



Quantifying Environmental and Resource Impacts of the Future UK Hydrogen-Fuelled Vehicle Fleet

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Abstract

The demand for hydrogen vehicles is increasing as part of the drive to decarbonise transportation, particularly in sectors where battery electric vehicles (BEVs) face limitations. Since hydrogen-powered vehicles are zero-emission at the tailpipe, the sustainability of these technologies depends on upstream supply chain impacts, including critical material needs, manufacturing emissions, and end-of-life recovery. To deploy these technologies sustainably, it is essential to understand both the current and future environmental impacts of the vehicles and their fuel sources. This thesis provides a system-level assessment of hydrogen mobility across three scales: the production of proton exchange membrane (PEM) electrolyzers, vehicle-level comparisons of hydrogen and electric powertrains, and fleet-level deployment scenarios in the UK through 2050.

This thesis evaluates the lifecycle greenhouse gas (GHG) emissions, cost and material demands of hydrogen technologies. On the production side, we focus on proton exchange membrane (PEM) electrolyzers for renewable, green hydrogen. While the vehicle and fleet analysis consider fuel cell electric vehicles (FCEVs), fuel cell range extended vehicles (FCREx) and hydrogen internal combustion engines (H_2 -ICEV).

Our results reveal a critical material bottleneck in the production of green hydrogen. Even with rapid reductions in iridium catalyst loading and 95% closed-loop recycling, cumulative demand for iridium is projected to reach 526–1,100 tonnes by 2050 - far exceeding current global supply capacity (7 t yr^{-1}). In terms of GHG emissions, manufacturing waste emerges as a dominant source in PEM systems, contributing up to $56 \text{ tCO}_2\text{eq/MWe}$ in 2022. Without recycling, emissions increase by 14-fold, underscoring the essential role of circular economy strategies. At the vehicle level, BEVs consistently demonstrate the lowest lifecycle emissions and cost, with projected costs of $\text{€}0.26\text{--}0.28$ per mile by 2050, compared to $\text{€}0.39\text{--}0.45$ for hydrogen vehicles. However, if deployed today, hybridised hydrogen vehicles such as FCREx would be cost-competitive with internal combustion engine vehicles due to reduced hydrogen consumption. In addition, sensitivity analysis indicates that hydrogen FCEVs can be more economical than BEVs without access to home charging due to the high cost of public electricity. Fleet-scale modelling confirms these trends. The Major H_2 scenario (30% hydrogen fleet market in 2050) has a 6% increase in cumulative GHG emissions and lifecycle costs compared to the BEV-dominant Base scenario, driven by higher hydrogen production and vehicle component costs. We find that early deployment of H_2 -ICE vehicles introduces efficiency penalties, with the Transitional scenario (7% hydrogen fleet in 2050) showing the highest energy demand despite lower hydrogen uptake. Finally, improvements in the manufacturing

phase such as a 42–45% reduction in fuel cell and tank production costs translate to just a 0.5% reduction in total cumulative cost, highlighting the dominance of upstream fuel production in shaping hydrogen vehicle economics.

This thesis advances current understanding by demonstrating that hydrogen’s role in transport decarbonisation relies not just on technological readiness but also on material supply chains, low-carbon production methods, and user behaviour. It highlights that circular economy approaches, though vital, need to be complemented with alternative technologies and better material efficiency to address systemic challenges.

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List of abbreviations

Table 1: List of abbreviations used throughout the thesis

Name	Abbreviation
AE	Alkaline electrolyser
BAU	Business as usual
BEV	Battery electric vehicle
BoP	Balance of plant
CCS	Carbon capture and storage
CFRP	Carbon fibre-reinforced plastic
CO ₂	Carbon dioxide
CO ₂ eq	Carbon dioxide equivalent
DOE	Department of Energy
EoL	End-of-life
EoL-rr	End-of-life recycling rate
EU	European Union
FCEV	Fuel cell electric vehicle
FCREx	Fuel cell range extended vehicle
GGE	Gasoline gallon equivalent
GHG	Greenhouse gases
REET	Greenhouse gases, Regulated Emissions, and Energy use in Technolo- gies
Gt	Gigatonnes
GW	Gigawatt
GWP	Global warming potential
H ₂	Hydrogen
HEV	Hybrid electric vehicle
ICEV	Internal combustion engine vehicle
ICEV-H ₂	Hydrogen internal combustion engine vehicle

Continued on next page

Table 1 continued from previous page

Name	Abbreviation
IEA	International Energy Agency
IPCC	Intergovernmental Panel on Climate Change
Ir	Iridium
kgCO ₂ eq	Kilograms of carbon dioxide equivalent
km	Kilometre
kW	Kilowatt
kWh	Kilowatt hour
L	Litre
LCA	Life cycle assessment
LCC	Life cycle cost
LCHS	Low carbon hydrogen standard
LCI	Life cycle inventory
LDV	Light-duty vehicle
MFA	Material flow analysis
MJ	Megajoule
Mt	Million tonnes
MtCO ₂ eq	Million tonnes of carbon dioxide equivalent
MW	Megawatt
NO _x	Nitrogen oxides
PEM	Proton exchange membrane
PGM	Platinum group metal
PHEV	Plug-in hybrid electric vehicle
Pt	Platinum
RED	Renewable Energy Directive
SMR	Steam methane reforming
SOE	Solid oxide electrolyser

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Table 1 continued from previous page

Name	Abbreviation
SUV	Sports utility vehicle
T&D	Transmission and distribution
TCO	Total cost of ownership
tkm	Tonne kilometre
TT	Tube trailer
UK	United Kingdom
US	United States
VAT	Value added tax
VKT	Vehicle kilometres travelled
VMT	Vehicle miles travelled
WTW	Well-to-wheel
ZEV	Zero emission vehicle

Chapter 1

Introduction

1.1 Background

Since the early 20th century, energy consumption has grown 80 times (Sharma et al., 2021). The combustion of fossil fuels continues to be the predominant way of fulfilling this unprecedented energy demand and currently more than 80% of the world's primary energy supply consists of coal, oil, and natural gas. Fossil fuel dependence comes with environmental challenges such as increased greenhouse gas (GHG) emissions, air pollution, resource depletion and climate change. The Intergovernmental Panel on Climate Change (IPCC) reported that the observed increase in global temperatures since 1850 is a result of human-induced CO₂ emissions (Allen et al., 2018). Without action, temperatures could rise above 4°C by 2100, causing irreversible damage such as rising sea levels, extreme weather events and loss of biodiversity (Allen et al., 2018). Global efforts, including the Paris Agreement, seek to *"keep warming below 2°C and ideally limit it to 1.5°C"* (Allen et al., 2018). Among energy-related sectors, transport is the second-largest producer of global CO₂ emissions, producing nearly 20% of global GHGs (Ritchie, 2020). Passenger cars alone produce over 45% of transport-related emissions. A similar trend is seen in the UK where transport is the largest emitting sector, responsible for 28% of all GHG emissions (Department for Transport (DfT), 2024). Within this sector, cars and taxis make up 52% of emissions, emphasising the need to focus on passenger vehicle decarbonisation (Department for Transport (DfT), 2023). The UK's legally binding target to net-zero emissions by 2050 (Climate Change Committee, 2025) puts significant pressure on the transport sector to deliver rapid and sustained emissions reductions. Meeting the target of net-zero emissions by 2050 requires not only the widespread adoption of zero-emission vehicles (ZEVs) but also the decarbonisation of upstream energy

supply chains, including electricity and hydrogen (Sims et al., 2014).

The Role of Hydrogen. Hydrogen is regarded as a ‘pillar’ of future net-zero energy systems (Kouchaki-Penchah et al., 2024), due to its versatility and potential to decarbonise hard-to-abate sectors. As a carbon-free energy carrier, hydrogen can be stored, transported, and converted into electricity or heat, making it suitable for a wide range of applications: from industrial processes, heating, to long-duration energy storage. Therefore, the system-level flexibility of hydrogen makes it an important part of the low-carbon energy transition, especially when integrated with renewable energy sources.

However, the climate benefits of hydrogen depend on its production method. Currently, about 96% of the world’s hydrogen supply comes from fossil fuels, mainly through steam methane reforming (SMR) without carbon capture (grey hydrogen). To meet climate goals, a rapid shift towards low-carbon production methods is essential. These include green hydrogen, produced via electrolysis powered by renewable electricity and blue hydrogen, produced from SMR of natural gas with carbon capture and storage (CCS). Globally, hydrogen deployment is increasing. By 2020, 15 countries had published national hydrogen strategies and global electrolyser capacity reached 1.2 GW in 2022, with planned new developments expected to exceed 150 GW by 2030 (International Energy Agency (IEA), 2022a,b). However, to achieve global net-zero emissions, the IEA estimates that electrolyser capacity must reach 850 GW by 2030 and 3,670 GW by 2050, a 3,000-fold increase (International Energy Agency (IEA), 2022a,b).

Among electrolyser technologies, proton exchange membrane (PEM) electrolyzers are widely regarded as a key production pathway for green hydrogen, offering high efficiency and rapid response to renewable energy variability (Chandrasekar et al., 2021). These systems are expected to dominate up to 65% of the future electrolyser market by 2050 (International Energy Agency (IEA), 2022a), driven by their compatibility with intermittent renewables and suitability for decentralised deployment. However, PEM electrolyzers rely on platinum group metals (PGMs) for catalysis - particularly iridium (Ir) and platinum (Pt) - which are costly, geologically scarce, and environmentally intensive to extract. Current development efforts aim to reduce system costs and resource intensity by lowering PGM content or substituting alternative catalysts such as nickel, cobalt, or iron oxides through nano-structuring and advanced material design (Miller et al., 2022).

This raises a critical research question: how sustainable is large-scale green hydrogen production when considering material supply and resource dependency? As electrolyzers are deployed at an accelerating rate, so too will the demand for the materials in these systems. Some of the metals used in PEM systems are classified as

critical raw materials, such as PGMs, by the European Commission due to their economic importance and supply vulnerability. Iridium, for instance, is one of the rarest elements in Earth’s crust and is mainly sourced as a by-product of platinum mining, with limited global production capacity. These materials are often concentrated in regions with complex supply chains, limited recycling infrastructure, and significant environmental and social impacts associated with extraction. Understanding the lifecycle implications and supply risks of these materials is essential to ensuring the sustainability of hydrogen production beyond carbon emissions.

Hydrogen in Transport. To date, decarbonisation efforts in transport have focused primarily on BEVs, which eliminate tailpipe emissions and reduce urban air pollution (Ellingsen et al., 2014). In 2024, over 382,000 BEVs were newly registered in the UK - 19% of all new car sales (Department for Transport (DfT), 2025a). In contrast, fewer than 300 hydrogen-powered vehicles were in use (DNV, 2023). Despite BEVs’ rapid growth, several challenges affect their adoption. ‘*Range anxiety*’ affects 78% of prospective buyers (ChargeSmart EV, 2025) and infrastructure remains uneven, with rural areas hosting only 35 chargers per 100,000 residents compared to 250 in London (Department for Transport (DfT), 2025a). Additional concerns include the environmental footprint of lithium-ion battery production, especially in carbon-intensive regions (Llamas-Orozco et al., 2023), grid stress from accelerated BEV uptake (Acharige et al., 2025), and high energy demands during battery manufacturing (Funke et al., 2019).

These limitations highlight the need for complementary low-emission technologies. Hydrogen-powered vehicles - including fuel cell electric vehicles (FCEVs), hydrogen internal combustion engines (H_2 -ICE), and fuel cell range extenders (FCREx) offer alternative pathways for decarbonising vehicles where BEVs face inherent constraints. However, for novel technologies (H_2 -ICE and FCREx) their lifecycle impacts, material demands, and cost competitiveness remain under-explored.

Quantifying Environmental, Resource use and Cost implications. The adoption of hydrogen-powered mobility introduces a series of sustainability questions. What are the environmental impacts of producing green hydrogen at scale? How much critical material is required to manufacture PEM electrolyzers, and can global supply chains meet this demand sustainably? How do hydrogen-powered vehicles compare to battery electric and conventional vehicles in terms of lifecycle emissions, resource use, and cost? And what are the implications of deploying hydrogen technologies across an entire national fleet?

Addressing these questions requires a systems-level approach that captures the full lifecycle of hydrogen technologies (from raw material extraction and manufacturing to use and end-of-life) (International Organization for Standardization (ISO),

2006). While Life Cycle Assessment (LCA) is widely used to evaluate environmental impacts, existing studies often focus on isolated components (e.g., fuel cells), specific fuel pathways (e.g., (Sinha and Brophy, 2021)), or single-vehicle comparisons e.g., (Lu et al., 2022) and frequently omit end-of-life stages or assume static supply chain conditions (Koj et al., 2024). These limitations hide the dynamic nature of hydrogen deployment, where fuel composition, technology mix, and infrastructure are continually developing.

Dynamic LCA approaches provide a more nuanced perspective by taking into account time-dependent changes in energy systems, technology performance, and material flows (Garcia and Freire, 2017; Milovanoff et al., 2019; Shui et al., 2024). However, hydrogen mobility remains under-explored within this class of models, especially in terms of material demand and cost scalability. To address this gap, material flow analysis (MFA) is used to quantify the material resource demands of hydrogen technologies, particularly critical metals for PEM electrolyzers. MFA enables the tracking of stock accumulation, supply constraints, and end-of-life recovery potential, all of which are crucial factors for evaluating long-term sustainability. This is especially relevant PGMs, which are geologically scarce, environmentally intensive to extract, and concentrated in geopolitically sensitive regions.

Life cycle costing (LCC) brings in an economic perspective, taking into account purchase costs, fuel and maintenance expenses, and operational factors that change with policy, market maturity, and technology advancements. When combined with environmental and material assessments, LCC helps to show trade-offs between financial viability and environmental performance. Together, these methods support a scenario-based evaluation of hydrogen’s role in future low-carbon mobility, quantifying emissions, resource demands and cost implications across both product-level and fleet-level deployment pathways.

1.2 Research Questions and Objectives

To address these gaps, this research develops a dynamic, scenario-based framework to explore the sustainability of hydrogen-powered mobility across multiple scales. It aims to quantify the environmental impacts of hydrogen production and use, assess the material demands and supply risks associated with key technologies and evaluate the long-term cost implications of hydrogen vehicle deployment. By capturing changes in technology, infrastructure and energy systems over time, the framework enables a comprehensive assessment of hydrogen’s potential role in future low-carbon transport. The research is guided by the following objectives:

- **Quantify critical material demands of green hydrogen production.** Assess the environmental impact of PEM electrolyser production, operation, and end-of-life, focusing on critical materials.
- **Evaluate the GHG environmental impact of green hydrogen production.** Evaluate emissions associated with the production, operation, and end-of-life stages of PEM electrolyzers, considering their overall carbon footprint within the hydrogen supply chain.
- **Explore advancements in hydrogen technology.** Investigate developments in electrolyser and hydrogen-powered vehicle technologies, evaluating their potential for efficiency improvements, emission reductions, and material sustainability. Analyse the potential impact of innovations in fuel cell technology, hydrogen storage, and vehicle design on future emissions and cost competitiveness.
- **Assess the environmental and economic implications of hydrogen-powered transport technologies in comparison with conventional vehicle systems.** Benchmark hydrogen-powered vehicles against battery electric vehicles (BEVs), hybrid vehicles, and conventional internal combustion engine (ICE) vehicles in terms of emissions, resource use, and cost. Consider real-world driving conditions, infrastructure constraints, and energy supply variations in the comparative analysis.
- **Assess cradle-to-grave fleet-level environmental and resource impacts of hydrogen vehicle deployment.** Extend vehicle-level LCA to a national scale to determine potential emissions reductions and resource implications under different adoption pathways.
- **Model fleet-level hydrogen deployment under various UK adoption scenarios.** Simulate light-duty fleet compositions under various policy, infrastructure, and market uptake assumptions.

These objectives are addressed through four core research questions:

- (1) What are the life cycle GHG emissions and material requirements PEM electrolyzers, and how do these evolve over time?
- (2) How do emerging hydrogen vehicle technologies compare to battery electric and conventional powertrains in terms of life cycle GHG emissions and life cycle cost?
- (3) What are the fleet-level GHG and cost implications of hydrogen vehicle deployment in the UK through to 2050?

(4) To what extent can circular economy strategies (material recovery and recycling) reduce GHG and material impacts of PEM and hydrogen vehicle production?

In this light, this thesis makes several key contributions to the field:

Methodological innovation: Development of an integrated dynamic LCA-MFA framework that captures technological and policy improvements over time. **Technology coverage:** First comprehensive cradle-to-grave assessment of H_2 -ICE, FCEV and FCREx vehicle architectures within a unified analytical framework. **Policy relevance:** Integration of realistic deployment scenarios and policy frameworks to assess hydrogen’s viable role in transport decarbonisation. **Temporal dynamics:** Incorporation of technology learning curves, energy system decarbonisation, and evolving material supply chains to provide forward-looking insights.

1.3 Outline of Thesis

This thesis is organised into seven chapters in relation to the research questions outlined above. **Chapter 1** introduces the research motivation, outlines the sustainability challenges of hydrogen-powered mobility and defines the scope and objectives. **Chapter 2** presents the literature review of hydrogen production pathways, transport applications, lifecycle assessment studies on hydrogen technologies, critical material supply risks, limitations in end-of-life modelling, the need for dynamic approaches to capture technological evolution, and gaps in fleet-level modelling of hydrogen vehicle deployment. **Chapter 3** details the methodological framework used throughout the thesis across three analytical scales: hydrogen production (PEM electrolyzers), vehicle-level, and fleet-level (UK light-duty vehicle deployment). The chapter outlines data sources, modelling assumptions, system boundaries, and sensitivity analysis approaches used to address the research questions. **Chapter 4** presents results from the analysis of PEM electrolyzers, quantifying lifecycle GHG emissions, material demand trajectories, and supply constraints under future deployment scenarios. It highlights the role of recycling, catalyst loading, and technology development in shaping environmental and resource outcomes. **Chapter 5** evaluates hydrogen vehicle technologies at the single-vehicle level, comparing environmental and economic performance across powertrains and use-phase conditions. The analysis includes real-world driving profiles, hydrogen transport and distribution pathways, life cycle costing, and sensitivity to infrastructure access and fuel pricing. **Chapter 6** extends the analysis to the national fleet scale, modelling UK light-duty vehicle deployment under multiple hydrogen adoption scenarios. It assesses fleet-wide emissions, energy demand, and lifecycle costs from 2020 to 2050, supported

by a range of future scenario pathways. **Chapter 7** synthesises the key findings, discusses their implications for hydrogen policy and technology development, and outlines future research directions.

Together, these chapters build a dynamic, multi-scale assessment of hydrogen’s role in UK transport decarbonisation, integrating environmental, resource, and economic dimensions across time and technology.

1.4 Journal Papers

Duke, C., Davies, B., Abdul-Manan, A. and McKechnie, J. (2025). Addressing the Material Gap in Gw-Scale Proton Exchange Membrane (PEM) Electrolyser Deployment is Key to Meeting the Global Green Hydrogen Ambition. *Pending review*.
Duke, C., Davies, B., Abdul-Manan, A., Gordillo, V., Mitschler, M., Saafi, M. and McKechnie, J. (2025). A dynamic and comparative lifecycle assessment and lifecycle cost analysis of hydrogen passenger vehicle technologies in Europe from 2020 to 2050. *Pending review*.

1.5 Conference Presentations

- Oral presentation on my behalf by Dr. Jorge Llamas-Orozco, “Materials for Off-shore Wind-Hydrogen Systems”, SPE Offshore Europe, Aberdeen, UK (2025)
- Oral presentation, “Quantifying environmental and resource impacts of the future UK hydrogen fuelled vehicle fleet”, China-UK Young Scholars Summer School on Sustainability, Shaoxing, China, (2025)
- Poster presentation & Exhibitor, “Get with the H2ype”, Women in STEM Conference, University of Nottingham, UK (2024 & 2025)
- Speaker & Exhibitor: “Get with the H2ype”, Science Futures at Glastonbury Festival, Glastonbury, UK (2023, 2024 & 2025)
- Poster presentation: “Quantifying environmental and resource impacts of the future UK hydrogen fuelled vehicle fleet”, Sustainable Hydrogen CDT Stakeholder Workshop, University of Nottingham and Ulster University, UK (2023 & 2024).

- Poster presentation: “Assessing the Resource and Greenhouse Gas Emissions Implications of Green Hydrogen Production Infrastructure”, International Conference on Circular Economy in Transport (ICCET), Cranfield University, Cranfield, UK (2023)
- Poster presentation: “Lifecycle assessment of the future UK hydrogen vehicle fleet”, SPT School of Engineering research showcase, The University of Nottingham, UK (2023)
- Joint European Summer School (JESS) on fuel cell, electrolyser, and battery technologies, Athens, Greece, (2023)
- Visit to University of Toronto as visiting researchers as part of the Electrification of the UK light duty vehicle fleet project, Toronto, Canada, (FLAME), (2022)

Chapter 2

Literature Review

This chapter provides an overview of hydrogen’s role in the transition to a low-carbon energy system. The urgency for this transition arises from the need to decarbonise to achieve global net-zero targets. The transport sector represents a particularly challenging sector due to the large contribution to global GHG emissions. To tackle this, hydrogen is as a potential solution to address hard-to-abate emissions. This chapter begins by examining hydrogen’s general role as a low-carbon fuel and the various production pathways, with particular emphasis on current development in electrolytic hydrogen production technologies and their materials requirements. It then explores hydrogen applications across different sectors before focusing specifically on light-duty vehicles. The chapter concludes by reviewing recent literature on the environmental, material, and economic dimensions of hydrogen production and transport utilisation to identify key research gaps and frame the contributions of this thesis.

2.1 Hydrogen Production Pathways

Hydrogen is recognised for its potential as a sustainable energy carrier and can be produced through various processes (Maniscalco et al., 2024). The hydrogen colour classification is commonly used to categorise production methods based on the energy resources and processes used (Ajanovic et al., 2022) and is used throughout this thesis:

- Brown: produced via coal gasification, followed by syngas processes and gas purification. Brown can also interchangeably be used for biomass gasification
- Grey: steam methane reforming (SMR)

- Blue: steam methane reforming (SMR) with added carbon capture and storage
- Turquoise: produced through methane pyrolysis
- Yellow: grid-connected electrolysis
- Green: electrolysis powered by renewable energy sources
- Pink: electrolysis powered by a nuclear plant

Not included in historical classifications, but unofficially dubbed as ‘white hydrogen’, recent findings of naturally occurring hydrogen in underground reserves may also emerge as a future primary energy source (Lollar et al., 2014; Zgonnik, 2020). Of these routes to produce hydrogen; SMR, gasification, and electrolysis processes have reached large-scale operation and are the most popular methods of hydrogen production (World Nuclear Association, 2024) and so are explored further.

2.1.1 Steam Methane Reforming

Steam methane reforming (SMR) is the most common production method for hydrogen production, accounting for 76% of the world’s hydrogen supply (Ochu et al., 2021). Favoured for low cost and high efficiency, SMR produces high-purity hydrogen primarily used in petroleum, heating and transportation. In this process, natural gas is first desulfurised and then reacted with steam at temperatures of 800-900°C and pressures of 30-40 bar in the presence of a nickel-based catalyst, producing carbon monoxide and hydrogen. Using a water-gas shift reaction, carbon monoxide can produce additional hydrogen. Hydrogen from this process is purified using pressure swing absorption, ensuring hydrogen outputs meet the high purity standards required for fuel cell applications (min. 99.99%). However, despite its technical advantages, SMR is highly energy-intensive and depends on fossil fuels to meet heat demands, which has raised concerns about its long-term sustainability and the potential to lock into fossil fuel use. Moreover, given the anticipated growth of the hydrogen industry, reliance solely on SMR may be insufficient to meet future demand. These issues have intensified industry interest in alternative approaches, such as blue and green hydrogen produced from renewable energy sources, which promise lower emissions and a more sustainable pathway toward net-zero targets.

2.1.2 Reforming with Carbon Capture

SMR with carbon capture and storage (CCS) or 'blue hydrogen' reduces the carbon intensity of grey hydrogen production while benefiting from the economic advantages of SMR. This approach modifies the SMR process by capturing CO₂, typically through absorption techniques employing solvents such as monoethanolamine, achieving capture rates reported to range between 56% and 90% (Roy et al., 2025). Like grey hydrogen, blue hydrogen risks fossil fuel lock-in, particularly as current plants may remain operational until net-zero targets become imminent. These factors, along with issues such as methane leakage and the uncertainty of long-term CO₂ sequestration security, have prompted calls for a shift toward emerging hydrogen production technologies that promise better economic and environmental performance.

2.1.3 Gasification

Gasification is a thermochemical process that converts coal into a hydrogen-rich gas through partial oxidation with oxygen (or air) and steam at high temperatures and pressures (1,200-1,500 °C; 20-30 bar) (Maniscalco et al., 2024). Despite its technical maturity and large-scale deployment, brown hydrogen is one of the most carbon-intensive hydrogen routes, emitting roughly 22-26 kg CO₂ per kg H_2 produced when not captured (International Energy Agency (IEA), 2023b). Retrofitting gasifiers with carbon capture and storage can abate up to 90% of these emissions, but the additional energy penalty for solvent regeneration, CO₂ compression, and transport can erode overall thermal efficiency (World Nuclear Association, 2021). Other challenges include feedstock logistics, ash and slag handling, tar management and the long-term liability of CO₂ sequestration sites. In response, biomass and waste- gasification are emerging as renewable alternatives: when paired with CCS, they can deliver carbon-neutral or even net-negative hydrogen, though sustainable feedstock supply chains, tar removal strategies and competitive project economics remain critical hurdles (Wu et al., 2024).

2.1.4 Electrolysis

Coupled with renewable electricity inputs, water electrolysis is the lowest-emitting commercial production route available to produce hydrogen (Castelvecchi, 2022). Electrolysis technology splits water into hydrogen (H_2) and oxygen (O) using an electricity input. There are currently two commercialised electrolyser technologies: Alkaline (AE) and Proton Exchange Membrane (PEM).

Alkaline electrolysis is the most technologically mature and widely deployed method for large-scale hydrogen production (Mittelsteadt et al., 2015). It uses a potassium hydroxide electrolyte with low-cost nickel electrodes and porous transport layers and can even substitute cheaper non-noble catalysts such as manganese for platinum or ruthenium, driving down capital costs (Mittelsteadt et al., 2015; Saba et al., 2018). Robust cell separators give AE systems exceptional durability and impurity tolerance, yielding the longest lifetimes and lowest maintenance costs among commercial electrolyzers (Koponen, 2015). However, their relatively low current densities, slow response times and high minimum-load requirements limit direct coupling with variable renewable sources, often necessitating auxiliary batteries or control systems to manage intermittent power (Ionomr Innovations, 2020; Mittelsteadt et al., 2015).

PEM electrolysis uses a solid polymer electrolyte that facilitates proton transport, enabling higher current densities and rapid responsiveness to intermittent power inputs. This operation makes PEM well suited for coupling with renewable energy inputs (Ionomr Innovations, 2020; Mittelsteadt et al., 2015; Russell et al., 1974). However, PEM stacks command higher capital costs due to reliance on PGM catalysts, perfluorinated membranes, and titanium current collectors (Ionomr Innovations, 2020), and raise material demand challenges for critical material catalysts. These material and cost constraints challenge PEM’s potential for widespread adoption and highlight the need for catalyst-loading reductions, advanced membrane chemistries or non-PGM catalysts to unlock its green hydrogen potential.

A less mature technology but reaching pre-commercial levels of development (TRL 7), Solid oxide electrolyzers (SOEs) operate at 650-1000°C with ceramic electrolytes that convert steam and electricity into hydrogen with efficiencies of 80-90%, well above alkaline or PEM systems and unconstrained by Carnot limits (Andersson and Sunden, 2017; Ursúa et al., 2012). Whilst these systems have improved energy efficiency due to the utilisation of waste heat, their high temperature operation introduces material challenges and durability issues (Ishihara et al., 2010; Koponen, 2015).

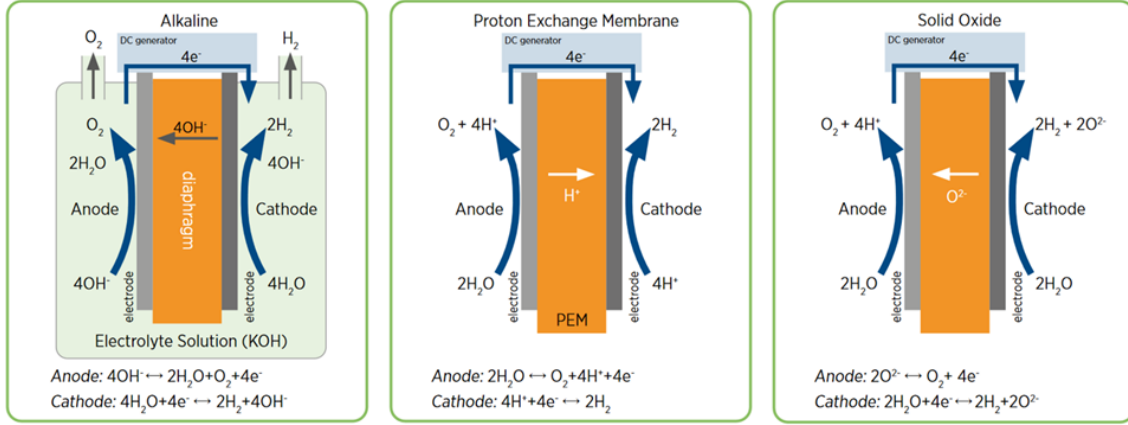


Figure 1: Electrolyser Operating Principles (FinH2, 2025)

2.2 Hydrogen in Transport

The transport sector remains one of the largest contributors to global GHG emissions, accounting for approximately 30% of total emissions (International Energy Agency (IEA), 2023a). Emissions from transportation have outpaced those from all other energy-consuming sectors (Pachauri et al., 2014). In the UK, transport is the largest single source of GHG emissions, responsible for 26% of the national total, with cars and taxis alone contributing 68% of these emissions (Department for Transport (DfT), 2023). Consequently, light-duty vehicles have become a central focus in national and international decarbonisation strategies aimed at achieving net-zero emissions by 2050.

Hydrogen is increasingly recognised as a viable alternative fuel for decarbonising light-duty vehicles, offering a clean and efficient substitute for fossil fuels. Its potential lies in its ability to reduce local air pollutants and contribute to broader climate mitigation goals (Staffell et al., 2019). While battery electric vehicles have dominated the early transition to low-emission mobility, they are not without limitations. BEVs face challenges related to range anxiety (Pevec et al., 2019), battery degradation (Timilsina et al., 2023) and the environmental impact of lithium-ion battery production, particularly in regions with carbon-intensive energy grids (Ellingsen et al., 2014; Llamas-Orozco et al., 2023). Furthermore, increased BEV adoption will place additional strain on electricity grids, requiring substantial upgrades to accommodate peak loads and intermittent renewable energy sources (Bastida-Molina et al., 2021). Moreover, although BEVs benefit from home charging convenience, gaps in public charging infrastructure (i.e. urban vs rural deployment) persist (Funke et al.,

2019).

These limitations highlight that BEVs may not be applicable across all transport scenarios, prompting interest in complementary technologies such as hydrogen vehicles. Hydrogen vehicles offer a number of advantages compared to BEVs, while still providing emission reductions. For example, FCEVs can refuel in under five minutes (comparable to ICEVs), making them attractive to consumers that prioritise refuelling times (Pagliaro et al., 2020). Also, hydrogen’s higher energy density (by mass) allows longer driving ranges than batteries, addressing BEV ‘range anxiety’ concerns (Li et al., 2023; Zhou et al., 2023). As a result, hydrogen vehicles are well-suited for high-utilisation fleets and long-distance travel (Apostolou and Xydis, 2019; Talebi et al., 2025).

Despite their promise, hydrogen powertrains remain in early-stage deployment, hindered by high costs and limited refuelling infrastructure (DNV, 2023). Also, hydrogen’s low volumetric energy density requires advanced storage solutions such as high-pressure tanks or cryogenic systems, which introduce technical and economic challenges (Xie et al., 2024). The existing fuel distribution infrastructure is incompatible with hydrogen, requiring significant investment to retrofit or develop new networks (André et al., 2014; Possémé et al., 2024). Additionally, the manufacturing of fuel cells presents a further challenge, particularly regarding the reliance on critical materials such as platinum and iridium. This dependency raises concerns over both availability and cost, as the scarcity of these materials could hinder the economic feasibility of mass fuel cell manufacture (Hosseini and Butler, 2020).

Given these dynamics, hydrogen-powered light-duty vehicles are viewed not as a replacement for BEVs, but as a complementary solution. The following section outlines the technological configurations of hydrogen vehicles.

