ~ 24 kWh are considered ideal for residential use, as the average home energy consumption per day is approximately 10 kWh (Casals, García & Canal 2019).

From an environmental feasibility standpoint, this reviewed literature offered some interesting results. A case study investigating the repurposing of a typical 16 kWh LIB into a stationary ESS that in turn charges a PHEV. The assumed second-life commences at 80% of the remaining capacity after 160,000 km over eight years, adjusting for 1% assumed cell failure rate, and 95%

pack recovery rate. Further assumptions include an average effective value of 7.2 kWh useable pack capacity, a charge/discharge cycle of 20 ~ 80% SOH, cycling once per day over ten years, and hypothetical degradation. The study found that an additional 1.4 tCO_2e was produced from LIB repurposing. Incorporating this value, the results still showed that charging the PHEV with the stationary ESS stored with off-peak renewables energy (nuclear and wind) reported a 56% reduction in GHG emissions compared to charging the PHEV with electricity from on-peak natural gas generation (Ahmadi et al. 2014). An evaluation of the potential environmental benefits from theoretically managing 1,000 EoL BEV battery packs in the USA indicated that under the assumption of sufficient technology and market support of battery reuse, approximately 56,000 kWh of CED would be recouped. The same amount of energy would have been consumed to produce 11 new 18-kWh EV LIBs. It was further stated that second use as stationary ESS applications would magnify the benefits nearly tenfold, and in so doing avoid new LIB production and use of inferior battery chemistries (such as lead-acid). As such, the results showed that for 1,000 EoL LIBs used in stationary ESS applications operating for five years, approximately 2.69 kWh of CED were saved due to the avoided process of production. Additionally, 130 tons of metal inputs would be potentially avoided, mainly from avoiding primary and secondary lead production (Richa, Babbitt & Gaustad 2017). In a study scenario, a repurposed LIB was installed in the place of a new stationary ESS and coupled with a solar PV system. The resulting life cycle impact analysis revealed that a reduction of 58% of life cycle GWP was achieved (Bobba et al. 2018).



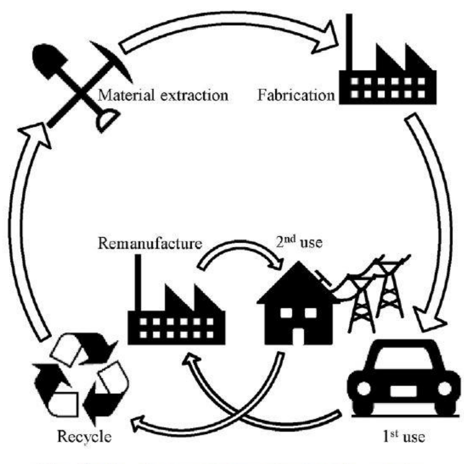


Figure 22 – (left) Theoretical waste management hierarchy for LIBs after automotive applications.

Figure 23 – (right) Circular economy of repurposing LIBs.

Source: Richa, Babbitt & Gaustad (2017); Casals, García & Canal (2019).

5.3.2.2 MITIGATION OPPORTUNITIES

This literature review has shown that LIB production has a serious impact on the environment, contradictory to their endorsement as absolute zero-emissions vehicles. Thus, this study proposes the following mitigation opportunities endorsed by the results obtained. First, **increase the LIB production efficiency and introduce cleaner manufacturing and production processes, enriched through the implementation of battery recycling strategies**. Kim et al. (2016) and Ellingsen et al. (2013) have shown that the LIB assembly process that is not completely manual contributed little to energy consumption and environmental impact. Additionally, if the battery assembly facilities operate at or near capacity, then the pack assembly process contributes to no more than 10% of total energy consumed (Dunn et al. 2015). This study's results have shown that the GHG emissions and total energy consumed from LIBs production in Europe are significantly lower than in Asia. Taking steel and aluminium, for example, Argonne's GREET® model (2019) reported the production CED and GWP of 8,582.18kWh/2,740.30 kgCO_2e and 33,461.91 kWh/7,242.57 kgCO_2e, respectively. The electricity generation method largely affects the GWP of LIB production processes. In their sensitivity analysis, Wang et al. (2016) determined that if 10% of China's coal-fired power is replaced by wind or hydropower, the decrease in environmental impacts of battery use reaches 7.9% and 8.2%, respectively. Ultimately the main driver for GWP reduction would be to decarbonise the grid-mix via the adoption of renewable energy generation. Consequently, a series of policies should be introduced by governments and policy makers to also regulate the emissions produced by energy generation. Then, **increase the battery energy density to reduce the LIBs life cycle environmental impacts on the EV power systems, including GHG and other air pollutant emissions** (Yu et al. 2018). The authors also concluded that increasing battery energy density by 0.1 kWh/kg can reduce air pollutant emissions by up to 20%. Low energy density LIBs require more frequent charging and increased weight to satisfy the energy demand, thus implying a greater energy loss in the BEVs life cycle (Wang et al. 2016). With the increased popularity and extensive promotion of BEVs, the key to improving LIB energy density (and consequently improving mileage per charge) ultimately lies in the breakthrough of battery material research and development. Next, **reduce LIB weight with alternative materials, consequently increasing operations efficiency**. Larger battery capacities will power the BEV over longer distances, but will also add to the total vehicle weight and production costs (Nealer, Reichmuth & Anair 2015). There are strategies to achieve adequate weight reduction: replacing heavier materials (such as metals) with lighter materials (e.g. carbon fibre composites) and modifying the production process (for example, welding joints together instead of using bolts). Last, **develop and operate a fully functional LIB EoL decommissioning industry**. The wastes generated from BEVs as they reach their EoL is a major concern to the environmental sustainability of LIBs. Without any EoL disposal strategies, the retired LIBs materials will ultimately end up in landfills. Researchers have urged governments and policy makers to facilitate and support research aimed at initiating and improving LIB recycling or repurposing processes (Ahmadi et al. 2014; Nealer, Reichmuth & Anair 2015; Richa, Babbitt & Gaustad 2017; Hao et al. 2017a). Repurposing LIBs into mobility or utility applications should first be considered before resolving to LIB recycling. The results from the research will assist with introducing regulatory laws to gain environmental benefits from LIB decommissioning. Currently, Canada, China, Germany, the UK, and the USA are the pioneering countries in the LIB recycling market (MarketsandMarkets 2020).

5.4 CONCLUSION

This study has reviewed an overall of 86 available LCA studies that assessed the environmental impact of automotive lithium-ion batteries. The evaluated LCA studies have been formally peer-reviewed and published in reputable academic journals, conference proceedings, and official government publications that provided detailed contribution analysis and a transparent inventory of LIBs. A total of 55 studies were identified that fulfilled the selection criteria, and accordingly, the data and inconsistent results on life cycle GHG emissions and energy impacts were thoroughly examined. Afterwards, the usefulness of LCA studies available to date was scrutinised to facilitate discussion among industry, governments, and policy makers seeking advice and guidance from LCA studies on the environmental impact of LIBs. When considering all LIB chemistries, this study found a range of emissions intensities, from a low of 5.40 $kgCO_2e/kWh$ to a high of 1,730.77 $kgCO_2e/kWh$, with an average value of 187.26 $kgCO_2e/kWh$ in terms of battery capacity. In terms of battery mass, it ranges from 0.21 ~ 96.96 $kgCO_2e/kg$, where the average value is 19.78 $kgCO_2e/kg$. In terms of CED, the average value is 42.49 kWh/kg from a range of 5.44 ~ 393.33 kWh/kg of battery production. Based on these results, the following conclusions are made.

First, the absence of available primary data caused inconsistencies of results found across the LCA data. The majority of the reviewed studies relied on using secondary data, as it is rather difficult to obtain real battery production data from the industry sector. Therefore, LCA studies often combined available industry and literature data into hypothetical designs or modelling tools tailored specifically for the study. Thus, the diversity of battery specifications from literature, reports, and industrial data often induce errors in GWP and CED calculations.

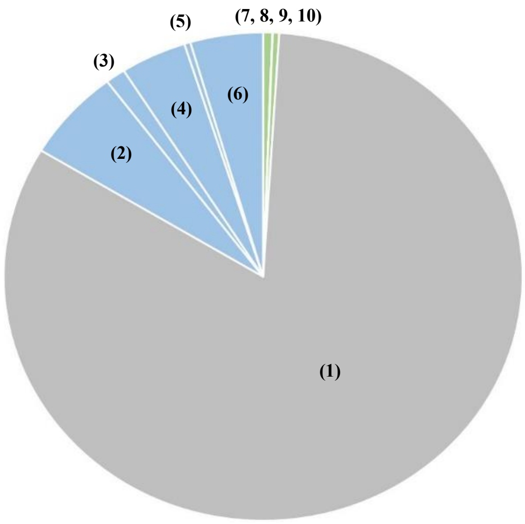
Second, battery chemistry played an important role in influencing GWP and CED results. We found production and processing of active cathode materials, together with high-energy consumption from metals production were the key contributors to life cycle energy and environmental impacts. Additionally, the assembly process energy intensity was influenced by facility capacity. Facilities operating at or near the capacity for which they were designed minimised energy intensity. There were more case studies and thorough investigations conducted on five mature battery chemistries (NMC-C, NCA-C, LCO-C, LFP-C, and LMO-C) compared to potential next-generation LIBs. Unsurprising, battery manufacturers have the ability to produce these five mature LIB chemistries on an industrial scale (as opposed to laboratory scale), hence their GWP and CED values were also reported to be lower than the new LIBs.

Third, the environmental impact varies significantly depending on geographic locations due to the variable production techniques, industrial practices, specific manufacturing processes, and regional-dependent electricity generation methods. This is exacerbated by a broader push towards low emissions energy production internationally. As these changes evolve, there will be an anticipated reduction in the variation.

Last, from the results obtained by this study, the following mitigation opportunities are suggested: 1) increase the LIB production efficiency and introduce cleaner manufacturing and production processes, 2) increase the battery energy density to reduce the LIBs life cycle environmental impacts on the EV power systems, including GHG and other air pollutant emissions, 3) reduce LIB weight with alternative materials to increase operations efficiency, and 4) develop and operate a fully functional LIB EoL decommissioning industry. Retired battery materials recovery and recycling may be achieved at lower energy intensities and emissions compared to producing and processing virgin materials. Literature has shown that repurposing LIBs with some capacity still remaining into mobility or utility applications extend their service lives and yield environmental, social, and economical benefits. Additionally, recycling reduces landfill waste and materials shortage. Therefore, after the LIB's initial conceived purpose of powering EVs propulsion systems, we strongly recommend repurposing the LIBs into second-use applications and then followed by recycling the materials at their second use EoL.

6 CHAPTER 6: SUMMARY & CONCLUSIONS

The aim of this research was to address the major research gap for developing a comprehensive understanding of the potential for and environmental impact of powertrain electrification in the Australian transport sector. The aim is achieved through the application of four alternative case studies investigating a specific research area. *Figure 24* graphically shows the accumulative life cycle GHG emissions produced from the equipment life cycle segments (charging infrastructure and bus production) and WTW life cycle segment (operations).



Life Cycle Segments	$kgCO_2e$
Operations (WTW)	
Electricity Generation **(1)**	682,344.0
Bus Production (Equipment)	
Production **(2)**	50,297.3
Assembly **(3)**	10,232.7
Maintenance **(4)**	34,002.7
Transportation **(5)**	3,112.3
Decommissioning **(6)**	37,387.1
Charging Infrastructure (Equipment)	
Production **(7)**	4,737.6
Transportation **(8)**	71.0
Installation **(9)**	86.3
Decommissioning **(10)**	3,310.7
Total	**825,581.7**

Figure 24 – Total Life Cycle GHG emissions ($kgCO_2e$) of a BEV bus in the Australian transport sector.

The following subsections summarise the results and achievements of this research.

6.1 LCA OF CHARGING INFRASTRUCTURES

This chapter addressed the charging infrastructure life cycle segment of the Complete Life Cycle model. A case study approach was applied for Australia and calculated the magnitude of GHGs produced from the implementation of electric bus charging stations into existing bus depots concurrent with the transitioning of the commuter bus fleets into electrified powertrains. Utilising the Australian-based fleets as a case study and baseline scenario, the greenhouse gas emissions from production, transportation, installation, operations, recycling, and disposal phases were estimated to establish a comprehensive and in-depth emissions LCA. To the study's best estimate, the total life cycle environmental load of a BEV bus charging station amounts to 690,549.6 $kgCO_2e$. The production, transportation, installation, operations, and decommissioning phases contributed to 0.7%, 0.01%, 0.01%, 98.8%, and 0.5% of the total emissions, respectively. Charging a BEV bus with the current Australian grid-mix yields a carbon intensity of 94.4 ~ 105.0

$kgCO_2e/100km$. In comparison, diesel and hybrid buses yield a carbon intensity of 76.5 ~ 79.9 $kgCO_2e/100km$ and 53.4 ~ 55.7 $kgCO_2e/100km$, respectively. This signifies that electricity generation in Australia produces approximately 1.2 ~ 1.3 times more GHG emissions than when combusting diesel fuel. Thus, the operations phase is heavily dependent on the electricity grid-mixes carbon intensity and produced the most greenhouse gas emissions. Consequently, net-zero emissions will not be achieved without substantial grid-mix decarbonisation. *Figure 25* compares the operations lifetime GHG emissions of a diesel, hybrid, and BEV bus.

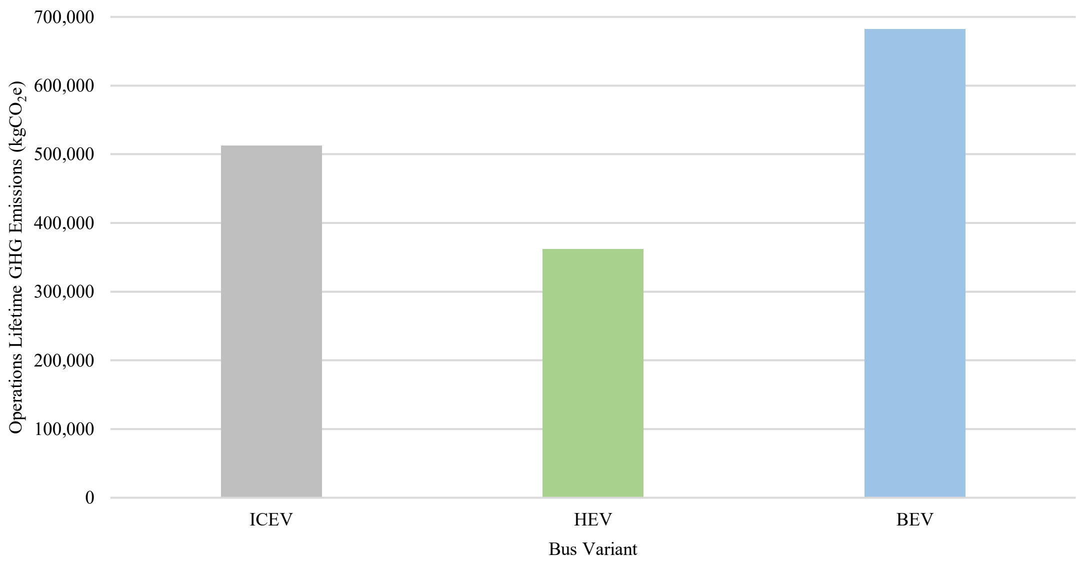


Figure 25 – Operations lifetime GHG emissions of a diesel, hybrid, and BEV bus[10].