2.3 Outline of Hydrogen Vehicles

Hydrogen-powered vehicles are a promising decarbonisation solution for light-duty transport. Among the available configurations, fuel cell electric vehicles are the only technology that has been commercialised. FCEVs power their motors by converting hydrogen into electricity in a proton exchange membrane fuel cell (PEMFC), the industry’s leading fuel cell type known for having high power density and low-temperature start-up (Mo et al., 2023; Wang et al., 2022b). Components within a PEMFC are: an anode, a cathode, a polymer electrolyte membrane and balance-of-plant. At the anode, hydrogen is split into protons and electrons; protons migrate through the membrane while electrons pass over an external circuit to deliver traction power, before recombining with oxygen at the cathode to form water (Mo et al.,

2023). The only outputs: electricity, heat, and water mean zero CO_2 or NO_x emissions during operation (Wang et al., 2022b).

In terms of performance characteristics, PEMFCs demonstrate electrical efficiencies of 60-65% under realistic driving cycles, surpassing internal combustion engines (Madheswaran et al., 2023). Today’s automotive PEMFC stacks cost approximately US \$75 per kilowatt at scale. To meet the U.S. Department of Energy targets of US\$ 30 per kilowatt, it is critical to reduce platinum usage and increase manufacturing volumes (Madheswaran et al., 2023; Wang et al., 2022b). As of 2024, FCEV uptake remains limited compared to BEVs due to high total system cost and sparse hydrogen refuelling infrastructure (Mo et al., 2023; Wang et al., 2022b).

For instance, in the UK, there were over 381,000 fully zero-emission vehicles (ZEVs) registered for the first time (Department for Transport (DfT), 2025a). But, hydrogen vehicle deployment remained low with 300 cars on the roads (ASM, 2023) - less than 0.1% market share of new ZEV sales. Nevertheless, projections suggest that hydrogen will play an important role in the transition towards ZEVs, although the demand and uptake levels remain highly uncertain. Future projections indicate that 5-50% of all vehicles will be hydrogen-powered by 2050 (Arena et al., 2017; Hydrogen Mobility Europe, 2020; International Energy Agency (IEA), 2020; McKinsey & Co., 2010), with a consensus at 5-10%.

Beyond full-stack FCEVs, other hydrogen vehicle configurations are being developed. Fuel cell range-extender (FCREx) is a hybrid vehicle aimed at extending the driving range of battery electric vehicles by pairing a smaller PEMFC with a larger battery. The cell charges the battery at a steady rate while the battery handles peaks, cutting hydrogen use by up to 25% and extending cell life (Fernández et al., 2016; Wu et al., 2019). This configuration leverages the high energy density of hydrogen while maintaining the efficiency and simplicity of an electric drivetrain (Álvarez and Corbera, 2018). Despite their promise, the added complexity of a hybrid powertrain introduces challenges related to cost, control strategy, and system efficiency (Luciani and Tonoli, 2022). Nonetheless, recent demonstrations have shown encouraging results, suggesting that FCREx could play a key role during the transition from conventional fossil fuel reliance to fully electric or next-generation fuel cell-powered mobility (Desantes et al., 2021; García et al., 2020).

Hydrogen internal combustion engines (H_2 -ICE) vehicles adapt conventional internal combustion engine technology to operate using hydrogen as a fuel instead of gasoline/diesel. In these engines, hydrogen combusts in a spark-ignited cycle, generating power in a manner akin to traditional engines but the combustion products are predominantly water. These vehicles benefit from compatibility with current manufacturing practices and reduced carbon emissions compared to gasoline/diesel. H_2 -

ICE vehicles have lower thermal efficiencies compared to fuel cells (35% vs 55%). Additionally, challenges remain in mitigating nitrogen oxide (NO_x) emissions, produced during combustion that require after-treatment e.g., catalytic converters (Wang and Fulton, 2024). In summary, novel hydrogen vehicles have the potential to act as transitional technologies leverage existing refuelling and manufacturing infrastructure before FCEV adoption.

2.4 Lifecycle Assessment

In recent decades, concerns surrounding environmental challenges have become increasingly valued in policy, business and individual decisions (Eckerberg and Nilsson, 2013). For this purpose, many tools for assessing and benchmarking environmental impacts have been developed (Ness et al., 2007), including Life Cycle Assessment (LCA).

LCA is a standardised methodology which is used to identify and quantify the potential environmental impacts of a product or a system for all life cycle phases from initial resource extraction, raw material processing, manufacture, use of the product, and end-of-life (International Organization for Standardization (ISO), 2006). The LCA methodology typically involves four phases: (1) goal and scope definition, (2) inventory analysis, (3) impact assessment, and (4) interpretation. While the ISO 14040/44 (International Organization for Standardization (ISO), 2006) standards provide a general framework, LCA studies often vary in their assumptions, system boundaries, and allocation methods, leading to diverse interpretations and results (Finnveden et al., 2009).

One of the main advantages of LCA compared to other tools is the focus on *systems thinking* with a life cycle perspective. For example, LCA is useful in avoiding problem-shifting (i.e. from one life cycle phase to another, one region to another, and even from one environmental problem to another) (Finnveden and Potting, 2014).

2.5 Lifecycle Assessment in the Transport Sector: Overview

The transport sector is responsible for 30% of global CO_2 emissions and is increasingly viewed as a priority area for decarbonisation. In this context, LCA is a key tool, providing structured insight into the environmental footprint of vehicles and transport fuels. Many researchers have contributed papers in this field, as well as

industry consultants (Ricardo, 2020). In the automotive sector, LCAs are typically divided into two 'cycles': the vehicle cycle (cradle-to-grave) and the fuel cycle (well-to-wheel). The vehicle (cradle-to-grave) analysis considers the entire lifecycle of the vehicle - from raw material extraction to material processing and manufacturing and assembly, usage including operation and maintenance and end-of-life disposal. The fuel cycle (well-to-wheel) focuses on the energy supply chain associated with vehicle operation, separated into two phases: well-to-tank (WTT) and tank-to-wheel (TTW). WTT covers the extraction, refining, and distribution of fuels, while TTW pertains to the combustion processes in internal combustion engines or the energy conversion processes in electric vehicles. Integrating both the vehicle and fuel cycles reveals the environmental impacts spanning the entire life cycle of both the vehicle and the energy inputs required for its operation, providing a comprehensive life cycle assessment.

2.6 Lifecycle Assessment in the Transport Sector: Studies

2.6.1 Hydrogen Vehicle Technologies

Researchers have focused significantly on FCEVs in the existing body of LCA literature (Bauer et al., 2015; Bieker, 2021; Moholkar et al., 2025; Valente et al., 2020; Yoo et al., 2018). Most studies concentrate on comparative analyses between FCEVs and conventional powertrains, including ICEVs, (P)HEVs, and BEVs. A key finding from the literature is the dependence of FCEVs environmental performance on upstream hydrogen supply chains. (Dulău, 2023) demonstrates that while FCEVs produce zero tailpipe emissions, their overall lifecycle GHG emissions are heavily influenced by the energy sources and processes used in hydrogen generation.

In addition, the geographical and temporal context of energy systems is a key factor in determining relative environmental benefits. (Dulău, 2023) illustrates this dynamic in their Romanian case study, where FCEVs initially show lower emissions than BEVs before the electricity grid decarbonises. However, the share of renewable energy connected to the grid increases, BEVs have the lowest CO₂ emissions profile. Thus showing the evolving nature of comparative environmental performance as energy systems transition toward cleaner electricity generation.

Similarly, (Bieker, 2021) demonstrate substantial differences in FCEV lifecycle carbon intensity across hydrogen supply chains. The author finds that compared to gasoline ICEVs, FCEVs can reduce emissions by approximately 25% when using

grey hydrogen, 68% with blue hydrogen, and up to 79% with green hydrogen. In comparison, BEVs achieve emission reductions of 63-81% depending on inputs from the electricity grid mix or 100% renewable energy sources. Using a comparative analysis, their findings reveal scenarios where FCEVs fail to provide decarbonisation advantages over BEVs.

The low emission profile of renewable hydrogen pathways emerges consistently across multiple studies. For example, (Valente et al., 2020) carry out a comparative analysis of FCEVs powered by different hydrogen sources, revealing that vehicles utilising SMR-based hydrogen emit significantly more CO₂/mile than those powered by wind-powered electrolysis. Notably, even when SMR incorporates CCS, renewable electrolysis maintains a clear environmental advantage.

Despite providing insight into FCEV performance under various hydrogen production scenarios, these studies fail to capture the environmental implications of alternative hydrogen vehicle configurations beyond FCEVs. This research bias has been explicitly acknowledged in recent study. For example, (Ajanovic et al., 2022) conducted a comprehensive review highlighting this limitation, noting that despite growing interest in alternative hydrogen powertrains, comparative LCA studies remain heavily skewed toward fuel cell systems, leaving other promising hydrogen technologies under explored in the environmental assessment literature.

One notable exception included (Candelaresi et al., 2021), who carry out one of the few comparative studies exploring FCEVs alongside H_2 -ICE vehicles, hydrogen-gasoline blends (Hythane), and hybrid hydrogen electric vehicles. Their findings revealed that H_2 -ICE vehicles powered by green hydrogen (wind electrolysis) achieved lower gCO₂/mile compared to FCEVs, suggesting a potential environmental advantage of alternative hydrogen technologies. This finding aligns with broader H_2 -ICE research showing efficiency gains of 15-25% compared to conventional ICEs when optimised for hydrogen combustion (Lyme et al., 2024; Verhelst and Wallner, 2009).

(Lombardi et al., 2017) carried out a comparative LCA of multiple hydrogen vehicle powertrains, including a plug-in hybrid fuel cell/battery configuration. The authors find that fuel cell/battery hybrid vehicles showed lower cumulative energy demand and climate change impacts than ICEV-gasoline vehicles, but higher impacts in categories such as human toxicity, acidification and eutrophication. Therefore, highlights the importance of considering impact categories outside of climate change and GHG emissions.

The lack of LCA studies on FCReX and H_2 -ICE vehicles represents a key research gap. These powertrains could serve as transitional technologies bridging challenges with current electric vehicle limitations and future hydrogen infrastructure development. However, this knowledge gap limits the understanding of the environmental

impact of these vehicles.

2.6.2 Environmental Impact of Real-world Driving

Current hydrogen vehicle LCA methodologies rely on standardised driving cycles that may not reflect real-world vehicle usage patterns. Recent research has revealed discrepancies between laboratory test cycles and the real-world performance of PHEVs e.g., (Plötz et al., 2017) and (Funke et al., 2019). Standardised assumptions such as consistent charging behaviour, fixed trip patterns, and high electric driving shares often fail to reflect actual user habits. The authors find that PHEV owners rely on internal combustion engines more than anticipated by manufacturers. This variation is how PHEV owners charge and operate their vehicles can misrepresent true GHG emission impacts. Thus, real-world CO₂ emissions often exceed official estimates (Plötz et al., 2022). The International Council on Clean Transportation (Plötz et al., 2022) found that real-world PHEV fuel consumption averages 4.0–4.4 L/100 km for private cars and 7.6–8.4 L/100 km for company cars, significantly higher than Worldwide Harmonised Light Vehicle Test Procedure (WLTP) estimates of 1.6–1.7 L/100 km. Onboard monitoring confirms frequent ICE reliance, widening the real-world gap.

In a detailed analysis, (Wang et al., 2022a) carry out an analysis considering emissions and fuel consumption of hybrid electric vehicles in real-world conditions for Beijing and Toronto. The authors note that greater CO₂ emission benefits are achieved in Beijing due to more low power and speed driving. This study reinforces the importance of localised comparisons.

Despite the well documented gap between official and real-world fuel consumption, the integration of real-world driving data into hydrogen vehicle LCA studies is limited. Current studies typically assume fixed operational parameters that may not reflect the dynamic nature of hybrid hydrogen system usage. For FCReX vehicles, factors such as battery state of charge and charging frequency are likely to influence their environmental performance. Without accounting for these variables, LCA studies may provide misleading comparisons between hydrogen and conventional vehicle technologies. As such, the novelty of FCReX vehicles presents a key area for future LCA research as real-world discrepancies in performance metrics have not yet been documented comprehensively. With the ongoing advancement and increasing adoption of hydrogen technologies, developing an understanding of how they perform in real-world driving conditions will be crucial for evaluating lifecycle environmental impacts.

2.6.3 End-of-Life Process

The current body of LCA research on hydrogen vehicles has a research gap for end-of-life (EoL) data for fuel cell systems and hydrogen storage tanks. This data gap prevents a complete cradle-to-grave assessment and introduces uncertainty into life-cycle emission impacts. Several studies have included ‘cradle-to-grave’ assessments, including (Bauer et al., 2015; Kelly et al., 2024; Nordelöf et al., 2019; Stropnik et al., 2019). However, these studies often provide limited detail on EoL assumptions. Specifically, the recycling pathways for hydrogen storage systems (carbon fibre and polymers) are inadequately defined, with the environmental implications of various disposal options largely unexplored (Stropnik et al., 2019). Consequently, reliance on theoretical recycling scenarios frequently introduces uncertainty into the LCA results of these technologies, thereby limiting their comparative assessments against alternatives (Nealer and Hendrickson, 2015).

2.6.4 Fleet Approaches

Existing LCA studies reviewed focus on hydrogen vehicles’ individual components e.g. fuel cells; (Mori et al., 2023), fuel pathways (Sinha and Brophy, 2021; Valente et al., 2020), or conduct single-vehicle comparisons (Bauer et al., 2015; Bieker, 2021; Moholkar et al., 2025; Valente et al., 2020; Yoo et al., 2018). While these studies offer valuable insight into system-level hotspots, they often assume static conditions and do not capture changes in supply chain emissions, fuel composition, or vehicle mix over time.

Dynamic LCA approaches, which model changing system conditions over time, are increasingly applied to battery-electric vehicles and power systems (Garcia et al., 2015). Dynamic, fleet-scale models have been applied extensively to BEVs e.g., (Davies et al., 2024; Milovanoff et al., 2019; Shui et al., 2024), enabling evaluation of future emission trajectories under different policy and technology pathways. However, hydrogen mobility has received far less attention within this class of models, especially in the context of national-scale deployment scenarios through 2050.

Recent work by (Gueniat and Maryam, 2024) takes a system dynamics approach to hydrogen powertrain adoption in the UK, focusing on diffusion rates and infrastructure investment. While their findings underscore the policy and cost barriers to hydrogen expansion, their framework does not quantify life cycle environmental impacts or include component-level emissions from fuel cells, tanks, and vehicle manufacturing.

2.7 Lifecycle Assessment in the Hydrogen Production Sector

Hydrogen is now a central pillar of decarbonisation roadmaps across the globe. However, the GHG emission profile of hydrogen depends on how it is produced. LCA has therefore become an essential lens for quantifying impacts and clarifying trade-offs among competing technologies. Thus, providing a sizeable body of literature to find methodological patterns, summarise impacts and highlight persistent research gaps, which are explored in the following section.

Most hydrogen LCAs adopt a functional unit of 1 kg H₂ produced, allowing for direct comparison across studies. (Maniscalco et al., 2024) observe that PEM electrolysis, SMR, gasification, and AE are the most commonly investigated technologies, reflecting both market relevance and data availability. Secondary data, especially from the Ecoinvent database (Maniscalco et al., 2024), is predominantly used in the literature, with only a few studies utilising primary data e.g. (Cho et al., 2022; Ji et al., 2023). The vast majority of hydrogen LCAs employ static approaches that assess technologies at a single point in time, with only a limited number adopting dynamic or prospective methodologies e.g. (Bauer et al., 2015; Schropp et al., 2022; Valente et al., 2020). A cradle-to-gate system boundary remains the default, partly due to the scarcity of downstream infrastructure, distribution, and end-of-life datasets (Bhandari et al., 2014; Lotrič et al., 2021). This exclusion of end-of-life assessments is often caused by a lack of empirical data (Bhandari et al., 2014; Lotrič et al., 2021). Only a small number of studies extend to cradle-to-grave scopes e.g. (Dufour et al., 2012; Gerhardt-Mörsdorf et al., 2024). Nonetheless, those that do consistently show that upstream production remains the primary contributor to impacts. However, the focus on static, cradle-to-gate assessments significantly limits the methodology used in the literature. These approaches cannot capture the technological advancements and manufacturing improvements that will influence hydrogen’s environmental impact over time.

For SMR (grey hydrogen), reported global warming potential (GWP) values range from approximately 9-13 kgCO₂eq per kgH₂, averaging around 11 kgCO₂eq/kg H₂ (Cetinkaya et al., 2012; Cho et al., 2022; Henriksen et al., 2024; Mio et al., 2024; Patel et al., 2024). Integrating CCS with SMR (blue hydrogen) can reduce average emissions to 3-9 kgCO₂eq/kg H₂ (Dufour et al., 2012; Henriksen et al., 2024; Mio et al., 2024; Patel et al., 2024; Simons and Bauer, 2011). Several crucial factors contribute to blue hydrogen having a lower GHG footprint. For instance, these include capture efficiency, upstream methane leakage, the electricity source used for compression, and regional gas composition.

The significance of these assumptions is illustrated by contrasting studies. (Howarth and Jacobson, 2021) found blue hydrogen emissions higher than grey hydrogen, citing CCS capture rates of 85% for SMR-derived CO₂ and 65% for combustion emissions, with upstream leakage rates of 3.5%. In contrast, newer plants achieve 90% efficiency in both streams (International Energy Agency (IEA), 2019), with modern ATR facilities reaching 95% capture rates (Zapantis, 2021), while Equinor’s UK blue hydrogen project reports only 0.03% upstream leakage (Equinor ASA, 2020). (Oni et al., 2022) demonstrate that increasing the capture rates lowers the total GHG emissions. Similarly, (Bauer et al., 2022) display the impact of these assumptions: with low methane leakage (0.2%) and high capture rates (93%), emissions are 8.5x lower than scenarios with 8% leakage and no carbon capture.

The dominant factor for lifecycle GHG emissions for electrolysis technologies is the carbon intensity of the electricity source. Studies show that 96% of PEM electrolysis emissions come from electricity input source (Bareiß et al., 2019). Renewable powered electrolysis achieves 2.35 - 4 kgCO₂eq/kg H₂ for solar PV (Cetinkaya et al., 2012; Mio et al., 2024; Simons and Bauer, 2011) and 0.97 - 2 kgCO₂eq/kg H₂ for wind (Cetinkaya et al., 2012; Simons and Bauer, 2011). In contrast, grid-powered electrolysis has greater variation and in carbon-heavy regions can exceed grey hydrogen impacts (Henriksen et al., 2024). The literature reports values of 13.4 - 34 kgCO₂eq/kg H₂ (Bareiß et al., 2019; Dufour et al., 2012; Gerhardt-Mörsdorf et al., 2024; Henriksen et al., 2024; Mio et al., 2024). For example, (Simons and Bauer, 2011) carried out a future-oriented cradle-to-gate LCA considering the years 2005 and 2035. The authors that EU grid-powered electrolysis drops from 34 to 19 kgCO₂eq/kg H₂ in 2035, based on the decarbonisation of the grid.

Although electricity input is a major factor in the lifecycle emissions of electrolyzers, the impact of electrolysis production should not be overlooked. Firstly, as electricity grids decarbonise and renewable energy becomes the dominant source, manufacturing emissions make up a larger proportion of total impacts. (Gerhardt-Mörsdorf et al., 2024) clearly illustrate this transition. For instance, a 5 MW PEM plant with a GWP impact of 34 kgCO₂/kg H₂ sees its impact reduced by 89% (3.74 kgCO₂) when the electricity grid is decarbonised. However, once the grid is decarbonised, manufacturing and infrastructure emissions become the limiting factor, and technological development reduces the impact by only a further 9% (3.4 kgCO₂), leaving manufacturing as the main remaining source of emissions.

Secondly, how environmental impacts are allocated in LCA studies are important in determining the GHG emissions of a technology. This is particularly important for PGMs where co-production of PGMs (e.g., platinum, palladium, iridium, etc.) and base metals (e.g., nickel, copper) is common. Thus, a single mining operation

yields multiple outputs. This creates a common methodological challenge known as multifunctionality – how do you fairly divide environmental burdens of a shared process across its different products? To address this challenge, LCA studies use allocation methods. For example, (Santero and Hendry, 2016) argue that value-based (also commonly known as economic allocation) should be used for PGMs as it better reflects the relationship between co-products and the financial drivers for undertaking the mining operation due to the high market value of PGM products. It is also recommended by standards such as BS EN 15804 when co-product values differ significantly. In contrast, mass-based allocation is often used when metals undergo similar processing and may misrepresent the environmental burden in systems dominated by high-value metals such as PGMs. This is shown by (Iyer et al., 2024), who conduct a comparative LCA of SOE, PEM and AE electrolyzers. The authors perform a sensitivity analysis to investigate the impact of switching allocation approach from mass to economic. Using the economic allocation approach substantially increases the GHG impact of PEM by 330% and reverses the results from PEM having the lowest lifecycle GHG emissions to the highest.

A similar allocation issue arises in the treatment of recycled PGMs. PGMs are frequently recovered due to their high economic value and scarcity. One widely applied method is the avoided burden approach (also known as substitution), which assigns a ‘credit’ to the system for displacing the need for primary production. In this approach, when a product containing PGMs reaches end-of-life and is recycled, the system is credited with the environmental benefits of avoiding primary extraction and refining in a future life cycle. This method is commonly used in consequential LCA and in studies aiming to reflect the broader system-level benefits of circularity e.g., (Ekvall and Weidema, 2004; Nuss and Eckelman, 2014). In contrast, the cut-off approach often used in attributional LCA and industry datasets assigns no upstream burden to recycled inputs and no credit at end-of-life. For example, in the (International Platinum Group Metals Association (IPA), 2025) methodology, secondary PGMs are modelled as burden-free inputs and no credits are given for recycling when assessing products made from primary PGMs. While this simplifies modelling and avoids double-counting, it may underrepresent the environmental value of high-recovery metals like PGMs in circular systems.

Also, manufacturing impacts vary significantly across technologies, influencing optimal technology selection beyond operational considerations. AE and SOE electrolyzers, which do not use PGMs, have lower manufacturing emissions than PEM systems (Häfele et al., 2016). However, these technologies face other challenges, for example, SOE systems suffer from stack degradation, driving higher ongoing material turnover and associated manufacturing emissions throughout their lifecycle

(ibid). This creates a trade-off between initial manufacturing impacts and replacement requirements that cradle-to-gate analyses do not include. The authors (Häfele et al., 2016) note that reducing stack degradation rates, improving current densities and extending the lifecycle of SOEs could reduce their emissions impact by up to 20%.

2.8 Limitations of current life cycle assessment studies

2.8.1 System Boundaries

Comparative LCA studies on hydrogen vehicles are mainly limited by narrow system boundaries, with most assessments using WTW due to data availability issues. These studies often compare FCEVs with conventional vehicles, considering several hydrogen production methods (Burchart and Przytuła, 2025). However, WTW frameworks exclude vehicle manufacturing and end-of-life phases. They ignore the environmental impacts of producing hydrogen vehicle components, such as fuel cell stacks and high-pressure hydrogen storage tanks. This omission can greatly underestimate upstream and downstream environmental burdens (Lee et al., 2019; Messagie et al., 2014).

On the other hand, cradle-to-grave assessments provide a wider systemic view by including manufacturing, maintenance, and end-of-life stages. However, cradle-to-grave assessments face challenges when considering early stage hydrogen vehicle technologies. End-of-life modelling often depends on speculative assumptions about future recycling infrastructure and processes. Key components like PGM fuel cell catalysts and carbon-fibre reinforced hydrogen storage tanks lack detailed material inventory data and established recycling pathways, forcing researchers to make simplified assumptions regarding their disposal (Evangelisti et al., 2017; Pehnt, 2001). These data gaps can result in incomplete evaluations that neglect resource recovery potential and environmental impacts associated with future recycling practices, particularly for critical metals vital to hydrogen propulsion systems.

2.8.2 Impact Assessment Method

Current LCA studies of hydrogen systems focus on climate change impacts, with GWP typically being the predominant (often sole) environmental indicator. While GWP remains an important metric, its prevalence masks other important dimen-

sions. Notably, LCA indicators may be limited when it comes to evaluating natural resource use and so can be complemented with MFA to quantify resource demands, for example, as relates to critical materials used in battery electric vehicles (e.g., (Tarabay et al., 2023)). Given hydrogen technologies’ reliance on critical materials such as platinum, iridium and rare earth elements (Eikeng et al., 2024; Graedel et al., 2015), it’s important to capture resource depletion indicators. There is a need for the integration of MFA approaches to track critical material stocks and flows throughout the hydrogen system lifecycle, offering additional insights that complement traditional LCA frameworks. MFA is further explored in Section 2.10.

2.8.3 Temporal Limitations

One major limitation in hydrogen LCA research is the widespread use of static assessments that give a snapshot picture of environmental performance based on current technologies and energy systems (Mehmeti et al., 2018; Zhao et al., 2020). These studies overlook the dynamic improvements in electrolyser efficiency, catalyst loading, changing electricity grid compositions, and ongoing technological innovations that will alter the environmental profiles of hydrogen systems throughout their lifetimes.

Future-oriented modelling can lead to significant reductions in environmental impacts compared to static projections, highlighting the crucial role of dynamic LCA frameworks. Recognising this limitation early on in automotive applications, (Pehnt, 2001) pioneered the use of dynamic LCA in fuel cell systems, stressing that conclusions drawn from immature technologies need regular updates to account for changes in energy systems and component design.

Recently, several notable studies have investigated these dynamic LCA approaches for hydrogen and fuel cell systems. For fuel cell systems, (Usai et al., 2021) compare current and state-of-the-art impacts across eight impact categories, including GWP. The authors (Usai et al., 2021) find that technological improvements, such as reducing stack size, decreasing platinum loading, and increasing the use of secondary platinum material in the most optimistic scenario (‘development stage 2’), can halve total GWP emissions compared to the reference scenario. For green hydrogen production, (Koj et al., 2025) focus on the prospective assessment of PEM water electrolysis. Their study incorporates learning curves for technological advancements up to 2050, highlighting the declining environmental impacts of green hydrogen production as efficiency improves.

Despite the valuable insights gained from studies like those of (Pehnt, 2001), (Usai et al., 2021), and (Koj et al., 2025), prospective and dynamic LCA methodologies,

which account for the evolving nature of technology and energy sources, are still much less adopted than traditional static LCA approaches. This trend, as noted by (Thonemann et al., 2020) highlights the need for wider integration of dynamic LCA in future research to better reflect the impacts of innovations and changing circumstances over time.

2.8.4 Fleet Limitations

Dynamic, fleet-scale LCA approaches are increasingly applied to BEVs e.g., (Davies et al., 2024; Milovanoff et al., 2019; Shui et al., 2024), enabling the assessment of future emission trajectories under various policy and technological pathways. However, hydrogen mobility has received considerably less attention within these models, and recent fleet models by (Gueniat and Maryam, 2024) omit life cycle environmental impacts, concentrating instead on policy and cost barriers. Therefore, there is a need for dynamic, fleet-wide modelling that includes future technology adoption, energy supply decarbonisation, material recycling, and vehicle manufacturing for hydrogen powertrains. Existing frameworks do not sufficiently capture the interactions between infrastructure deployment, fuel supply routes, and vehicle fleet transitions at the national level.

2.9 Policy Context

Globally, hydrogen policy is driven by efforts to decarbonise energy systems and reduce reliance on fossil fuels. Many countries have introduced national hydrogen strategies. In fact, in 2020, fifteen countries had formalised hydrogen strategies, a number that has since risen to 60 governments worldwide (International Energy Agency (IEA), 2025). These strategies focus on scaling up green hydrogen production, developing infrastructure and integrating hydrogen into transport and industry sectors.

From a European perspective, hydrogen output in 2023 reached 7.94 Mt, of which over 99% came from steam methane reforming (SMR) of natural gas. Electrolysis contributed just 31 kt and reforming with carbon capture 42 kt (European Hydrogen Observatory, 2024). To shift this grey-dominated mix, the EU plans to install 17.5 GW of electrolyser capacity by 2025 and to deliver 10 Mt of renewable hydrogen plus 10 Mt of imports by 2030 under the European Hydrogen Strategy and RePowerEU framework (European Parliament and Council, 2023).

Zooming into the UK, domestic hydrogen production is roughly 1 Mt per year, almost entirely via SMR of natural gas (UK Government (UK Gov), 2021). This grey-

heavy profile is projected to shift with the UK Government’s *‘twin-track’* approach: 5 GW of CCS-enabled blue hydrogen alongside 5 GW of electrolytic green hydrogen by 2030 (Department for Energy Security & Net Zero (DESNZ), 2023). Over 100 low-carbon hydrogen projects are under development, leveraging abundant offshore wind, established industrial clusters, and a clear policy framework to pave the way for a clean hydrogen economy (National Grid, 2023).

To ensure consistency and credibility across markets, hydrogen regulations increasingly rely on standardised LCA methodologies and certification schemes. Initiatives such as the European Renewable Energy Directive, RED III (European Parliament and Council, 2023) and the UK’s Low Carbon Hydrogen Standard (Department for Energy Security & Net Zero (DESNZ), 2023) define thresholds for carbon intensity (3.38 and 2.4 kgCO₂e/kg H_2 , respectively) and require transparent reporting of emissions across the supply chain. These standards are essential for enabling hydrogen trade, securing investment, and aligning national strategies with climate goals.

2.10 Material Flow Analysis

Material Flow Analysis (MFA) is a systematic method used to quantify the stocks and flows of materials within a defined system boundary, providing insights into resource use, accumulation, and loss over time (Brunner and Rechberger, 2016). In this context, flows refer to the movement of materials into, through, and out of a system, such as annual platinum inputs into the automotive sector, while stocks represent the accumulation of materials within the system, such as platinum embedded in in-use vehicles. These concepts underpin assessments of primary material demand, which measures the amount of primary resources needed to meet system requirements, accounting for existing stocks, losses, and recycling potential. MFA can be performed using either static or dynamic approaches. Static MFA captures material flows at a specific point in time, whereas dynamic MFA includes temporal modelling to project historical trends and future scenarios (Müller et al., 2014). This dynamic approach is especially useful for evaluating long-term sustainability strategies, technological shifts, and policy measures.

A range of literature reviews have analysed MFA applications across different regions (Bao et al., 2010), explored how they may align with broader sustainable development agendas (Huang et al., 2012), and assessed evolving techniques used in dynamic modelling of material flows (Müller et al., 2014). (Huang et al., 2012) note that the combination of MFA and LCA can be seen as a potential future research area. However, they are limited by data availability and uncertainties to successfully

couple these methods (Binder, 2007).

The integration of MFA and LCA has been investigated in recent methodological advancements, resulting in integrated analytical frameworks that improve the comprehensiveness of environmental assessments. To establish a proof of concept for sequential industrial ecology tools that offer thorough evaluations of future resource demands and environmental impacts, (Van der Voet et al., 2019) used metal demand projections from MFA models as input parameters for other LCA studies. This ‘MFA to Scenario to LCA’ sequence was also used by (Kayo et al., 2019) to measure the environmental effects of wood consumption in Japan. These studies lay the methodological groundwork for integrated approaches that can use a single analytical framework to address both environmental impact assessment and critical material demands at the same time. Nevertheless, (Withanage and Habib, 2021) find that in the area of e-waste, few studies have successfully coupled MFA with LCA, but there is significant potential for future research to provide valuable insight into the effective management of critical raw materials.

Within the field of hydrogen technologies, MFA studies have concentrated almost solely on PGMs, due to the enabling role of these materials in fuel cell and electrolysis applications, their high cost, and limited availability. At the global level, (Rasmussen et al., 2019a,b) constructed dynamic MFA models to predict platinum flows and stocks from 1975 to 2050, finding that the automotive and jewellery industries were the dominant historical consumers of primary platinum but large increases in electrolysis and fuel cell stocks are projected by 2050. Their study showed that only under high-demand scenarios involving extensive fuel cell market penetration would cumulative platinum demand to 2050 outstrip global reserves. However, the authors noted limitations in accounting for electrolysis technology progress beyond platinum content reduction from 1 kg/MW to 0.25 kg/MW by 2050. Similarly, (Alonso et al., 2012) project that future FCEV expansion is a key driver for platinum demand within the automotive industry, directly attributing vehicle technology development to critical material demand. Additionally, (Minke et al., 2021) examine the risk of global iridium supply to produce hydrogen by 2050. The authors identified possible supply bottlenecks that could impede the growth of the global hydrogen economy.

Regional MFA investigations narrow platinum flows and losses. Within Europe, (Saurat, 2006) constructed detailed MFA models of PGMs, illustrating that the automotive sector accounted for the largest single consumer of primary PGMs, with catalytic converters being the predominant end-use application. (Saidani et al., 2019) built on this work to quantify platinum losses in European catalytic converter systems, concluding that inadequate collection systems and unregulated exports caused

65% of platinum losses, and 25% by in-use dissipation, emphasising the need for end-of-life management strategies. These results highlight the imperative role of recycling pathways in ensuring resource security and reducing environmental impacts, highly pertinent to modelling prospective recycling and disposal routes of PEM electrolysis and fuel cell systems.