6.2 OPERATIONS EMISSIONS ASSESSMENT & EVALUATION

This chapter addressed the WTW life cycle segment within the Complete Life Cycle model (vertical flow in *Figure 1*). First, the study analysed the GHG emissions produced from electricity generation of 49 different countries from the available data in the public domain. A break-even analysis was conducted for six countries that had reported high carbon-intense electricity grid-mixes. Considering the environment, the break-even analysis shows that it would be intuitive to promote and implement BEV buses in countries with low carbon-intense grid-mixes, as the operations environmental load of a BEV bus will be equivalent to or less than that of a diesel bus. For Australia, this signifies reducing the grid-mix's fossil fuel share to 74.5% ~ 81.5%. Implementing renewable technologies will decarbonise the grid-mix, however, it should be noted that no technologies can be 100% emissions-free.

Then, considering the logistics of the operation, the optimal arrangement in the urban and suburban settings is to deploy a BEV bus with a small battery capacity and charge with an

[10] See *Section 2.2.6.5* for operations lifetime emissions calculations.

opportunity pantograph charger at the terminals during operational hours and at the depots overnight. For highway settings, the optimal arrangement is to deploy a BEV bus with a larger battery capacity and charge at the depot or in a dedicated charging complex.

6.3 LCA OF BUS PRODUCTION

This chapter addressed the vehicle life cycle segment of the Complete Life Cycle model and applied a case study approach and calculated the life cycle GHG emissions produced from the production, assembly, maintenance, transportation, and EoL phases of diesel, hybrid, and BEV buses. The study showed that from a life cycle perspective, the BEV bus had the highest environmental load compared to its diesel and hybrid counterparts. Results from the study's calculations showed that the total life cycle environmental load of a BEV bus amounted to 135,032.1 $kgCO_2e$. The production, transportation, installation, operations, and decommissioning phases contributed to 37.2%, 7.6%, 25.2%, 2.3%, and 27.7% of the total emissions, respectively. Comparatively, the life cycle environmental load of a diesel and hybrid bus amounted to 101,439.1 $kgCO_2e$ and 105,254.8 $kgCO_2e$, respectively. However, there are many opportunities to reduce product life cycle emissions, such as improvement in manufacturing efficiency, developing new battery technology, and production in regions with low carbon-intense grid-mixes.

6.4 LCA OF LITHIUM-ION BATTERIES

This chapter addressed the last item of the equipment life cycle segment in the Complete Life Cycle model. The study critically reviewed an overall of 76 available life cycle studies that have assessed the environmental impact of LIBs and have also provided detailed contribution analyses and transported inventories. A total of 55 studies were identified that investigated the four notable product life cycle phases of materials and parts production, cell manufacturing, battery pack assembly, and EoL. Based on the results from the reviewed studies, the average values for Global Warming Potential (GWP) and Cumulative Energy Demand (CED) from LIB production were found to be 187.26 $kgCO_2e/kWh$ or 19.78 $kgCO_2e/kg$, and 42.49 kWh/kg, respectively. This provided evidence to expose the fact that from a life cycle perspective EVs are not emissions-free and contribute to GWP. An examination into the disparity in GWP and CED estimates revealed that the results were influenced by LIB chemistry, active materials, production volume, regional manufacturing, and various assumptions adopted by the life cycle studies. Most studies claimed that the magnitude of EoL contributions to total environmental impact is relatively small and consequently omitted the EoL phase from their investigation. Further investigations into LIB second-life applications presented the argument that repurposing LIBs into mobility or utility applications extend their service lives and yield environmental, social, and economical benefits.

Also, recycling reduces landfill waste and materials shortage. Therefore, the study recommended more research efforts and implementation of industrial practices on LIB decommissioning through repurposing and recycling.

6.5 FUTURE RESEARCH

As a result of this research, the life cycle environmental impacts of electric buses in the Australian transport sector have been properly quantified and assessed. Although the aim of this research has been addressed and an extensive range of topics have been covered in this thesis, there are still many challenges yet to be overcome. In an endeavour to advocate clean transport technologies developments to combat climate change, the following are topics that require further investigation.

First, the results of this research have shown that the electricity generation process in Australia is substantially carbon-intense, consequently, there is a need to decarbonise the electricity grid-mix. From the break-even analysis conducted in Chapter 3, the fossil fuel percentage of the average Australian grid-mix must be reduced to 72.0% ~ 81.5% in order for the operations environmental load of a BEV bus to be equivalent to that of a diesel bus. Implementing renewable technologies will vary across the country and are dependent on geography, climate, local and federal government incentives, and energy generation efficiency. At this point in time as this research is being authored, only Western Australia, South Australia, Tasmania, and the Northern Territory can properly implement BEV buses from an energy and environmental sustainably perspective. Much effort would be required for governments and policy makers to continuously scrutinise and evaluate the environmental load of electricity generation, and then take the proper measures to mitigate these impacts.

Second, the operations emissions data calculated in Chapter 2 can be experimentally verified in real-world traffic conditions. The data collection process involves temporarily attaching a Portable Emissions Measurement System (PEMS) unit to a bus with an ICEV powertrain (diesel or hybrid), and then collecting data under Real-World Driving Emissions (RDE) testing conditions. The raw data collected is a direct representative of the real-world driving carried out under Australian conditions, such as weather, traffic, driving behaviour, and road topology. Consequently, the experimental data will continuously provide updates on ICEV powertrain environmental performances, which in turn will assist with GWP mitigation processes in the Australian transport sector. For BEV buses, the real-world energy consumption data will influence the planning for charging infrastructure implementation (Chapter 2), optimising charging schedules, and determining charging strategies (Chapter 3).

Third, extend this research to conduct a complete emissions LCA for hydrogen buses. The primary outcomes of this research will be to comprehensively evaluate the life cycle environmental costs and benefits of incorporating hydrogen Fuel Cell Vehicle (FCV) buses into the Australian transport bus fleet during the transitional phase to electrified powertrains. At present when this dissertation is authored, FCV buses are still undergoing rapid development and are not commercially available in the Australian transport sector as the cost to operate FCV buses readily exceeds ICEV, HEV, and BEV buses. Additionally, the research will provide an accurate environmental impact assessment of the hydrogen fuel supply life cycle in the operations phase. The current and most common method of producing hydrogen is steam-methane reforming (EIA 2021), and the main methane source is natural gas. Another method is electrolysis, where on large commercial scale hydrogen is created by splitting water into hydrogen and oxygen. However, this is an energy-intensive process and from an environmental perspective, it would not be environmentally sustainable (or to claim zero emissions) if the electricity was generated from heavy shares of fossil fuels. There are also additional barriers to overcome that are similar in nature to BEV buses, such as high purchasing costs and lack of supporting refuelling infrastructures. Ultimately, the complete emissions LCA for FCV buses will entail a near-identical methodology as this thesis had conducted for BEV buses, which is to investigate the equipment life cycle and WTW life cycle specific to FCV buses. The conclusion of this research will assist in facilitating the discussion among industry, governments, and policy makers seeking advice and guidance regarding the true environmental impact of FCV buses.

7 APPENDIX

7.1 APPENDIX A: LCA OF CHARGING INFRASTRUCTURES

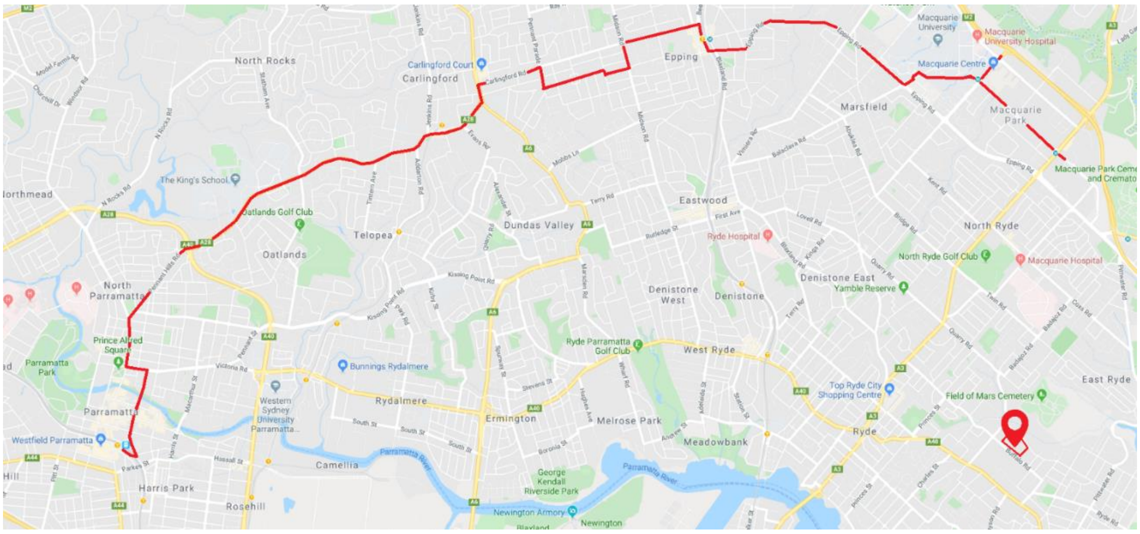


Figure A1 – Route 550 and depot location.

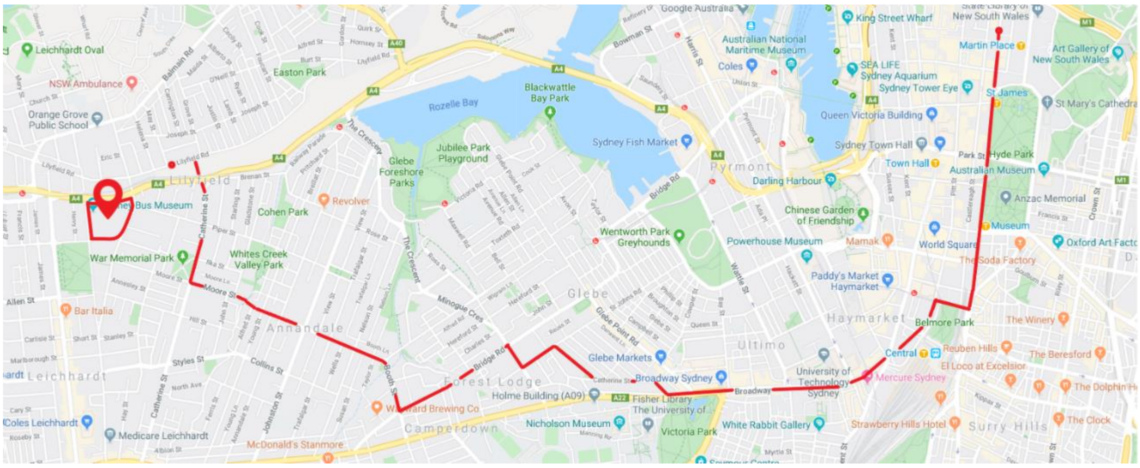


Figure A2 – Route 470 and depot location.

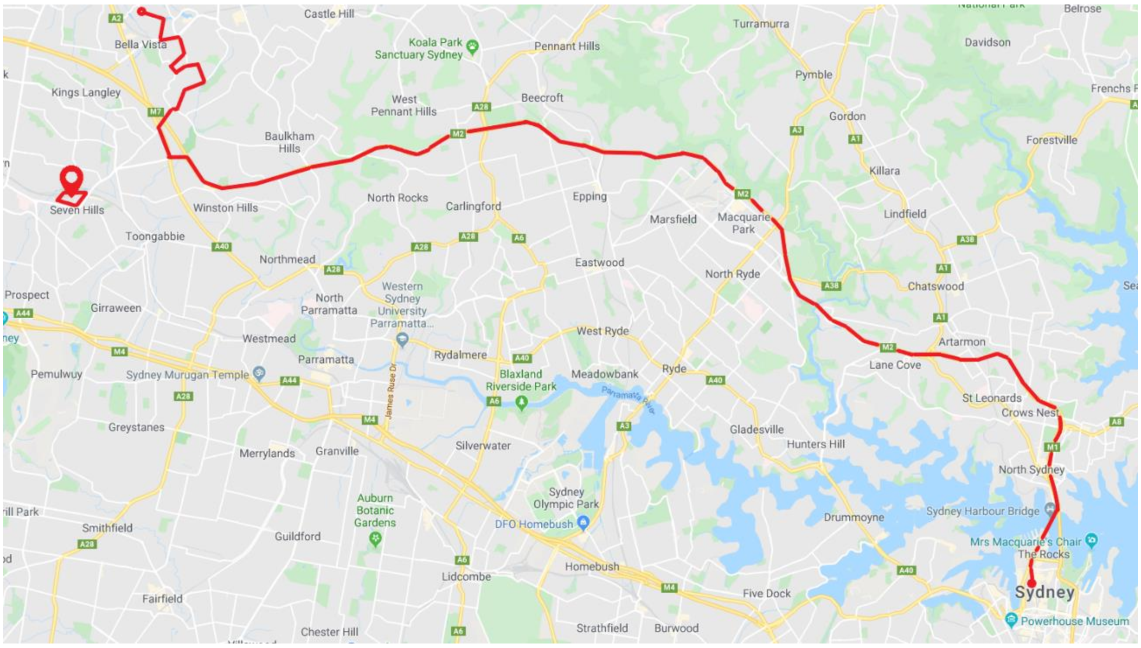


Figure A3 – Route 607X and depot location.

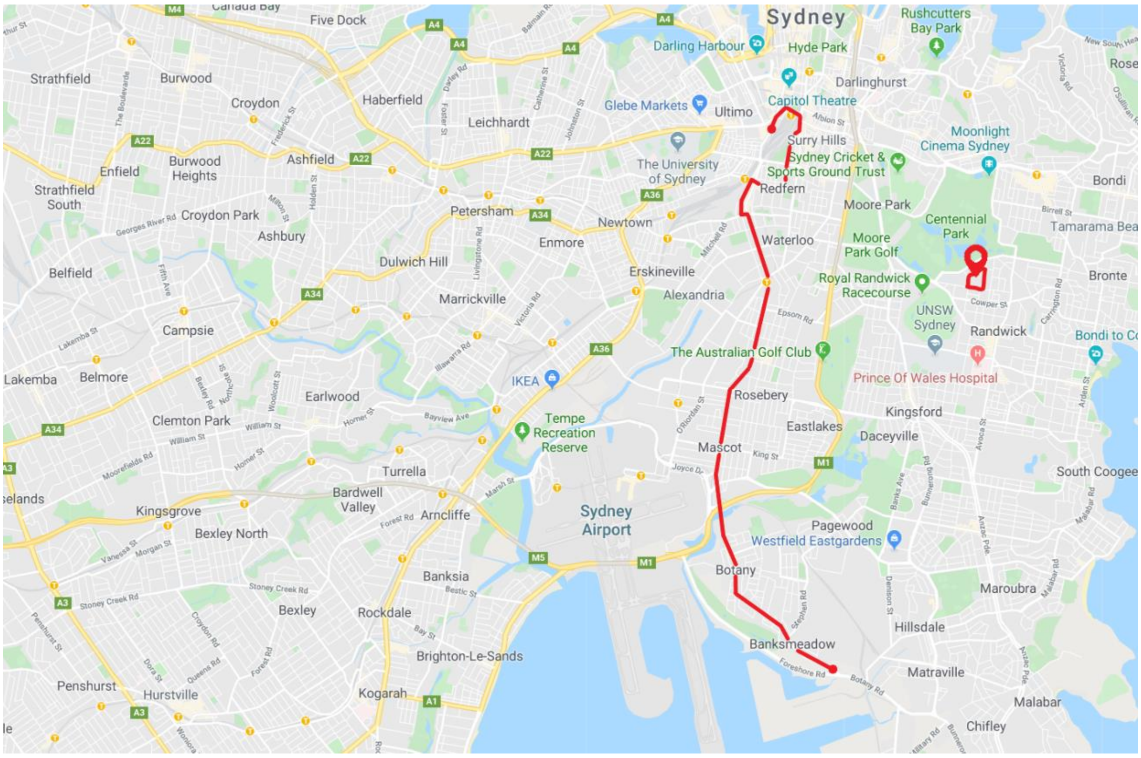


Figure A4 – Route 309 and depot location.

Figure A5 – BEV bus with BYD chassis and Gemilang body (Transit Systems 2019).

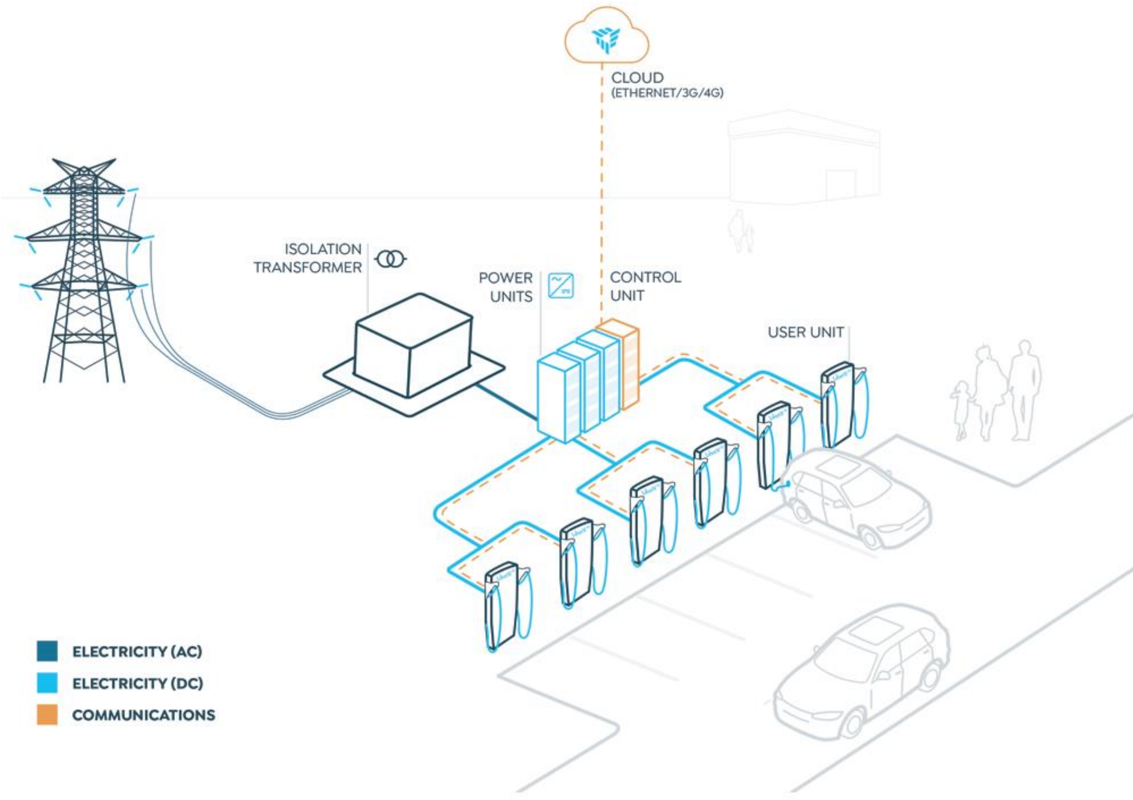


Figure A6 – Tritium system components layout (Tritium 2018).

7.2 APPENDIX B: OPERATIONS EMISSIONS ASSESSMENT & EVALUATION

OPERATIONS MODEL

DATA INPUT	VALUE	UNIT	
Operation Hours	18.00	hours/day	
Service Frequency	**5.00**	min./bus	**User Input**
Service Frequency	12.00	buses/hour	
Service Frequency	216.00	buses/day	
Route Type	**Urban**	**User Input**	
Average Speed	40.00	km/h	
Route Length	**5.00**	km	
Number of Stops	**30.00**	-	
Distance between Stops	166.67	m	
Idling Time at Stops	**30.00**	sec.	
Traffic	**15.00**	min.	
Route Time	37.50	min.	
BEV Bus	**Volvo 7900**	**User Input**	
Battery Capacity	150.00	kWh	
Energy Consumption	0.75	1.25	kWh/km
Operational Range	100.00	200.00	km
Operational Range	20.00	40.00	routes/charge
Route Frequency	1.60	trips/hour/bus	
Required Buses	7.50	buses/day	
Required Buses	28.80	trips/day/bus	
Charging Requirements	1.44	0.72	charges/day/bus
Charging Method	**Opportunity Pantograph Charging**	**User Input**	
Charging Capacity	300.00	kW	
Charging Time	0.50	hours	
Emissions Factor	0.74	$kgCO_2e/kWh$	
Installation Emissions	8,182.30	8,538.60	$kgCO_2e$
GHG Emissions	159.84	79.92	$kgCO_2e$/bus/day
	1,198.80	599.40	$kgCO_2e$/day
	437,562.00	218,781.00	$kgCO_2e$/year

Figure B1 – Operations model used to analyse the GHG emissions produced from the different operation charging strategies.

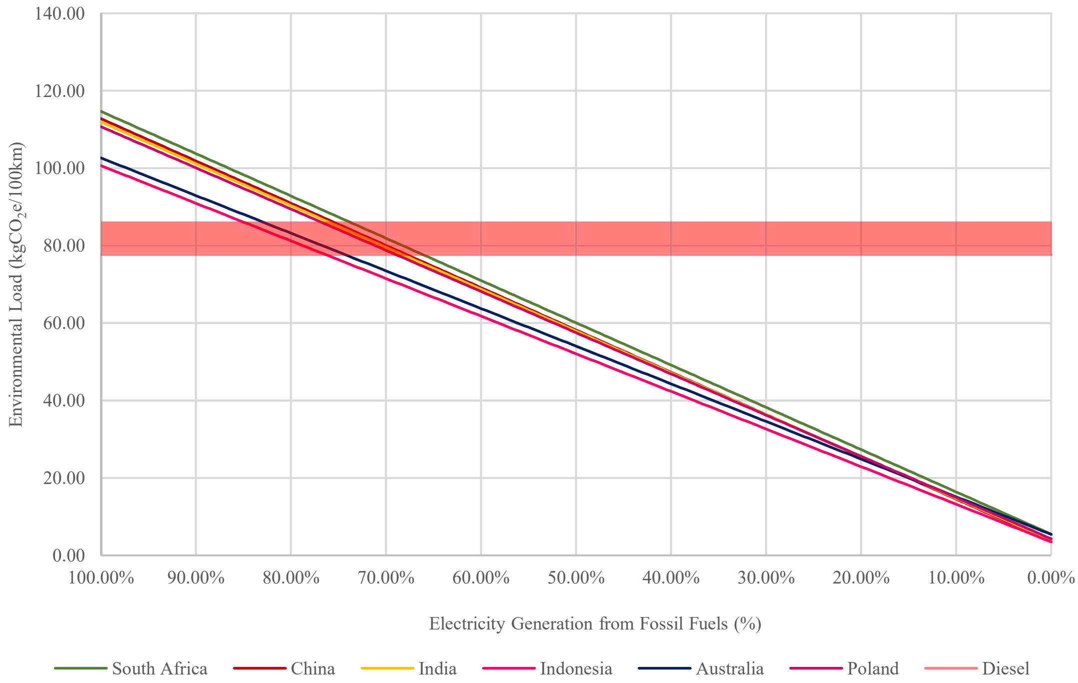


Figure B2 – Emissions break-even analysis from electricity generation.

Source: BP Statistical Review of World Energy (2019); DEE (2019b).

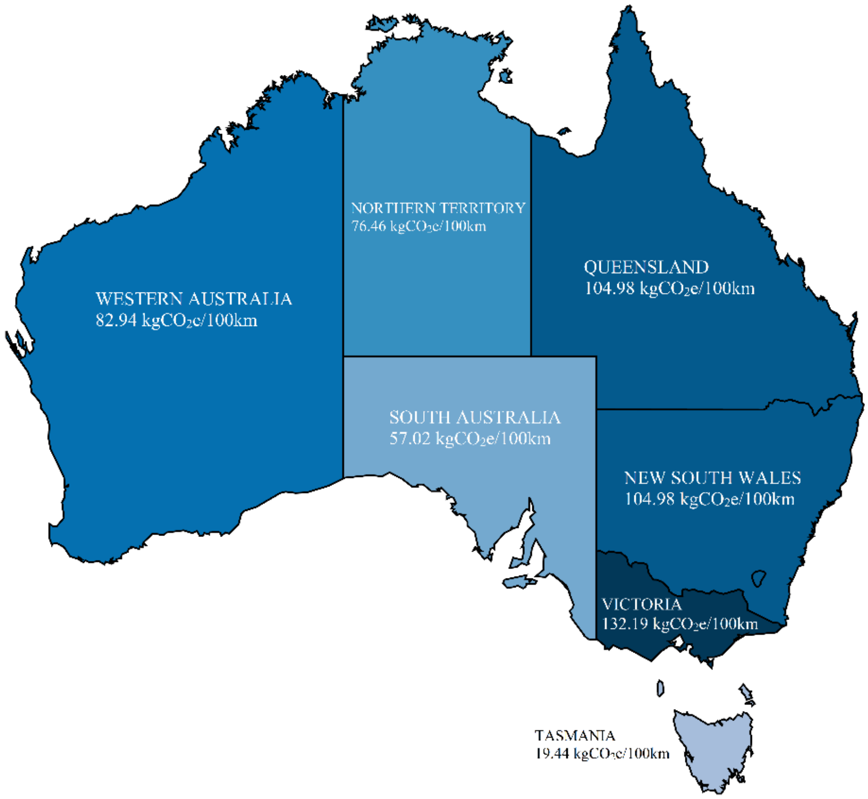


Figure B3 – Grid-mix emissions factor variation between the states of Australia.

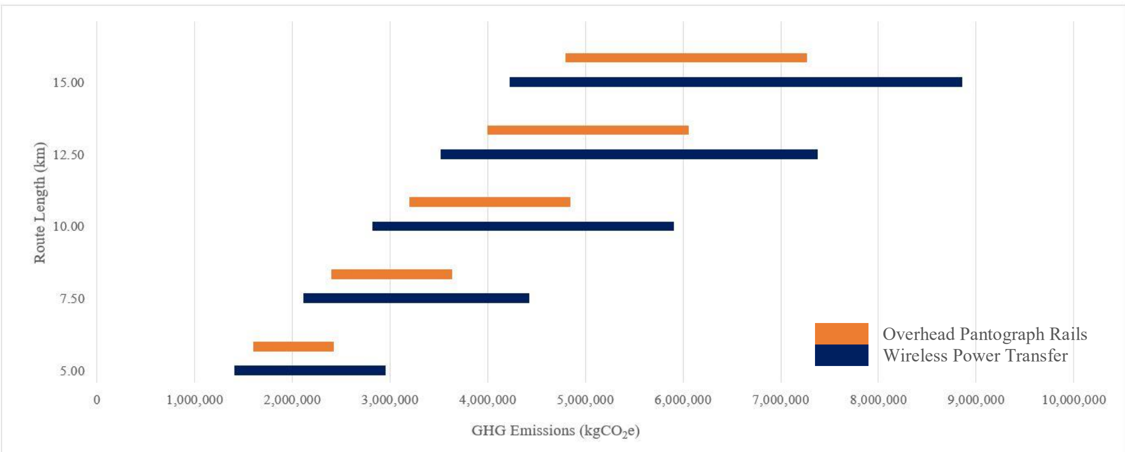


Figure B4 – Charging emissions for WTP and OPR in urban settings.

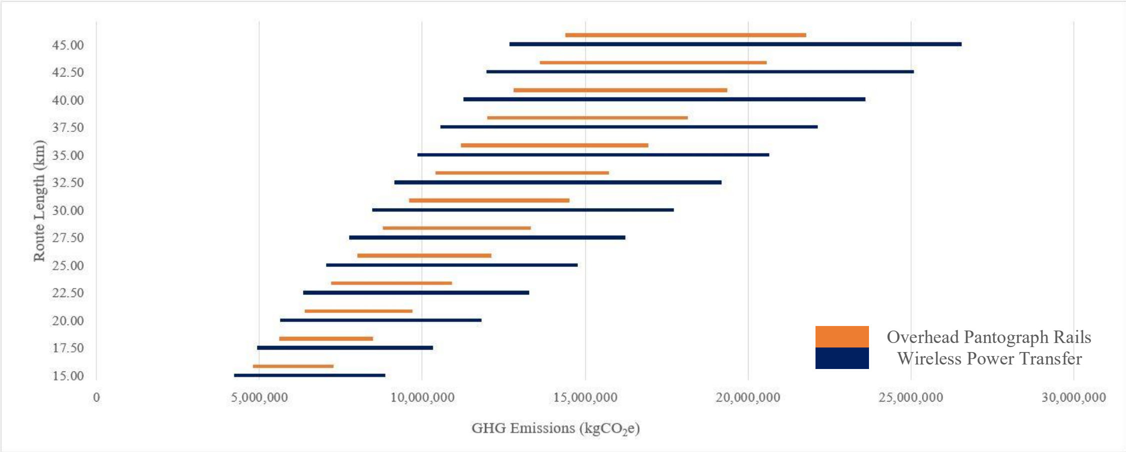


Figure B5 – Charging emissions for WTP and OPR in suburban settings.

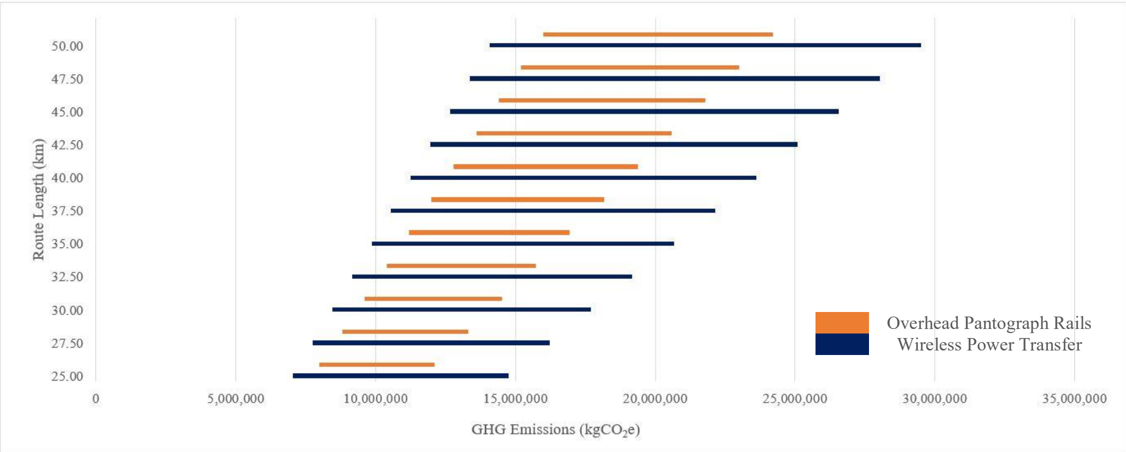


Figure B6 – Charging emissions for WTP and OPR in highway settings.