At the national level, in China, (Zhang et al., 2021) conducted a quantitative analysis of platinum flows in the automotive sector, projecting to 2050 from 2009. Their findings are linked to meeting two climate target scenarios: under 2°C and carbon neutral. The authors (Zhang et al., 2021) report that platinum stocks range from 346-713 tonnes and 214-441 tonnes, respectively, in 2050, with the deployment of FCEVs driving large increases. (Zhang et al., 2021) note that between 2009 and 2019, the accumulation of platinum within in-use vehicles increased from 37 tonnes to 134 tonnes, highlighting the sharp rise in platinum applications in vehicle technology as the market shifts towards more sustainable automotive options. Crucially, their analysis shows that recovering platinum from end-of-life conventional vehicle stocks could provide 50-61% of the required platinum for fuel cell and hybrid vehicles in 2050, further underscoring the importance of closed-loop material management and dynamic modelling. (Wang et al., 2024) built upon this analysis by investigating platinum demand for hydrogen technologies using dynamic MFA, concluding that domestic Chinese platinum resources would be insufficient to meet national hydrogen development objectives under carbon neutrality pledges without enhanced recycling systems.

The extension of MFA beyond platinum materials in hydrogen systems is scarce, with (Minke et al., 2021) being one of the limited attempts to address iridium supply risk. This limited material and geographical focus identifies important gaps in the holistic critical material evaluation of hydrogen technologies, especially for other electrolysis systems that have alternative material inventories or assessing the environmental impact of hydrogen production from a broader geographical lens. Existing MFA research on hydrogen systems present a number of methodology limitations that offer opportunities for future research. Most of the current research focuses on individual materials or geographic regions, which limits its usefulness for overall sustainability evaluation. The widespread application of static methods restricts the potential for integrating technological change, policy effects, and infrastructure development dynamics. Also, the limited integration with environmental impact assessment frameworks hinders the potential to quantify material requirements and environmental impacts. These omissions are key opportunities for developing integrated MFA-LCA methodologies that can quantify material requirements, environmental impacts and dynamic technological change within integrated analytical

frameworks.

2.11 Life Cycle Cost Analysis

2.11.1 Hydrogen Production Costs

An increasing number of studies apply life cycle costing (LCC) to hydrogen production, storage, transport, and utilisation (Ishimoto et al., 2024). Many authors compare levelised costs across various production routes. For example, (Valente et al., 2019) demonstrate that biomass-derived hydrogen outperforms SMR in GHG emissions but remains more expensive than SMR on a levelised cost basis. (Tenhumberg and B ker, 2020) compare AE, PEM and SOE, finding AE to be the least costly due to lower capital expenditure, however cost reductions for PEM remain uncertain. (Gerloff, 2021) carried out an LCA of these three systems incorporating changing national energy mixes but did not include any LCC analyses. (Wilkinson et al., 2023) report that only 15% of green hydrogen production LCAs incorporate cost metrics, and just 10% evaluate environmental, technological, and economic factors simultaneously. (Nicita et al., 2024) further observe that most studies focus on individual components (i.e., electrolyser stacks or fuel cell catalysts) without considering complete system integration or end-of-life costs.

2.11.2 Vehicle Technology Costs

While environmental LCAs of FCEVs are becoming increasingly common in the literature (Ahmadi, 2019; Li et al., 2016), comprehensive LCC studies remain limited. (Chen and Melaina, 2019) conduct a techno-economic comparison of conventional, hybrid, battery electric, and FCEV powertrains, identifying hydrogen fuel and fuel cell system costs as the primary contributors. (Zhu et al., 2023) compare FCEVs, ICEVs, BEVs, and ammonia-fuelled FCEVs under various hydrogen supply pathways, finding that BEVs are currently more cost-competitive, but FCEV economics improve over time with decreasing renewable electricity prices. (Sengupta and Cohan, 2017) demonstrate that BEVs carry a 32% higher average ownership cost than conventional vehicles. (M ndez et al., 2025) incorporate LCA outputs into LCC models to evaluate how taxes and subsidies influence competitiveness, showing that a 20% purchase subsidy can make FCEVs more cost-effective than BEVs. Emerging systems (FCREx and H_2 -ICE) remain virtually unexamined within LCC literature despite their potential as transitional technologies.

2.11.3 Cost-driven Innovation

Hydrogen production faces economic hurdles, mainly due to the high costs of PGMs, which are used in both fuel cells and electrolyzers. Reductions in PGM loading in fuel cells and electrolysis systems are largely driven by the need to reduce costs (IRENA, 2020). While material availability concerns are becoming increasingly important, economic pressure has led to significant research into optimising catalysts and exploring alternative materials. Current development trajectories in both fuel cell and PEM electrolysis applications show a convergence of approaches to reducing PGMs. In fuel cells, the industry aims to reduce platinum loading from around 20-25 g/FCEV (GREET2, 2022; Sun et al., 2010) to the U.S. Department of Energy’s ultimate target of 6 g/FCEV (U.S. Department of Energy, 2020). Similarly, PEM electrolysis research focuses on replacing or reducing iridium and platinum catalysts with nickel, cobalt, and iron oxide alternatives, supported by nano-structuring and hybrid catalyst support innovations (Miller et al., 2022).

Despite the increase of environmental LCAs for hydrogen vehicle technologies e.g., (Ahmadi, 2019; Li et al., 2016), comprehensive LCC studies are sparse in the literature. This analytical gap limits understanding of how critical material reduction strategies translate into economic benefits across vehicle lifecycles and technology development pathways that could simultaneously minimise costs and environmental impacts.

The existing LCC literature highlights patterns regarding cost drivers and technology competitiveness. (Chen and Melaina, 2019) provide one of the most comprehensive techno-economic analyses, comparing conventional, hybrid, battery electric, and fuel cell electric vehicle powertrains. Their study identifies hydrogen fuel costs and fuel cell system capital costs (both directly linked to critical material requirements) as the primary economic barriers to FCEV adoption. The authors demonstrate that material cost reductions in fuel cell stacks could fundamentally alter vehicle cost competitiveness, though they do not quantify specific material reduction scenarios.

Recent studies have attempted to compare hydrogen vehicles against alternative powertrains under dynamic cost assumptions. For example, (Zhu et al., 2023) compare FCEVs, ICEVs, BEVs, and ammonia-fuelled FCEVs across various hydrogen supply pathways. Their analysis reveals that while BEVs currently maintain cost advantages, FCEV economics improve substantially over time with decreasing renewable electricity prices and system cost reductions from technological learning. However, the study does not model critical material cost reductions, limiting insights into how PGM reduction strategies might accelerate cost competitiveness.

In contrast, (Sengupta and Cohan, 2017) demonstrate that BEVs carry a 32%

higher average cost of ownership than ICEVs, suggesting that cost competitiveness also impact electrified vehicles. Their analysis, however, predates many material reduction strategies currently under development and may underestimate future improvement potential from critical material optimisation.

2.12 Summary

The literature reviewed highlights several opportunities that directly align with the objectives of this thesis. Table 2 summarises these, linking each relevant section of the review to a specific research objective and expected contribution.

Table 2: Literature Review Objectives Summary

Literature Review Section	Thesis Objective	Literature Gap	Opportunity for Thesis	Impact
Section 2.8; Section 2.10	(1) Quantify critical material demands of green hydrogen production	Existing studies focus on cost and supply risk but lack environmental impact data for critical materials used in H_2 technologies	Assess environmental impacts of critical materials for H_2 technologies	Offer a broader sustainability perspective
Section 2.6.3	(2) Evaluate the GHG environmental impact of green hydrogen production	Limited end-of-life data for PEM electrolyzers, fuel cells, and hydrogen tanks	Model potential recycling/disposal pathways for PEM electrolysis, fuel cells, hydrogen tanks	Enable full cradle-to-grave LCA
Section 2.7	(3) Explore advancements in hydrogen technology	Most LCAs use static inventories and do not reflect technological evolution or future deployment scenarios	Increase dynamic elements to H_2 production and vehicle technologies	Future-proof findings/assess future impacts

Section 2.6.1	(4) Compare the environmental and economic viability of hydrogen vehicles, including emerging/novel systems (FCREx, H_2 -ICE) with conventional powertrains	Few LCA studies exist for transitional technologies like FCREx and H_2 -ICE; comparative data is lacking	Conduct LCA for FCREx and H_2 -ICE systems	Contribute novel insights on transitional vehicle technologies
Section 2.6.2		Real-world driving behaviours (e.g., for PHEVs) are rarely modelled in LCA studies, limiting real-world applicability	Simulate variable use patterns (real-world behaviours for PHEVs in LCA)	Make vehicle comparisons more realistic and policy-relevant
Section 2.6.4	(5) Assess cradle-to-grave fleet-level environmental and resource impacts of hydrogen vehicle deployment	Limited integration of national-scale fleet composition and infrastructure dynamics in LCA studies	Assess hydrogen adoption trends and fleet compositions under different policy and infrastructure scenarios	Provide insights into national-scale feasibility and potential emissions reduction pathways

Chapter 3

Methodology

This chapter outlines the methodological framework that structures the thesis across three interconnected tiers. While Section 1.3 maps the progression of topics, this section clarifies how each methodological tier contributes to the thesis objectives, linking global hydrogen supply chains, vehicle-level assessments and national fleet-scale modelling.

As shown in Figure 2, this research integrates: **Tier 1:** investigates the global supply chain for green hydrogen production (PEM electrolysis). This tier addresses Objectives 1 and 2 by conducting a cradle-to-grave LCA and dynamic MFA to quantify both GHG emissions and critical material demand. This tier establishes the upstream GHG intensity of green hydrogen production and the material requirements, providing data inputs for subsequent tiers. **Tier 2:** assesses vehicle-level impacts, responding to Objectives 3 and 4. It applies a cradle-to-grave LCA and LCC to explore the environmental and economic lifecycle impacts. This level allows for a direct comparison of technology-specific performance. **Tier 3:** extrapolates the findings from the first two tiers to the national scale, supporting Objectives 5 and 6. It models the UK passenger vehicle fleet to 2050. This fleet-scale analysis evaluates emissions and total cost trajectories across various deployment scenarios.

The three-tier approach provides a comprehensive methodology and system-wide relevance. By linking the global supply chain to individual vehicle performance and then to fleet-level outcomes, the framework captures interactions and trade-offs that would otherwise remain hidden in single-scale analyses. It also maintains methodological consistency by applying common functional units, temporal boundaries, and scenario assumptions across all tiers. Throughout the study, multiple impact categories are assessed, including electrolyser scale expressed as 1 MWe capacity based on electrical input, hydrogen output expressed as 1kg H_2 , primary material demand

and secondary material availability in kilograms or tonnes, and GHG emissions calculated using 100-year global warming potentials (GWP) for GHGs as reported in the Intergovernmental Panel on Climate Change Fifth Assessment Report (Pachauri et al., 2014). Emissions are reported both in kilograms of CO₂ equivalent (kgCO₂eq) and in kilograms of CO₂ equivalent per vehicle kilometre travelled (kgCO₂eq/vkm). All cost-related results are reported in great british pounds (£), consistent with the UK-focused scope of the fleet-scale modelling. Where international comparisons are made, euros (€) are used and clearly indicated. Currency conversions, where applicable, are based on average exchange rates for the year of analysis.

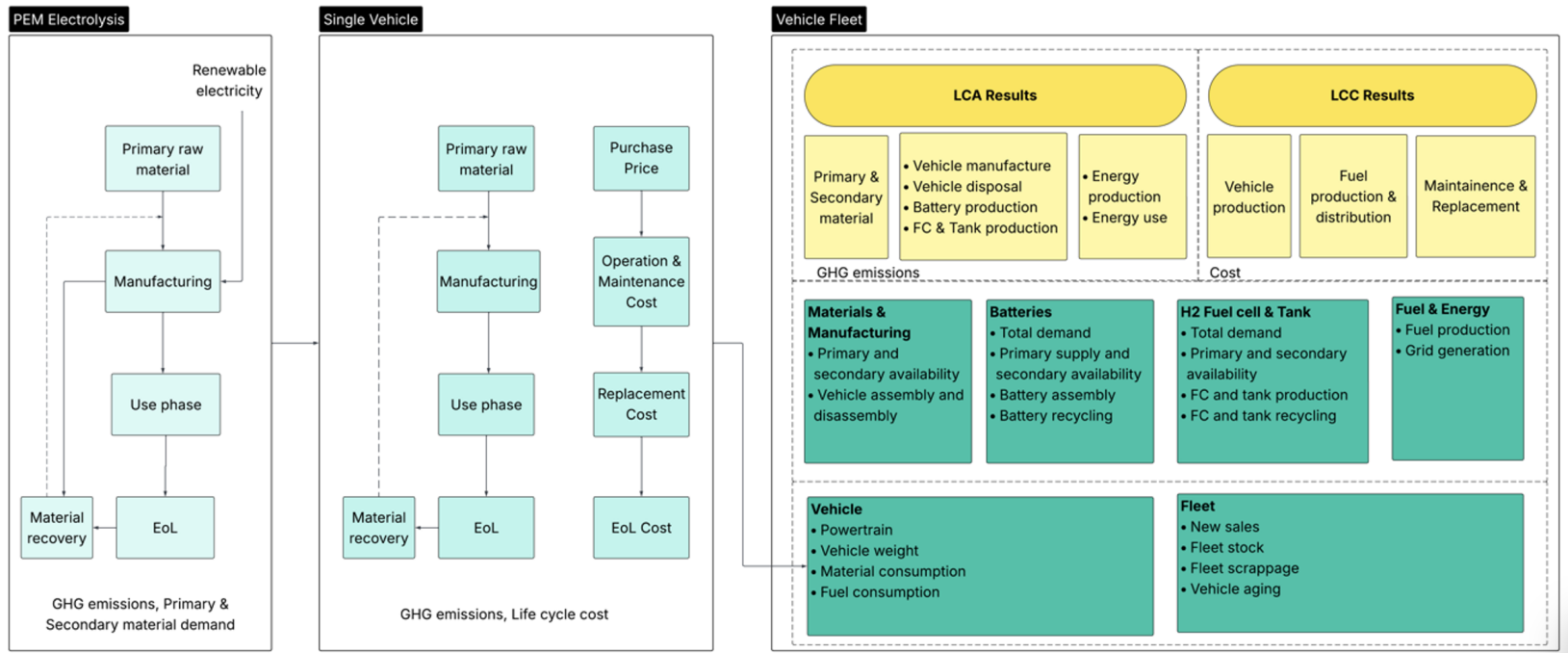


Figure 2: Overview of system boundary flowchart used in this thesis (PEM electrolysis and vehicles) to 2050

3.1 Tier 1: Global Green Hydrogen Supply Chain Analysis

This section begins by examining global hydrogen deployment trajectories to 2050, with a particular emphasis on electrolysis technologies. As nations accelerate their decarbonisation strategies, electrolyser deployment has emerged as a key technology for green hydrogen production. The IEA projects that installed electrolysis capacity must grow from 1.2 GW today to 850 GW by 2030 and over 3,600 GW by 2050 to meet net zero targets (International Energy Agency (IEA), 2022b).

Among the available technologies, PEM electrolyzers are increasingly favoured for their compatibility with intermittent renewable energy sources such as wind and solar (International Energy Agency (IEA), 2019, 2022a). Unlike AE, which have historically dominated industrial hydrogen production, PEM systems offer faster response times and higher current densities, making them well-suited to dynamic grid conditions. Their projected growth in market share reflects this operational advantage, particularly in regions investing heavily in renewables, see Table 3. This thesis focuses on PEM electrolysis due to its expected market dominance (Ehlers et al., 2023; International Energy Agency (IEA), 2022a) and limited prior studies on its material requirements.

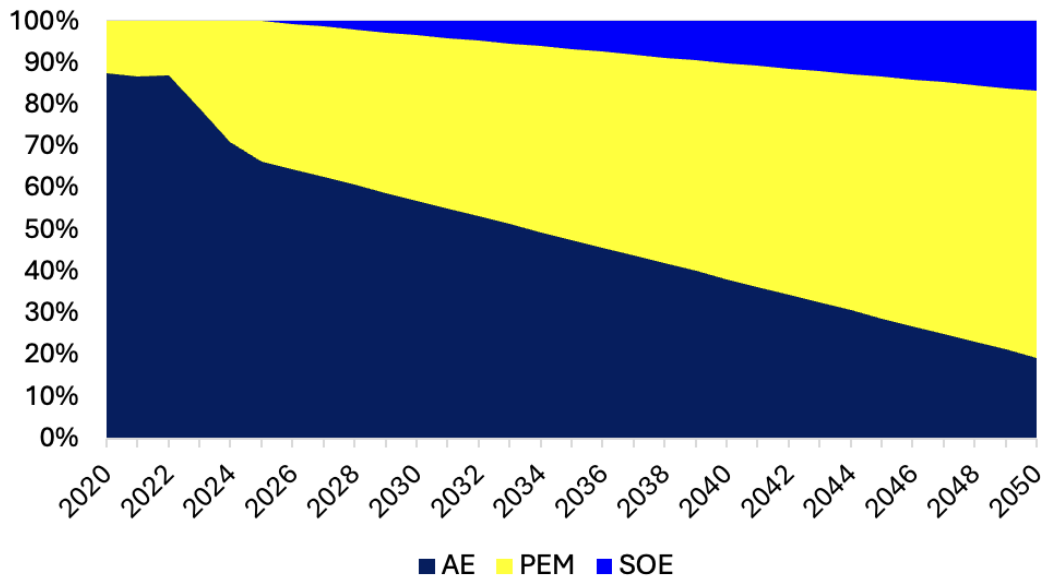


Figure 3: Electrolysis market share scenario (%), (IEA, 2022a; IEA, 2022b)

3.1.1 Life Cycle Assessment and Material Flow Analysis of PEM Electrolysis

This subsection integrates material flow analysis and life cycle assessment models to assess global GHG emissions and critical material demands for PEM electrolysis from 2020 to 2050. PEM technology is selected due to its expected market dominance (Ehlers et al., 2023; International Energy Agency (IEA), 2022a) and limited prior studies on its material requirements. The analysis includes raw material production, use-phase impacts (including maintenance and component replacement), and end-of-life processes. The modelling approach is outlined in Figure 4 and incorporates future deployment scenarios based on (International Energy Agency (IEA), 2022a,b) projections, as well as alternative scenarios that consider potential constraints in material availability. A comparison with lifecycle GHG emissions of alternative electrolysis technologies (AE, SOE) for green hydrogen production from a range of renewable energy sources is used to contextualise the impacts of PEM electrolysis with competitor renewable hydrogen production routes.

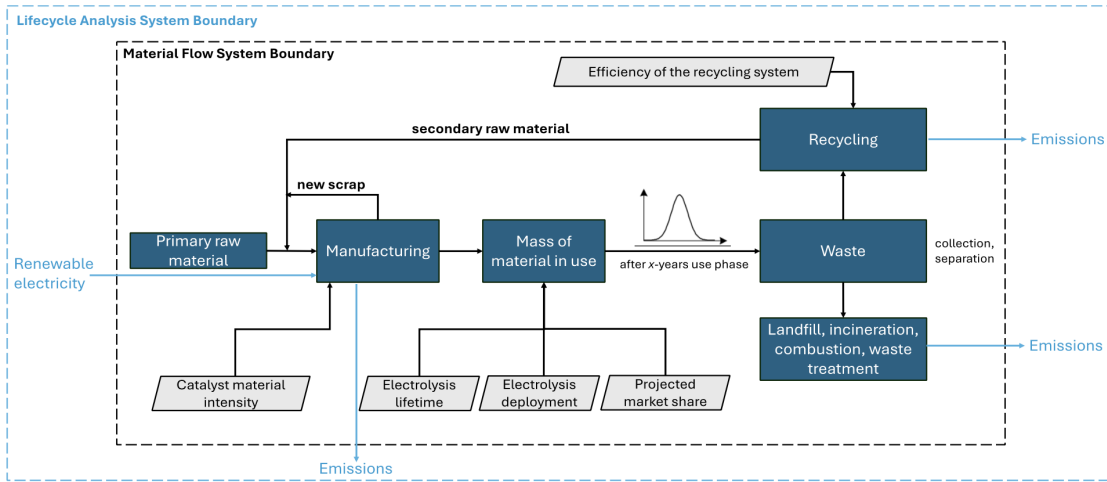


Figure 4: Overview of the MFA and LCA modelling approaches used for quantifying the GHG and material impacts of PEM electrolysis

The MFA model is bottom-up, detailing the specific components of electrolysis and production processes. Trends in dynamic parameters are considered, including catalyst loading rates, efficiency improvements, lifetime, and market adoption rates. The models track demand for two key minerals – platinum and iridium.

The LCA module is a process-based, attributional, cradle to grave assessment

using 100-year global warming potentials (IPCC, 2022) and avoided burden credits for closed-loop recycling to quantify the GHG emissions over the electrolyser lifecycle. Two functional units are used: 1MWe electrolysis capacity (based on electrical input) and 1kg H_2 (hydrogen output). The LCA module consists of two key phases: (1) Technology emission factor update – this phase accounts for technology improvements over time, including factors such as stack lifetime and hydrogen production efficiency. For catalyst loading, we assume a reduction in the material quantity over time without other changes to the life cycle inventory, providing a first order estimate of how decreasing catalyst demand may impact GHG emissions from electrolyser manufacturing. This is further discussed in Section 3.1.2. (2) Hydrogen energy emissions assessment, evaluating the emissions related to hydrogen production, incorporating dynamic parameters to reflect the evolving efficiency of electrolysis. We employ data for PEM materials inventory from the GREET2022 model (GREET2, 2022) and GHG emissions factors from both GREET2022 and the Ecoinvent database (Ecoinvent v3.7., 2020; GREET2, 2022). GREET and Ecoinvent provide lifecycle emissions data on materials, processes, energy, and waste. Additional public data fills gaps as needed (Ember, 2024; Graedel et al., 2015; United Nations Economic Commission for Europe (UNECE), 2022), with sources for scenarios detailed in subsequent subsections. Figure 4 shows the system boundary and major material/energy flows. Tables 3–5 list the key components and material inputs of AE, PEM and SOE technologies sourced from GREET (GREET2, 2022). Whilst Tables 6–8 show the recycling rates of materials.

Table 3: AE key input materials by component. All values presented in kg/MW (GREET2, 2022)

Component	Materials	Weight (kg)	Manufacturing Waste (kg)
Anode	Nickel	2999	247
	Ni(NO ₃) ₂	0.39	0.0076
	Co(NO ₃) ₂	0.65	0.0129
	Ru(NO ₃) ₂	0.22	0.0048
	NiAlMo	19	38
Cathode	Stainless Steel	540	109
Gasket / Seals	PTFE	1	0
	EPDM	404	603
Diaphragm	AGFA Zirfon Perl UTP 500	35	0
Others	Polypropylene	23	126
	Copper	18	3
	PEEK	302	12
	Water	0	0.036
	3% Sulfuric Acid	0	57
BoP	Cold-rolled steel	1494	—
	Steel	326	—
	Wrought Al	23	—
	Copper wire	90	—
	Concrete	2705	—
	Elastomer	30	—
	Ethylene Glycol	1	—
	Extruded HDPE	68	—
	Injection molded PP	0.44	—
	Glass fiber	68	—
	Stainless steel	98	—

Table 4: PEM key input materials by component. All values presented in kg/MW (GREET2, 2022)

Component	Materials	Weight (kg)	Manufacturing Waste (kg)
Membrane	Nafion	12	7
Anode	Iridium powder	0.55	2.17
	Au coating	0.01	0.01
Cathode	Pt/C powder	0.3	1.83
	Pt coating	0.13	0.08
	Copper	5.38	0.8
GDL	SGL Carbon: GDL 34 BA (Non-Woven)	2.18	1.45
	PTFE	1.32	0.41
	Adhesive Powder (Polyurethane)	0.26	0
	Titanium powder	56.72	0
BPP	Stainless steel	71.96	28.55
	Titanium Grade 2	107.87	55.18
	Polypropylene	8.05	45.72
Seals and gas-kets	Lubricant Powder (Zinc Stearate)	0.29	0
	Mylar/PET	20.31	53.75
Others	Miscellaneous	16.6	601.03
BoP	Stainless steel	1909	—
	Steel	930	—
	Cold-rolled steel	2374	—
	Al sheet	115	—
	Copper wire	152.17	—
	Concrete	2470	—
	Elastomer	50.72	—

Table 4 continued

Component	Materials	Weight (kg)	Manufacturing Waste (kg)
	Ethylene glycol	3.09	—
	Extruded HDPE	204.92	—
	Injection moulded PP	132.32	—
	Lubricating oil	44.11	—
	Zeolite	44.11	—
	Electronics	44.11	—

Table 5: SOE key input materials by component. All values presented in kg/MW (GREET2, 2022)

Component	Materials	Weight (kg)	Manufacturing Waste (kg)
GDL	Polypropylene	0	236.98
Electrolyte	Sodium nitrate/sodium chloride mixture	0	9.40
	Nitric acid	0	11.03
	8 mol.% Yttria-stabilized Zirconia	9.35	0
	3 mol.% Yttria-stabilized Zirconia	79.64	0
	Gadolinia doped ceria	25.30	0
Electrode	Alumina	4.00	0
Cathode	Lanthanum strontium cobalt ferrite	10.80	0
	Cobalt carbonate	2.08	1.39
Anode	Nickel oxide	144.42	0
	Manganese carbonate	1.89	1.26
Interconnect	Ni-Cr-Fe alloy (Alloy 600)	258.80	51.92
	Stainless steel	115.88	4.60
	Fe-24% Cr alloy	545.20	162.40
Insulation	Aluminium silicate fiber	15.24	1.98
Others	2% Hydrogen in nitrogen	0	12.76
	Heraeus V-006A Lower Viscosity Vehicle	0	3.16
	Glycine	0	13.86
	Glass powder	8.13	0
BoP	Hot-rolled stainless steel	3680	–
	Cold-rolled steel	2177	–

Table 5 continued

Component	Materials	Weight (kg)	Manufacturing Waste (kg)
	Hot-rolled steel	498.13	—
	Steel	2303	—
	Wrought Al	66.64	—
	Al sheet	22.14	—
	Copper wire	94.86	—
	Concrete	1240	—
	Controller	22.14	—
	Acrylonitrile-Butadiene (Injection moulding)	0.31	—
	Elastomer	39.10	—
	Extruded LDPE	118.22	—
	Ethylene glycol	7.75	—

Table 6: AE EoL recycling rate (%) (ELGGF, 2017; Graedel et al., 2011)

Component	Materials	Recycling Rate (%)
Anode	Nickel	100
	Ni(NO ₃) ₂	0
	Co(NO ₃) ₂	0
	Ru(NO ₃) ₂	0
	NiAlMo	0
Cathode	Stainless steel	100
Gasket / Seals	PTFE	0
	EPDM	0
Diaphragm	AGFA Zirfon Perl UTP 500	0
Others	Polypropylene	100
	Copper	100
	PEEK	50
	Water	0
	3% Sulfuric Acid	0
BoP	Cold-rolled steel	90
	Steel	90
	Wrought Al	95
	Copper wire	95
	Concrete	100
	Elastomer	10
	Ethylene Glycol	0
	Extruded HDPE	100
	Injection moulded PP	100
	Stainless steel	88

Table 7: PEM EoL recycling rate (%) (ELGGF, 2017; Graedel et al., 2011)

Component	Materials	Recycling Rate (%)
Membrane	Nafion	0
Anode	Iridium powder	95
	Au Coating	95
Cathode	Pt/C powder	95
	Pt coating	95
	Copper	45
GDL	SGL Carbon: GDL 34 BA (Non-Woven)	90
	PTFE	0
	Adhesive Powder (Polyurethane)	—
	Titanium powder	40
BPP	Stainless steel	88
	Titanium Grade 2	40
	Polypropylene	1
Seals and gas-kets	Lubricant Powder (Zinc Stearate)	—
	Mylar/PET	1
Others	Miscellaneous	—
BoP	Stainless steel	88
	Steel	90
	Cold-rolled steel	90
	Al sheet	95
	Copper wire	95
	Concrete	100
	Elastomer	10

Table 7 continued

Component	Materials	Recycling Rate (%)
	Ethylene glycol	0
	Extruded HDPE	100
	Injection moulded PP	100
	Lubricating oil	0
	Zeolite	0
	Electronics	91.5

Table 8: SOE EoL recycling rate (%) (ELGGF, 2017; Graedel et al., 2011)

Component	Materials	Recycling Rate (%)
Electrolyte	8 mol.% Yttria-stabilized Zirconia	0
	3 mol.% Yttria-stabilized Zirconia	0
	Gadolinia doped ceria	0
	Alumina	90
Cathode	Lanthanum strontium cobalt ferrite	0
	Cobalt carbonate	0
Anode	Nickel oxide	30
	Manganese carbonate	0
Interconnect	Ni-Cr-Fe alloy (Alloy 600)	0
	Stainless steel	90
	Fe-24% Cr alloy	0
Insulation	Aluminium silicate fiber	0
Others	Glass powder	0
BoP	Hot-rolled stainless steel	90
	Cold-rolled steel	90
	Hot-rolled steel	90
	Steel	90
	Wrought Al	95
	Al sheet	95
	Copper wire	95
	Concrete	100
	Controller	0
	Acrylonitrile-Butadiene (Injection molding)	0
	Elastomer	10
	Extruded LDPE	100
	Ethylene glycol	0

3.1.2 PEM Electrolyser Materials and Manufacturing

The PEM materials inventory is based on the (GREET2, 2022) model, covering key components like membranes, anode, cathode, gas diffusion layers, bipolar plates, seals, and gaskets, along with balance of plant (BoP) materials and manufacturing wastes (Table 4). Catalyst loading scenarios for platinum and iridium are drawn from projections (Heraeus Precious Metals, 2020; Mayyas et al., 2019; U.S. Department of Energy, 2022a) and pre-commercial data (Toshiba, 2018), addressing material availability (Figure 5)

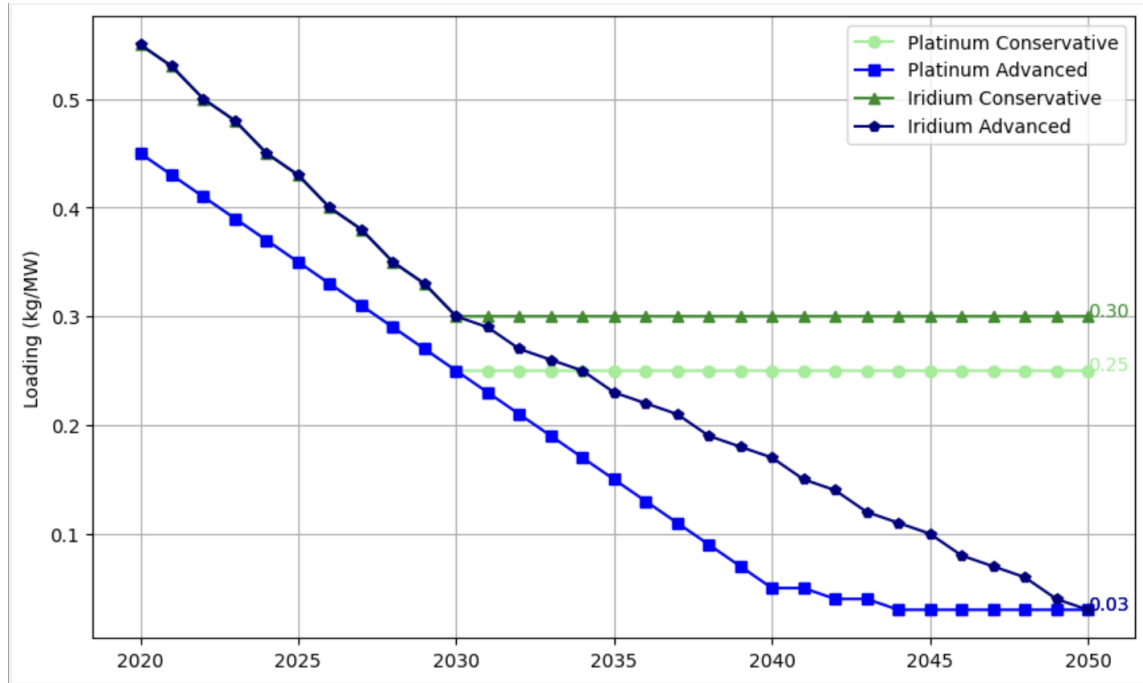


Figure 5: PEM Platinum and Iridium catalyst material loading – Conservative and Advanced scenarios

Current platinum and iridium loadings for PEM in 2022 are 0.45 and 0.55 kg/MW, respectively (GREET2, 2022). These intensities are projected to decline, with platinum reaching 0.03 kg/MW by 2050 (Mayyas, 2019). A conservative scenario assumes the (U.S. Department of Energy, 2022a) platinum target (0.25 kg/MW by 2030) is met with no further reductions by 2050, while iridium intensity follows projections from (Heraeus Precious Metals, 2020), declining to 0.3 kg/MW by 2030. An advanced scenario, based on emerging technologies like Toshiba’s ‘ultra-low’ iridium catalyst (Toshiba, 2018) (0.003 kg/MW), assumes commercial availability by 2050.

Under potential material constraints, achieving IEA PEM deployment goals may require further catalyst loading reductions. (Clapp et al., 2023) suggest a reasonable allocation of primary iridium supply for electrolyzers is 1.5 tonnes per year (20% of global supply), explored further in Section 4.2.