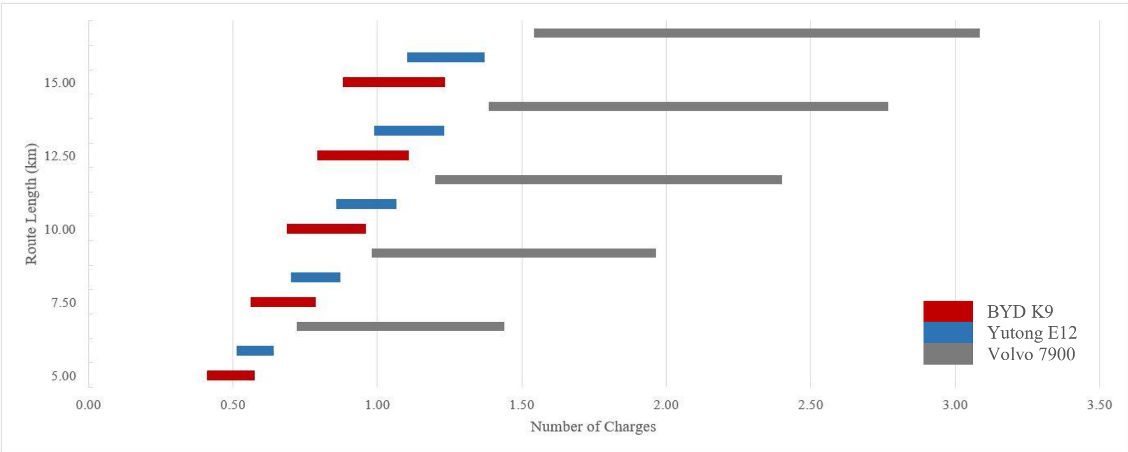


Figure B7 – Charging requirements per individual BEV bus per day in urban traffic conditions.

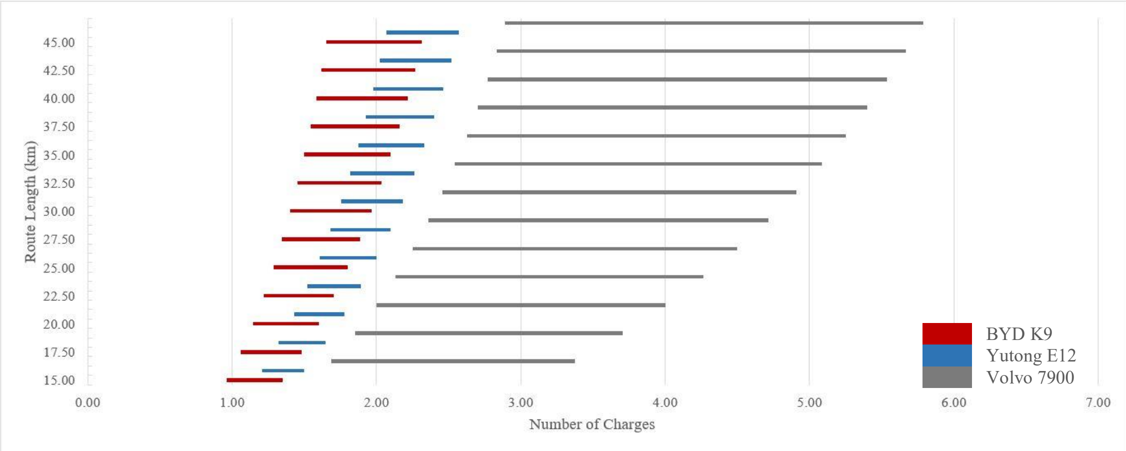


Figure B8 – Charging requirements per individual BEV bus per day in suburban traffic conditions.

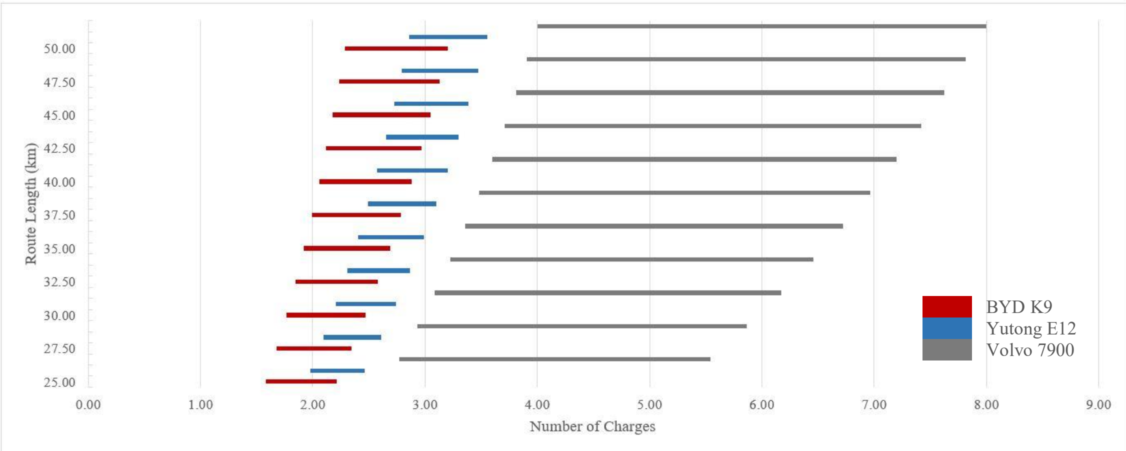


Figure B9 – Charging requirements per individual BEV bus per day in highway traffic conditions.

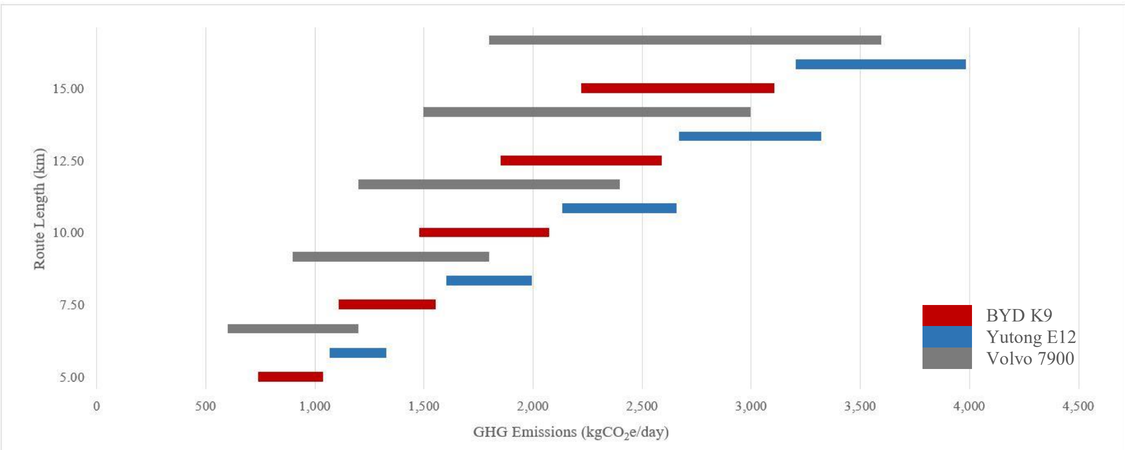


Figure B10 – GHG emissions in the urban traffic conditions with a service frequency of 5 min/bus.

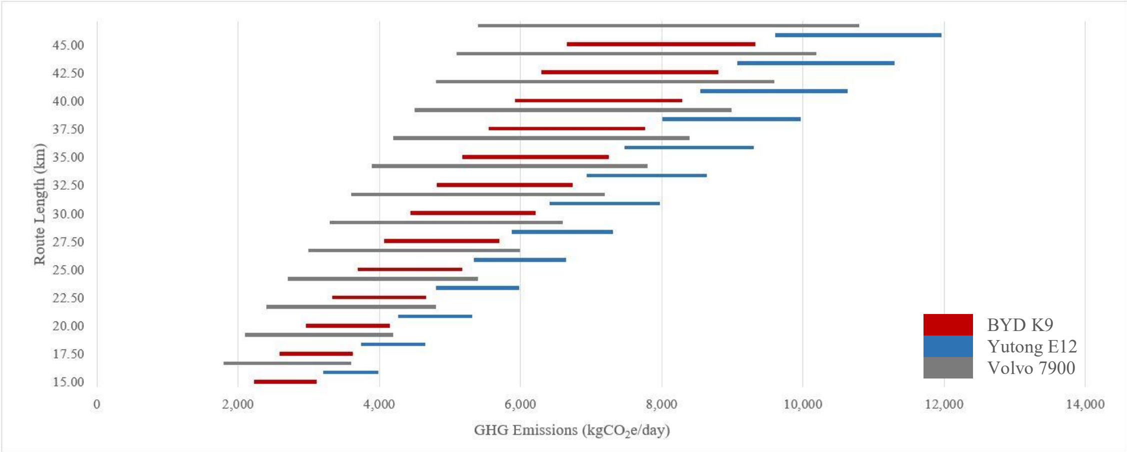


Figure B11 – GHG emissions in the suburban traffic conditions with a service frequency of 5 min/bus.

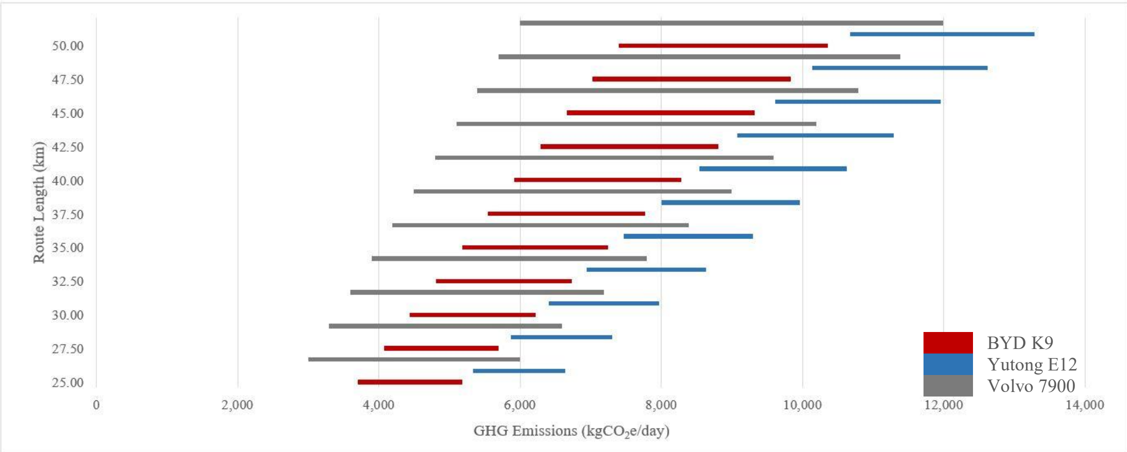


Figure B12 – GHG emissions in the highway traffic conditions with a service frequency of 5 min/bus.

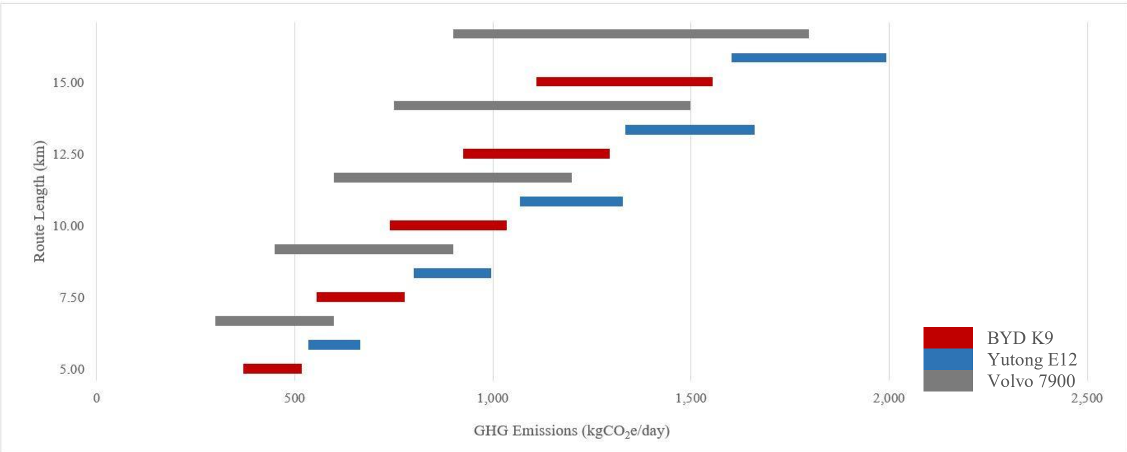


Figure B13 – GHG emissions in the urban traffic conditions with a service frequency of 10 min/bus.

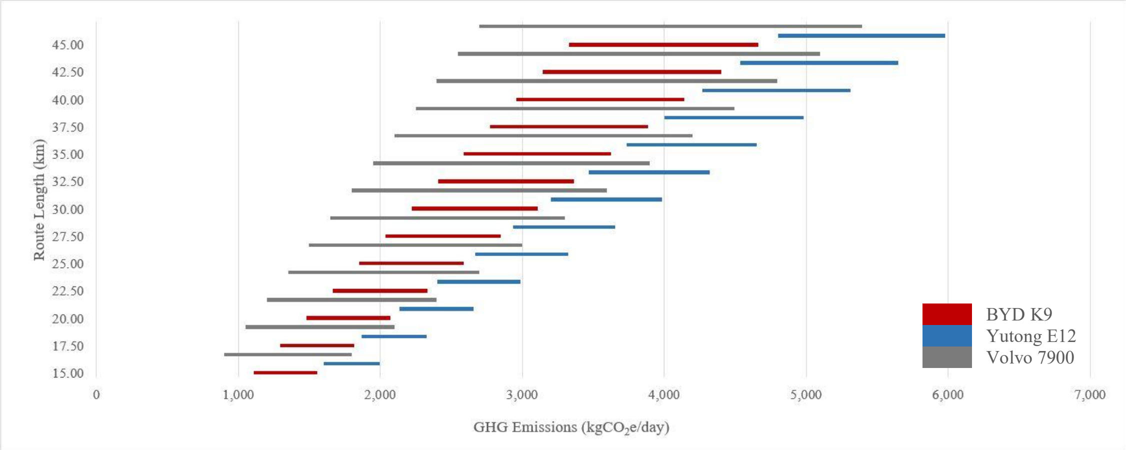


Figure B14 – GHG emissions in the suburban traffic conditions with a service frequency of 10 min/bus.

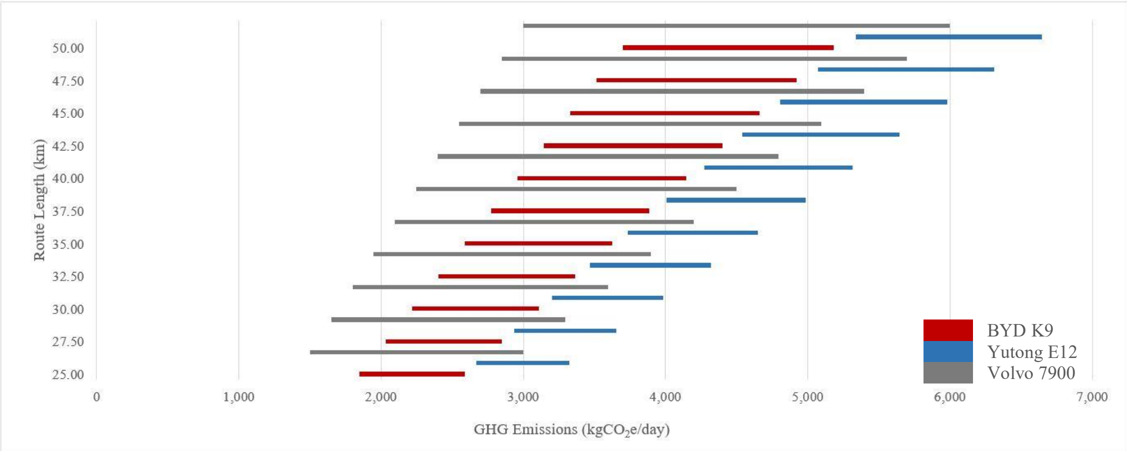


Figure B15 – GHG emissions in the highway traffic conditions with a service frequency of 10 min/bus.

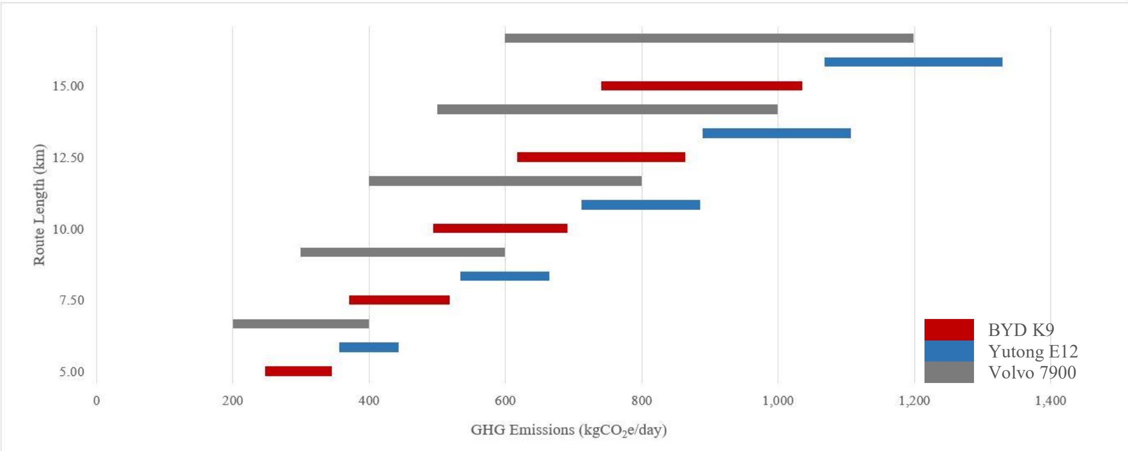


Figure B16 – GHG emissions in the urban traffic conditions with a service frequency of 15 min/bus.

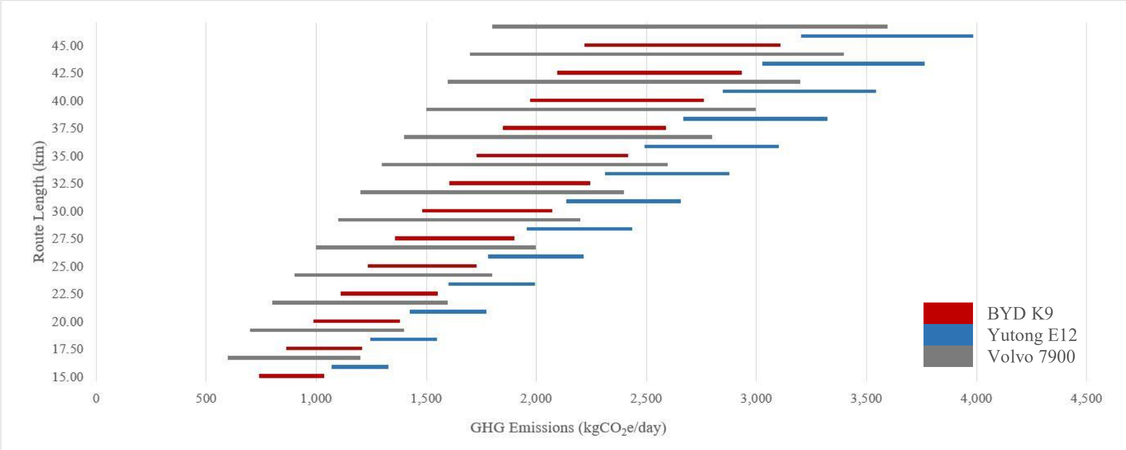


Figure B17 – GHG emissions in the suburban traffic conditions with a service frequency of 15 min/bus.

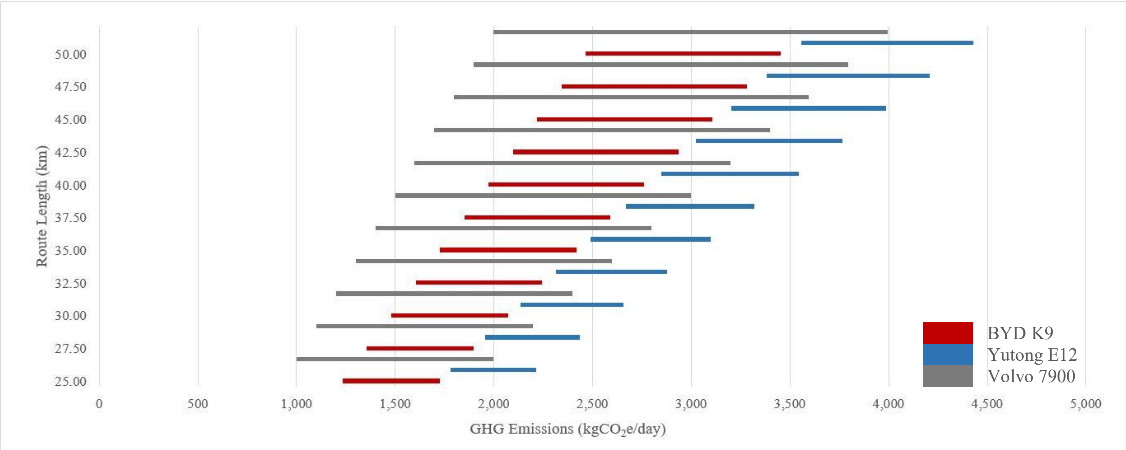


Figure B18 – GHG emissions in the highway traffic conditions with a service frequency of 15 min/bus.

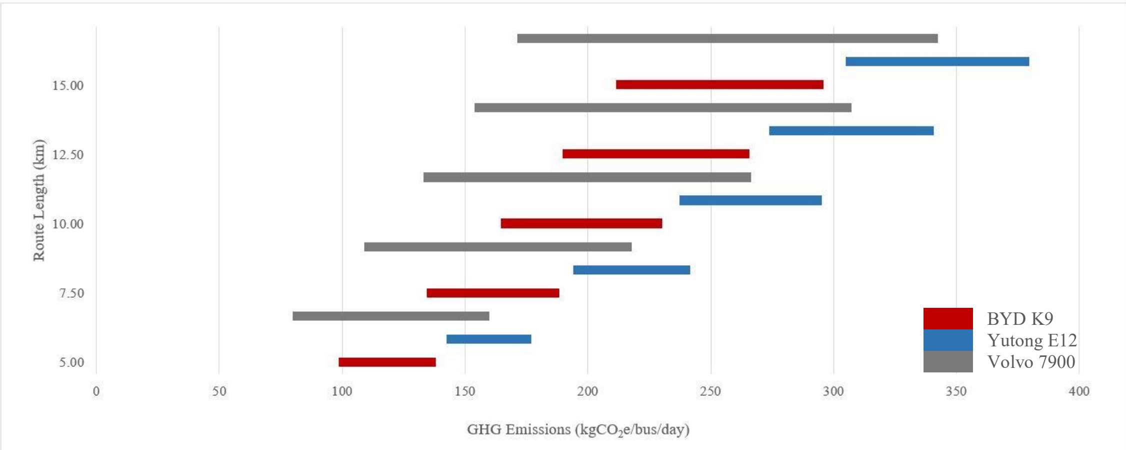


Figure B19 – GHG emissions per individual BEV bus per day in urban traffic conditions.

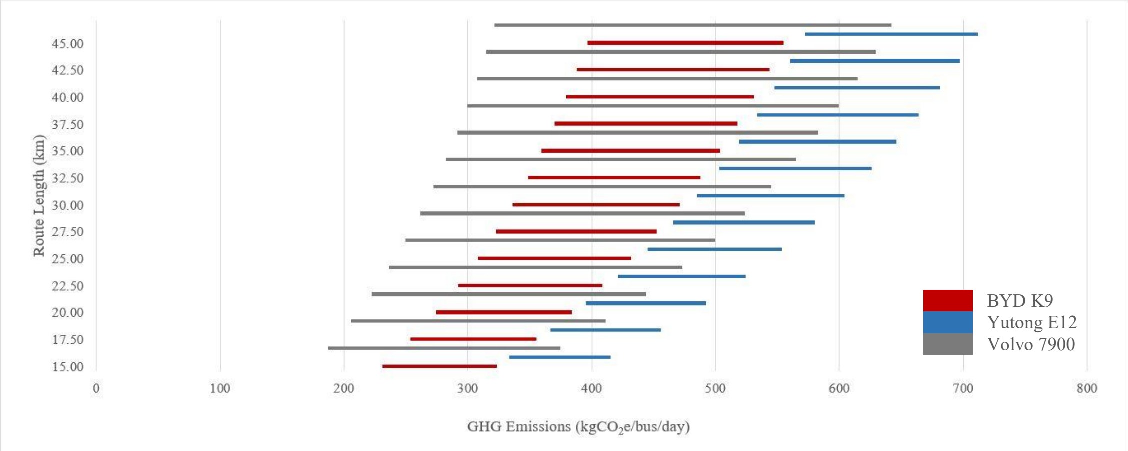


Figure B20 – GHG emissions per individual BEV bus per day in suburban traffic conditions.

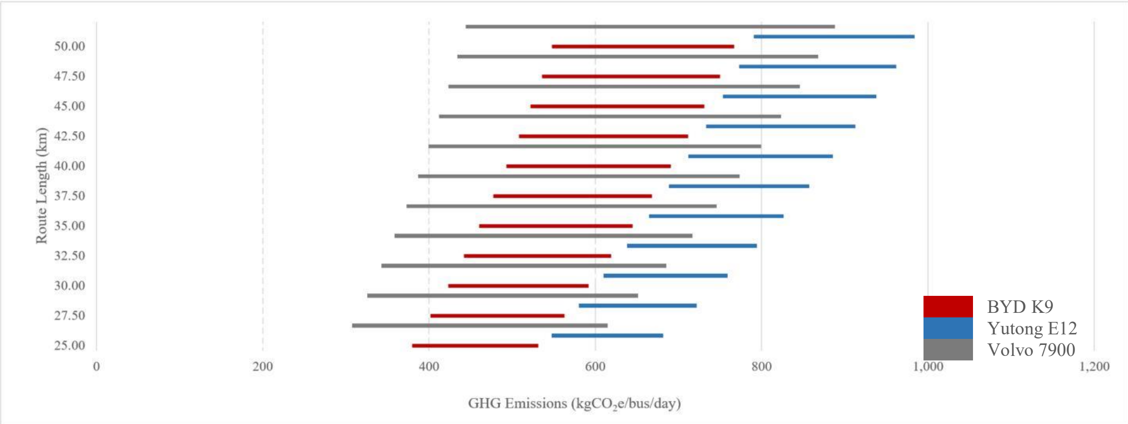


Figure B21 – GHG emissions per individual BEV bus per day in highway traffic conditions.

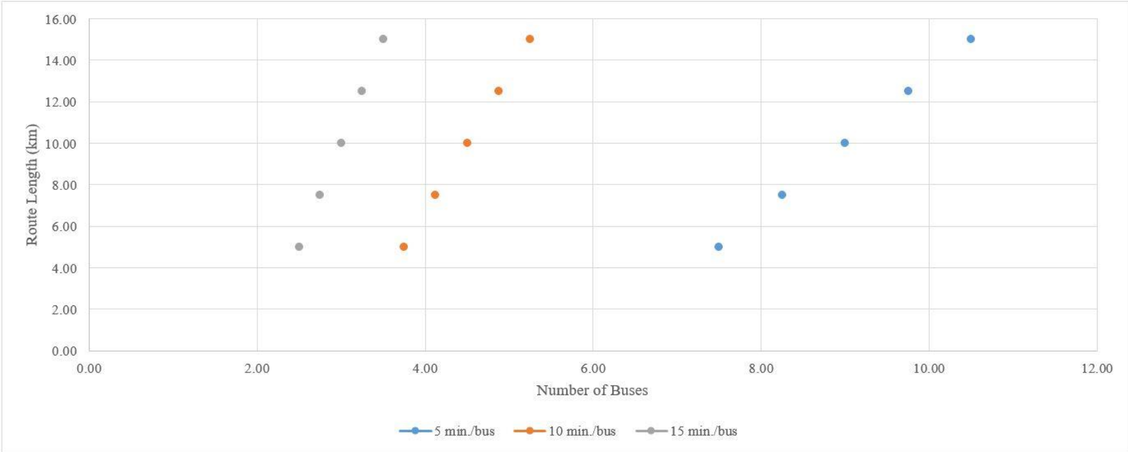


Figure B22 – Required number of BEV buses with respect to route length and service frequency in urban traffic conditions.

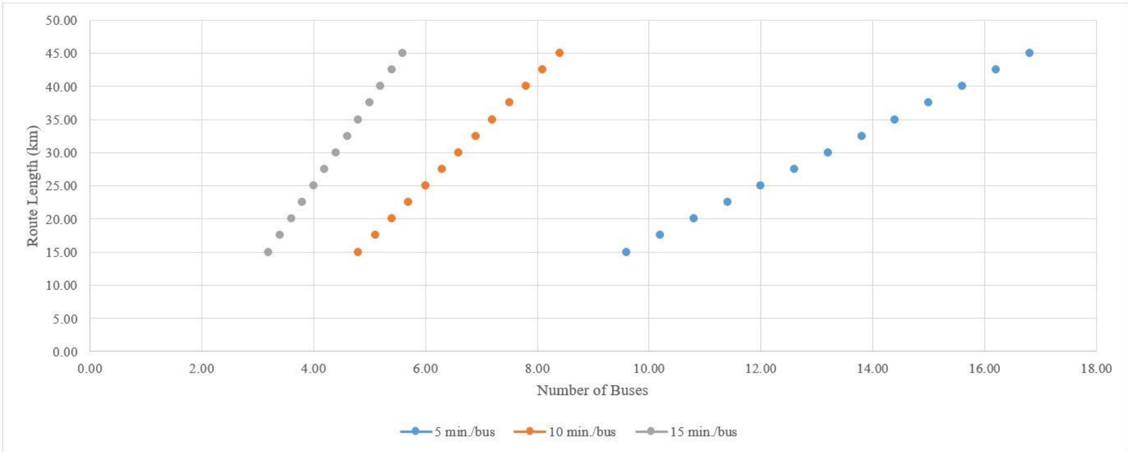


Figure B23 – Required number of BEV buses with respect to route length and service frequency in suburban traffic conditions.