The PEM bill of materials highlights significant material waste in manufacturing (GREET2, 2022). Producing a 1MW PEM stack generates 1.9 kg of platinum and 2.2 kg of iridium as scrap in the slot die coating process – 80% of the PGM requirement (Iyer et al., 2022)(Table 4). As manufacturing scraps are confined to the production facility, we assume a 95% recovery rate offsetting primary material demand. Numerous studies on PEM fuel cell recycling report technical recovery efficiencies > 95% for PGMs under controlled conditions (Sharma and Andersen, 2021; Sharma et al., 2021; Wittstock et al., 2016; Zeng et al., 2022). While current recycling rates are typically 50-60% (Graedel et al., 2011; Hao et al., 2019; Johnson Matthey, 2025), these figures are based on automotive catalysts or other industries, where recovery is constrained by collection rates rather than technical limitations. In contrast, PEM electrolyzers are deployed in industrial closed-loop systems, where full stack collection at EoL is expected. To account for uncertainty, we conduct sensitivity analyses using 75% and to reflect current day recovery rates, 50% (Graedel et al., 2011). Additionally, we assume PGM scrap generation is proportional to catalyst loading and thus decline following trends indicated in (Figure 6). This reduction in scrap generation is due to the decreasing material usage, which aligns with the reduction in catalyst loading. However, it is also possible that advancements in manufacturing efficiency are contributing to this reduction in waste. As the manufacturing process evolves, better practices such as improved recycling, more precise material cutting, and reduced off-cuts could lead to less material wastage for the same amount of electrolysis capacity produced. Consequently, the decrease in waste is not solely a result of lower material requirements but may also reflect greater process efficiency. Electricity for stack and BoP manufacturing is sourced from (GREET2, 2022) (Table 9).

Allocation in LCA plays a critical role in determining the GHG emissions of a given technology. For base metals, we follow (Santero and Hendry, 2016), using mass allocation where co-mined metals have similar market values. This method distributes environmental impacts based on the mass of each metal extracted, ensuring a balanced allocation among co-products. However, for precious metals like PGMs, where platinum’s economic value far outweighs other co-mined metals, economic allocation is used. This approach assigns burdens based on market value, aligning with the primary economic drivers of mining these ores (Amatayakul and Ramnäs, 2001).

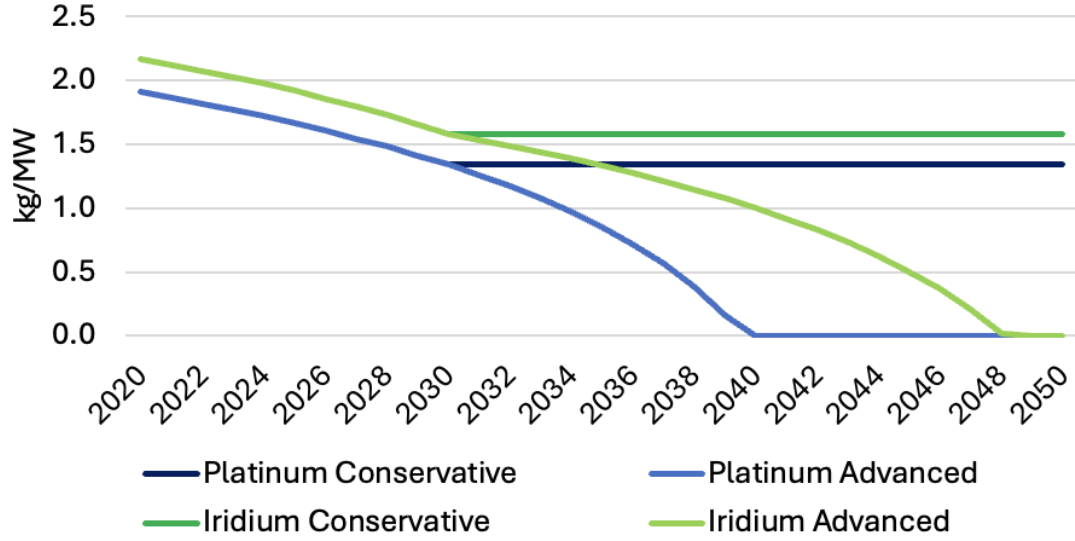


Figure 6: Manufacturing waste reduction scenario to 2050. Projected in line with linear percentage decrease in catalyst material loadings in the stack

Table 9: Capacity and manufacturing-related energy consumption of electrolyser stack and BoP per technology (GREET2, 2022)

Parameter	Electrolyser Technologies		
	AE	PEM	SOE
System (stack) capacity (kW)	3,836	998	50
Stack manufacturing energy consumption (kWh)	77,326	5,302	770
Energy consumed for stack manufacturing (kWh/kW)	220	238	279
Energy consumed for BoP manufacturing (kWh/kW)	38	50	76

3.1.3 PEM Electrolysis Installed Capacity

This section outlines the approach for modelling PEM electrolyser performance, focusing on changing installed capacity over time. The model assumes sufficient manufacturing capacity to replace end-of-life stacks and meet growth targets. It uses static parameters (e.g. 1 MW system capacity, 20-year BoP lifetime) and dynamic parameters reflecting efficiency improvements, such as reduced input power (1,500 kW in 2020 to 1,200 kW in 2050) (GREET2, 2022; U.S. Department of Energy, 2022a). The capacity factor is projected to rise from 34% to 50% as electrolyser utilisation improves (U.S. Department of Energy, 2022a). The overall system capacity factor is calculated using the capacity factor of the renewable energy input and the efficiency of the electrolyser, as detailed in Table 10, Table 11 and Table 12. Detailed parameters of the electrolyser systems are shown in Table 13 (AE), Table 14 (PEM) and Table 15 (SOE).

Table 10: Energy input source capacity factor (%) (Broadbent, 2016; González-Pabón et al., 2019)

Year	Offshore Wind	Onshore Wind	Solar PV	Other Renewables	Grid
2020	50%	40%	25%	50%	56%
2030	55%	45%	30%	60%	56%
2050	60%	50%	35%	70%	60%

Table 11: Renewable electricity share by electrolyser technology across selected years

Year	AE	PEM	SOE
2020	61%	67%	79%
2030	67%	74%	83%
2050	74%	83%	83%

Table 12: Calculated capacity factor (%) of the overall electrolysis system

Year	Offshore Wind	Onshore Wind	Solar PV	Grid
AE				
2020	30.3%	24.2%	15.2%	34.1%
2030	36.7%	29.9%	19.9%	37.0%
2050	44.4%	37.0%	25.9%	44.5%
PEM				
2020	33.3%	26.7%	15.2%	37.5%
2030	40.7%	33.3%	22.2%	41.1%
2050	49.9%	41.7%	29.2%	50.2%
SOE				
2020	39.7%	31.7%	19.8%	44.6%
2030	45.7%	37.5%	24.9%	46.3%
2050	49.9%	41.6%	29.2%	50.2%

To calculate the overall system capacity factor for renewable technologies coupled with electrolysis systems we first find the capacity factor of the energy input source (Table 10) and the efficiency of electrolyzers (Table 11). The overall system capacity factor for each combination of renewable source and electrolyser technology was calculated using the following formula:

$$\text{Overall Capacity Factor} = CF_{\text{Renewable}} \times \text{Efficiency}_{\text{Electrolyser}} \quad (3.1)$$

To maintain and expand electrolysis capacity, the model accounts for stack lifespans increasing from 7 to 10 years by 2030 (U.S. Department of Energy, 2016, 2023). This gradual shift is implemented as a linear change over 5 years, avoiding sudden drops in deployment during the transitional period. The overall capacity at any given year can be expressed by the following equations:

$$\text{Capacity}_y = \text{Capacity}_{y-1} - \text{Scrap}_y + \text{New}_y \quad (3.2)$$

Where:

- Capacity_y is the total system capacity in year y ,
- Scrap_y is the capacity scrapped in year y ,
- New_y is the new capacity added in year y .

Table 13: Static and dynamic modelled AE electrolyser parameters (British Plastics Federation (BPF), 2023; GREET2, 2022; U.S. Department of Energy, 2022a)

Parameter	2020	2030	2050
Static Parameters			
System capacity (kW)	1,000	1,000	1,000
BoP lifetime (years)	20	20	20
Dynamic Parameters			
System (stack) input power (kW)	1,650	1,500	1,350
Capacity factor (%)	34%	37%	45%
Hydrogen production capacity (kg/kWh)	0.0182	0.022	0.022
Electrolyser electricity consumption (kWh/kg)	55	50	45
Electrolyser market share (%)	88%	57%	19%
Stack lifetime (years)	10	10	10

Table 14: Static and dynamic modelled PEM electrolyser parameters (British Plastics Federation (BPF), 2023; GREET2, 2022; U.S. Department of Energy, 2022a)

Parameter	2020	2030	2050
Static Parameters			
System capacity (kW electricity input)	1,000	1,000	1,000
BoP lifetime (years)	20	20	20
Dynamic Parameters			
System (stack) input power (kW)	1,500	1,350	1,200
Capacity factor (%)	34%	42%	50%
Hydrogen production capacity (kg/kWh)	0.020	0.022	0.025
Efficiency (%)	67%	74%	83%
Electricity consumption (kWh/kg H ₂)	50	45	40
PEM market share (%)	12%	40%	65%
Stack lifetime (years)	7	10	10

Table 15: Static and dynamic modelled SOE electrolyser parameters (British Plastics Federation (BPF), 2023; GREET2, 2022; U.S. Department of Energy, 2022a)

Parameter	2020	2030	2050
Static Parameters			
System capacity (kW electricity input)	1,000	1,000	1,000
BoP lifetime (years)	20	20	20
Dynamic Parameters			
System (stack) input power (kW)	1,260	1,200	1,200
Capacity factor (%)	45%	46%	50%
Hydrogen production capacity (kg/kWh)	0.0238	0.025	0.025
Electricity consumption (kWh/kg H ₂)	42	40	40
SOE market share (%)	0%	3%	17%
Stack lifetime (years)	4	4	7

$$\text{Scrap}_y = \sum_a \text{Scrap}_{y,a} \quad (3.3)$$

Here, a represents the age of the electrolyzers, calculated as the model year minus the manufacturing year.

$$\text{Scrap}_{y,a} = \lambda_{y,a} \cdot \text{Capacity}_{y,a} \quad (3.4)$$

Where:

- $\lambda_{y,a}$ is the failure rate for electrolyzers of age a in year y , assumed to follow a normal distribution,
- $\text{Capacity}_{y,a}$ is the capacity of electrolyzers of age a in year y .

We assess the lifecycle impacts of green hydrogen production using renewable energy sources such as onshore and offshore wind, and solar PV, based on (United Nations Economic Commission for Europe (UNECE), 2022) data. These include regional variations and embodied emissions, with global renewable emission factors detailed in Table 16. As little variation is noted in the emission factors for offshore and onshore wind across the globe, a global average is used for all regions (Table 16). As nations commit to reducing their carbon emissions, GHG emissions attributed to the electricity grid are anticipated to fall namely due to the increased energy mix of

renewables. Europe as a region is leading the way in terms of wind and solar PV fed into the grid which in 2022 contributed to 22% of EU electricity – double the global average (Ember, 2022). Whilst other nations have a longer way to go in achieving emissions reductions such as China where the electricity grid remains dominated by coal power, accounting for 62% of electricity generation in 2021 (Statista, 2022). Figure 7 provides the emissions intensity of regional electricity grids considered.

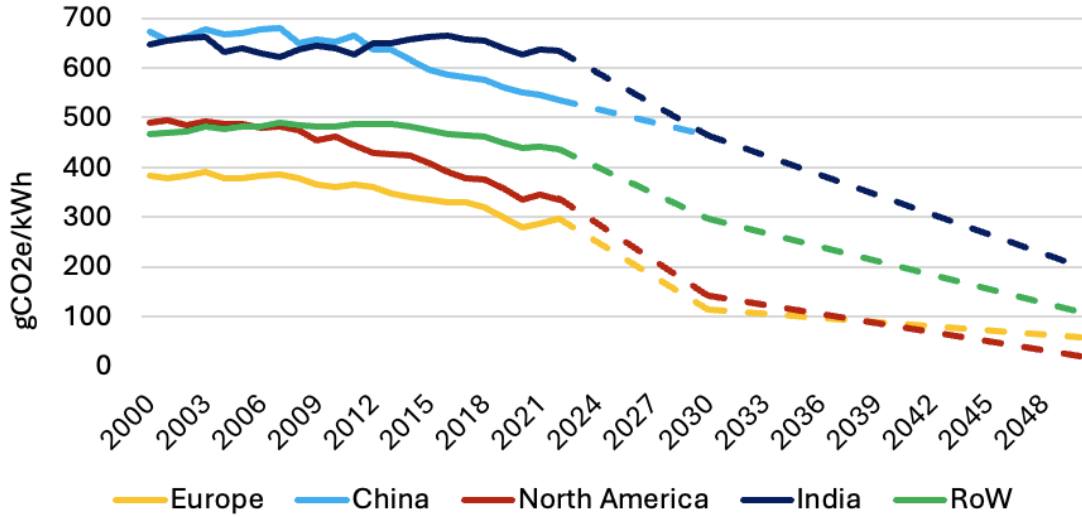


Figure 7: Emissions intensity of regional electricity grids, historical and future projections (Enerdata, 2023)

Table 16: Renewable energy input and lifecycle GHG emissions per region (2020), in gCO₂eq/kWh (Bonou et al., 2016; EEA, 2023; Ember, 2024; United Nations Economic Commission for Europe (UNECE), 2022)

Technology	Europe	China	North America	India	Other (RoW)
Offshore Wind	11.0	11.0	11.0	11.0	11.0
Onshore Wind	7.0	7.0	7.0	7.0	7.0
Solar PV	21.5	21.5	27.0	21.5	30.0

Electricity consumption per kilogram of hydrogen is shown in Tables 14, 15, and 13. Use-phase emissions from electricity generation are calculated using:

$$\text{Use-phase emissions (kg CO}_2\text{eq)} = \frac{(E_{\text{EF}}/1000) \times E_{\text{EC}}}{E_{\text{SE}}} \quad (3.5)$$

Where:

- E_{EF} is the emission factor of the electricity source (in g CO₂eq/kWh),
- E_{EC} is the electricity consumption of the electrolyser (in kWh/kg H₂),
- E_{SE} is the system efficiency of the electrolyser (expressed as a percentage).

This approach provides a comprehensive view of emissions across different electrolyser technologies.

3.1.4 Material Recovery from Manufacturing Waste and End of Life Components

The end-of-life (EoL) analysis was modelled by creating a process flow chart for the EoL procedure for electrolysis technologies. The process begins with electrolysers being collected, disassembled (manually) into the key components. Manufacturing waste materials are assumed to not require disassembly. For each component, it is assumed that the materials will enter into a recycling stream. Where this is not possible, the material will instead enter into a combustion, landfill, or incineration process. Any additional waste treatments that are required are also considered. Material recycling rates (RR) were gathered from an extensive literature review (Ballard, 2023; Deutsches Kupferinstitut, 2018; Graedel et al., 2011; Isa et al., 2022; Lotrič et al., 2021; Wang et al., 2021), while waste disposal impacts were collected from (Ecoinvent v3.7., 2020; ELGGF, 2017). This study assumes a closed-loop recycling process (Rasmussen et al., 2019b) for manufacturing waste, stack components, and EoL electrolysers. Recovered materials directly offset primary material use, creating a secondary material source. Recycling credits, which reduce lifecycle GHG emissions, are calculated as:

$$\text{Recycling Credit} = RR \cdot (P_V - P_R) + P_W \quad (3.6)$$

Where:

- RR — Material recycling rate
- P_V — GHG impact of primary (virgin) material production

- P_R — GHG impact of the material recycling process
- P_W — GHG impact of waste treatment in the absence of recycling

Material and waste flows are shown in Figure ???. Manufacturing waste is assumed to be readily recoverable without disassembly (Duclos et al., 2017; Moschovi et al., 2021; Wittstock et al., 2016), with recycling and supply of secondary material taking place during the same year. While EoL recovery occurs at the end of lifespan and includes manual disassembly into key components (Figure 9. For PEM systems, this includes anodes, cathodes, bipolar plates, gas diffusion layers, and membranes, while less valuable materials like seals and gaskets are discarded (Duclos et al., 2017). Each component is assumed to enter a recycling stream; if not, it undergoes landfill or incineration. Emission impact of recycling processes is obtained from (Ecoinvent v3.7., 2020) and industry data (ELGGF, 2017). Special waste treatments, like scrubbing hydrogen fluoride from incinerated Nafion membranes, are also considered (González-Pabón et al., 2019). Material recycling rates are derived from an extensive literature review (Deutsches Kupferinstitut, 2018; Graedel et al., 2011; GREET2, 2022; Isa et al., 2022; Lotrič et al., 2021; Wang et al., 2021), with upper estimates used since recycling rates are modelled to remain constant over time. The collection rate of materials is critical. We assume full collection of manufacturing waste, stack components, and EoL systems given the high material value, minimising process losses at EoL.

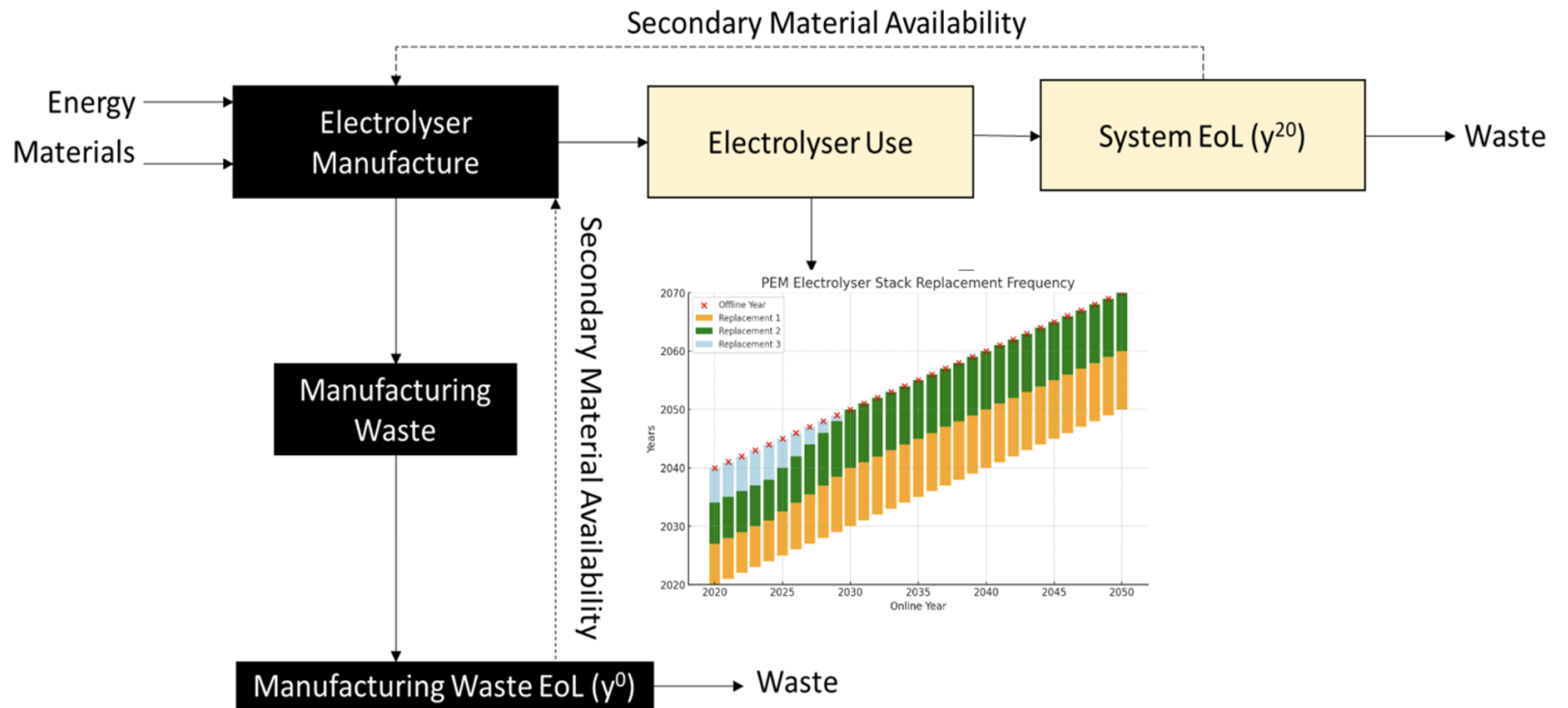


Figure 8: Material Flow Map of Electrolysis Production and Lifecycle

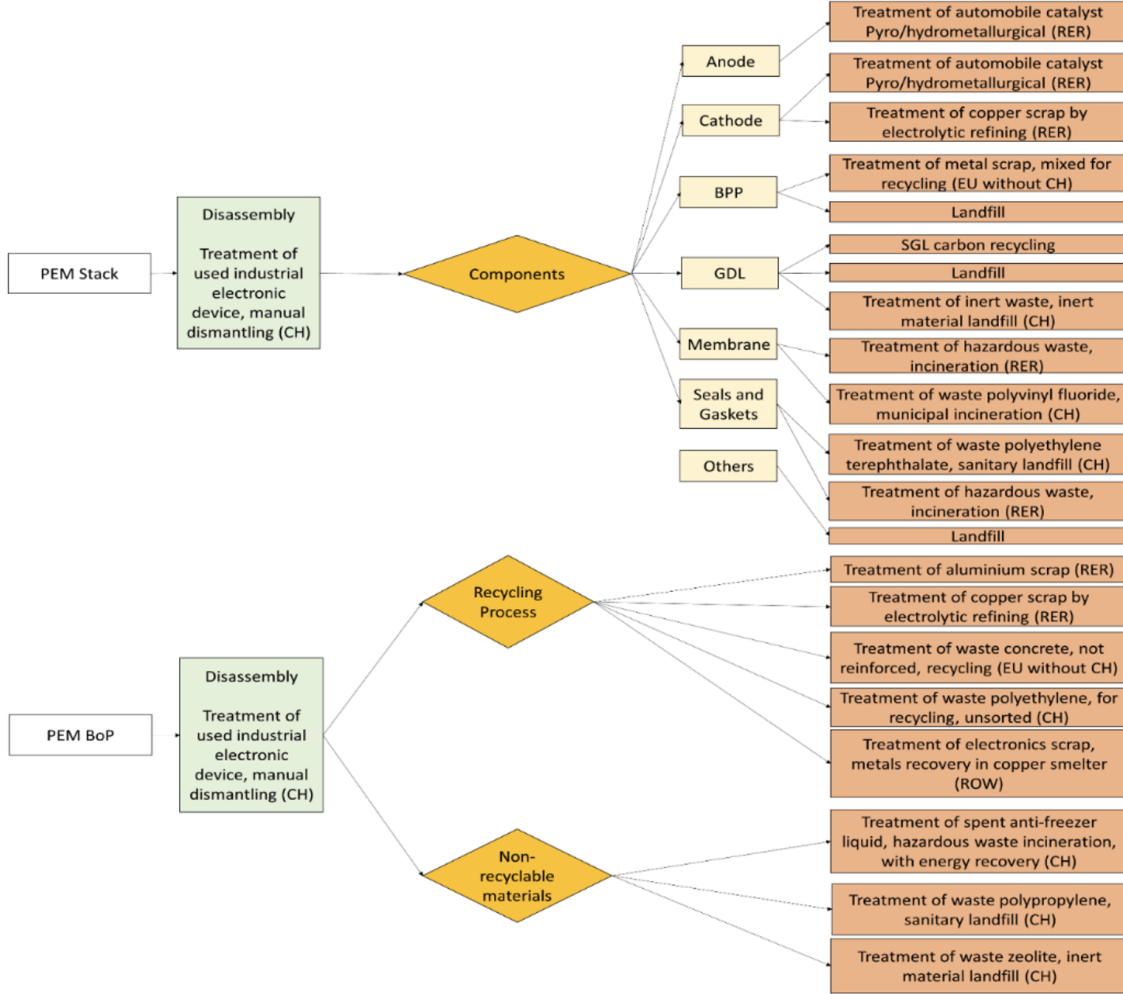


Figure 9: Process flow map of PEM stack and BoP disposal methods. Impact of recycling data sourced from (Ecoinvent v3.7., 2020; Lotrič et al., 2021)

3.1.5 Scenario definitions

We model demand-driven PEM deployment to assess manufacturing requirements and associated Platinum and Iridium material demands. Future projections are based on the (International Energy Agency (IEA), 2022b) announced pledges (APS) and net zero emissions (NZE) scenarios, which identify installed electrolysis capacity and technology mix to 2050. Additionally, we consider supply-constrained PEM

deployment, assuming 20% of annual iridium production is available to PEM electrolyzers (Clapp et al., 2023). A third set of model results determine catalyst loading rates required to achieve the IEA deployment projections under the iridium supply constraint (20% of annual production) for recycling rate data (Xun et al., 2020).

3.2 Tier 2: Single-vehicle Life Cycle Assessment and Life Cycle Cost Analysis of Hydrogen Vehicles (2020-2050)

The second part of the research methodology involves a cradle-to-grave LCA and LCC assessment of various sport utility vehicles (SUVs) currently in use or anticipated for future adoption in Europe. The analysis concentrates on SUVs for several reasons. Firstly, market demand – SUVs accounted for 51% of new car sales in Europe in 2023, reflecting a strong consumer preference (Smith, 2024). Secondly, higher energy demands – due to their larger size and weight, SUVs require more energy, making battery electrification challenging. Hydrogen technologies may better meet their range and performance requirements (Hassan et al., 2023). Lastly, design suitability for hydrogen storage – hydrogen tanks require more space than conventional fuel tanks, and SUVs larger dimensions allow for easier integration without compromising passenger or cargo space (Reuß et al., 2021).

The LCA evaluates environmental impacts across all life cycle stages, from raw material extraction to end-of-life disposal (Baptista et al., 2011). The system boundary is detailed in Figure 10. LCA models, developed in Microsoft Excel, incorporate data from (GREET2, 2022) and (Ecoinvent v3.7., 2020), supplemented with publicly available sources to ensure accurate emissions calculations. The study employs two functional units: GHG emissions per vehicle (kg CO₂eq/vehicle) and GHG emissions per kilometre travelled (kg CO₂eq/vkm), based on a vehicle lifetime of 15 years and a total distance of 295,000 km. The primary emissions indicator is Global Warming Potential (GWP-100) over a 100-year horizon, measured in kg CO₂-equivalent (IPCC, 2023).

For economic evaluation, the LCC analysis calculates lifetime costs, including purchase, operation, maintenance, and disposal. Additionally, abatement costs are estimated by determining the cost per ton of CO₂ reduced compared to ICEVs. To account for real-world variability, the study incorporates scenario-based models reflecting different driving patterns, impacting fuel consumption and emissions across powertrain types. It also assesses hydrogen distribution methods (pipeline vs. road transport) to evaluate logistical and environmental trade-offs in fuelling infrastruc-

ture. Following this, a policy analysis examines the impact of taxes and subsidies such as VAT, registration fees, road tax and fuel duties on the LCC of each powertrain. Incentives, including purchase subsidies and tax exemptions, are assessed separately from core cost modelling to isolate technological factors from policy-driven affordability.

The combination of environmental and economic metrics offers a holistic assessment of the environmental and economic feasibility of various hydrogen and battery-electric powertrains in the European SUV market.

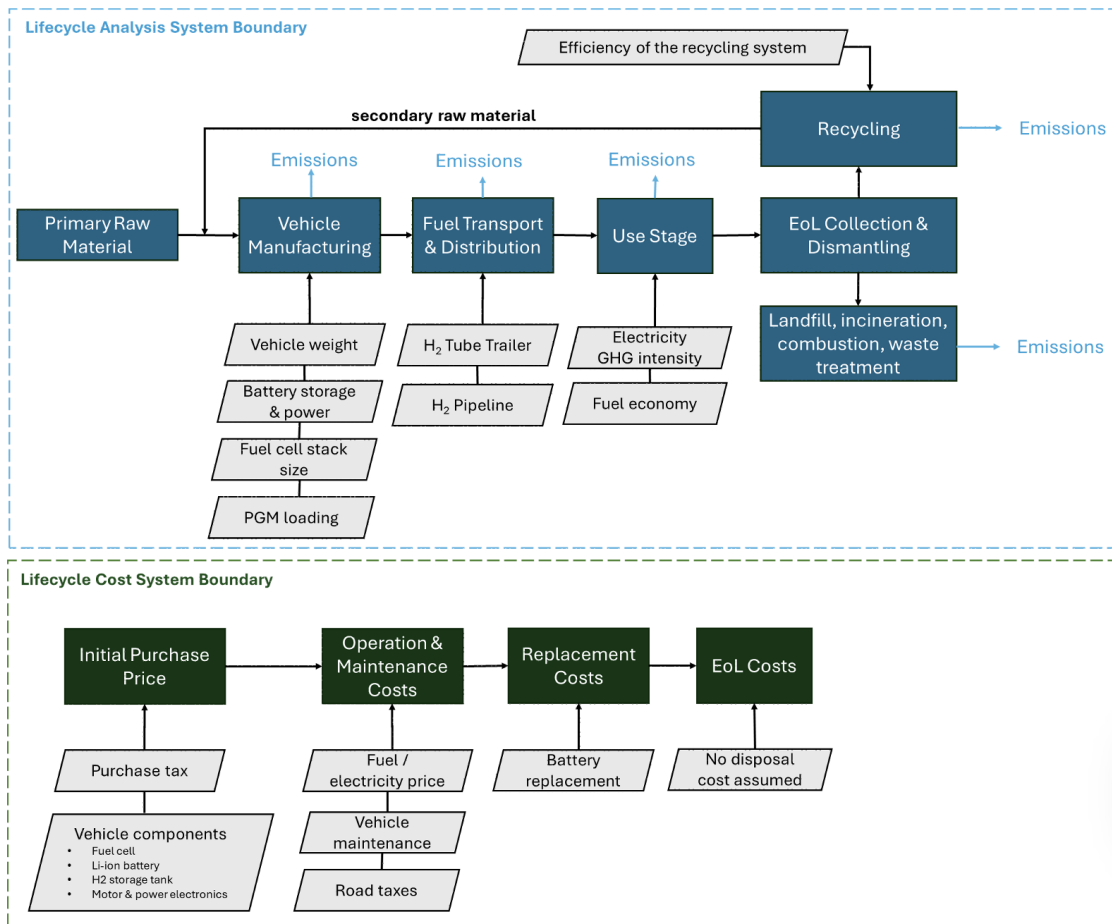


Figure 10: System boundaries of lifecycle analysis and lifecycle cost analysis

3.2.1 Vehicle Cycle

3.2.1.1 Vehicle Technologies Analysed

SUV powertrain variants currently in use or anticipated for future adoption in Europe are outlined in Figure 11, with further detail provided in Tables 17 to 24. As defined by the (European Commission, 1999), the passenger car classification used is the SUV, with an average weight estimated at 1,760–2,477 kg following the GREET model (GREET2, 2022). Vehicle data for the examined powertrain technologies were sourced from the GREET2 model (GREET2, 2022). For H_2 -ICEs and FCEVs, we adopt GREET2’s on-board storage specification of 5.5 kg H_2 . FCREx vehicles use a smaller tank (4.5 kg H_2 on average), scaling between 4.2 and 4.7 kg from 2020 to 2050, based on the 0.045 kg H_2 per kg fuel-cell stack scaling factor from (Desantes et al., 2021). We note that, without a large battery or primary fuel-cell stack, H_2 -ICE vehicles could in principle accommodate a larger hydrogen tank, but storage capacity is held constant across all hydrogen powertrains for a controlled comparison.

FCREx vehicles are an innovative type of hybrid hydrogen-powered vehicle that combine elements of BEVs and FCEVs to optimise range and energy efficiency. Operating primarily as BEVs, they employ a fuel cell system to generate electricity at a high efficiency operating point, recharging the battery rather than directly powering the vehicle. This setup reduces charging frequency while maintaining flexibility. The FCREx model incorporates a medium-sized 60 kWh lithium-ion battery and a fuel cell stack with a 100-kW power output (approximately 20% lower than a full FCEV) since its role is auxiliary rather than primary ((Desantes et al., 2021)). Key performance parameters, including vehicle kilometres travelled (VKT), vehicle life expectancy and battery replacement cycles are assumed constant in this analysis.

Due to the absence of H_2 -ICE and FCREx models in GREET, custom configurations were developed. The H_2 -ICE model is based on LCI data from (GREET2, 2022) with additional insights from (Candelaresi et al., 2021), who note that minor structural modifications, such as tungsten carbide-hardened valve seats, are needed to adapt internal combustion engines to hydrogen fuel. These modifications are minimal and do not significantly impact LCA outcomes, so they were not specifically modelled. (Stepien, Zbigniew, 2021) further confirms that spark ignition engines can use hydrogen with few adjustments, which is assumed to be a reasonable approach. Additionally, conventional gasoline tanks were replaced with hydrogen storage tanks using FCEV tank specifications available in (GREET2, 2022). This substitution ensures accurate modelling of vehicle mass, material composition, and refuelling requirements for hydrogen-powered configurations.

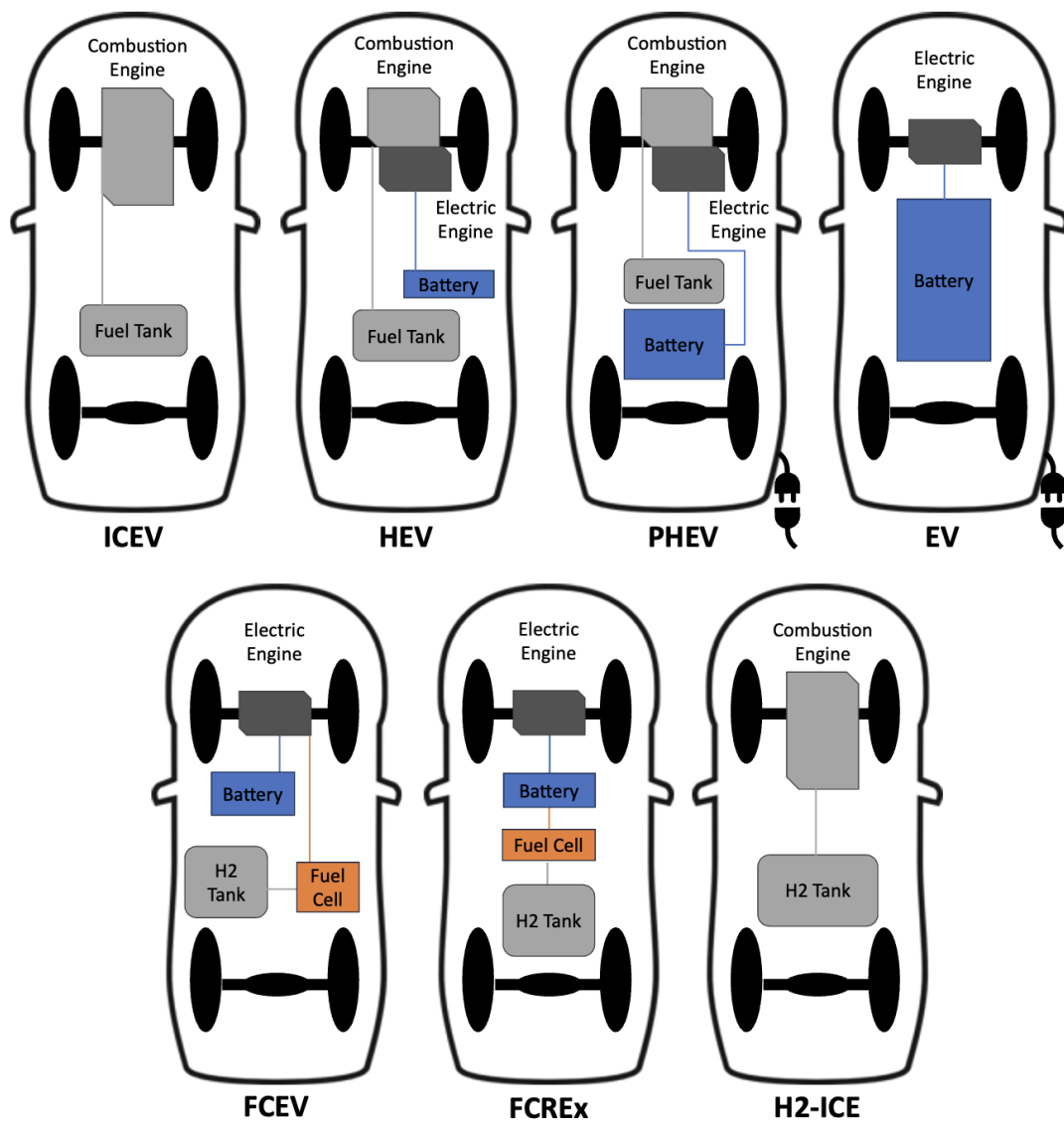


Figure 11: Basic outline of technology powertrains considered

Table 17: Specifications of ICEV-P vehicles from 2020 to 2050 (Bieker, 2021; Desantes et al., 2021; GREET2, 2022)

Materials	2020	2025	2030	2035	2050
Fuel economy (MPGGE)	22.65	23.96	24.70	27.39	29.59
Engine power (kW)	201	201	201	201	201
Tyre replacements	4	4	4	4	4
Range (miles)	380	380	380	380	380
Total VMT	183,363	183,363	183,363	183,363	183,363
Vehicle life (years)	15	15	15	15	15
Fuel consumption (L/100km)	12.47	11.79	11.44	10.31	9.55

Table 18: Specifications of ICEV-D vehicles from 2020 to 2050 (Bieker, 2021; Desantes et al., 2021; GREET2, 2022)

Materials	2020	2025	2030	2035	2050
Fuel economy (MPGGE)	27.29	28.75	29.76	31.34	33.38
Engine power (kW)	201	201	201	201	201
Tyre replacements	4	4	4	4	4
Range (miles)	420	420	420	420	420
Total VMT	183,363	183,363	183,363	183,363	183,363
Fuel consumption (L/100km)	9.00	8.54	8.25	7.84	7.36

Table 19: Specifications of HEV vehicles from 2020 to 2050 (Bieker, 2021; Desantes et al., 2021; GREET2, 2022)

Materials	2020	2025	2030	2035	2050
Fuel economy (MPGGE)	36.01	39.29	41.99	44.37	51.78
Engine power (kW)	132	132	132	132	132
Battery energy (kWh)	13.80	13.80	13.80	13.80	13.80
Battery technology	NCM111	NCM111	NCM111	NCM111	NCM111
Tyre replacements	4	4	4	4	4
Range (miles)	400	400	400	400	400
Battery life (miles)	183,363	183,363	183,363	183,363	183,363
Total VMT	183,363	183,363	183,363	183,363	183,363
Vehicle life (years)	15	15	15	15	15
Fuel consumption (L/100km)	9.00	7.19	6.73	6.37	5.46

Table 20: Specifications of PHEV vehicles from 2020 to 2050 (Bieker, 2021; Desantes et al., 2021; GREET2, 2022)

Materials	2020	2025	2030	2035	2050
Fuel economy (MPGGE)	69.08	74.04	77.06	79.16	84.92
Engine power (kW)	195	195	195	195	195
Motor power (kW)	67	67	67	67	67
Battery energy (kWh)	20.67	19.73	18.31	16.78	14.90
Battery technology	NCM111	NCM111	NCM111	NCM111	NCM111
Tyre replacements	4	4	4	4	4
Range (miles)	400	400	400	400	400
Battery life (miles)	183,363	183,363	183,363	183,363	183,363
Total VMT	183,363	183,363	183,363	183,363	183,363
Vehicle life (years)	15	15	15	15	15
Electricity consumption (kWh/100km)	30.32	28.29	27.18	26.46	24.66
Fuel consumption (L/100km)	4.09	3.82	3.67	3.57	3.33

Table 21: Specifications of BEV vehicles from 2020 to 2050 (Bieker, 2021; Desantes et al., 2021; GREET2, 2022)

Materials	2020	2025	2030	2035	2050
Fuel economy (MPGGE)	81.99	87.21	91.39	93.67	100.01
Motor power (kW)	147	147	147	147	147
Battery energy (kWh)	84	82	83	75	66
Battery technology	NCA	NCA	NCA	NCA	NCA
Tyre replacements	4	4	4	4	4
Range (miles)	400	400	400	400	400
Battery life (miles)	183,363	183,363	183,363	183,363	183,363
Total VMT	183,363	183,363	183,363	183,363	183,363
Vehicle life (years)	15	15	15	15	15
Electricity consumption (kWh/100km)	25.54	24.01	22.92	22.36	20.94

Table 22: Specifications of FCEV vehicles from 2020 to 2050 (Bieker, 2021; Desantes et al., 2021; GREET2, 2022)

Materials	2020	2025	2030	2035	2050
Fuel economy (MPGGE)	44.62	51.99	54.83	57.79	64.21
Motor power (kW)	120	120	120	120	120
Fuel cell power (kW)	121	95	96	90	86
Battery energy (kWh)	40	40	40	40	40
Battery technology	NCM111	NCM111	NCM111	NCM111	NCM111
Tyre replacements	4	4	4	4	4
Hydrogen storage (kg H_2)	5.50	5.50	5.50	5.50	5.50
Hydrogen storage pressure (bar)	350 or 700	350 or 700	350 or 700	350 or 700	350 or 700
Range (miles)	413	413	413	413	413
Battery life (miles)	183,363	183,363	183,363	183,363	183,363
Total VMT	183,363	183,363	183,363	183,363	183,363
Vehicle life (years)	15	15	15	15	15
H_2 consumption (kg/100km)	1.40	1.20	1.14	1.08	0.97
H_2 consumption (kg/mile)	0.0087	0.0075	0.0071	0.0067	0.0061

Table 23: Specifications of FCREx vehicles from 2020 to 2050 (Bieker, 2021; Desantes et al., 2021; GREET2, 2022)

Materials	2020	2025	2030	2035	2050
Fuel economy (MPGGE)	64.33	69.24	72.12	74.77	81.08
Engine power (kW)	0	0	0	0	0
Motor power (kW)	120	120	120	120	120
Fuel cell power (kW)	100	76	76.80	72	68.80
Battery energy (kWh)	60	60	60	60	60
Battery technology	NCM111	NCM111	NCM111	NCM111	NCM111
Tyre replacements	4	4	4	4	4
Hydrogen storage (kg H_2)	4.50	4.18	4.22	4.68	4.47
Hydrogen storage pressure (bar)	350 or 700	350 or 700	350 or 700	350 or 700	350 or 700
Range (miles)	435	435	435	435	435
Battery life (miles)	183,363	183,363	183,363	183,363	183,363
Total VMT	183,363	183,363	183,363	183,363	183,363
Vehicle life (years)	15	15	15	15	15
Electricity consumption (kWh/100km)	32.56	30.25	29.04	28.01	25.83
H_2 consumption (kg/100km)	0.97	0.90	0.87	0.84	0.77
H_2 consumption (kg/mile)	0.0060	0.0056	0.0054	0.0052	0.0048

Table 24: Specifications of H_2 -ICE vehicles from 2020 to 2050 (Bieker, 2021; Desantes et al., 2021; GREET2, 2022)

Materials	2020	2025	2030	2035	2050
Fuel economy (MPGGE)	27.18	28.75	29.64	31.22	33.44
Engine power (kW)	300	300	300	300	300
Tyre replacements	4	4	4	4	4
Hydrogen storage (kg H_2)	5.50	5.50	5.50	5.50	5.50
Hydrogen storage pressure (bar)	350 or 700	350 or 700	350 or 700	350 or 700	350 or 700
Range (miles)	400	400	400	400	400
Total VMT	183,363	183,363	183,363	183,363	183,363
Vehicle life (years)	15	15	15	15	15
H_2 consumption (kg/100km)	2.30	2.18	2.11	2.00	1.87
H_2 consumption (kg/mile)	0.0143	0.0135	0.0131	0.0125	0.0116

3.2.1.2 Manufacturing and Assembly

Vehicle manufacturing data is detailed in Table 25 to 29. Assembly emissions, based on (GREET, 2023) average 729 kg CO₂eq per vehicle, with major contributors including painting (232 kg CO₂eq /vehicle) and industrial heating processes (218 kg CO₂eq/vehicle), such as curing, drying and metal treatment.

In addition to assembly-related emissions, powertrain-specific components play a major role in determining emissions. Across the eight examined vehicle technologies, lithium-ion battery production for BEVs is estimated to contribute an average of 6,015 kg CO₂eq per vehicle, while hydrogen storage systems for FCEVs add 9,592 kg CO₂eq per vehicle (Figure 30). These values, sourced from 2020 emissions estimates, highlight the considerable impact of powertrain manufacturing on life cycle assessments. A comprehensive analysis ensures that these variations are accurately represented in the LCA.

Table 25: Vehicle Manufacturing and Assembly GHG emissions – 2020, in kgCO₂eq.

	Total GHG emissions kgCO ₂ eq.						
	ICEV	HEV	PHEV	BEV	FCEV	FCREx	H ₂ -ICE
Body	3,200	2,988	3,067	3,975	2,922	3,693	3,262
Powertrain System	1,335	1,400	1,601	218	948	570	1,532
Transmission System/Gearbox	580	414	425	550	314	637	671
Chassis	2,165	2,021	2,074	2,689	2,771	2,290	1,736
Traction Motor	0	259	427	1,398	552	549	0
Generator	0	250	404	0	0	0	0
Electronic Controller	0	248	250	902	536	532	0
Hydrogen Storage	0	0	0	0	9,592	9,887	8,731
Li-ion Battery	0	412	3,496	6,015	405	2,885	0
Assembly	729	729	729	729	729	729	729

Table 26: Vehicle Manufacturing and Assembly GHG emissions – 2025, in kgCO₂eq.

	Total GHG emissions kgCO ₂ eq.						
	ICEV	HEV	PHEV	BEV	FCEV	FCREx	H ₂ -ICE
Body	3,122	2,906	2,973	3,841	2,831	3,585	3,188
Powertrain System	1,303	1,361	1,552	211	917	552	1,497
Transmission System/Gearbox	566	402	412	532	304	618	656
Chassis	2,112	1,966	2,011	2,598	2,685	2,223	1,697
Traction Motor	0	252	414	1,351	535	533	0
Generator	0	243	392	0	0	0	0
Electronic Controller	0	241	242	872	520	516	0
Hydrogen Storage	0	0	0	0	9,294	9,597	8,534
Li-ion Battery	0	360	2,975	4,890	345	2,529	0
Assembly	729	729	729	729	729	729	729

Table 27: Vehicle Manufacturing and Assembly GHG emissions – 2030, in kgCO₂eq.

	Total GHG emissions kgCO ₂ eq.						
	ICEV	HEV	PHEV	BEV	FCEV	FCREx	H ₂ -ICE
Body	3,045	2,824	2,880	3,707	2,740	3,477	3,114
Powertrain System	1,270	1,323	1,503	204	886	535	1,462
Transmission System/Gearbox	552	391	399	513	294	600	641
Chassis	2,060	1,910	1,948	2,507	2,599	2,156	1,658
Traction Motor	0	245	401	1,304	518	517	0
Generator	0	237	379	0	0	0	0
Electronic Controller	0	234	235	841	503	501	0
Hydrogen Storage	0	0	0	0	8,997	9,308	8,337
Li-ion Battery	0	322	2,605	4,731	297	2,333	0
Assembly	729	729	729	729	729	729	729

Table 28: Vehicle Manufacturing and Assembly GHG emissions – 2035, in kgCO₂eq.

	Total GHG emissions kgCO ₂ eq.						
	ICEV	HEV	PHEV	BEV	FCEV	FCREx	H ₂ -ICE
Body	2,967	2,742	2,787	3,573	2,650	3,369	3,040
Powertrain System	1,238	1,284	1,455	196	857	518	1,428
Transmission System/Gearbox	538	380	386	495	285	581	626
Chassis	2,007	1,855	1,885	2,417	2,513	2,089	1,619
Traction Motor	0	238	388	1,257	500	501	0
Generator	0	230	367	0	0	0	0
Electronic Controller	0	227	227	811	486	485	0
Hydrogen Storage	0	0	0	0	8,699	9,018	8,139
Li-ion Battery	0	290	2,416	4,468	275	2,166	0
Assembly	729	729	729	729	729	729	729

Table 29: Vehicle Manufacturing and Assembly GHG emissions - 2050, in kgCO₂eq.

	Total GHG emissions kgCO ₂ eq.						
	ICEV	HEV	PHEV	BEV	FCEV	FCREx	H ₂ -ICE
Body	2,733	2,495	2,507	3,170	2,378	3,044	2,819
Powertrain System	1,140	1,169	1,309	174	769	469	1,324
Transmission System/Gearbox	495	346	347	439	256	525	580
Chassis	1,849	1,688	1,696	2,145	2,255	1,888	1,501
Traction Motor	0	217	349	1,115	449	453	0
Generator	0	209	330	0	0	0	0
Electronic Controller	0	207	204	720	437	439	0
Hydrogen Storage	0	0	0	0	7,807	8,150	7,547
Li-ion Battery	0	215	1,856	3,679	212	1,666	0
Assembly	729	729	729	729	729	729	729

3.2.1.3 End-of-Life

In this study, the end-of-life (EoL) treatment of vehicles is modelled to reflect European averages, using data from (Chen and Melaina, 2019). Vehicle manufacturing is based on data from (GREET2, 2022), with the assumption that manufacturing waste is recycled, thereby reducing reliance on primary materials. This recycling process is assessed within the LCA using the closed-loop method, which accounts for the avoided burden of producing primary materials.

At the EoL stage, vehicles undergo dismantling, sorting and targeted treatment of various waste streams. Metallic components are recovered and reintegrated as secondary raw materials for comparable-value applications in the automotive sector. Non-recyclable materials are classified into metallic, combustible and non-combustible fractions for appropriate processing. Combustible materials are incinerated for energy recovery, while non-combustible waste is sent to landfill. Certain components require specialised treatment. For example, lithium-ion batteries are extracted for dedicated recycling processes, while parts such as catalytic converters, oil filters, and airbags follow established recycling or disposal pathways.

The treatment of carbon fibre reinforced plastic (CFRP) components in hydrogen storage tanks presents a challenge at EoL due to the limitations of existing recycling technologies. CFRP production methods - mechanical, thermal (pyrolysis, fluidised bed), and chemical (solvolysis) all require the material to be fragmented, leading to a substantial reduction in tensile strength (Zhu et al., 2019). While chemical solvolysis can retain the most strength of all processes (85-98%), it involves toxic solvents and remains technologically immature ((Zhang et al., 2020)). Given hydrogen storage tanks' stringent durability and pressure requirements, the degraded mechanical properties of recycled CFRP make it unsuitable for re-use in like for like applications. Additionally, strict regulatory standards (European Parliament and Council, 2009), (United Nations, 2013), (Vincent and Dissanayake, 2023) and variability in the strength of material outputs in different CFRP recovery processes (Zhu et al., 2019), (Zhang et al., 2020) further complicates the closed-loop recycling systems. Therefore, in this study we assume that hydrogen tanks follow existing EoL pathways (landfill or incineration) and use 100% primary material inputs. Detailed assumptions and processes are provided in Figure 12.

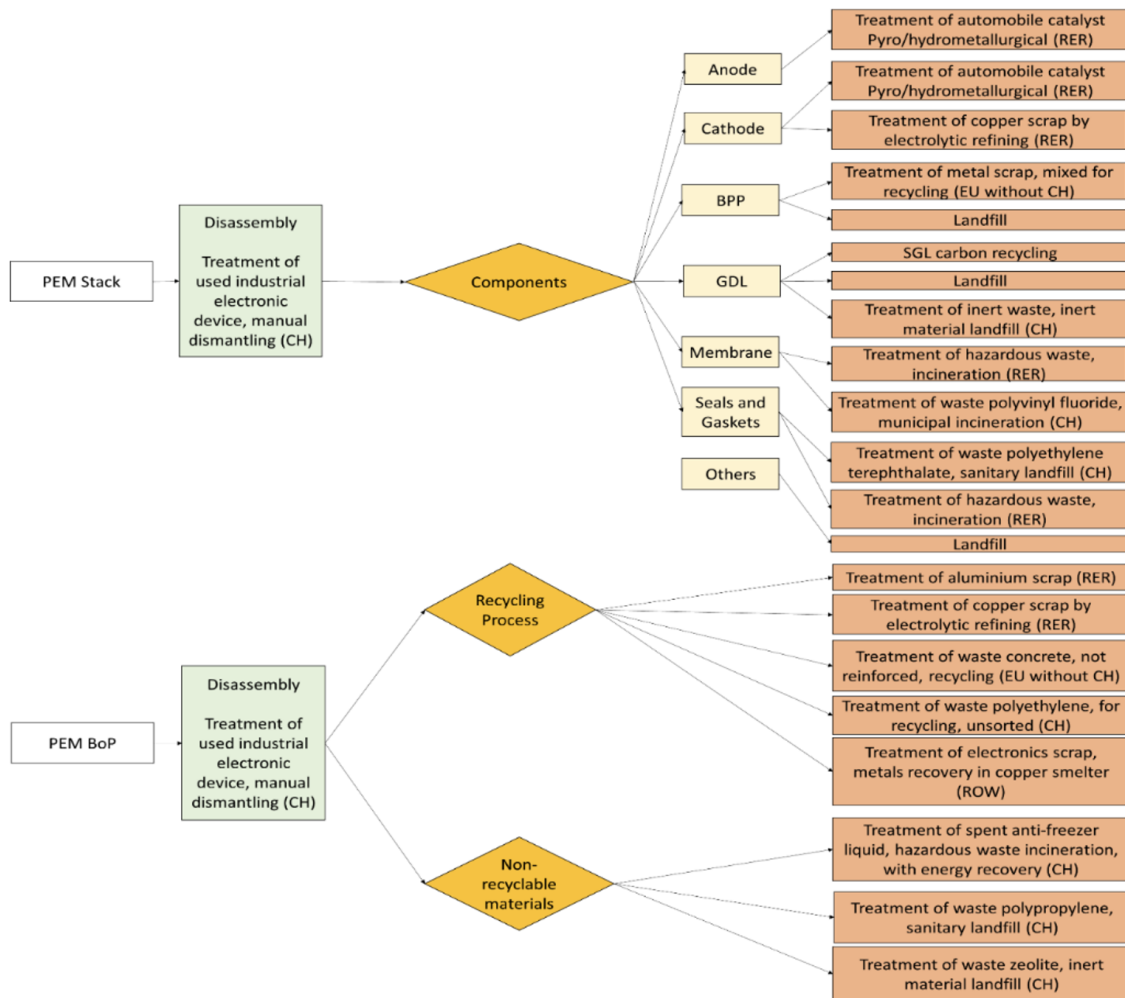


Figure 12: End-of-life boundary (inspired from (Bolin, 2020))

3.2.2 Fuel Cycle

3.2.2.1 Fuel and Electricity Production

Fuel and electricity production is a key part of the cradle-to-grave LCA of SUVs. This study models the production of gasoline, diesel, hydrogen, and electricity, integrating data from (GREET2, 2022) and (Ecoinvent v3.7., 2020).

The carbon intensity of electricity production plays a significant role in determining the lifecycle emissions of BEVs and PHEVs (Bieker, 2021). In 2020, electricity grid emissions in Europe averaged 280 gCO₂/kWh. However, the decarbonisation of the European energy grid, driven by increased penetration of renewable energy sources such as wind and solar, is projected to reduce the carbon intensity to 57 gCO₂/kWh by 2050.

The environmental footprint of hydrogen depends on its production pathway. Green hydrogen, produced via electrolysis powered by renewable energy, has the lowest emissions intensity at approximately 1.06 kgCO₂eq/kg *H*₂ (Duke et al., 2025). In contrast, blue hydrogen, derived from natural gas with carbon capture and storage, results in emissions of 3.4 kgCO₂/kg *H*₂ (GREET2, 2022), while grey hydrogen, produced from natural gas without carbon capture, emits 11.6 kgCO₂/kg *H*₂ (GREET2, 2022).

For ICEVs and hybrid vehicles, gasoline production emits approximately 2.35 kgCO₂/L (GREET2, 2022), while diesel emits 2.66 kgCO₂/L (GREET2, 2022). Biofuels offer a lower-emission alternative to conventional fuels (Department for Transport (DfT), 2025b). Although biofuels are already used as drop-in fuels blended with gasoline/diesel and their share is expected to grow, the lack of a definitive long-term mandate for on-road biofuel use led us to exclude them from this analysis. Additionally, there is no definitive mandate for biofuels to become standard at European fuel pumps (European Parliament and Council, 2023). Due to these limitations, biofuels are omitted from future modelling scenarios.

3.2.2.2 Real-World Driving Impact

Fuel economy across different powertrain technologies were evaluated, considering both regulatory test-cycle figures and real-world driving conditions. Official fuel consumption data is sourced from (GREET2, 2022), while real-world performance adjustments are informed by recent research.

A well-documented gap exists between official and real-world fuel consumption for PHEVs, with real-world CO₂ emissions often exceeding official estimates due to lower-than-expected electric driving shares and suboptimal charging habits (Euro-

pean Commission, 2022). The International Council on Clean Transportation (International Council on Clean Transportation (ICCT), 2022) found that real-world PHEV fuel consumption averages 4.0–4.4 L/100 km for private cars and 7.6–8.4 L/100 km for company cars, significantly higher than Worldwide Harmonised Light Vehicle Test Procedure (WLTP) estimates of 1.6–1.7 L/100 km. Onboard monitoring confirms frequent ICE reliance, widening the real-world gap.

To simulate real-world usage conditions in our scenario analysis, a 250% uplift is applied to official PHEV fuel consumption figures (European Commission, 2022), using the following annual reduction rate formula to project future fuel consumption:

$$r = \left(\frac{F_{\text{end}}}{F_{\text{start}}} \right)^{\frac{1}{n}} - 1 \quad (3.7)$$

Where:

- F_{start} — Fuel consumption in the starting year
- F_{end} — Fuel consumption in the ending year
- n — Number of years between starting and ending year
- r — Annual reduction rate

Table 30 summarises the stated vs real-world adjusted values for PHEV fuel consumption.

Table 30: Stated vs Real-world driving fuel consumption data

Year	Stated (L/100 km)	Real-world (L/100 km)
2020	4.09	10.30
2030	3.67	9.62
2050	3.33	8.39

Since FCREx vehicles operate primarily as BEVs with occasional fuel cell use, their real-world efficiency is expected to align closely with official estimates, unlike PHEVs which exhibit greater variability due to mixed-mode operation. However, due to the novelty of these vehicles, real-world discrepancies have not yet been reported and so provide a key area of future work.

To estimate FCREx fuel economy, we adjusted the gasoline-equivalent fuel economy miles per gallon gasoline equivalent (MPGGE) of a conventional ICEV (22.65

mpgge) using a 284% uplift, based on (GREET, 2018) projections for charge-depleting mode in hydrogen-powered plug-in hybrid fuel cell vehicles (FC PHEV H_2). This adjustment accounts for the efficiency of battery electric operation before the fuel cell takes over to sustain propulsion.

However, GREET-based uplift factors rely on US drive cycles (US Environmental Protection Agency, EPA test conditions), where trip lengths, charging behaviours, and regulatory assumptions differ. Due to the absence of European-specific uplift factors for FCRe vehicles, GREET-based values are used, with the acknowledgement that real-world performance in European contexts may vary. This limitation is considered in the interpretation of results.

3.2.2.3 Fuel Transport and Distribution

This study applies the framework of the (GREET2, 2022) fuel transport and distribution (T&D) model but adjusts key data inputs to reflect European conditions, ensuring alignment with regional fuel distribution practices, emission factors, and electricity grid characteristics. To enhance accuracy, data from the (Ecoinvent v3.7., 2020) database, providing environmental inventory data specific to Europe, was integrated ensuring the model is representative of European infrastructure and supply chain dynamics.

The fuel supply chain for gasoline, diesel, hydrogen, and electricity is comprehensively modelled to capture the emissions associated with their extraction, processing, transportation, and distribution. Gasoline and diesel emissions are calculated using a combination of the (GREET2, 2022) and (Ecoinvent v3.7., 2020) data. Gasoline is transported entirely by truck, with an emissions factor of 0.04 kgCO₂eq/km, based on European logistics data. Diesel transportation uses a mixed-mode approach, with 50% transported by barge and 50% by pipeline, reflecting higher emissions due to longer transport distances and energy-intensive barge operations. Last-mile delivery for both fuels occur exclusively by truck, with similar emissions factors.

Hydrogen supply chains focus on gaseous hydrogen, incorporating emissions from tube-trailer transport and pipeline delivery. Liquid hydrogen-fuelled trucks are excluded due to high energy requirements for liquefaction and boil-off losses, which significantly affect overall supply chain efficiency (Hydrogen Europe, 2024). Additionally, gaseous hydrogen remains the dominant distribution method in Europe, making it a more relevant focus for future emissions modelling (Hydrogen Europe, 2024), (European Parliament, 2021). Hydrogen production emissions are based on steam methane reforming and electrolysis, adjusted for European energy grid factors using (Ecoinvent v3.7., 2020) data.

Electricity supply chains account for grid carbon intensity, which dynamically decreases from 280 gCO₂eq/kWh in 2020 to 57 gCO₂eq/kWh in 2050, based on projections from the (European Environment Agency, 2024). T&D losses of 6–8% are included, with emissions calculated by combining energy consumption per vehicle-kilometre and grid intensity. To calculate the transmission and distribution emissions (kgCO₂eq/km) for electric powertrains, the following formula was applied:

$$\text{T\&D Emissions} = E_{\text{cons}} \cdot \text{Loss}_{\text{T\&D}} \cdot EF_{\text{CO}_2} \quad (3.8)$$

Where:

- E_{cons} — Energy consumption per kilometre (kWh/km)
- $\text{Loss}_{\text{T\&D}}$ — Transmission and distribution loss rate (%)
- EF_{CO_2} — CO₂ emissions factor (kgCO₂/kWh)

While the existing transport methods largely rely on conventional energy sources such as diesel, opportunities for decarbonisation arise through fuel switching, particularly in truck transport. Given the focus of this study on hydrogen transportation and the development of hydrogen powered vehicles, the following section examines hydrogen as the primary alternative fuel for logistics decarbonisation.

3.2.2.4 Fuel Switching for Transport Decarbonisation

The Hydrogen Delivery Scenario Analysis Model (U.S. Department of Energy, 2015) was adapted for European logistics by adjusting delivery distances, grid emissions factors and vehicle transport characteristics. Hydrogen is assumed to be distributed via tube trailer (500 bar high-pressure tanks) initially using conventional diesel-fuelled trucks. However, hydrogen fuel is considered for the future decarbonisation of transport logistics. Additionally, pipelines of 100km or 400km are included to capture both centralised and decentralised hydrogen production (Reuß et al., 2017). For localised hydrogen distribution, an additional 3 km of distribution pipeline per refuelling station is assumed (Reuß et al., 2017).

Hydrogen pipeline construction is modelled using (Ecoinvent v3.7., 2020). Data for natural gas pipelines are adapted to incorporate material specifications specifically designed for hydrogen transport. This includes the application of GALVALUME® coatings (an alloy of aluminium, zinc, and silicon) to prevent hydrogen diffusion (Krieg, 2012). The lifetime of these pipelines is assumed to be 40 years (Wulf et al.,

2018), with associated GHG emissions calculated based on material inputs and energy requirements.

The GHG emissions for hydrogen T&D are calculated using (3.9):

$$\text{T\&D GHG Emissions} = \text{tkm} \cdot H_{2D} \cdot T_w \cdot \left(\frac{\sum (F + H_{2T} + H_{2C} + RM)}{TT_{SC}} \right) \quad (3.9)$$

Where:

- tkm — Mass of hydrogen transported multiplied by distance travelled (ton-kilometre)
- H_{2D} — Emissions associated with lorry fuel consumption for hydrogen delivery
- T_w — Weight of the truck used for transport
- F — GHG emissions from the type of fuel used (e.g., diesel, hydrogen)
- H_{2T} — Emissions related to the hydrogen storage tank
- H_{2C} — Emissions from compressing hydrogen for storage and transport
- RM — Emissions associated with road maintenance
- TT_{SC} — Storage capacity of the tube trailer used to transport hydrogen

3.2.3 Lifecycle Cost Analysis

Life Cycle Cost (LCC) methodology is a comprehensive approach used to evaluate the total cost of a product or system over its entire life cycle (Greeff and Ghoshal, 2004). This includes all costs incurred from the initial acquisition to the disposal of the product. The main objective of LCC analysis is to offer a detailed insight into the generation of costs, thereby facilitating more informed decision-making that considers both short-term and long-term financial impacts. The LCC of vehicles is calculated using Equation 3.10:

$$\text{LCC} = \frac{C_P + C_M + C_O + C_D}{\text{VMT}} \quad (3.10)$$

- C_P — Vehicle purchase cost

- C_M — Maintenance cost
- C_O — Operational costs
- C_D — Vehicle disposal costs
- VMT — Vehicle miles travelled

Operating costs (C_O) are determined by the costs of fuel (C_F), tyres (C_T), and road taxes (C_{RT}), as expressed in Equation 3.11:

$$C_O = C_F + C_T + C_{RT} \quad (3.11)$$

- C_F — Fuel cost
- C_T — Tyre cost
- C_{RT} — Road tax cost

Fuel costs (C_F) are influenced by the fuel economy of the vehicle, as expressed in Equation 3.12.

$$C_F = \frac{F_P}{F_E} \cdot \text{VMT} \quad (3.12)$$

- F_E — Vehicle fuel economy (miles per gallon or miles per GGE)
- VMT — Vehicle miles travelled
- F_P — Fuel price (€/gal or €/gge)

The cost of tyres (C_T) is expressed by Equation 3.13.

$$C_T = \frac{\text{VMT}}{T_l} \cdot n_T \cdot P_T \quad (3.13)$$

- T_l — Tyre lifetime (miles)
- n_T — Number of tyres per vehicle
- P_T — Price per tyre

It is worth noting that vehicle resale values are not considered in the model as we model the total life cycle cost of the vehicle rather than total cost of ownership. Therefore, the assumed value for the vehicle at end-of-life is €0.