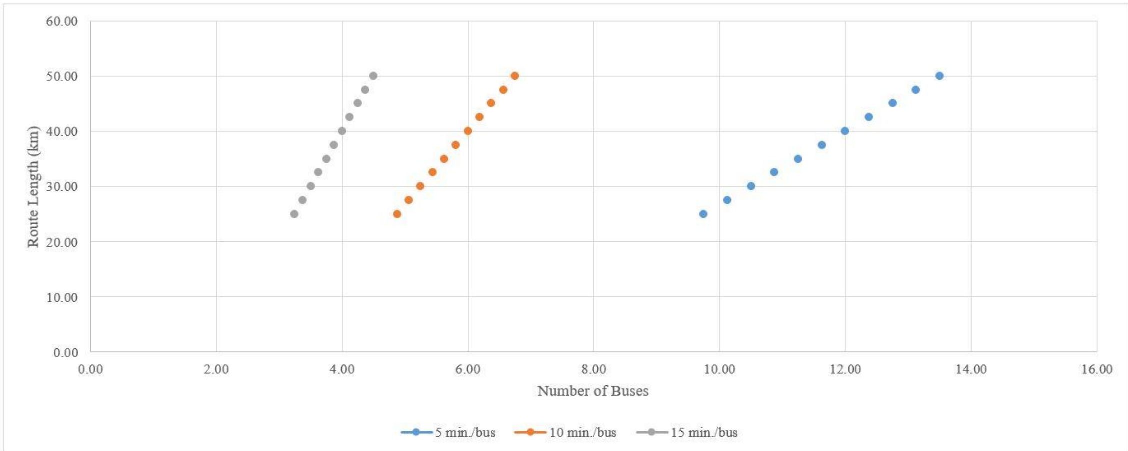


Figure B23 – Required number of BEV buses with respect to route length and service frequency in highway traffic conditions.

7.3 APPENDIX C: LCA OF BUS PRODUCTION

Table C1 – Emissions intensity per unit of material weight.

Emissions/Material (kg/ton)	CO_2	CO	NOx	SOx	CH_4	N_2O
Aluminium (Recycled)	1,545.21	0.95	1.58	2.80	4.06	0.04
Aluminium (Virgin)	7,085.32	2.72	5.86	29.30	12.63	0.11
Battery Management System	21,600.00	10.32	17.72	28.76	50.44	0.43
Cast Iron	498.87	0.70	1.19	3.33	3.60	0.00
Copper	2,570.56	2.30	6.05	131.84	5.13	0.05
Epoxy Resin	5,059.20	4.22	7.30	9.46	19.55	0.16
Ethylene Glycol	2,870.23	1.72	4.25	11.23	7.12	0.06
Nylon 66	5,180.07	4.83	7.58	10.56	17.96	0.76
Fiberglass Composites	5,244.69	3.45	7.33	9.51	16.22	0.13
Fluids & Lubricants (ICEV)	2,241.97	1.11	4.31	13.16	4.80	0.05
Fluids & Lubricants (HEV)	2,194.24	1.12	4.20	12.80	4.83	0.05
Fluids & Lubricants (BEV)	801.86	0.98	1.32	3.75	3.92	0.03
Glass	1,065.89	0.59	1.61	1.09	2.19	0.02
Lead (Recycled)	420.77	0.11	0.56	5.91	0.62	0.00
Lead (Virgin)	542.40	0.62	1.10	27.66	6.62	0.01
Lithium-ion Battery	3,836.77	2.39	6.00	42.72	8.34	0.07
Magnesium	7,842.68	4.28	7.15	6.94	18.64	0.17
Polyethylene	1,965.78	4.82	3.18	23.79	25.16	0.09
Polyurethane Flexible Foam	2,543.84	6.42	4.65	15.21	18.04	0.08
Rare Earth	14,990.00	5.37	10.52	25.86	29.74	0.23
Rubber	3,283.33	2.03	4.56	12.47	6.97	0.81
Stainless Steel	1,596.87	5.53	1.33	2.83	3.51	0.03
Steel (Recycled Production)	1,159.97	3.65	0.97	1.97	2.59	0.02
Steel (Virgin Production)	2,623.88	21.98	2.59	10.72	4.35	0.02
Zinc	2,400.66	0.93	1.81	3.89	4.80	37.87

Source: GREET® model (2019).

Table C2 – BEV battery production emissions.

Studies	Production Emissions (kgCO2e/kWh)
Emilsson & Dahllöf (2019)	61 – 106
Philippot et al. (2019)	123
Hao et al. (2017a)	96 – 127
Messagie (2017)	56
Romare & Dahllöf (2017)	150 – 200
Wolfram & Wiedmann (2017)	106
Ambrose & Kendall (2016)	194 – 494
Ellingsen, Singh & Strømman (2016)	157
Kim et al. (2016)	140
Peters et al. (2016)	110
Nealer, Reichmuth & Anair (2015)	73
Majeau-Bettez, Hawkins & Strømman (2011)	200 – 250

Table C3 – Battery bill of materials.

Components (%)	HEV	BEV
Active Material	15.5%	23.8%
Graphite/Carbon	9.1%	13.8%
Binder	1.3%	2.0%
Copper	24.3%	10.4%
Wrought Aluminium	20.1%	23.1%
Electrolyte ($LiPF_6$)	1.9%	2.5%
Electrolyte (Ethylene Carbonate)	5.4%	6.8%
Electrolyte (Dimethyl Carbonate)	5.4%	6.8%
Plastic (Polypropylene)	2.0%	1.0%
Plastic (Polyethylene)	0.5%	0.3%
Plastic (Polyethylene Terephthalate)	0.3%	0.2%
Steel	1.4%	0.7%
Thermal Insulation	0.7%	0.5%
Coolant (Glycol)	5.7%	5.1%
Electronic Parts	6.4%	3.0%

Table C4 – Emissions from production.

Emissions (kg)	ICEV		HEV		BEV	
	Virgin	Recycled	Virgin	Recycled	Virgin	Recycled
CO_2	34,824.05	21,560.72	36,165.31	23,827.15	46,929.61	34,486.53
CO	161.10	37.98	154.24	37.94	155.93	44.06
NO_x	39.62	26.11	42.48	29.87	58.64	46.07
SO_x	142.47	66.87	207.09	137.08	366.03	295.96
CH_4	84.35	67.05	86.53	70.62	108.24	92.04
N_2O	1.03	0.98	1.07	1.02	1.27	1.22

Table C5 – Emissions from assembly.

Emissions (kg)	ICEV	HEV	BEV
CO_2	8,336.79	8,139.86	9,452.74
CO	5.39	5.27	6.12
NO_x	8.20	8.01	9.30
SO_x	7.62	7.44	8.64
CH_4	22.61	22.07	25.63
N_2O	0.21	0.20	0.23

Table C6 – Emissions intensity from transportation.

Source	Emissions Intensity ($kgCO_2e$/tonne-km)
OECD (2008)	0.013
COSCO Group[11]	0.013

[11] Abbasov et al. (2019)

ONE (Ocean Network Express)[11]	0.015
MSC[11]	0.020
CMA CGM Group[11]	0.020
Evergreen Line[11]	0.021
Hapag-Lloyd[11]	0.021
Yang Ming[11]	0.021
APM-Maersk[11]	0.022
UniFeeder[11]	0.039
X-Press Feeders Group[11]	0.043
Average	0.023
Port of Gothenburg to Port of Sydney	26,237.28 km (14,167 NM)
Port of Shenzhen to Port of Sydney	9,587.80 km (5,177 NM)

Table C7 – Emissions from maintenance.

Emissions (kg)	ICEV	HEV	BEV
CO_2	21,475.13	22,330.47	20,757.19
CO	33.82	34.34	33.52
NO_x	30.02	31.34	29.39
SO_x	90.28	98.73	86.04
CH_4	44.42	46.01	42.86
N_2O	3.81	3.83	3.80

Legend for *Figure C1*. Materials manufacturing method, Service Life*, Maintenance Frequency

A	Recycled, Short, Infrequent	J	Virgin, Long, Infrequent
B	Recycled, Base, Infrequent	K	Recycled, Short, Frequent
C	Recycled, Short, Base	L	Virgin, Base, Base
D	Recycled, Long, Infrequent	M	Virgin, Long, Base
E	Recycled, Base, Base	N	Recycled, Base, Frequent
F	Virgin, Short, Infrequent	O	Virgin, Short, Frequent
G	Recycled, Long, Base	P	Recycled, Long, Frequent
H	Virgin, Base, Infrequent	Q	Virgin, Base, Frequent
I	Virgin, Short, Base	R	Virgin, Long, Frequent

*Short: 500,000 km; Base: 650,000 km; Long: 800,000 km

For the HEV and BEV bus, one battery replacement is assumed in the base case scenario. An additional battery replacement is incorporated for scenarios with long service life and/or frequent maintenance frequency. This study assumes no more than two battery replacements.

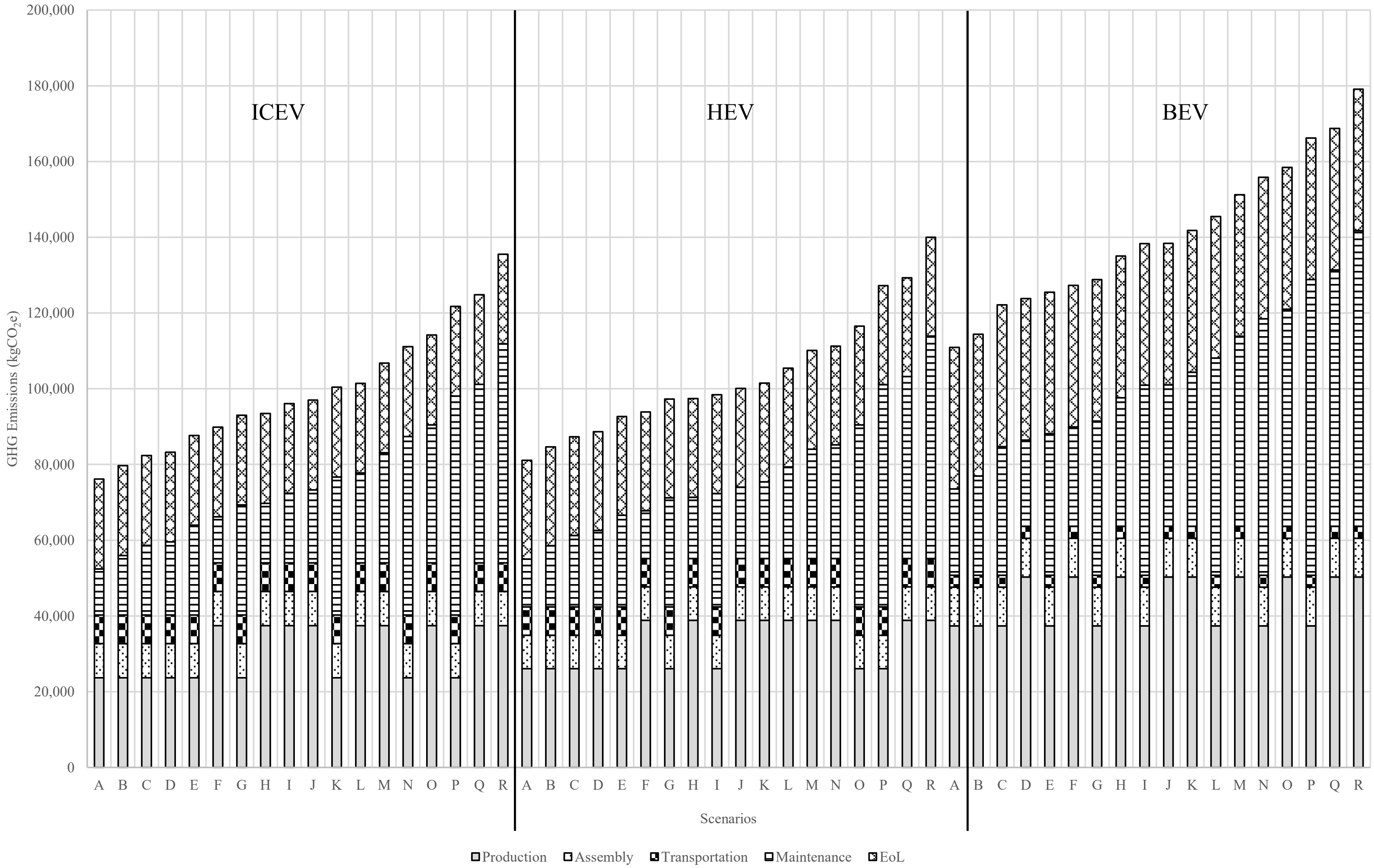


Figure C1 – Sensitivity of GHG emissions to the key parameters.

7.4 APPENDIX D: LCA OF LITHIUM-ION BATTERIES

Table D1 – LCA studies on LIB batteries assessed in the literature review.

ER: Energy Reference; BC: Battery Capacity (kWh); BW: Battery Weight (kg); EC: Energy Consumption (kWh/100km); SE: Specific Energy (kWh/kg); EE: Charge-Discharge Efficiency (%); CC: Lifetime Charge Cycles; SS: Solid State; HT: Hydrothermal; "-": no value given.

Author(s)	Year	Location	Approach	LIB Chem	ER	LCI Data Source	BC	BW	EC	SE	EE	CC
Sun et al.	2020	Beijing, China	Process-Based	NMC622-C	China	Cell: Primary data (industry) Materials: Primary data (industry), Ecoinvent 3.0, GREET 2018 Assembly: Primary data (industry)	72.5	630.0	-	0.12	-	2,000
Kelly, Dai & Wang	2020	Illinois, USA	Process-Based	NMC111-C	USA	Cell: GREET 2018 (Primary data) Materials: GREET 2018 (Primary data) Assembly: GREET 2018 (Primary data)	27.0	188.7	-	-	-	-
Kallitsis et al.	2020	London, UK	Process-Based	NMC333-C NMC333-SiC NMC622-SiC NMC811-SiC	South Korea Norway China	Cell: Majeau-Bettez et al. (2011), Dai et al. (2019) Materials: Ecoinvent 3.5, Dai et al. (2019) Assembly: Ellingsen et al. (2014), Dai et al. (2019)	26.6 40.9 46.2 52.9	253.0	-	0.11 0.16 0.18 0.21	-	-
Raugei & Winfield	2019	Wheatley, UK	Process-Based	LCP-C	Average European	Cell: Primary data (industry), BatPac Materials: Primary data (industry), Ecoinvent 3.3 Assembly: BatPac	17.0	108.0	-	0.16	-	-
Cusenza et al.	2019	Ispra, Italy	Process-Based	LMO-NMC	Japan Average European	Cell: Primary data (industry), Kim et al. (2016) Materials: Notter et al. (2010), Majeau-Bettez et al. (2011), Ellingsen et al. (2014), Kim et al. (2016) Assembly: Ellingsen et al. (2014)	11.4	175.0	19.2	0.07	95	-
Yin, Hu & Yang	2019	Beijing, China	Top-Down	NMC333-C NCA-C LMO-C LFP-C	China	Cell: N/A Materials: Primary data (industry), Ecoinvent 3.0, GaBi 6.0 (Cell materials only) Assembly: N/A	-	-	-	-	-	-
Philippot et al.	2019	Brussel, Belgium	Process-Based	NCA-C	Korea	Cell: BatPac 3.1 Materials: Primary data (Samsung) Assembly: Ellingsen et al. (2014)	20.0	154.0	-	0.25	-	-
Deng et al.	2019	Suzhou, China Wisconsin, USA Ohio, USA	Process-Based	NMC-SiNT NMC-C	USA	Cell: Primary data (industry), BatPac 2015 Materials: Primary data (industry), Li et al. (2014), Ellingsen et al. (2014) Assembly: Primary data (industry), Ellingsen et al. (2014) Disposal: Hawkins et al. (2013), Dunn et al. (2014)	63.0 66.0	320.0 417.0	19.7 20.6	0.20 0.16	93	-
Dai et al.	2019	Illinois, USA	Process-Based	NMC111-C	USA	Cell: GREET 2018 Materials: BatPac 2018 Assembly: GREET 2018	23.5	165.0	-	0.14	-	-