3.2.3.1 Abatement Costs

GHG emission abatement cost, expressed as € per tonne of CO₂ avoided, is calculated by Equation 3.14:

$$\text{Abatement Cost} \left(\frac{\text{€}}{\text{tonne CO}_2} \right) = \frac{[C_{\text{capex,ZEV}} - C_{\text{capex,ICEVg}}] + \sum_{t=1}^T \frac{(C_{\text{fuel,ZEV},t} + C_{\text{tax,ZEV},t}) - (C_{\text{fuel,ICEV},t} + C_{\text{tax,ICEV},t})}{(1+r)^t}}{(E_{\text{ICEVg}} - E_{\text{ZEV}}) \cdot \sum_{t=1}^{15} \frac{1}{(1+r)^t}} \quad (3.14)$$

- C_{capex} : Upfront cost at $t = 0$, including purchase price, VAT, and embedded emissions levy.
- $C_{\text{fuel},t}$: Annual fuel cost of the vehicle in year t .
- $C_{\text{tax},t}$: Annual road tax and insurance for the vehicle in year t .
- E_t : Lifecycle CO₂ emissions of the vehicle in year t .
- r : Discount rate (3%).
- T : Time horizon (15 years).

This formulation captures both the cost and emissions differences between zero-emission vehicles (ZEVs) and a baseline gasoline internal combustion engine vehicle (ICEV-g) over a 15-year operational lifespan. The numerator reflects the total life-time cost difference between the ZEV and the ICEV-g. Manufacturing emissions are monetised and incorporated into the total vehicle cost. The denominator represents the discounted cumulative CO₂ emissions savings over time, applying a 3% discount rate to reflect the time value of emissions reductions. This method ensures a more realistic valuation of future environmental benefits. In this analysis, ICEV-g is used as the reference vehicle for comparing abatement costs across BEVs, FCEVs, FCREx, and H₂-ICE configurations.

3.2.3.2 Taxes and Subsidies

In alignment with the European Union's policy direction to mitigate the climate impact of fossil-fuelled vehicles, member states have implemented various strategies, including tax benefits and purchase incentives, to promote the adoption of ZEVs.

Currently, 26 of 27 EU member states offer incentive schemes, with Estonia being the sole exception (ACEA, 2024). This study considers acquisition taxes (VAT and registration cost), ownership taxes (road tax and insurance) and fossil fuel taxes.

While the VAT rate varies across countries, the average rate is approximately 21% so we use this value as a European average. Vehicle registration costs across the EU vary significantly depending on each member state’s taxation policies. These costs are often influenced by factors such as vehicle weight, CO₂ emissions, engine size, or purchase price. Most EU countries do not have registration costs for ZEVs and thus are excluded for these technologies. For hybrid vehicles, fees average at €2,000, while costs for ICEVs are estimated to be €3,500 (Danielis, 2023), (ACEA, 2022). Direct purchase subsidies for ZEVs vary widely but average around €5,000 for BEVs and FCEVs (Danielis, 2023), (ACEA, 2024).

Road tax benefits are common across countries, with ZEVs often paying minimal (typically €30-50) or no annual tax (ACEA, 2024). For ICEVs this is approximately €500 and €200 for hybrids (Carpenter and Antich, 2022). While most countries do not require road taxes for ZEVs, we assumed €0 road tax. However, as the BEV market matures, nations like Germany and the UK are phasing out subsidies and incentives, marking a shift toward market-driven adoption. Consequently, it is assumed that purchase subsidies and tax exemptions will be eliminated across Europe by 2035 and BEVs will begin contributing to road tax (Jacobs, 2024). Hybrid vehicles incur an average road tax of approximately €200 per month (ACEA, 2022) which we assume to be the same for BEVs from 2035 onwards. As hydrogen technologies are more novel, we assume that these technologies will still benefit from subsidies and tax benefits until 2050. Insurance costs across Europe average at €885 (Statista, 2024), (Jacobs, 2024), (Walker, 2024) and are not assumed to change based on technology type.

Finally, fossil fuel costs are heavily influenced by fuel taxation. Taxes constitute a significant portion of fossil fuel prices in Europe, with the average taxation on gasoline and diesel respectively of €0.548/L and €0.445/L (Macumber-Rosin and Hoffer, 2024). In this analysis, fossil fuel taxes are already included in the fuel price at pump. The range in tax levels in electricity taxes between European countries is much wider than fuel excise duties (Carpenter and Antich, 2022). However, the most common taxes are between €0.03 and €0.06/kWh (Carpenter and Antich, 2022), thus we use a median of €0.045/kWh.

To ensure a clear focus on the intrinsic technical costs of vehicle technologies, subsidies and tax incentives are excluded from the core cost modelling to isolate technological factors. Subsidies and taxes predominantly affect consumer costs rather than the fundamental costs of technology deployment. Including them in abatement

cost calculations could inadvertently create misleading conclusions about the cost-effectiveness of a technology, especially when these costs are heavily influenced by policy interventions. Instead, this study considers the impact of subsidies and taxes separately within a broader analysis of total cost of ownership (TCO) from the consumer perspective. This approach enables an assessment of financial support levels required to make low-carbon technologies cost-competitive without conflating policy-driven effects with underlying technical costs.

3.2.4 Scenario Framework and Key Assumptions

This section provides an overview of the key scenario trajectories modelled in the LCA and LCC analysis. All scenarios are applied across the whole modelling period (2020-2050). A complete dataset, including incremental yearly adjustments, detailed cost breakdowns and supporting calculations, is provided in the subsequent section.

3.2.4.1 Vehicle Weight

This scenario focuses on the adoption of lightweight materials and advanced manufacturing techniques to reduce vehicle weight. By substituting traditional materials with high-strength steel, aluminium, and carbon fibre composites, vehicles can achieve significant weight reductions. This leads to improved fuel efficiency, enhanced performance, and reduced greenhouse gas emissions over the vehicle’s lifespan. LCI data is collected from (GREET2, 2022). Overall, the weight of vehicles is projected to decrease by 25-35% in total vehicle weight, with the largest reductions seen in EVs and FCEVs (Table 31).

Table 31: Scenario showing the reduction in vehicle weight over time (kg), sans battery and fluids (GREET2, 2022)

Model Year	ICEV	HEV	PHEV	EV	FCEV
2015	1.736	1.827	1.922	1.645	1.954
2020	1.514	1.571	1.580	1.356	1.480
2025	1.444	1.491	1.499	1.277	1.402
2030	1.416	1.459	1.466	1.245	1.370
2035	1.381	1.411	1.419	1.199	1.324
2050	1.293	1.362	1.369	1.151	1.278

3.2.4.2 Vehicle Battery Storage & Power

As battery chemistries and technology improve, higher capacities, longer mileage range and faster charging times are encompassed in this scenario. In particular, advances in lithium-ion battery technologies are expected to enhance the performance and durability of EV batteries. See Table 32 and 33 for data assumptions.

Table 32: Scenario showing the future changes in battery storage (kWh) for PHEV and EVs (GREET2, 2022)

Year	PHEV	EV
2020	21	84
2025	20	82
2030	18	83
2035	17	75
2050	15	66

Table 33: Scenario showing the reduction in future battery power (kW) for HEVs and FCEVs

Year	HEV	FCEV
2015	46	46
2020	33	40
2025	33	37
2030	33	40
2035	35	38
2050	35	32

3.2.4.3 PGM in Fuel Cells

Advancements in fuel cell designs are part of an ongoing effort to reduce PGMs driven by cost savings motivations (IRENA, 2020) and to address material availability. This scenario tracks the reduction on the reliance of precious metals, particularly platinum which is used as a catalyst. By developing more efficient catalysts and optimising fuel cell architectures, the automotive industry aims to lower production costs and increase the accessibility of fuel cell vehicles.

A literature review was conducted to validate the projections made by (Sun et al., 2011) (see Table 35). As the average current loading of all literature is 25 g/FCEV,

the (GREET, 2023) model is 21 g/FCEV and the future DOE ultimate target is 6 g/FCEV we anticipate the projection made by the authors to be a likely path of reduction (U.S. Department of Energy, 2022b), (U.S. Department of Energy, 2022c). However, we apply a 3-year delay as current (2023) loadings have not yet begun to decrease as anticipated (GREET, 2023) (see Table 34).

Table 34: Target platinum loading (g/FCEV)

Year	(Sun et al., 2011)	Adjusted (3-year delay)
2020	20	20
2021	18	20
2022	16	20
2023	12	20
2024	10	18
2025	9	16
2026	8	12
2027	7	10
2028	6	9
2029	6	8
2030	6	7
2031	6	6
2050	6	6

*Note: values remain the same from 2031 to 2050, hence the values are not shown here.

Table 35: Current and future target of platinum loading, PEMFC vehicles (g/FCEV)

No.	Source	Loading (g/unit)
1	Toyota Mirai	30
2	(Banham et al., 2021)	25
3	(Banham et al., 2021)	35
4	(Hou et al., 2020)	13
5	(Notter et al., 2015)	14
6	(Wittstock et al., 2016)	11
7	(Wittstock et al., 2016)	42
8	(Bauer et al., 2015)	34
9	(GREET, 2023)	21
	<i>Average</i>	<i>25</i>
	<i>DOE 'Ultimate' Target (U.S. Department of Energy, 2022b,c)</i>	<i>6</i>

3.2.4.4 Fuel Cell Stack Size

This scenario considers innovations in fuel cell stack design that make these systems smaller, lighter, and more powerful. Improvements in membrane materials, thermal management, and overall stack configuration contribute to more efficient and compact fuel cells. Projections from the (GREET2, 2022) model are used to assess the reduction in stack size in SUVs from 105 kW (2020) to 86 kW (2050) (see Figure 13).

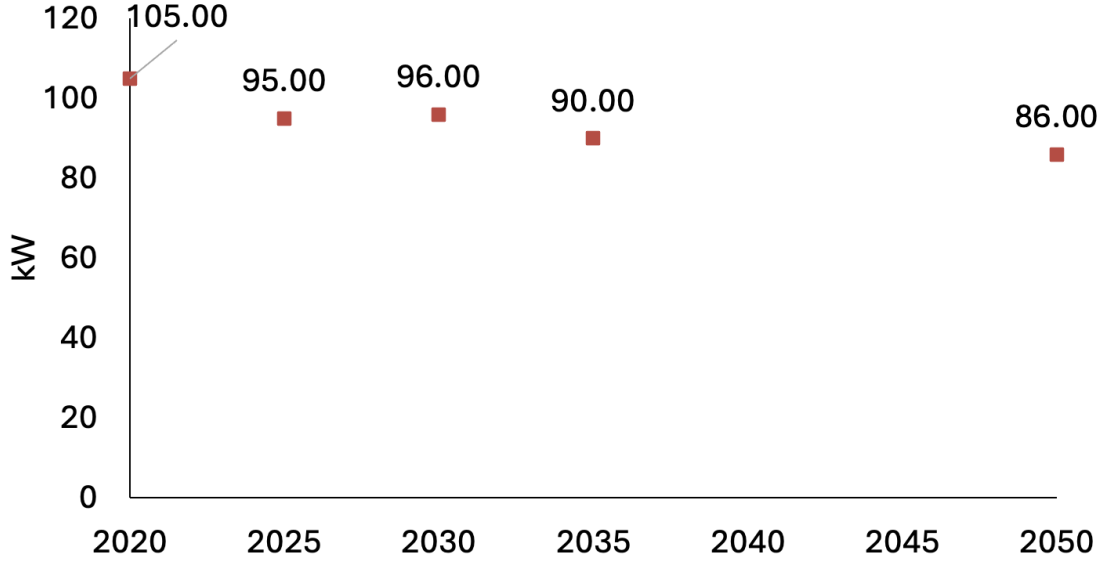


Figure 13: Scenario showing the reduction of fuel cell stacks for FCEVs (kW)

3.2.4.5 Hydrogen Storage Density

Hydrogen storage system parameters were implemented as time-varying inputs in the LCA model, aligned with scenario targets for system gravimetric and volumetric energy capacity. For each modeling year (2020, 2025, 2050), the required storage system mass and volume were calculated based on the amount of usable hydrogen stored onboard (assumed to be 6.33 kg H_2 as aligned with the Hyundai Nexo (H2Live, 2025)). The following relationship was applied:

$$\text{System Mass} = \frac{\text{Usable } H_2 \text{ (kg)}}{\text{Gravimetric Capacity} \left(\frac{\text{kg } H_2}{\text{kg system}} \right)} \quad (3.15)$$

$$\text{System Volume} = \frac{\text{Usable } H_2 \text{ (kg)}}{\text{Volumetric Capacity} \left(\frac{\text{kg } H_2}{\text{L system}} \right)} \quad (3.16)$$

The following targets in Table 36 reflect projected improvements in high-pressure composite tank designs (e.g., 700 bar Type IV) (GREET2, 2022), these values are also closely aligned to the (U.S. Department of Energy, 2017) targets, but fall just short, providing a conservative future outlook.

Table 36: Hydrogen Storage Density for FCEVs (GREET2, 2022); (H2Live, 2025)

Year	Usable H ₂ (kg)	System Mass (kg)	Gravimetric Capacity (kg H ₂ /kg)	Volumetric Capacity (kg/L)	System Volume (L)
2020	6.33	180	0.0352	0.0352	180
2025	6.33	169	0.0375	0.0375	169
2050	6.33	147	0.0431	0.0431	147

3.2.4.6 Vehicle Fuel Economy

This scenario includes innovations in engine efficiency, hybrid and electric powertrains, and aerodynamic designs to enhance fuel economy. The adoption of advanced combustion techniques, energy recovery systems, and lightweight design principles will result in vehicles with lower fuel consumption and reduced operational emissions. (GREET, 2018) provides a time series of improvements in fuel economy for all vehicle powertrains from 2020 to 2050. These projections used in the model are described in Table 37.

Table 37: Fuel economy time series for SUVs (GREET, 2018)

Fuel/Vehicle Type	2020	2025	2030	2035	2050
Gasoline (mpgge)	22.65	23.96	24.70	27.39	29.59
Diesel (mpgge)	27.29	28.75	29.76	31.34	33.38
CIDI Vehicles	120%				
		120%	120%	114%	113%
SI HEV	159%	164%	170%	162%	175%
PHEV Gasoline (CD)	305%	309%	312%	289%	287%
PHEV Gasoline (CS)	145%	161%	164%	154%	158%
CIDI HEV	147%	150%	155%	148%	147%
CIDI PHEV (CD)	299%	303%	305%	283%	282%
CIDI PHEV (CS)	136%	145%	153%	147%	149%
EVs	362%	364%	370%	342%	338%
H ₂ FCV	197%	217%	222%	211%	217%
FC PHEV H ₂ (CD)	284%	289%	292%	273%	274%
FC PHEV H ₂ (CS)	189%	210%	217%	207%	213%

3.2.4.7 Component Cost

This scenario highlights the expected decline in the cost of key automotive components due to economies of scale, technological advancements, and improved manufacturing processes. Lower costs for batteries, electric motors, and fuel cells will make advanced vehicle technologies more affordable, facilitating their broader adoption and reducing overall life cycle costs.

Two trajectories are envisaged – business as usual (BAU) and aggressive. The BAU projection assumes that technology improves over time but not fast enough that vehicle components meet DOE (U.S. Department of Energy, 2022b), (U.S. Department of Energy, 2022c) goals. The aggressive scenario has a more rapid progression meaning that future technology improvements and cost reductions meet DOE (U.S. Department of Energy, 2022b), (U.S. Department of Energy, 2022c) goals.

Price projections for vehicle components are largely sourced from (Grube et al., 2021) and FASTSim ((National Renewable Energy Laboratory, 2018)), with additional information from (Nassif and de Almeida, 2020) and (Advanced Propulsion Centre UK, 2023) where necessary. Cost projections are outlined for BAU in Table 38 and aggressive trajectories in Table 39

Table 38: Cost of various vehicle components following a business as usual (BAU) trajectory (Advanced Propulsion Centre UK, 2023; Grube et al., 2021; Nassif and de Almeida, 2020; National Renewable Energy Laboratory, 2018, 2021)

Year	Fuel Cell (€/kW)	BEV battery (€/kW)	HEV battery (€/kW)	Motor (€/kW)	H ₂ storage (€/kWh)	Transmission (ICEV) (€/kW)
2020	140	150	650	18	18	16
2021	132	147	630	18	18	16
2022	125	144	610	17	16	16
2023	118	142	592	17	14	15
2024	112	139	574	16	12	15
2025	106	136	557	16	11	15
2026	100	133	541	16	10	15
2027	95	130	526	15	10	15
2028	90	127	512	15	10	14
2029	85	125	498	14	10	14
2030	81	122	485	14	10	14
2031	77	119	472	14	10	14
2032	73	116	461	13	10	14
2033	70	113	449	13	9	14
2034	67	110	439	13	9	14
2035	63	108	428	12	9	14

2036	61	105	419	12	9	14
2037	58	102	410	12	9	14
2038	55	99	401	11	9	14
2039	53	96	392	11	9	14
2040	51	93	385	11	9	14
2041	49	91	377	10	9	14
2042	47	88	370	10	9	14
2043	45	85	363	9	9	14
2044	44	82	357	9	9	14
2045	42	79	350	9	9	14
2046	41	76	344	8	9	14
2047	39	74	339	8	9	14
2048	38	71	334	8	9	14
2049	37	68	329	7	9	14
2050	36	65	324	7	9	14

Table 39: Cost of various vehicle components following an aggressive (AGG) reduction trajectory (Advanced Propulsion Centre UK, 2023; Grube et al., 2021; Nassif and de Almeida, 2020; National Renewable Energy Laboratory, 2018, 2021)

Year	Fuel Cell (€/kW)	BEV battery (€/kW)	HEV battery (€/kW)	Motor (€/kW)	H ₂ storage (€/kWh)	Transmission (ICEV) (€/kW)
2020	140	150	650	8	18	14
2021	122	143	606	8	17	14
2022	107	136	567	8	16	14
2023	94	129	532	8	14	14
2024	83	122	500	8	12	13
2025	74	115	472	8	11	13
2026	66	108	446	7	10	13
2027	59	101	423	7	10	13
2028	53	94	402	7	10	13
2029	48	87	383	7	10	13
2030	44	80	366	7	10	13
2031	41	79	351	7	10	13
2032	38	77	338	7	10	12
2033	35	76	325	7	9	12
2034	33	74	314	7	9	12
2035	31	73	304	7	9	12
2036	29	71	295	6	9	12
2037	28	70	287	6	9	12
2038	27	68	279	6	9	12

2039	26	67	273	6	9	11
2040	25	65	267	6	9	11
2041	24	64	261	6	9	11
2042	24	63	257	6	8	11
2043	23	62	252	6	8	11
2044	23	61	248	6	8	11
2045	22	60	245	6	7	11
2046	22	59	242	5	7	11
2047	22	58	239	5	6	10
2048	21	57	236	5	6	10
2049	21	56	234	5	5	10
2050	21	55	232	5	5	10

3.2.4.8 Fuel Prices

This scenario examines the projected fluctuations in fuel prices, encompassing both conventional fossil fuels and alternative energy sources such as hydrogen and renewable fuels. Analysing these trends is essential for evaluating the economic viability and environmental impact of various vehicle types across their entire lifecycle.

Current fuel prices for gasoline and diesel were obtained from the (RAC, 2023). Future projections are informed by the (European Climate Foundation, 2014). (Zhou and Searle, 2022) provide the price of hydrogen mid-level and optimistic scenarios; however, we do not include the 3-euro per kg subsidy that is analysed in this study due to their currently being no policy commitments to suggest this subsidy. Additionally, the AEO (U.S. Energy Information Administration (EIA), 2022), (U.S. Energy Information Administration (EIA), 2023) projections for electricity prices in the transportation sector are utilised. Conventional fuel, electricity and hydrogen price projections are outlined in Tables 40, 42 and 41 respectively.

Two scenarios are presented: a conservative scenario and an aggressive scenario. The conservative scenario assumes a BAU trajectory, characterised by moderate technological advancement (see Table 43). In contrast, the aggressive scenario anticipates significant technological progress, where elevated oil prices are accompanied by a strong emphasis on CO₂ reduction. This focus results in more substantial decreases in the prices of blended and alternative fuels, as well as electricity.

Table 40: Conventional fuel price projections (€/gal) (European Climate Foundation, 2014; RAC, 2023)

Year	Petrol (Conser- vative)	Petrol (Aggres- sive)	Diesel (Conser- vative)	Diesel (Aggres- sive)
2020	6.39	6.83	6.15	6.65
2025	6.54	7.41	6.32	7.15
2030	6.83	7.85	6.48	7.65
2035	6.90	8.14	6.65	8.15
2040	6.98	8.57	6.82	8.65
2050	7.27	9.59	6.98	9.64

Table 41: Hydrogen price projections (€/kg) (Department for Energy Security & Net Zero (DESNZ), 2023; European Parliamentary Research Service, 2025; International Council on Clean Transportation (ICCT), 2022; Zhou and Searle, 2022)

Year	Blue Hydrogen	Green Hydrogen (Conservative)	Green Hydrogen (Aggressive)
2020	11.00	11.00	11.00
2025	8.65	9.00	9.00
2030	6.80	7.00	6.00
2035	5.20	6.50	5.00
2040	4.20	6.00	5.00
2050	3.50	5.00	4.50

Note that at-pump-price of blue hydrogen costs are based on the following assumptions; a LCOH of €6.30 in 2020 (European Parliamentary Research Service, 2025); €3.80 in 2025 (European Parliamentary Research Service, 2025), a transport and compression price of €2.87 (Department for Energy Security & Net Zero (DESNZ), 2023), storage cost of €0.85 (Department for Energy Security & Net Zero (DESNZ), 2023) and 15% tax rate. A future projection of €3.50 is considered feasible based on study by (International Council on Clean Transportation (ICCT), 2022) (due to a lack of data quality and uncertainty this is number is for India not Europe, but is considered reasonable as the (International Council on Clean Transportation (ICCT), 2022) costs for green hydrogen in India are comparable to Europe). Future iterations should update this assumption where data quality allows. Price projections for green hydrogen are provided by (Zhou and Searle, 2022).

3.2.4.9 Electricity Emissions, Utilisation and Price

This scenario addresses the reduction in emissions associated with electric vehicle charging due to the increasing use of renewable energy sources in electricity generation. In 2020 the EU-average grid emission intensity was 280 gCO₂ /kWh and is anticipated to reduce to 114 gCO₂ kWh in 2030 and 57 gCO₂ /kWh by 2050 (European Environment Agency, 2024). As the electricity grid incorporates more renewable energy sources, the carbon footprint of vehicles charged using the grid will decrease, enhancing their overall environmental performance.

In this study we consider varying scenarios centred on the utilisation of public or private electric vehicle charging stations. (ZapMap, 2024) envisage the following consumer charging conditions: (1) mostly at home, (2) often at home and (3) no home charger. The level of utilisation is presented in Figure 14, while the associated costs are detailed in Table 42.

Table 42: Assumed future costs of home and public charging in €/kWh (Shell, 2024; ZapMap, 2024)

Year	At home (price cap)	At home (off- peak)	Public rapid charger (≥50 kW)	Public slow/fast charger (<50 kW)
2020	0.30	0.10	0.95	0.65
2025	0.30	0.10	0.95	0.65
2030	0.30	0.11	0.95	0.65
2035	0.30	0.11	0.95	0.65
2040	0.30	0.11	0.95	0.65
2050	0.30	0.11	0.95	0.65

Table 43: Electricity / Electric fuel charging price projections (€/kWh) (U.S. Energy Information Administration (EIA), 2023; ZapMap, 2024)

Year	Conservative	Aggressive	Price Cap	Off-Peak
2020	0.29	0.29	0.30	0.10
2025	0.29	0.29	0.30	0.10
2030	0.30	0.33	0.30	0.10
2035	0.33	0.34	0.30	0.10
2040	0.34	0.36	0.30	0.10

2050	0.35	0.38	0.30	0.10
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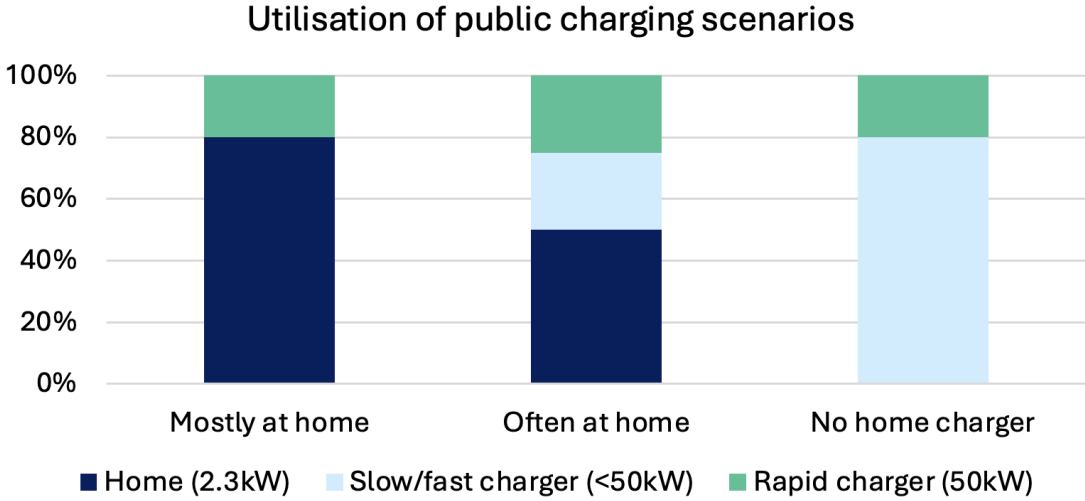


Figure 14: Utilisation of public charger scenarios from (ZapMap, 2024)

3.2.4.10 Hydrogen Transport and Distribution

The (GREET2, 2022) model is used to model the LCI for gaseous hydrogen pipelines and was adapted to reflect a European context. The model considers a pipeline transmission length of 100 km and 400 km, with an additional 3 km of distribution pipeline required per hydrogen refuelling station. For the tube trailer transport, the LCI is based on the (GREET2, 2022) model. EU-average grid electricity is assumed for emissions related to compression processes. Characterisation data were sourced from (Ecoinvent v3.7., 2020). The study employs a functional unit of $\text{kgCO}_2 / \text{kg } H_2$ to ensure consistency in the assessment of environmental impacts. The main process parameters utilised in the study are detailed in Table 44.

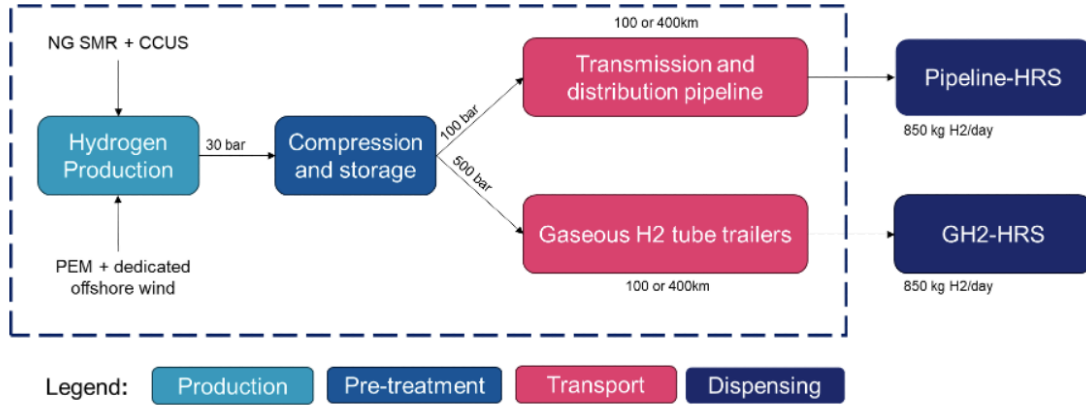


Figure 15: Modelling flowchart for the transport and distribution of gaseous hydrogen

Table 44: Main process parameters for T&D modelling (Akhtar et al., 2021; U.S. Department of Energy, 2022b,c; Wulf et al., 2018)

Parameter	Value	Unit
Electrolysis outlet pressure	30	bar
H_2 pipeline pressure	70–100	bar
Electricity consumption, pipeline compression (30 to 100 bar)	1.05	kWh/kg H_2
Electricity consumption, recompression (every 100 km)	0.02	kWh/kg H_2
Transmission pipeline length	100; 400	km
Distribution pipeline length	3	km/HRS
GH_2 tube trailer distance travelled	100; 400	km
GH_2 tank pressure for trailer	500	bar
Electricity consumption, tube trailer compression (30 to 500 bar)	2.1	kWh/kg H_2
H_2 storage capacity, tube trailer	600	kg H_2
HRS capacity	500	kg H_2 /day

3.2.4.11 Vehicle Production Recyclates

To address sustainability challenges, several automotive manufacturers have committed to enhancing material circularity in vehicle production. For instance, BMW aims to incorporate 50% secondary materials in new models by 2030, supported by closed-loop recycling systems, disassembly-oriented design and sourcing post-consumer recyclates (BMW Group, 2025). These efforts align with emerging regulatory frameworks such as the 2023 EU Battery Regulation (EU 2023/1542), which mandates a digital

battery passport for EV batteries over 2 kWh and sets minimum thresholds for recycled content of 16% cobalt, 85% lead, 6% lithium, and 6% nickel from non-primary sources by 2031. Achieving these targets will depend on significant advances in battery recycling technologies and infrastructure (Harper et al., 2019). Table 45 shows the energy use and GHG emissions of primary and secondary material production for materials used in vehicle production (GREET2, 2022). This study assumes that 50% recycled materials are incorporated into vehicle production by 2050, following a more conservative approach compared to ambitious manufacturers (e.g., BMW).

Table 45: Energy use and GHG emissions of primary and secondary (recycled) material production (GREET2, 2022)

Material	Total Energy (MJ/kg)	Fossil Energy (MJ/kg)	GHG (gCO₂eq/kg)	CO₂ (gCO₂eq/kg)
Virgin Steel	34.35	32.32	3032.44	2833.03
Recycled Steel	11.20	9.56	702.05	645.95
Average Steel	28.24	26.31	2417.22	2255.64
Steel Credit	0.00	0.00	0.00	0.00
Stainless Steel	13.71	11.62	864.53	795.10
Final Iron Product Mix	32.00	31.51	922.63	786.46
Virgin Wrought Aluminum	202.70	140.45	13505.83	12304.73
Recycled Wrought Aluminum	40.90	35.09	2590.49	2361.43
Average Wrought Aluminum	72.74	53.93	4770.31	4347.85
Virgin Cast Aluminum	145.91	99.18	9739.11	8880.29
Recycled Cast Aluminum	24.44	20.08	1544.18	1415.14
Average Cast Aluminum	48.73	35.90	3183.17	2908.17
Cast Aluminum Credit	0.00	0.00	0.00	0.00
Virgin Lead	24.46	23.33	711.35	497.30
Recycled Lead	4.92	4.91	485.34	463.67
Average Lead	4.92	4.91	485.34	463.67
Virgin Nickel	318.59	297.60	22178.70	20434.10
Recycled Nickel	22.41	21.50	1502.07	1385.64
Average Nickel	188.27	176.12	13080.98	12052.78
Virgin Nickel Hydroxide	210.37	195.96	14570.18	13424.56
Recycled Nickel Hydroxide	3.46	3.32	231.65	213.69
Average Nickel Hydroxide	119.33	111.20	8261.22	7611.78
Refined Nickel	318.59	297.60	22178.70	20434.10
Virgin Cobalt Oxide	261.68	235.86	17549.91	16314.99

Virgin Cobalt Chloride	123.88	110.07	8074.60	7512.49
Virgin Cobalt Metal	302.71	266.78	20167.88	18802.28
Copper	59.84	47.95	4031.29	3546.15
Zinc	42.26	33.65	2578.60	2411.75
Virgin Magnesium	256.20	228.48	38046.79	16335.63
Recycled Magnesium	15.93	14.29	25976.85	896.77
Average Magnesium	131.02	116.89	31758.35	8291.98
Sulfuric Acid	0.63	0.59	45.09	42.70
Glass	22.85	21.64	1717.20	1594.50
Fiber Glass	26.58	24.15	1814.84	1680.99
High-Density Polyethylene	78.16	75.19	2049.06	1754.96
Average Plastic	87.00	84.26	3543.74	2944.18
Glass Fiber Composite Plastic	85.66	79.75	4636.82	4201.11
Carbon Fiber Composite Plastic for General Use	250.73	231.19	12178.85	11052.87
Carbon Fiber Composite Plastic for High Pressure Vessels	371.55	333.22	77804.00	16130.70
Rubber	54.77	53.34	3863.58	3593.64
Platinum	1348.25	1291.42	126433.61	119948.82
Vanadium	0.00	0.00	0.00	0.00
Zirconium	209.50	192.07	12877.46	11789.48
Titanium	817.55	726.60	54241.87	50515.17
Chromium	0.00	0.00	0.00	0.00
Rare Earth (Ni-MH)	515.91	465.43	43658.94	37451.83
Final Manganese	23.07	18.66	1432.43	1345.10
Manganese	120.69	118.86	8663.75	8122.11
Thermal Insulation	26.58	24.15	1814.84	1680.99

Carbon Fiber	404.91	358.04	18790.60	17230.57
Carbon Paper	1417.19	1253.13	65767.10	60307.01
Nafion117 Sheet	21.30	18.70	1303.06	1200.45
Nafion Dry Polymer	20.85	18.31	1274.00	1173.54
PTFE	111.98	109.06	7965.35	7465.07
Graphite	99.93	95.01	9005.26	3425.38
Ethylene Carbonate	10.20	9.91	365.07	325.43
Dimethyl Carbonate	37.48	36.45	1355.66	1208.68
Silicon	1622.46	1366.74	123703.34	115507.12
Silicon as anode material	741.71	617.87	56949.21	53134.10
Molybdenum	337.93	325.51	24818.25	23760.26
Nylon 6	123.78	119.83	9633.08	6559.18
Epoxy resin	126.01	122.49	6188.28	5508.90
Alumina	12.27	11.94	976.61	907.32
Gold	1348.25	1291.42	126433.61	119948.82
Zinc oxide	82.83	75.74	6226.38	5921.50
Polyurethane	74.38	71.92	3371.00	2751.44

3.2.4.12 Scenario Framework and Key Assumptions Summary

In summary, the scenarios used throughout this analysis are detailed in Table 46.

Table 46: Summary of scenario trajectories used in the LCA/LCC models

Scenario	Summary	Detailed Data
Vehicle Weight	Major reductions over time e.g., FCEVs: 1954 (2020) $\rightarrow$ 1278 kg (2050)	Table 31
Battery Storage Power	EV battery size declines: 84 kWh (2020) $\rightarrow$ 66 kWh (2050)	Tables 32 and 33
PGM in Fuel Cells	Platinum loading reduces e.g., 20 g/kW (2020) $\rightarrow$ 6 g/kW (2028–2050)	Table 34
Fuel Cell Stack Size	Compact and efficient designs: SUV FCEVs 105 kW (2020) $\rightarrow$ 86 kW (2050)	Figure 13
Hydrogen Storage Density	Improved system efficiency (system mass: 180 kg $\rightarrow$ 147 kg) for 5.5 kg usable H_2 (FCEVs)	Table 36
Vehicle Fuel Economy	Efficiency gains across powertrains e.g., H_2 -ICE 120% (2020) $\rightarrow$ 113% (2050) vs ICEV-g baseline	Table 37
Component Costs	Fuel cell €140/kW (2020) $\rightarrow$ €35/kW (2050) in BAU trajectory, reducing to €21/kW (2050) in aggressive trajectory	Tables 38 and 39
Fuel Prices	Conservative vs. aggressive paths for gasoline, diesel, H_2 , electricity	Tables 40, 42, and 41
Electricity Emissions Charging	CO ₂ intensity decrease: 280 gCO ₂ /kWh (2020) $\rightarrow$ 57 gCO ₂ /kWh (2050)	Figure 14
Hydrogen Transport Distribution	Transition from diesel to hydrogen-powered tube trailers and pipeline delivery systems	Table 44, Figure 15
Vehicle Production	50% recycled materials incorporated into vehicle production by 2050	Table 45

3.3 Tier 3: UK Light-Duty Vehicle Fleet Modelling

This study introduces the UK- H_2 -FLAME model, developed to evaluate the life cycle GHG emissions and life cycle costs associated with the deployment of hydrogen-powered vehicles in the United Kingdom (UK) light-duty vehicle (LDV) fleet between 2025 and 2050. Building on the existing UK-FLAME framework (Davies et al., 2024), this model introduces hydrogen powertrains, hydrogen fuel pathways and hydrogen-specific vehicle components, while also integrating cost analysis into the existing model. The UK- H_2 -FLAME model has eight interconnected modules (Figure 16), enabling dynamic, scenario-based simulations aligned with key UK policy instruments including the Zero Emission Vehicle (ZEV) mandate and the National Hydrogen Strategy (Department for Transport (DfT), 2023), (Department for Energy Security & Net Zero (DESNZ), 2021). To provide a clear foundation for the scenarios that follow, Figure 16 illustrates the overall model framework. The original UK-FLAME model, developed by Davies et al. (2025) consists of six core modules: vehicle, fleet, materials and manufacturing, batteries, fuel and energy, and LCA results. In this study, two additional modules are introduced: a life cycle cost (LCC) module and a hydrogen fuel cell and tank module, which mirrors the structure of the battery module. In addition, key datasets and assumptions have been updated across several existing modules including; vehicles, materials and manufacturing, fuel and energy, and LCA results to reflect hydrogen vehicles in the fleet.

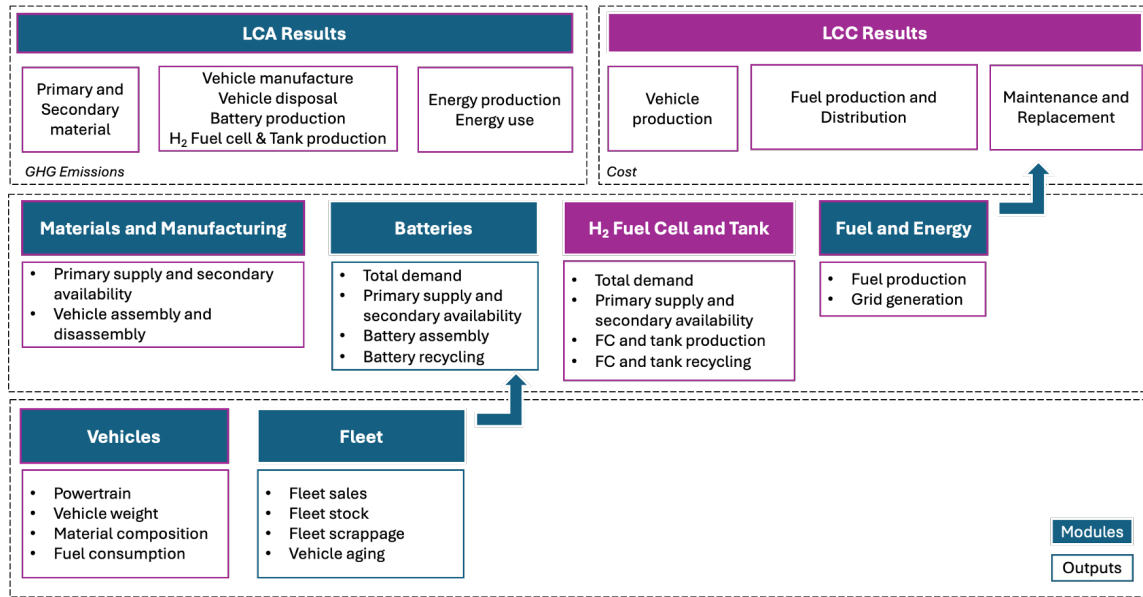


Figure 16: The modular framework of the UK-H₂-FLAME model. Note: new modules are highlighted in purple, while updated modules are outlined in purple.

3.3.1 Technical Model Modules

3.3.1.1 Fleet Composition and Vehicle Technologies

The vehicle module considers eight powertrains for light-duty vehicles: ICEV-petrol, ICEV-diesel, hybrid electric vehicles (HEVs), plug-in hybrids (PHEVs), battery electric vehicles (BEVs), fuel cell electric vehicles (FCEVs), fuel cell range extenders (FCREx), and hydrogen internal combustion engine vehicles (ICEV-H₂). Each vehicle type is characterised by average kerb weight, structural material composition, battery and hydrogen systems sizing (where appropriate) and fuel or energy consumption per 100 km (litres, kWh, or kg H₂). Vehicle characteristics are detailed in Tables 47, 48, 49 and 50.

The fleet module projects the composition of the UK LDV fleet from 2025 to 2050, capturing key trends in vehicle adoption, retirement, and technological transitions. The modelling draws on historic fleet composition data from the UK Driver and Vehicle Licensing Agency (DVLA) vehicle statistics (Department for Transport (DfT), 2022), which provide a comprehensive record of vehicle registrations. Vehicle survival rates are based on observed historical attrition patterns, reflecting the gradual retirement of older vehicles from the fleet due to age-related wear, obsolescence, or scrappage.

Meanwhile, new vehicle sales projections are informed by a combination of policy targets and market assumptions. For the BEV scenario, future uptake is aligned with the UK government’s ZEV mandate, which sets minimum sales targets for zero-emission vehicles through to 2035 (Department for Transport (DfT), 2021). Projections for hydrogen powertrains are derived from scenario assumptions based on infrastructure availability, technology costs, and deployment strategies informed by industry reports and previous studies, e.g. (DNV, 2023). This approach enables the model to capture divergent adoption pathways for BEVs and hydrogen vehicles, reflecting both regulatory commitments and the more speculative nature of hydrogen market development.

Table 47: Kerb weight by vehicle technology, in kg (GREET2, 2022)

Vehicle technology		2020	2030	2050
Car	ICEV	1,444	1,370	1,221
	HEV	1,555	1,466	1,288
	PHEV	1,684	1,577	1,364
	BEV	1,826	1,700	1,448
	FCEV	1,653	1,545	1,329
	FCREx	1,773	1,665	1,449
	ICEV-H ₂	1,580	1,506	1,357
SUV	ICEV	1,761	1,675	1,504
	HEV	1,901	1,797	1,588
	PHEV	2,053	1,928	1,678
	BEV	2,277	2,123	1,816
	FCEV	2,028	1,902	1,651
	FCREx	2,148	2,022	1,771
	ICEV-H ₂	1,897	1,811	1,640

Table 48: Materials composition by vehicle technology, in %(GREET2, 2022)

Material	ICEV	HEV	PHEV	BEV	FCEV	FCREx	ICEV-H ₂
Steel	60.31%	59.81%	59.49%	59.88%	53.97%	55.68%	55.22%
Stainless Steel	0.00%	0.04%	0.04%	0.11%	3.30%	2.18%	0.70%
Cast Iron	1.76%	4.93%	4.99%	0.00%	0.01%	0.00%	1.89%
Wrought Al	4.49%	2.57%	2.45%	2.86%	3.68%	3.78%	4.48%
Cast Al	8.30%	10.00%	10.50%	11.27%	9.20%	7.77%	7.23%
Copper	2.33%	3.75%	4.25%	5.40%	3.41%	3.83%	2.25%
Platinum	0.0003%	0.0004%	0.0004%	0.0000%	0.0014%	0.0008%	0.0003%
Plastics	15.12%	12.33%	11.95%	13.00%	12.21%	13.21%	15.22%
Rubber	4.01%	3.40%	3.27%	3.32%	4.05%	3.36%	3.37%
Glass	2.35%	2.03%	1.93%	2.26%	1.87%	2.23%	2.22%
CFRP	0.00%	0.00%	0.00%	0.00%	5.87%	5.71%	5.69%
Others	1.33%	1.13%	1.13%	1.90%	2.44%	2.24%	1.71%

Table 49: Fuel and energy consumption by vehicle technology in 2025
(Vehicle Certification Agency, 2023), (Candelaresi et al., 2021), (Carpenter and Antich, 2022),(GREET2, 2022)

Vehicle Technology	Fuel (L/100km)	Vehicle Technology	Electricity (kWh/100km)	Vehicle Technology	Hydro-gen H ₂ /100km (kg)
Car					
ICEV-p	6.38	PHEV	19.14	FCEV	1.02
ICEV-d	5.40	BEV	16.10	FCREx	0.66
HEV	5.26	FCREx	22.09	ICEV-H ₂	1.69
PHEV	6.90				
SUV					
ICEV-d	6.70	PHEV	22.26	FCEV	1.20
PHEV	8.64	BEV	17.70	FCREx	0.90
		FCREx	30.25	ICEV-H ₂	2.18

Table 50: Fuel and energy consumption by vehicle technology in 2050
(Vehicle Certification Agency, 2023), (Candelaresi et al., 2021), (Carpenter and Antich, 2022),(GREET2, 2022)

Vehicle Technology	Fuel (L/100km)	Vehicle Technology	Electricity (kWh/100km)	Vehicle Technology	Hydro-gen H ₂ /100km (kg)
<i>Car</i>					
ICEV-p	6.38	PHEV	19.14	FCEV	0.63
ICEV-d	5.40	BEV	16.10	FCREx	0.51
HEV	5.26	FCREx	22.09	ICEV-H ₂	1.38
PHEV	6.90				
<i>SUV</i>					
ICEV-d	6.70	PHEV	22.26	FCEV	0.97
PHEV	8.64	BEV	17.70	FCREx	0.77
		FCREx	30.25	ICEV-H ₂	1.87

3.3.1.2 Materials, Manufacturing and Recycling

Material and manufacturing emissions are linked to the year of vehicle production, while end-of-life treatment is based on scrappage projections. The model includes both primary and secondary material sources, integrating open and closed-loop recycling strategies based on material type and recycling maturity. Structural materials such as aluminium and steel are treated under open-loop assumptions, with end-of-life materials re-entering the general supply chain rather than returning to their original application (Oakdene Hollins, 2021), (Aluminium Association, 2021).

For hydrogen-specific components, PGMs used in fuel cells follow a closed-loop recycling process, with recovered metals reused in new systems. Emissions from PGM recycling are allocated based on market value, reflecting the economic drivers of recovery (Amatayakul and Ramnäs, 2001). Fuel cell manufacturing initially relies heavily on primary PGMs, but over time transitions toward greater use of recycled metals as FCEVs reach retirement age.

For carbon fibre reinforced polymer (CFRP) components, particularly high-pressure hydrogen storage tanks, recycling remains limited due to technical and economic barriers. In this study, a fixed open-loop recycling rate of 5% is applied, reflecting current commercial recycling practices for CFRP materials. This assumes that recovered carbon fibres are downcycled into lower-grade applications (e.g., fillers or non-structural composites), rather than re-used in closed-loop systems for new hy-

drogen storage tanks. Emission factors for primary and recycled CFRP production are sourced from (Ecoinvent v3.7., 2020), with recycled material reflecting secondary processing of carbon fibres for non-structural use. Battery-related methods follow (Davies et al., 2024).

3.3.1.3 Vehicle Operation and Energy Demand

All LDVs are assumed to travel a fleet-average of 14,000 km per year, based on recent travel data (Department for Transport (DfT), 2023). Energy consumption is calculated using vehicle-specific energy efficiencies, differentiated by powertrain and fuel type. Operational emissions are calculated on an annual basis by combining energy demand with fuel-specific emission factors, drawing on the supply-side assumptions outlined in Section 3.3.1.1. This approach ensures consistency between vehicle energy demand and the dynamic emissions characteristics of upstream fuel supply, capturing both technological change in the vehicle fleet and decarbonisation within the energy system.

3.3.2 Scenario Framework and Assumptions

3.3.2.1 Hydrogen Vehicle Technologies

This study extends the UK-FLAME fleet model to include three hydrogen-powered vehicle configurations: hydrogen internal combustion engine (ICEV- H_2), fuel cell electric vehicles (FCEV) and fuel cell range extender (FCREx) vehicles. These technologies vary in drivetrain configuration, material requirements and components. While the previous vehicle-level analysis in Section 3.2 focused on SUVs, the present study expands the scope to include both cars and SUVs. Vehicle specifications have been adjusted accordingly to reflect segment-specific differences in kerb weight, fuel and energy consumption, material requirements and component sizing. Assumptions from the SUV analysis are used as inputs into the fleet model and key differences between cars and SUVs are highlighted in subsequent sections. These specifications form the basis of the life cycle inventory for each vehicle type, which feeds directly into the vehicles, H_2 fuel cell and tank, batteries, and fuel and energy modules. The modelling approach builds on the UK-FLAME framework by adapting its structure and input parameters to accommodate an expanded technology scope. Specifically, the original definitions of vehicle categories, fleet composition, material flows and energy demand are used as foundational inputs. These include kerb weight, material composition, and fuel economy values for five conventional powertrain types (ICEV-p, ICEV-d, HEV, PHEV, and BEV), along with DVLA fleet statistics, manufacturing

and end-of-life treatment assumptions, and fuel and electricity consumption profiles. To reflect emerging hydrogen technologies, this study introduces hydrogen vehicles, updates material inventories and adds two new modules: life cycle cost and hydrogen fuel cell and tank systems. These adaptations preserve consistency with the original UK-FLAME model, which offers a robust and comprehensive structure for fleet-level analysis. However, as the previous model did not include hydrogen technologies, this study extends its scope to enable a more holistic assessment of the UK’s light-duty vehicle fleet transition. Fuel cells rely on PGMs, primarily platinum (Pt) and palladium (Pd) as catalysts. Reducing PGM use is therefore a key technological priority. Current research focuses on lowering Pt loading through alloy catalysts, non-precious metal alternatives, and improved membrane designs. Based on projections by (Sun et al., 2011), platinum loading per FCEV is expected to decline from 20 g in 2020 to 6 g by 2030. However, reflecting slower-than-anticipated market development and technological maturity, this study applies an adjusted scenario with a three-year delay, in which Pt loading reaches 6 g/FCEV only by 2033. See Figure 17 for more detail.

Unlike hydrogen FCEVs, ICEV- H_2 s are not entirely emissions-free as they produce nitrogen oxides (NOx) during the combustion process, which react with other atmospheric compounds to form ozone and particulate matter, which can negatively impact air quality (World Platinum Investment Council, 2024). Like diesel engines, ICEV- H_2 vehicles are fitted with catalytic converters to reduce NOx emissions. ICEV- H_2 adapt traditional ICEs to run on hydrogen, requiring minor modifications such as tungsten carbide-hardened valve seats (Candelaresi et al., 2021). However, as minimal modifications are required and do not significantly impact LCA outcomes, they are not modelled (Stepien, Zbigniew, 2021).

All hydrogen vehicle types use high-pressure CFRP tanks. The recycling rate for carbon fibre reinforced polymer (CFRP) used in hydrogen tanks is currently very low, with estimates placing it at 5-10% due to the technical challenges involved with effectively separating and reusing the carbon fibres from the resin matrix (Zhang et al., 2020). As there is little insight on the technical improvements related to the recyclability of CFRP for hydrogen vehicle applications, we keep this fixed at 5%. The reduction of PGMs focuses on non-PGM or alloy catalysts to lower loading and improve recycling and recovery of PGMs to reduce dependence on primary materials (Miller et al., 2022). Two main recycling approaches are used to recover PGMs from end-of-life fuel cells: pyrometallurgical and hydrometallurgical. Pyrometallurgy is an emission-intensive process, while hydrometallurgy can recover more PGM material with lower emissions (Yakoumis et al., 2021). Mass allocation is used to distribute emission impacts across recovered materials (Santero and Hendry, 2016). However,

for precious metals such as PGMs, a market-based allocation assigns the environmental burden based on the economic value of the materials, which aligns with the primary motivation for mining PGM ores (Amatayakul and Ramnäs, 2001).

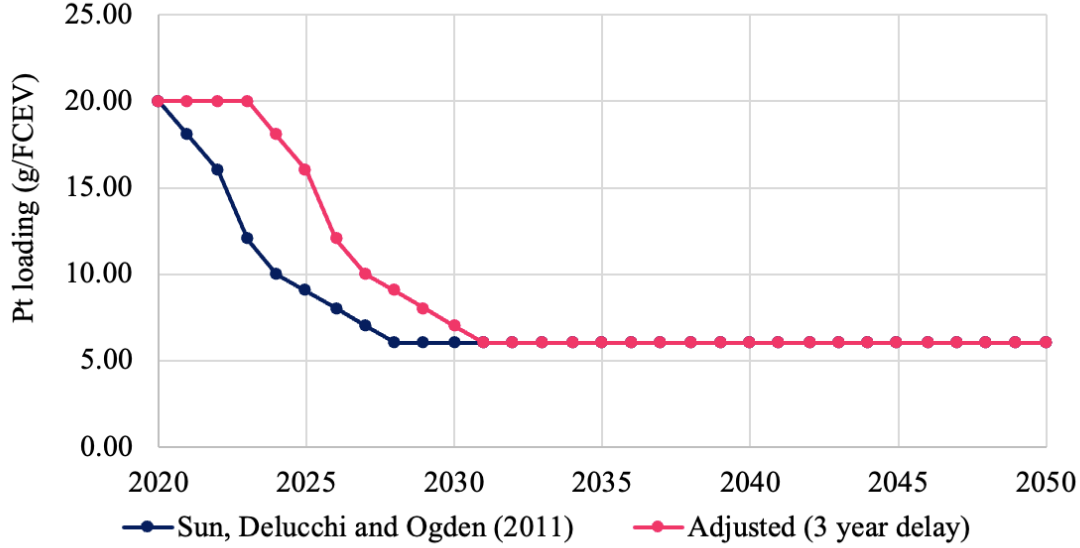


Figure 17: Target platinum loading in fuel cells for vehicles

3.3.2.2 Fleet Composition Scenarios

This section outlines the scenarios considered in the model, including fleet projections, technological advancements, cost evolution, and the decarbonisation of energy sources. The model is designed to accommodate various input parameters and scenario combinations, allowing for a flexible assessment of potential outcomes for the UK LDV fleet. Previous study has examined the dynamics of fleet electrification and the demand for traction batteries (Davies et al., 2024). Consequently, this analysis focuses on the integration of hydrogen-powered vehicle powertrains within the UK LDV fleet, assessing their implications for GHG emissions and total cost.

To explore how different vehicle technologies might evolve under changing policies and market conditions, we outline four fleet market scenarios: Base Case (A), High Hydrogen (B), Major Hydrogen (C), and Transitionary (D), shown in Figure 18 and Table 51. These scenarios draw on projections from (DNV, 2023), (UKH2Mobility, 2013), and the ZEV mandate, which sets a ban on new internal combustion engine

vehicles from 2035 (Department for Transport (DfT), 2023). Each scenario reflects a different pathway for the adoption of technologies like BEVs, FCEVs and other hydrogen-based powertrains. In the Base Case (A), the market follows the ZEV mandate closely. BEV uptake grows steadily, reaching 79% by 2030 and 95% by 2050, while FCEVs remain a niche technology, making up just 5% of the market by 2050 (DNV, 2023). This scenario aligns with current trends: UK FCEV registrations are still extremely low, with only 10 in 2022, 25 in 2023, and 65 in 2024; out of roughly 2.5 million new car registrations each year (Clean Hydrogen Observatory, 2025). Reflecting this, the model starts with 0% FCEV share in 2025 and rises only slightly to 0.002% in 2026 (Base Case). The High Hydrogen (B) scenario assumes stronger support for hydrogen vehicles, reaching a 10% market share by 2050, split evenly between FCEVs and FCREx. The Major Hydrogen (C) scenario takes this further, with BEV adoption peaking at 74% in 2030 before declining to 70% by 2050, as hydrogen vehicles, including FCEVs, FCREx, and ICEV- H_2 grow to 30%. This scenario builds on earlier projections from (European Climate Foundation, 2014), adjusted with a 10-year delay to reflect slower than expected rollout. Finally, the Transitional (D) scenario maps a more gradual shift, with ICEV- H_2 vehicles peaking at 8% by 2040, followed by a modest rise in FCREx and FCEVs to a combined 7% by 2050.

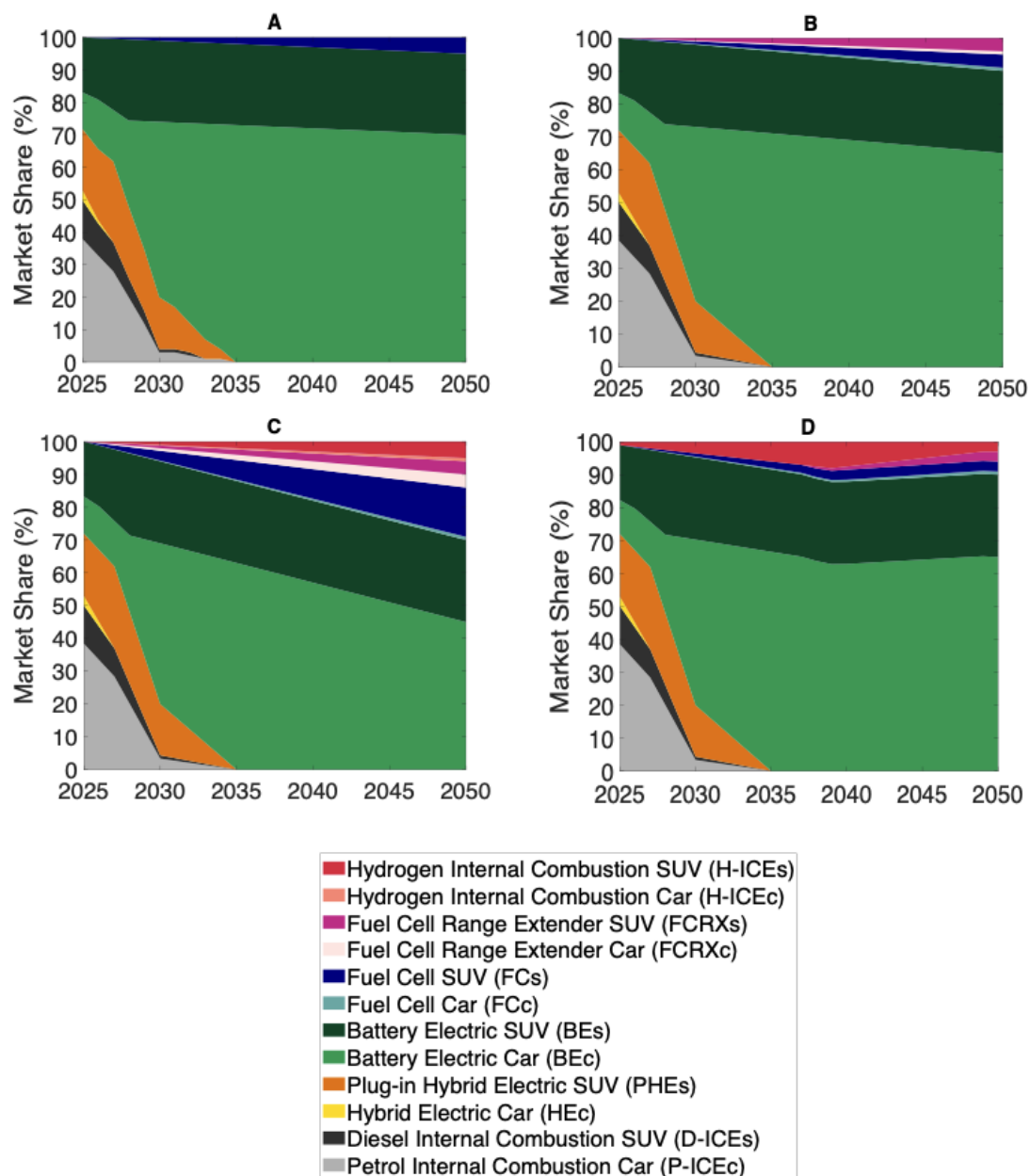


Figure 18: Projection of the UK LDV fleet composition, by vehicle technology

Table 51: Summary of fleet decarbonisation scenarios modelled in UK- H_2 -FLAME

Scenario name	H_2 vehicle sales share (2050)	Hydrogen supply	Grid mix	Technology pathway
Base	5%	Hydrogen Evolution Scenario (see Section 3.3.2.5)	Hydrogen Evolution Scenario (see Figure 19)	Niche FCEV car uptake
Accelerated H_2	10%			Gradual H_2 vehicle deployment (all technologies)
Major H_2	30%			Large-scale H_2 vehicle rollout (all technologies)
Transitional	7%			ICEV- H_2 as a transitional technology enabling early adoption and ramp-up to FCEV/FCREx

3.3.2.3 Vehicle Component Scenarios

The GHG emissions associated with key vehicle components were assessed from 2025 to 2050, expressed per functional unit relevant to each component (e.g., kgCO₂ per kW or per unit). Emissions data were primarily sourced from Ecoinvent 3.7 and academic literature. Table 52 and Table 53 details component-specific emissions across the time horizon. In addition to the baseline trajectory, an Improved Manufacturing scenario was modelled for fuel cells and hydrogen storage tanks (Table 53), reflecting potential reductions in embodied emissions through manufacturing process optimisation, critical material reductions and end-of-life recovery. This alternative pathway enables sensitivity testing of upstream manufacturing impacts on total vehicle emissions, particularly for components with high embodied carbon.

Cost factors for vehicle components were derived from multiple sources, including (Grube et al., 2021), (Houchins and James, 2019), and (Saxena et al., 2023). Maintenance and repair costs were based on (Grube et al., 2021), disaggregated by powertrain type and vehicle class. Detailed assumptions are summarised in Table 54.

Table 52: Summary of vehicle component GHG emission factors used in UK-*H*₂-FLAME, static from 2025 to 2050

Vehicle Component	Emissions Factor	Unit
Motor	3.75	kgCO ₂ /kW
Glider	5	kgCO ₂ /kg
Engine	10	kgCO ₂ /kW
Tyres	3.44	kgCO ₂ /replacement

Table 53: GHG emission factors for hydrogen vehicle components (2020 to 2050)

Scenario	Component	2020	2030	2050	Unit	Data source
Base	Fuel Cell	30.00	25.33	16.00	kgCO ₂ /kW	(Sternberg et al., 2019)
	H ₂ Storage Tank	29.68	28.16	25.11	kgCO ₂ /kWh	(Ye and Lu, 2023) (2023)
Improved Manufacturing	Fuel Cell	24.00	20.67	14.01	kgCO ₂ /kW	(Sternberg et al., 2019)
	H ₂ Storage Tank	20.10	19.10	17.10	kgCO ₂ /kWh	(Sternberg et al., 2019)

Table 54: Cost assumptions for hydrogen vehicle components and maintenance (2020–2050)

Scenario	Component	2020	2030	2050	Unit	Data source
Vehicle Components						
Base	Fuel Cell	89.00	50.00	16.00	£/kW	(Advanced Propulsion Centre UK, 2020)
	H ₂ Storage Tank	15.80	7.86	7.51	£/kWh	(Houchins et al., 2023), (Advanced Propulsion Centre UK, 2020)
Improved Manufacturing	Fuel Cell	89.00	31.60	15.80	£/kW	(Advanced Propulsion Centre UK, 2020)
	H ₂ Storage Tank	15.66	8.35	4.35	£/kWh	(National Renewable Energy Laboratory, 2018), (National Renewable Energy Laboratory, 2021)
Maintenance Costs (Fixed from 2020 to 2050)						
Vehicle Type	Car (£)	SUV (£)		Unit		Data Source
ICEV-g	887	1,103		£/annum		Grube (2021)
ICEV-d	1,021	1,269		£/annum		Grube (2021)
HEV	824	975		£/annum		Grube (2021)
PHEV	785	911		£/annum		Grube (2021)
BEV	680	1,081		£/annum		Grube (2021)
FCEV	900	1,395		£/annum		Grube (2021)
FCREx	785	911		£/annum		Assumed same as PHEV
ICEV-H ₂	887	1,103		£/annum		Assumed same as ICEV-g/d

3.3.2.4 Hydrogen, Electricity and Fuel Supply

Detailed representations of upstream energy supply pathways are incorporated in the models to account for the evolving emissions intensity of different fuels over time. Emissions factors for hydrogen and electricity supply are modelled dynamically, reflecting anticipated technological transitions and decarbonisation trajectories, while conventional fuels are assumed to exhibit static emissions due to technological maturity.

Hydrogen production is modelled using six distinct pathways: brown (biomass gasification), grey (natural gas reforming), blue (natural gas with carbon capture and storage), green (renewable electrolysis), yellow (grid-connected electrolysis), and pink (nuclear electrolysis). Each pathway is assigned a specific emission factor, derived from Oxford Institute for Energy Studies projections (Patonia and Poudineh, 2022) and supplemented with recent literature sources to ensure accuracy and technological relevance. The supply mix of hydrogen evolves over the simulation period (2025-2050), reflecting infrastructure deployment, market developments, and policy direction outlined in the National Energy System Operator (NESO) future energy scenarios (FES) (National Energy System Operator, 2024). Thus, the model captures the transition to low-carbon production technologies.

Electricity supply is modelled using NESO FES (National Energy System Operator, 2024), capturing decarbonisation of the UK grid with increasing shares of wind, solar, and nuclear power. Grid intensity and price signals are applied dynamically over the simulation period and used both for direct BEV charging and electricity-based hydrogen production (yellow and pink hydrogen).

Emissions factors for petrol and diesel are held constant, consistent with their technological maturity and limited scope for further decarbonisation. Cost values are likewise fixed in real terms after 2030, aligning with market expectations of price stabilisation. Further information on input values, assumptions, and data sources is provided in Section 3.3.2.5.

3.3.2.5 Fuel and Energy Supply Scenarios

NESO outlines four decarbonisation pathways in its future energy scenarios, projecting potential transitions to net-zero by 2050 (National Energy System Operator, 2024). Of these, three - Holistic Transition, Electric Engagement, and Hydrogen Evolution align with national climate targets, while the Counterfactual scenario assumes minimal climate action and fails to meet emissions goals.

This study adopts the Hydrogen Evolution scenario, representing a high hydrogen uptake pathway supported by extensive infrastructure rollout and broad application

of hydrogen across transport, industry, and heating sectors. The scenario is characterised by an early reliance on yellow hydrogen (grid-connected electrolysis) and blue hydrogen (natural gas SMR with CCS) to scale supply rapidly and meet initial demand. This reflects the relative commercial readiness of electrolysis using existing electricity grids and the availability of natural gas reforming infrastructure with CCS retrofitting. In contrast, green hydrogen (electrolysis powered by dedicated renewables) sees more limited deployment.