Ioakimidis et al.	2019	Mons, Belgium	Process-Based	LFP-LTO	Spain	Cell: Majeau-Bettez et al. (2011) Materials: Majeau-Bettez et al. (2011) Assembly: N/A	24.0	-	-	0.13	80	4,000
Yu et al.	2018	Xiamen, China	Process-Based	LFP-C	China	Cell: Primary data (industry), GaBi 6.0 Materials: Primary data (industry), GaBi 6.0 Assembly: GaBi 6.0, Zackrisson, Avellán & Orlenius (2010), Li (2015)	39.4	218.8	15.0	0.18	95	-
				NMC-C			27.0	150.2	13.0	0.18	99	
Dai et al.	2018	Illinois, USA	Process-Based	NMC442	China	**Cathode materials production only.** Cell: Primary data (industry) Materials: Primary data (industry) Assembly: Primary data (industry)	-	-	-	-	-	-
				NMC111								
				NMC622								
				NMC811								
				NCA								
				LCO								
Wu & Kong	2018	Guangzhou, China	Process-Based	NMC-C	-	Cell: Ecoinvent 3.3, Ellingsen et al. (2014) Materials: Ellingsen et al. (2014), Li et al. (2014), Zackrisson et al. (2016) Assembly: Ellingsen et al. (2014), Li et al. (2014), Zackrisson et al. (2016)		790.0		0.13		2,000
				NMC-Li			100.0	470.9	-	0.21	-	-
				NMC-SiNWs				611.5		0.16		-
de Souza et al.	2018	Minas Gerais, Brazil	Process-Based	Li-Ion	Brazil	Cell: Majeau-Bettez et al. (2011) Materials: Majeau-Bettez et al. (2011) Assembly: Primary data (industry)	24.0	300.0	14.0	0.08	-	-
Giordano, Fischbeck & Matthews	2018	Pennsylvania, USA	Process-Based	NCA-C	Average European	Cell: BatPac 2.2, GREET 2014, Ecoinvent 3.0 Materials: BatPac 2.2, GREET 2014, Ecoinvent 3.0 Assembly: BatPac 2.2, GREET 2014, Ecoinvent 3.0	23.4 46.8 70.2	158.3 316.5 474.8	-	0.15	89	-
				NMC333-C			23.4 46.8 70.2	164.6 329.1 493.7		0.14		
				NMC441-C			23.4 46.8 70.2	151.2 302.4 453.6		0.15		
				LFP-C			23.4 46.8 70.2	221.3 442.5 663.8		0.11		
				LMO-C			23.4 46.8 70.2	199.7 399.3 599.0		0.12		
Zackrisson, M.	2017	Mölndal, Sweden	Top-Down	LFP-Li	Western European	Cell: Primary data (laboratory), Ellingsen et al. (2014) Materials: Ecoinvent 3.1 Assembly: Ellingsen et al. (2014)	25.2	173.0	18.1	0.15	80	2,000
				LFP-Li			84.0	583.0	23.6	0.14		
				LFP-C			76.0	599.0	123.0	0.13		
				NMC-Li			25.2	132.0	17.9	0.18		
				NMC-Li			84.0	445.0	22.9	0.19		
				NMC-C			76.0	457.0	123.0	0.17		
Qiao et al.	2017b			NMC-C	China		-	171.0	-	-	-	-

		Beijing, China	Process-Based	LFP-C		Cell: GREET 2016, Primary data (industry) Materials: Brown (1996), Burnham et al. (2006), GREET 2.7, Keoleian et al. (2012), GREET 2016 Assembly: Primary data (industry), Lu et al. (2016)		228.0				
Yuan et al.	2017	Ohio, USA Wisconsin, USA	Process-Based	LMO-C	USA	Cell: Primary data (industry) Materials: BatPac, Ecoinvent 3.0, GaBi 7.3 Assembly: Primary data (industry)	24.0	221.6	-	0.11	-	-
Qiao et al.	2017a	Beijing, China	Process-Based	NMC-C	China	Cell: GREET 2015 Materials: GREET 2015 Assembly: GREET 2015	-	230.0	-	-	-	-
				LFP-C				170.0				
Hao et al.	2017a	Beijing, China	Process-Based	LFP-C	China	Cell: Primary data (industry), Lu (2013) Materials: BatPac 2015, Majeau-Bettez et al. (2011) Assembly: Primary data (industry), Lu (2013)	28.0	230.0	-	0.12	-	-
				NMC-C				170.0	-	0.17		
				LMO-C				210.0	-	0.13		
Wolfram & Wiedmann	2017	Connecticut, USA NSW, Australia	Top-Down	Li-Ion	Australia	Cell: Ecoinvent 3.1, Bauer et al. (2015) Materials: Ecoinvent 3.1, Bauer et al. (2015) Assembly: Ecoinvent 3.1, Bauer et al. (2015)	1.3	30.0	40.0	0.12	-	-
							18.0	150.0	29.0			
							42.0	350.0	15.0			
Vandepaer, Cloutier & Amor	2017	Quebec, Canada	Process-Based	LFP-C	China	Cell: Primary data (industry), Majeau-Bettez et al. (2011), Gaines et al. (2011), Amarakoon, Smith & Segal (2013) Materials: Primary data (industry), Majeau-Bettez et al. (2011), Gaines et al. (2011), Amarakoon, Smith & Segal (2013) Assembly: Primary data (industry), Majeau-Bettez et al. (2011), Gaines et al. (2011), Amarakoon, Smith & Segal (2013)	75.0 6,000	-	-	-	96	5,000
Deng et al.	2017b	Suzhou, China Wisconsin, USA Ohio, USA	Process-Based	NMC-MoS_2	USA	Cell: Primary data (industry), BatPac 2015 Materials: Primary data (industry), Majeau-Bettez et al. (2011), Simon & Weil (2013), Ellingsen et al. (2014) Assembly: Ellingsen et al. (2014)	49.4	380.0	15.4	0.13	96	-
				NMC-C			51.0	485.0	15.9	0.11		
Deng et al.	2017a	Suzhou, China Wisconsin, USA Ohio, USA	Process-Based	Li-S	USA	Cell: Primary data (industry), BatPac 2015 Materials: Primary data (industry), Ecoinvent Assembly: Primary data (industry), Ellingsen et al. (2014) Disposal: Hawkins et al. (2013), Dunn et al. (2014)	61.3	279.0	19.2	0.22	86	-
				NMC-C			63.8	531.0	19.9	0.12		
Ambrose & Kendall	2016	California, USA	Top-Down	NCA-C	USA	Cell: BatPac 3.1 Materials: Primary data (industry), GREET 2014, Dunn et al. (2012) Assembly: Notter et al. (2010), Dunn et al. (2012), Ellingsen et al. (2014)	-	-	-	0.13	90	1,000
				NMC-C						0.12		1,700
				LMO-C						0.11		685
				LFP-C						0.10		3,200
				LMO-LTO						0.07		5,000
Lu et al.	2016	Beijing, China	Process-Based	NMC-C	China	Cell: Primary data (industry) Materials: Dunn et al. (2012) Assembly: Primary data (industry)	20.0	125.0	-	0.16	-	-
				LFP-C				167.0		0.12		
				LCO-C				118.0		0.17		
Wang et al.	2016	Beijing, China	Process-Based	L(R)NMO-C NMC-C	China	Cell: Ecoinvent 3.0 Materials: Ecoinvent 3.0 Assembly: Ecoinvent 3.0	16.0	84.7	16.0	0.19	90	1,600

Tagliaferri et al.	2016	London, UK	Process-Based	NMC-C	Average European	Cell: Majeau-Bettez et al. (2011), Ellingsen et al. (2014) Materials: Majeau-Bettez et al. (2011), Ellingsen et al. (2014) Assembly: Majeau-Bettez et al. (2011), Ellingsen et al. (2014)	24.0	214.0	17.0	0.11	-	3,000
								250.0		0.11		
Zackrisson, M	2016	Mölndal, Sweden	Top-Down	LFP-Li	Average European	Cell: Primary data (laboratory), Zackrisson, Avellán & Orlenius (2010), Dunn et al. (2012) Materials: Ecoinvent 3.1 Assembly: Zackrisson, Avellán & Orlenius (2010)	2E-2	0.2	0.2	0.11	90	4,000
Ellingsen t al.	2016	Trondheim, Norway	Top-Down	NMC-C	Average European	Cell: Primary data (industry), Ellingsen et al. (2014) Materials: Ellingsen et al. (2014) Assembly: Ellingsen et al. (2014) Disposal: Dewulf et al. (2010), Hawkins et al. (2012)	17.7	177.0	13.3	0.10	95	-
							24.4	253.0	14.3	0.10		
							42.1	393.0	16.9	0.11		
							59.9	553.0	18.9	0.11		
Troy et al.	2016	Jülich, Germany	Process-Based	LCO-Li (SS)	Germany	Cell: Primary data (industry) Materials: GaBi 6.0 Assembly: Primary data (industry), Ellingsen et al. (2014)	1.8E-4	4.2E-3	-	0.04	-	-
Kim et al.	2016	Michigan, USA	Process-Based	LMO/NMC-C	Korea	Cell: Ecoinvent 3.1 Materials: Primary data (industry), GREET 2014 Assembly: Ecoinvent 3.1	24.0	303.0	19.9	0.08	-	-
				LCO-C						0.15		1,000
				LCO-C (SS)						0.30		1,000
				LMO-C						0.12		1,000
Lastoskie & Dai	2015	Michigan, USA	Process-Based	LMO-C (SS)	USA	Cell: Primary data (industry), Literature data (multiple) Materials: Primary data (industry), Ecoinvent 2.2 Assembly: Dunn et al. (2012)	-	-	-	0.23	90	1,000
				NMC-C						0.14		1,300
				NMC-C (SS)						0.27		1,300
				NCA-C (SS)						0.22		-
Hammond & Hazeldine	2015	Bath, United Kingdom	-	LCO-C	-	Cell: Rydh & Sandén (2005) Materials: Rydh & Sandén (2005) Assembly: Rydh & Sandén (2005)	30.0	-	-	0.12	90	1,500
				LCO-C (Polymer)						0.14		400
Bauer et al.	2015	Villigen, Switzerland	Process-Based	Li-Ion	Average European	Cell: Notter et al. (2010) Materials: Notter et al. (2010) Assembly: Notter et al. (2010)	50.0	448.0	21.6	0.11	-	-
				NMC-C (SS)				180.0		0.16		
				LMR-NMC (SS)				160.0		0.18		
				LCO-C (SS)				170.0		0.16		
Dunn et al.	2015	Illinois, USA	Process-Based	LCO-C (HT)	USA	Cell: Dunn et al. (2012) Materials: Dunn et al. (2012), BatPac 2011 Assembly: Dunn et al. (2012)	28.0	170.0	13.6	0.16	-	-
				LFP-C (HT)				230.0		0.12		
				LFP-C (SS)				230.0		0.12		
				LMO-C (SS)				210.0		0.13		
Dunn et al.	2014	Illinois, USA		NMC-C	USA		28.0	170.0	13.7	0.16	-	-

			Process-Based	LMR-NMC		Cell: BatPac, Primary data (industry) Materials: Primary data (industry), GREET 2.7, Majeau-Bettez et al. (2011) Assembly: BatPac		170.0		0.16		
				LCO-C (SS)				160.0		0.18		
				LCO-C (HT)				160.0		0.18		
				LFP-C (HT)				230.0		0.12		
				LFP-C (SS)				230.0		0.12		
				LMO-C				210.0		0.13		
Li et al.	2014	Wisconsin, USA	Process-Based	NMC-SiNW	USA	Cell: Primary data (industry) Materials: Primary data (industry), GaBi 6 Assembly: GaBi 6	43.2	120.0	41.2	0.36	90	-
Faria et al.	2014	Coimbra, Portugal	Process-Based	LMO-C	Poland Portugal France	Cell: Notter et al. (2010) Materials: Hischier et al. (2007) Assembly: Notter et al. (2010)	24.0	300.0	0.0	0.11	93	1,500
Ellingsen et al.	2013	Trondheim, Norway	Top-Down	NMC-C	Sweden	Cell: Primary data (industry), Majeau-Bettez et al. (2011), EcoInvent 2.2 Materials: Majeau-Bettez et al. (2011), Hischier et al. (2007) Assembly: Primary data (industry), Notter et al. (2010)	26.6	253.0	14.0	0.11	96	3,000
Hawkins et al.	2013	Trondheim, Norway	Top-Down	NMC-C	Average European	Cell: Majeau-Bettez et al. (2011) Materials: Majeau-Bettez et al. (2011) Assembly: Majeau-Bettez et al. (2011)	24.0	214.0	13.4	0.11	-	-
				LFP-C				273.0		0.09		
Simon & Weil	2013	Ulm, Germany	Top-Down	LFP-C	-	Cell: Notter et al. (2010), Majeau-Bettez et al. (2011) Materials: Zackrisson, Avellán & Orlenius (2010) Assembly: Hischier et al. (2007), Notter et al. (2010), Zackrisson, Avellán & Orlenius (2010), Majeau-Bettez et al. (2011)	21.1	195.0	20.0	0.11	-	-
				NMC-C				175.0		0.12		
Amarakoon, Smith & Segal	2013	USA	Process-Based	LMO-C	USA	Cell: Notter et al. (2010), Majeau-Bettez et al. (2011), Hawkins et al. (2013) Materials: Gabi4, Notter et al. (2010), Majeau-Bettez et al. (2011) Assembly: Primary data (industry), Notter et al. (2010), Majeau-Bettez et al. (2011)	25.8	-	-	0.09	85	1,053
				NMC-C								
				LFP-C								
McManus, M.	2012	Bath, United Kingdom	Process-Based	LFP-C (NMP solvent)	United Kingdom	Cell: Rydh & Sandén (2005), Samaras & Meisterling (2008), Zackrisson, Avellán & Orlenius (2010) Materials: Hischier et al. (2007) Assembly: Rydh & Sandén (2005), Samaras & Meisterling (2008), Zackrisson, Avellán & Orlenius (2010)	-	-	-	0.17	-	600
				LFP-C (water solvent)								
Aguirre et al.	2012	California, USA	Top-Down	NCA-C	USA	Cell: GREET 1.8b Materials: GREET 1.8b Assembly: GREET 1.8b	24.0	300.0	21.0	0.08	-	-
Sullivan & Gaines	2012	Illinois, USA	-	Li-Ion	USA	Cell: Rydh & Sandén (2005), Notter et al. (2010) Materials: Ishihara, Nishimura & Uchiyama (1999) Assembly: Rydh & Sandén (2005), Notter et al. (2010)	-	-	-	0.11	90	5,100
Dunn et al.	2012a	Illinois, USA	Process-Based	LMO-C	USA	Cell: BatPac 2012 Materials: GREET 2012 Assembly: Primary data (industry)	28.0	210.0	17.5	0.13	-	-
Majeau-Bettez et al.	2011	Trondheim, Norway	Top-Down	NMC-C	Average European	Cell: Primary data (industry), Gaines & Cucenca (2000), Schexnayder et al. (2001) Materials: Gaines & Cucenca (2000), Schexnayder et al. (2001) Assembly: Rydh & Sandén (2005)	13.9	-	19.0	0.11	90	3,000
				LFP-C					14.7	0.09		6,000
Kushnir & Sandén	2011	Göteborg, Sweden	Top-Down	LCO-C	Average European	Cell: Gaines & Nelson (2000), Gaines & Nelson (2009) Materials: N/A Assembly: N/A	-	-	-	0.13	90	950
				LCN-C						0.16		950

				LFP-C						0.10		3,000
				LCN-LTO						0.08		10,000
				LFP-LTO						0.06		8,750
Gaines et al.	2011	Illinois, USA	-	NCA-C	USA	Cell: Gaines & Nelson (2009) Materials: N/A Assembly: N/A	-	75.9	-	-	-	-
Sullivan, Burnham & Wang	2010	Illinois, USA	Top-Down	NCA-C	USA	Cell: Rydh & Sandén (2005) Materials: GREET 2.7 Assembly: Rydh & Sandén (2005), GREET 2.7	-	138.9	-	-	-	-
Notter et al.	2010	Duebendorf, Switzerland	Process-Based	LMO-C	Average European	Cell: Primary data (industry) Materials: Ecoinvent 2.0 Assembly: Primary data (industry)	35.8	300.0	17.0	0.11	80	-
Zackrisson, Avellán & Orlenius	2010	Stockholm, Sweden	Top-Down	LFP-C	Western European	Cell: Gaines & Cucenca (2000) Materials: EcoInvent 2.0, Venugopal et al. (1999), Gaines & Cucenca (2000) Assembly: Saft (2008)	10.0	107.0	16.7	0.09	90	3,000
							1.3	16.0	-		-	
Samaras & Meisterling	2008	Pennsylvania, USA	Top-Down	Li-ion	USA	Cell: Rydh & Sandén (2005) Materials: Rydh & Sandén (2005) Assembly: Rydh & Sandén (2005)	6.7	84.0	22.3	0.08	85	2,500
							13.4	168.0	22.3		80	
							20.1	252.0	22.3		80	
Rydh & Sandén	2005a	Kalmar, Sweden	Top-Down	NCA-C	-	Cell: Primary data (industry) Materials: Primary data (industry) Assembly: Primary data (industry)	450.0	5,050	-	0.10	90	4,000

Table D2 – Results range reported by all LCA studies assessed by this study.

Materials/Parts Manufacturing, Cell Manufacturing, Battery Pack Assembly, Decommissioning, Other: $kgCO_2e/kWh$; GWP_1: Total GHG emissions ($kgCO_2e/kWh$); GWP_2: Total GHG Emissions ($kgCO_2e/kg$); CED: kWh/kg.

Author(s)	Year	Battery Chemistry	Materials/Parts Manufacturing	Cell Manufacturing	Battery Pack Assembly	Decommissioning	Other	GWP_1	GWP_2	CED
Sun et al.	2020	NMC622-C	105.47	19.01	-	-30.91	-	93.57	10.49	32.70
Kelly, Dai & Wang	2020	NMC111-C	48.00	14.00	-	-	12.00	74.00	10.59	46.00
		NMC333-C		104.96				262.41	27.59	
Kallitsis et al.	2020	NMC333-SiC	-	75.48	-	-	-	179.71	29.05	130.20
		NMC622-SiC		67.27				160.17	29.25	
		NMC811-SiC		58.91				140.26	29.33	
Raugei & Winfield	2019	LCP-C	68.10	4.90	3.10	5.80	-	81.90	12.89	45.09
Cusenza et al.	2019	LMO-NMC	312.44			16.26	67.80	396.49	25.83	17.51
		NMC333-C	131.00					131.00	30.82	133.20
Yin, Hu & Yang	2019	NCA-C	81.00	-	-	-	-	81.00	27.65	117.53
		LMO-C	78.00					78.00	6.69	33.13
		LFP-C	82.00					82.00	11.16	53.76
Philippot et al.	2019	NCA-C	77.00	45.00	1.00	-	-	123.00	15.97	16.70
Deng et al.	2019	NMC-SiNT	101.27	96.51	13.94	4.79	-	216.54	-	20.40
		NMC-C	146.06	-	-	6.12	-	152.18		
Dai et al.	2019	NMC111-C	58.32	13.85	-	-	0.73	72.90	10.38	44.90
								1,253.62		
Ioakimidis et al.	2019	LFP-LTO	-	-	-	-	-	1,175.78	-	-
								1,050.23		
Yu et al.	2018	LFP-C	68.56	24.73	-	18.64	1.58	113.50	56.22	14.16
		NMC-C	69.54	24.82		11.65	1.72	107.73	57.39	14.15
		NMC442	0.21						0.21	17.67
		NMC111	0.21						0.21	17.67
Dai et al.	2018	NMC622	0.21	-	-	-	-	-	0.21	17.67
		NMC811	-						-	17.67
		NCA	-						-	17.67

		LCO	0.20						0.20	5.44
		NMC-C	101.21		53.98	-	13.49	168.68	21.35	
Wu & Kong	2018	NMC-Li	66.82		35.64		8.91	111.37	23.65	28.84
		NMC-SiNWs	133.28		71.08		17.77	222.13	36.33	
de Souza et al.	2018	Li-Ion						133.60	10.69	
		NCA-C						184.62	27.29	
								92.31	13.65	
								61.54	9.10	
								184.62	26.25	
		NMC333-C						92.31	13.13	
								61.54	8.75	
								164.10	25.40	
Giordano, Fischbeck & Matthews	2018	NMC441-C	-	-	-	-	-	82.05	12.70	29.09
								54.70	8.47	
								205.13	21.69	
		LFP-C						102.56	10.85	
								68.38	7.23	
								143.59	16.83	
		LMO-C						71.79	8.41	
								47.86	5.61	
		LFP-Li	101.53	5.10			-2.08	106.63	15.53	
		LFP-Li	40.22	2.02			-0.81	42.23	6.08	
Zackrission, M.	2017	LFP-C	319.65	10.62	-	-	-7.67	330.27	41.90	20.72
		NMC-Li	178.41	12.40			-2.80	190.82	36.43	
		NMC-Li	33.12	1.95			-1.07	35.07	6.62	
		NMC-C	263.11	105.45			-9.08	368.56	61.29	
Qiao et al.	2017b	NMC-C	-	-	-	-	-	-	16.94	16.57
		LFP-C							13.45	14.82
Yuan et al.	2017	LMO-C	-	-	-	-	-	-	-	50.17
Qiao et al.	2017a	NMC-C	-	-	-	-	-	-	12.13	-
		LFP-C							17.01	

		LFP-C	103.80	5.50				109.30	13.31	
Hao et al.	2017a	NMC-C	99.90	4.10	-	-	-	104.00	17.13	-
		LMO-C	91.50	5.10				96.60	12.88	
								1,730.77	75.00	
Wolfram & Wiedmann	2017	Li-Ion	-	-	-	-	-	166.67	15.00	-
								89.29	6.43	
								130.73		
Vandepaer, Cloutier & Amor	2017	LFP-C	-	-	-	-	-		-	-
								101.80		
Deng et al.	2017b	NMC-MoS2	205.26	10.28	4.70	7.00	-	227.13	29.53	28.00
		NMC-C	172.16	-	-	8.75		180.78	19.01	
Deng et al.	2017a	Li-S	57.15	86.46	2.75	4.18	-	150.34	33.03	22.80
		NMC-C	172.10	-	-	8.06		180.16	21.65	
		NCA-C	50.50	208.80			1.70	261.00		
		NMC-C	43.61	203.20			7.19	254.00		
Ambrose & Kendall	2016	LMO-C	47.47	206.40	-	-	4.13	258.00	-	-
		LFP-C	40.17	201.60			10.24	252.00		
		LMO-LTO	40.72	208.80			11.49	261.00		
Lu et al.	2016	NMC-C	43.27	-	2.28	-	-	45.55	7.29	30.55
		LFP-C	72.86		2.18			75.04	8.99	34.14
		LCO-C	48.51		2.17			50.68	8.59	33.79
Wang et al.	2016	L(R)NMO-C	-	-	-	-	-	79.81	79.81	57.92
		NMC-C						68.17	68.17	52.46
										15.96
Tagliaferri et al.	2016	NMC-C	-	-	-	-	-	-	-	
										28.22
Zackrission, M	2016	LFP-Li	151.75	63.75		-	-	215.63	23.05	20.72
			124.29			5.65		129.94	15.82	
Ellingsen et al.	2016	NMC-C	131.15	-	-	4.10	-	135.25	15.42	-
			116.39			2.38		118.76	14.50	
			115.19			3.34		118.53	14.29	
Troy et al.	2016	LCO-Li (SS)	-	-	-	-	-	1,045.00	47.62	393.33
Kim et al.	2016	LMO/NMC-C	71.40	63.00	1.70	-	4.10	140.20	11.10	33.60

		LCO-C						50.00		
		LCO-C (SS)						35.00		
		LMO-C						65.00		
Lastoskie & Dai	2015	LMO-C (SS)	-	-	-	-	-	45.00	-	-
		NMC-C						85.00		
		NMC-C (SS)						45.00		
		NCA-C (SS)						40.00		
Hammond & Hazeldine	2015	LCO-C	2.60	-	-	-	2.80	5.40	-	-
		LCO-C (Polymer)								
Bauer et al.	2015	Li-Ion	-	-	-	-	-	52.20	5.83	-
		NMC-C (SS)								43.40
		LMR-NMC (SS)								32.48
		LCO-C (SS)								48.44
Dunn et al.	2015	LCO-C (HT)	-	-	-	-	-	-	-	87.64
		LFP-C (HT)								15.68
		LFP-C (SS)								9.52
		LMO-C (SS)								9.80
		NMC-C								39.56
		LMR-NMC								29.31
		LCO-C (SS)								43.96
Dunn et al.	2014	LCO-C (HT)	-	-	-	-	-	-	-	73.56
		LFP-C (HT)								14.07
		LFP-C (SS)								11.43
		LMO-C								7.62
Li et al.	2014	NMC-SiNW	193.64	4.53	6.26	26.94	37.97	269.34	96.96	25.38
Faria et al.	2014	LMO-C	-	45.63	19.60	16.21	5.69	87.13	6.97	-
Ellingsen et al.	2013	NMC-C	64.70	106.77	0.87	-	-	172.34	18.12	17.25
Hawkins et al.	2013	NMC-C	-	-	-	-	-	192.50	21.59	-
		LFP-C						246.67	21.68	
Simon & Weil	2013	LFP-C	-	-	-	-	-	-	-	14.69
		NMC-C								14.58

		LMO-C	61.50	1.80	0.06			63.36	5.00	17.16
Amarakoon, Smith & Segal	2013	NMC-C	86.70	0.00	34.00	-	-	120.70	-	-
		LFP-C	90.80	60.20	-			151.00	16.00	61.85
McManus, M.C.	2012	LFP-C (NMP solvent)	-	-	-	-	-	6.16	12.50	25.20
		LFP-C (water solvent)							4.40	24.64
Aguirre et al.	2012	NCA-C	-	-	-	-	-	245.00	19.60	89.91
Sullivan & Gaines	2012	Li-Ion	-	-	-	-	-	-	0.56	50.43
Dunn et al.	2012a	LMO-C	37.00	2.10	-	-	-	39.00	5.10	20.83
Majeau-Bettez et al.	2011	NMC-C	56.00	88.00	36.00	-	20.00	200.00	22.00	58.52
		LFP-C	70.00	110.00	45.00		25.00	250.00		57.40
		LCO-C								14.00
		LCN-C								14.56
Kushnir & Sandén	2011	LFP-C	-	-	-	-	-	-	-	17.36
		LCN-LTO								31.92
		LFP-LTO								38.36
Gaines et al.	2011	NCA-C	-	-	-	-	-	-	-	54.98
Sullivan, Burnham & Wang	2010	NCA-C	-	-	-	-	-	-	7.20	25.48
Notter et al.	2010	LMO-C	51.00	0.90	0.60	-	-	53.00	6.00	29.20
Zackrisson, Avellán & Orlenius	2010	LFP-C	179.18	111.18	49.78	-	-	340.00	25.00	20.72
								120.00	9.75	38.68
Samaras & Meisterling	2008	Li-ion	-	-	-	-	-	120.90	9.64	38.00
								120.15	9.58	38.00
								120.40	9.60	38.00
Rydh & Sandén	2005a	NCA-C	-	-	-	-	-	-	-	217.50

Table D3 – Excluded life cycle studies based upon relevance.

Author(s)	Year
Aichberger & Jungmeier	2020
Emilsson & Dahllöf	2019
Yin, Hu & Yang	2019
Bauer et al.	2018
Del Pero, Delogu & Pierini	2018
Hall & Lutsey	2018
Hicks et al.	2018
Moro & Lonza	2018
Peters & Weil	2018
Rupp, Schulze & Kuperjans	2018
Wang et al.	2018
Ellingsen, Hung & Strømman	2017
Messagie, M.	2017
Mierlo, Messagie & Rangaraju	2017
Peters et al.	2017
Romare & Dahllöf	2017
Lajunen & Lipman	2016
Egede et al.	2015
Kabakian, McManus & Harajli	2015
Nealer & Hendrickson	2015
Nealer, Reichmuth & Anair	2015
Oliveira et al.	2015
Nordelöf et al.	2014
Cooney, Hawkins & Marriott	2013
Genikomsakis et al.	2013
Lu et al.	2013
Gerssen-Gondelach & Faaij	2012
Hawkins, Gausen & Strømman	2012
Matheys et al.	2009
Bossche et al.	2006
Rydh & Sandén	2005b
Castro, Remmerswaal & Reuter	2003
Schexnayder et al.	2001

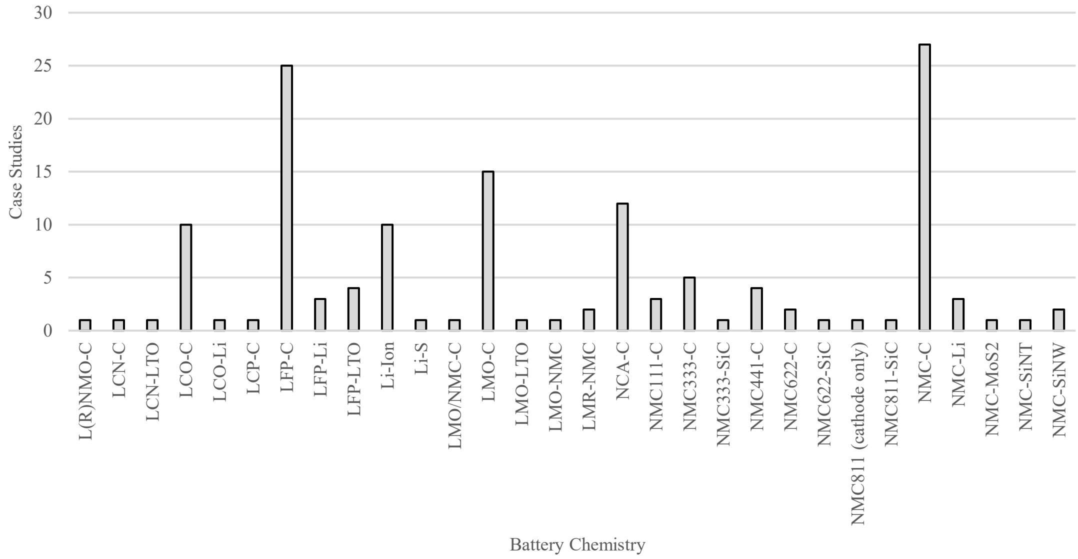


Figure D1 – Number of case studies with respect to battery chemistry.

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