Figure 19 illustrates the projected hydrogen production mix under this scenario, showing the dominance of yellow and blue hydrogen in early years, with green hydrogen remaining marginal until 2050. These shares directly inform the lifecycle emissions and cost modelling in UK- H_2 -FLAME. As the UK electricity grid progressively decarbonises over the study period, the emissions intensity of yellow hydrogen (produced via grid-connected electrolysis) declines accordingly. While initially reliant on a mixed generation portfolio, yellow hydrogen effectively ‘*greens*’ over time, reflecting the rising share of renewables in the grid mix (Figure 21). These dynamics are captured in the lifecycle modelling, where yellow hydrogen transitions from a moderately carbon-intensive input to a near-zero emissions source by 2050.

Electricity supply assumptions are derived from the same FES projections. Figure 20 shows the evolving UK electricity generation mix from 2025 to 2050, highlighting the declining role of fossil fuels and the increasing contribution of wind, solar, nuclear, and biomass. This shift is further quantified in Figure 21, which tracks the share of renewable energy in the grid mix, rising from under 60% in 2025 to over 80% by 2050. These trends directly affect the emissions intensity of electricity used for hydrogen production and vehicle charging, and are reflected in the scenario-specific lifecycle modelling framework.

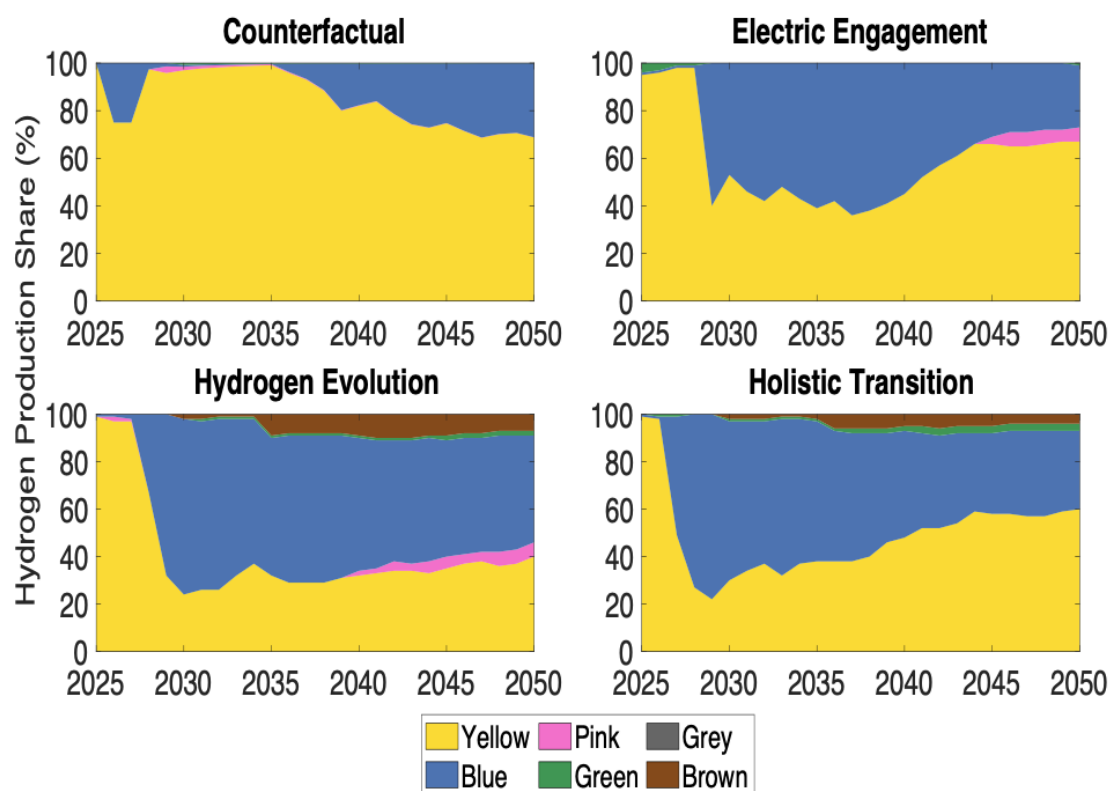


Figure 19: Hydrogen production share by technology type (2025–2050) under the NESO FES (National Energy System Operator, 2024). Note: yellow hydrogen is networked (grid connected) electrolysis and green hydrogen is non-networked electrolysis

Table 55: GHG Emission Factors for hydrogen and fuel production methods (2025–2050).
Values shown in kgCO₂ per kg H₂, litre of fuel, or kWh of electricity

Year	Hydrogen Production Method	GHG Emissions (kgCO ₂ /kg H ₂)
2025	Blue	3.85
2030	Blue	3.85
2050	Blue	3.85
2025	Green	1.20
2030	Green	1.20
2050	Green	1.20
2025	Yellow	4.81
2030	Yellow	2.66
2050	Yellow	0.36
2025	Brown	2.50
2030	Brown	2.50
2050	Brown	2.50
2025	Pink	0.20
2030	Pink	0.20
2050	Pink	0.20
2025	Grey	11.00
2030	Grey	11.00
2050	Grey	11.00
Year	Fuel Production Method	GHG Emissions (kgCO ₂ /L or kgCO ₂ /kWh)
2025	Petrol	0.681
2030	Petrol	0.681

2050	Petrol	0.681
2025	Diesel	0.532
2030	Diesel	0.532
2050	Diesel	0.532
2025	Electricity	0.087
2030	Electricity	0.061
2050	Electricity	0.025

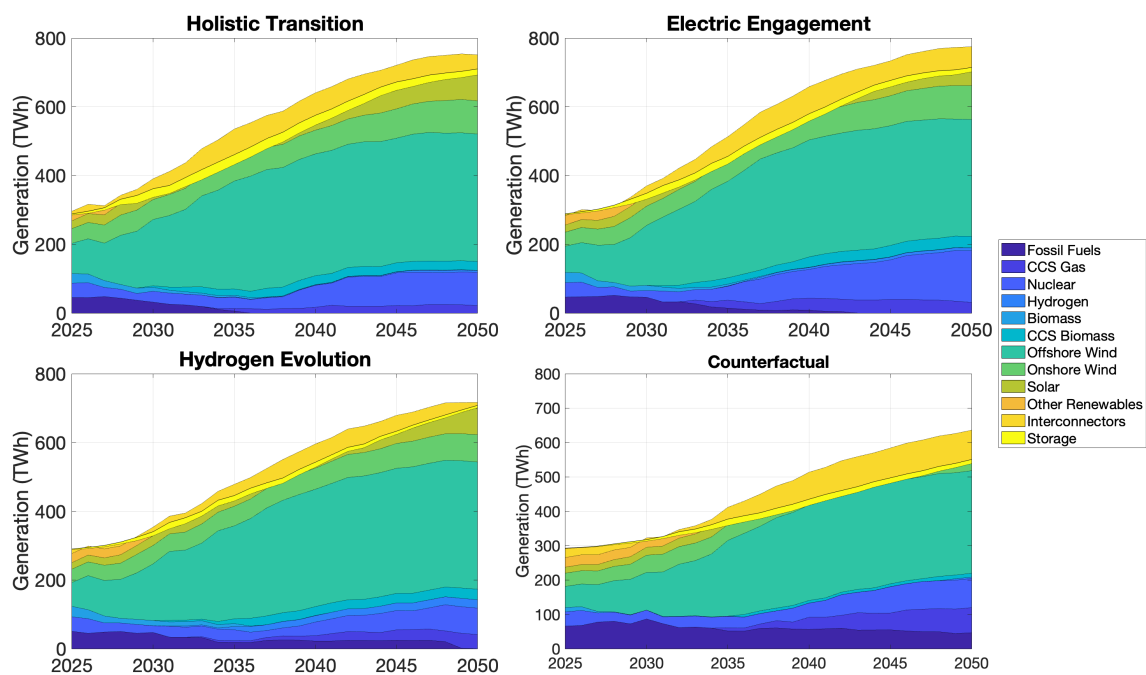


Figure 20: UK electricity generation mix from 2025 to 2050 (data sourced from (National Energy System Operator, 2024))

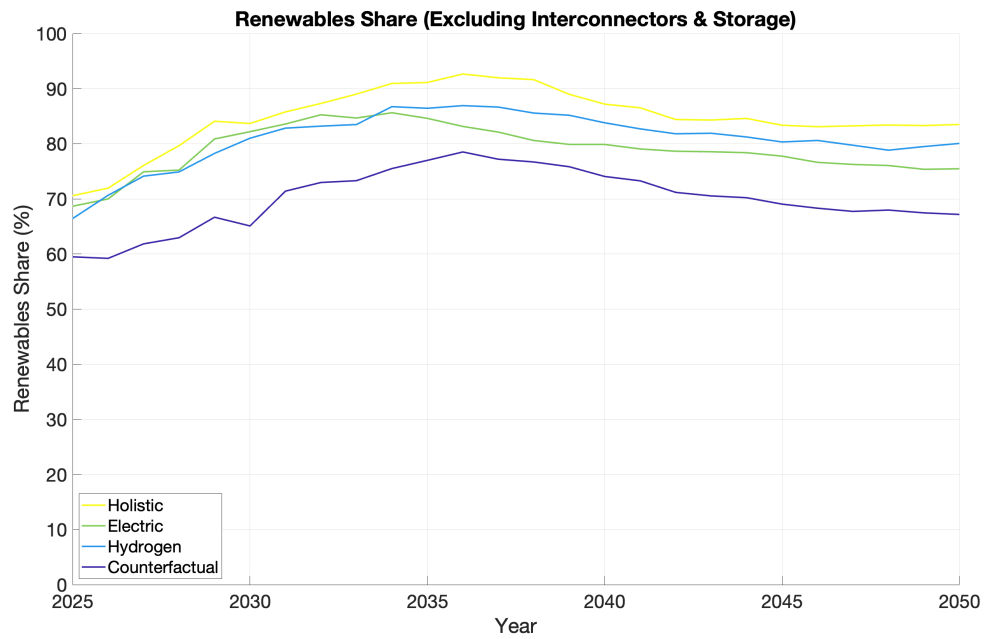


Figure 21: UK electricity generation share of renewable energy in the grid mix from 2025 to 2050 (data sourced from (National Energy System Operator, 2024))

3.3.2.6 Fuel and Energy Costs

In parallel with emissions modelling, UK- H_2 -FLAME dynamically tracks fuel cost trajectories to inform operational expenditure estimates across all powertrain types. These cost inputs are scenario-dependent and vary by fuel type and production method. A recent analysis by (Department for Energy Security and Net Zero and Department for Business, Energy and Industrial Strategy, 2021) highlights that hydrogen fuel costs differ significantly depending on the production pathway (see Table 57). In the early years, grid-powered electrolysis (yellow hydrogen) is the most expensive, costing approximately £6.10/kg in 2025. As renewable energy deployment accelerates and the costs of electrolyzers and wind turbines decline, green hydrogen becomes increasingly competitive, dropping from £3.30/kg in 2030 to £2.50/kg by 2050. In the model, blue hydrogen, produced via natural gas reforming with carbon capture, remains relatively stable at around £3/kg by 2050, although its future cost may be sensitive to carbon pricing and infrastructure investment ((UK Government (UK Gov), 2021), (International Council on Clean Transportation (ICCT), 2022)). Due to limited data on current and projected refuelling costs for hydrogen vehicles in the UK, production prices are used to estimate “at-pump” refuelling costs. According to (Cadent, 2022), production accounts for the largest share of the final price (37%), followed by VAT (17%), large-scale storage (14%), and refuelling station infrastructure (13%). The remaining 19% comprises purification, compression, distribution, and on-site storage. To reflect this breakdown, UK- H_2 -FLAME separates hydrogen fuel costs into production costs and use-phase costs. Production costs represent the upstream cost of generating hydrogen (Table 56). While use-phase costs are incurred after the hydrogen is produced and include storage, transport and distribution, dispensing infrastructure, taxes and VAT. These costs are linked directly to the final price paid by consumers. Separating these costs in the modelling allows us to isolate costs from production and downstream logistics. Historical fuel production costs were sourced from the (RAC Foundation, 2025), with future projections from (Autotraveler, 2025). Petrol and diesel prices are assumed to plateau from 2030 onwards at £1.53/L and £1.63/L respectively, consistent with (RAC Foundation, 2025) projections and (UK Government (UK Gov), 2021) market outlooks. This assumption reflects fuel phase-out policies and reduced consumption volumes under the ZEV mandate.

Electricity costs, which affect both BEV charging and grid-powered hydrogen production, are based on (National Energy System Operator, 2024) Hydrogen Evolution pathway and reflect average retail tariffs. Prices rise slightly to £0.22/kWh in 2030 due to peak demand and infrastructure costs, then fall back to £0.19/kWh by 2050 as decarbonisation and renewables expansion increase efficiency ((ZapMap,

2025), (European Commission, 2024)). Electricity tariffs for home charging were based on (European Commission, 2024) data, while public charging prices reflect (ZapMap, 2025) averages. Table 56 summarises all fuel and energy cost inputs used in the model. These values are applied consistently across scenarios and years to calculate annualised life cycle costs (see Section 6.2).

Table 56: Cost factors for fuel and electricity production methods (2025–2050)

Fuel Type	2020	2030	2050	Unit
Petrol	0.56	0.53	0.53	£/litre
Diesel	0.57	0.53	0.53	£/litre
Electricity (grid)	0.06	0.07	0.08	£/kWh
Blue H ₂	3.00	3.00	3.00	£/kg H ₂
Green H ₂	4.50	3.32	2.52	£/kg H ₂
Yellow H ₂	6.11	5.82	5.57	£/kg H ₂
Brown H ₂	2.49	2.49	-0.16	£/kg H ₂
Pink H ₂	2.49	2.49	2.49	£/kg H ₂
Grey H ₂	1.30	1.30	1.30	£/kg H ₂

Table 57: Refuelling price for the consumer ('at pump') in £/unit

Fuel Type	2020	2030	2050	Unit	Source
Petrol	1.56	1.53	1.53	£/litre	RAC (2025)
Diesel	1.67	1.63	1.63		
Electricity (domestic)	0.19	0.22	0.19	£/kWh	ZapMap (2025)
Electricity (public, slow/fast)	0.52	0.52	0.52		ZapMap (2025)
Electricity (public, rapid)	0.76	0.76	0.76		ZapMap (2025)
Blue H ₂	11.60	10.74	7.30	£/kg H ₂	UK Government (2021); Cadent (2022)
Green H ₂	13.10	11.06	6.82		
Yellow H ₂	14.71	13.56	9.87		
Brown H ₂	11.09	10.23	4.14		
Pink H ₂	11.09	10.23	6.79		
Grey H ₂	9.90	9.04	5.60		

3.3.3 Lifecycle Assessment and Cost Calculations

Fleet-wide life cycle GHG emissions are quantified in metric tonnes CO₂-equivalent, applying 100-year global warming potentials from the IPCC Fifth Assessment Report (Stocker et al., 2013). Emissions factors for materials and processes are primarily sourced from ecoinvent 3.7 (Ecoinvent v3.7., 2020). Battery manufacturing emissions are adopted from (Llamas-Orozco et al., 2023), while hydrogen-specific production factors are drawn from Oxford Institute (2024). Time-varying grid electricity emissions factors are applied based on National Grid statistics (National Energy System Operator, 2024). Fleet-level life cycle costs are calculated in GBP (£) and reported annually, encompassing both operational expenditures (OPEX) and vehicle purchase costs attributed to the year of manufacture. This analysis includes costs associated with manufacturing, energy consumption, and maintenance (Khizouz et al., 2020). End-of-life costs were excluded from the current LCC model. While EoL costs are increasingly relevant, particularly in the context of battery recycling for BEVs their inclusion was beyond the scope of this study due to data limitations in early-stage hydrogen vehicle technologies. While the model includes GHG emissions across the vehicle lifecycle, the financial costs associated with decommissioning, recycling, or disposal were not incorporated. This reflects the current lack of reliable data and standardised methodologies for estimating EoL costs for hydrogen vehicles. Unlike

BEVs, which are subject to established regulatory frameworks such as the EU Battery Directive and Extended Producer Responsibility, hydrogen systems do not yet have comparable end-of-life pathways or publicly available cost benchmarks. Future research should aim to address this gap by incorporating EoL costs as data availability improves and hydrogen systems mature, enabling a more comprehensive assessment of life cycle cost. The vehicle purchase price includes technology-specific components such as battery packs, fuel cell systems, and hydrogen storage. Operational costs are projected using fuel price trajectories for petrol, diesel, electricity, and hydrogen, reflecting production costs and electricity tariffs. Maintenance costs are based on literature-derived estimates for different powertrains (Grube et al., 2021), (Burnham et al., 2021). Alongside core LCC modelling, a sensitivity analysis was conducted to explore the conditions under which the Major H_2 scenario could achieve cost parity with the Base case. Rather than attempting to predict future market conditions, this analysis examines the break-even thresholds for electricity charging price. Following on from analysis carried out in Section 5.4.2, the present analysis also considers the impact of charging behaviour, specifically the proportion of home versus public charging on overall fleet-wide economic outcomes. By varying the home charging share from 100%, 90%, 45% and 0% (Figure 14) the fleet model demonstrates how electricity cost could shift the relative competitiveness of electrification and hydrogen decarbonisation pathways.

Chapter 4

Tier 1 Results: Global Green Hydrogen Supply Chain Analysis

This chapter presents a comprehensive assessment of PEM electrolyser systems, focusing on critical material demand, supply chain vulnerabilities and lifecycle GHG emissions under projected deployment and technological advancement scenarios. Through the integration of life cycle assessment and dynamic material flow analysis, this chapter quantifies the environmental and resource implications of scaling hydrogen production. The analysis quantifies GHG emissions and material demand profiles that serve as inputs for broader system-level modelling and directly supporting the evaluation of Research Questions 1 and 4.

4.1 PEM electrolyser material demand

The projected demand for platinum in PEM electrolysis presents a critical challenge. Under the conservative catalyst loading scenario, primary platinum demand peaks at 16 tonnes/year by 2050 in the APS scenario, while the more ambitious NZE trajectory reaches a demand of 46 tonnes/year by 2044. However, the global annual primary supply (190 tonnes) is unlikely to grow in the future (Matthey, 2023). Furthermore, competing uses, such as fuel cells, may intensify pressure on these limited resources (Matthey, 2023). The discrepancy between growing demand and a stable supply raises concerns about potential supply constraints.

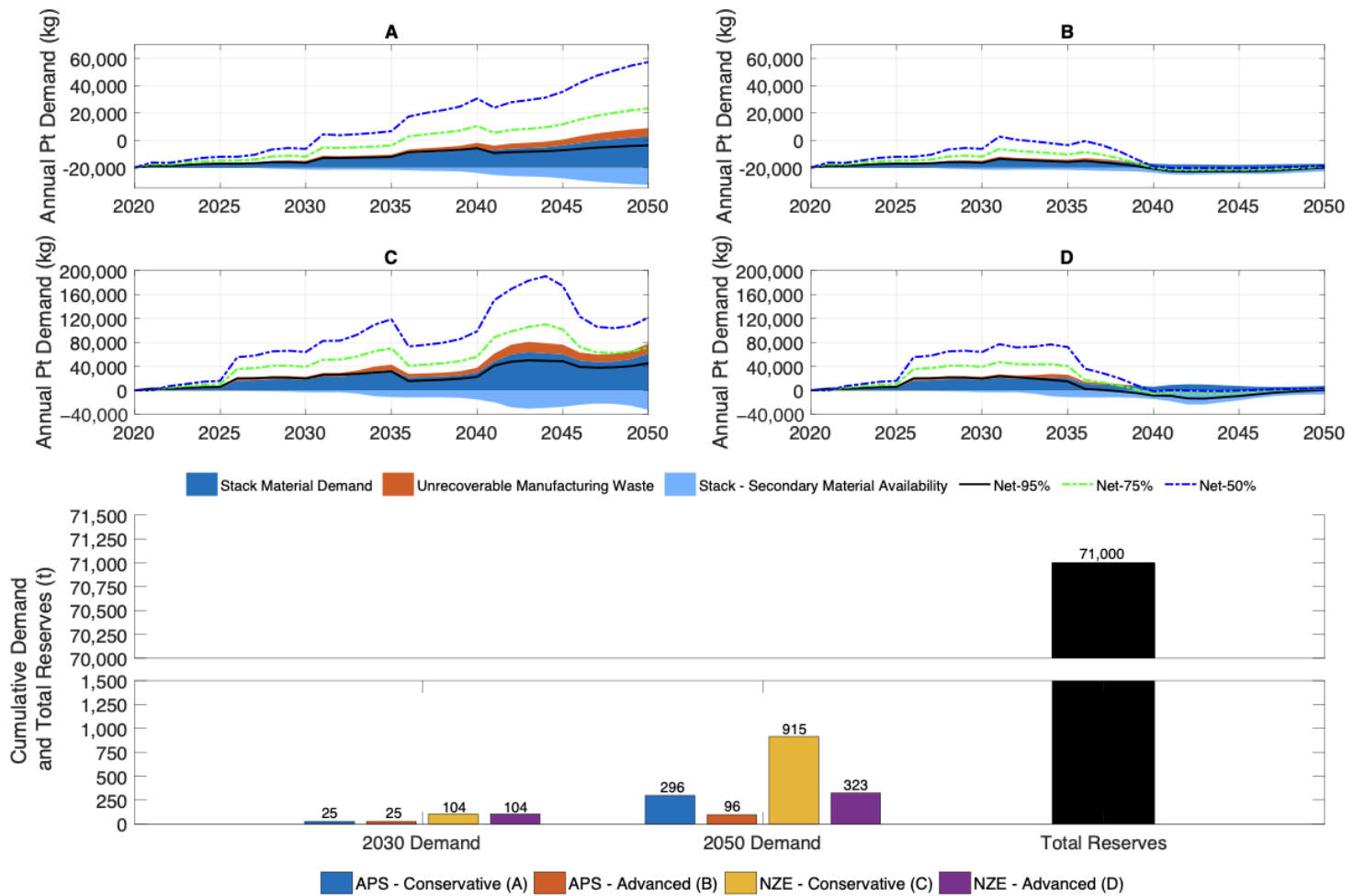


Figure 22: Annual Platinum demand and secondary supply for PEM electrolyzers under APS and NZE scenarios to 2050: (A–B) APS with conservative and advanced catalyst loadings; (C–D) NZE with conservative and advanced loadings. Current primary supply: 7 t/year. Bar chart shows cumulative demand vs. global reserves. Scenarios assume a 95% end-of-life recycling rate; sensitivity analyses for 75% and 50% recycling rates are also shown

However, the material flow analysis results offer a more optimistic outlook. Figure 22 show that in some scenarios, the supply of platinum could exceed the demand, as indicated by the negative values in Figure 22B and Figure 22D. By 2040 and beyond, recycling and EoL components and manufacturing waste could generate enough platinum to meet demand, potentially even creating a surplus. This surplus demonstrates the potential for secondary sources to reduce dependence on primary platinum production. By 2050, recycled EoL components could even surpass the demand for new platinum, reducing or even eliminating the need for additional primary material in some scenarios.

These positive results underscore the potential for effective circular management in the platinum supply chain. (Lotrič et al., 2021) estimate a potential 478 tonnes of platinum per year could become available as internal combustion engine vehicles are replaced by battery electric vehicles, further bolstering the secondary supply. This increase in secondary supply could play a crucial role in meeting the growing demand for PEM electrolysis technologies and other applications of platinum, alleviating some of the pressure on primary resources.

Interestingly, secondary supply from EoL recycling and manufacturing waste could account for up to 44% of annual demand under conservative scenarios. In more advanced cases, reductions in catalyst loading could further decrease primary platinum dependence. By 2040, recycled EoL components alone could exceed primary platinum demand, potentially allowing future needs to be met without significant increases in primary production.

The surplus of secondary material raises an important question for the industry: will there be a focus on stockpiling excess platinum, or will attention shift to other technologies with less dependence on platinum? Regions with strong recycling capabilities could become net suppliers of platinum, enabling them to support the widespread deployment of PEM electrolysis technologies and other platinum-dependent applications. This shift in material flow could also reduce the overall reliance on primary platinum mining, which is a positive step toward sustainable resource management in the transition to a green hydrogen economy.

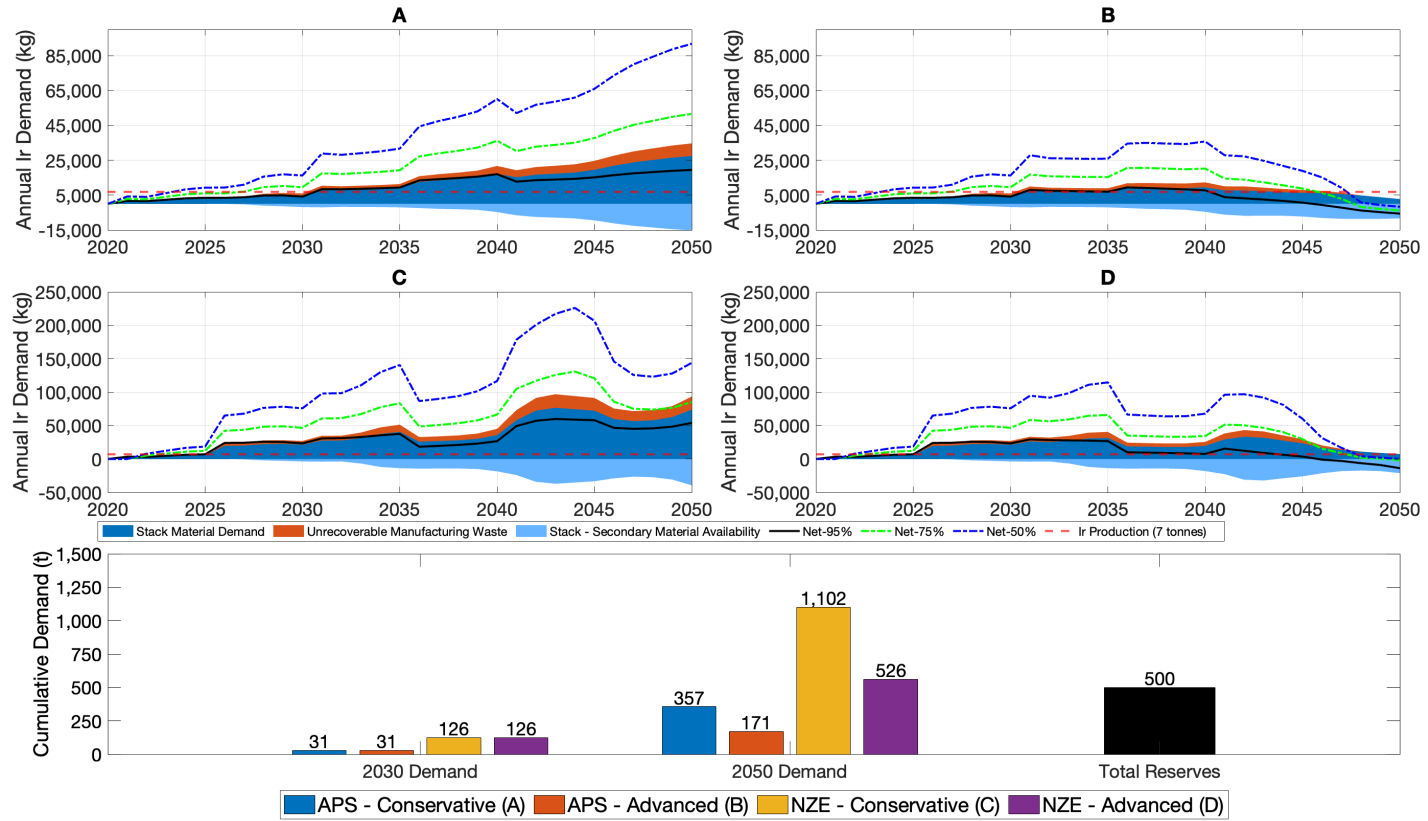


Figure 23: Annual Iridium demand and secondary supply for PEM electrolyzers under APS and NZE scenarios to 2050: (A–B) APS with conservative and advanced catalyst loadings; (C–D) NZE with conservative and advanced loadings. Current primary supply: 7 t/year. Bar chart shows cumulative demand vs. global reserves. Scenarios assume a 95% end-of-life recycling rate; sensitivity analyses for 75% and 50% recycling rates are also shown

Iridium demand for PEM electrolysis is projected to exceed current production levels (7 tonnes/year) by 2031, even under optimistic catalyst loading scenarios (Figure 23). In the conservative scenario (Figure 23A and Figure 23C), declines in catalyst intensity are insufficient to offset rising PEM deployment, pushing annual iridium demand to 19.5-55 tonnes by 2050.

In contrast, the advanced catalyst scenario (Figure 23B and Figure 23D) shows a more favourable trajectory, with net demand falling below zero by 2046 at the earliest (Figure 23B). In this case, peak demand remains under 10 tonnes per year by 2036, driven by reduced catalyst intensities and increased recycling of end-of-life stacks. By 2046, the net material demand becomes negative because secondary material availability exceeds demand. This shift is attributed to reduced waste generation during manufacturing and the higher concentration of iridium in recovered end-of-life stacks compared to new stacks, making iridium recovery sufficient to meet future PEM manufacturing needs (Figure 23B).

Achieving the IEA’s net zero (NZE) trajectory with projected PEM market share risks depleting most iridium reserves, even with advanced catalyst loading. Cumulative iridium demand could reach 126 tonnes by 2030 under NZE, about 30% of total global reserves (500t). By 2050, iridium demand may exceed global reserves by over 70% if modest catalyst loading rates are used, making such PEM deployment unsustainable. In contrast, the stated policy (APS) scenario, with more conservative green hydrogen demand, could be more feasible, requiring 21% of global iridium reserves by 2050 with advanced loading, or 64% with conservative loading.

Our cumulative iridium demand estimates for PEM electrolysis climb from 31-126 t by 2030 to 526-1,100 t by 2050 (Figure 23), versus (Minke et al., 2021) 10 t and 12-30 t for the same horizons. Three key factors drive this divergence. First, we assume a global PEM market share of up to 65%, whereas (Minke et al., 2021) cap deployment at 40%. Second, our catalyst loading trajectories are more conservative remaining at 0.23-0.30 kg/MW by 2035, compared with the authors rapid decline to 0.05 kg Ir/MW. Our literature review indicates that sub-0.05 kg/MW loadings are unlikely before 2050 (GREET2, 2022; Mayyas et al., 2019; Toshiba, 2018; U.S. Department of Energy, 2022a), whereas (Minke et al., 2021) assume such intensities are achievable by 2035. Lastly, the authors (Minke et al., 2021) assume a smaller cumulative installed electrolysis capacity by 2050 (100GW), whereas the IEA APS and NZE (2022) scenarios used in the present study are 1,200-3,600 GW by 2050. Collectively, these assumptions and the deep uncertainty around electrolysis deployment, future catalyst reductions and market roll-out, drive wide variation in iridium forecasts. To anchor our analysis in resource reality, Section 4.2 then applies a cap on iridium availability ($1.5 \text{ t } y^{-1}$) to quantify the maximum PEM capacity and market

share achievable under such supply constraints.

4.2 Estimating PEM deployment under plausible iridium supply constraints

Given the iridium supply constraints in Section 4.1, we use the MFA model to assess maximum PEM capacity and market share under a plausible iridium use limit of 1.5 t/yr (Clapp et al., 2023) (Figure 24). While near-term PEM electrolysis demand can be met, our results indicate reductions in catalyst loading rates are not sufficiently rapid to enable projected deployments from the mid-2020s onwards. During this period, escalating demand and fixed iridium supply lead to a substantial decline in achievable market share. In the APS and NZE conservative scenarios (Figure 24A, Figure 24C), high and rising iridium demand significantly outpaces the constrained supply, causing potential PEM deployment shares to fall sharply, particularly from the late 2020s onward, reaching as low as 8% by 2050 (Figure 24A) and remaining around 4-6% from 2026 (Figure 24C). Considering these constraints, we expect the remaining demand for electrolyser capacity will be largely filled by alternative technologies, particularly AE systems, which do not rely on iridium.

With ongoing reductions in iridium loading under the advanced catalyst loading scenarios (Figure 24B, Figure 24D), the availability of secondary material from EoL components becomes sufficient for large increases in PEM electrolyser capacity in the 2040s, potentially delivering 100% market share by 2050. In 2045, when secondary material becomes sufficient to meet the demand for PEM capacity, secondary iridium supply contributes 90% (Figure 24B) and 78% (Figure 24D) of total iridium availability, with the remaining contribution from primary sources. By 2050, the secondary stock of material is greater than demand, exceeding 100%; in the APS advanced scenario (Figure 24B), secondary availability is >300% of primary demand. This outcome depends on maximum PEM electrolyser production during the intervening years, accumulating a material stock within in-use electrolysers, and an ongoing flow of secondary material available from EoL components.

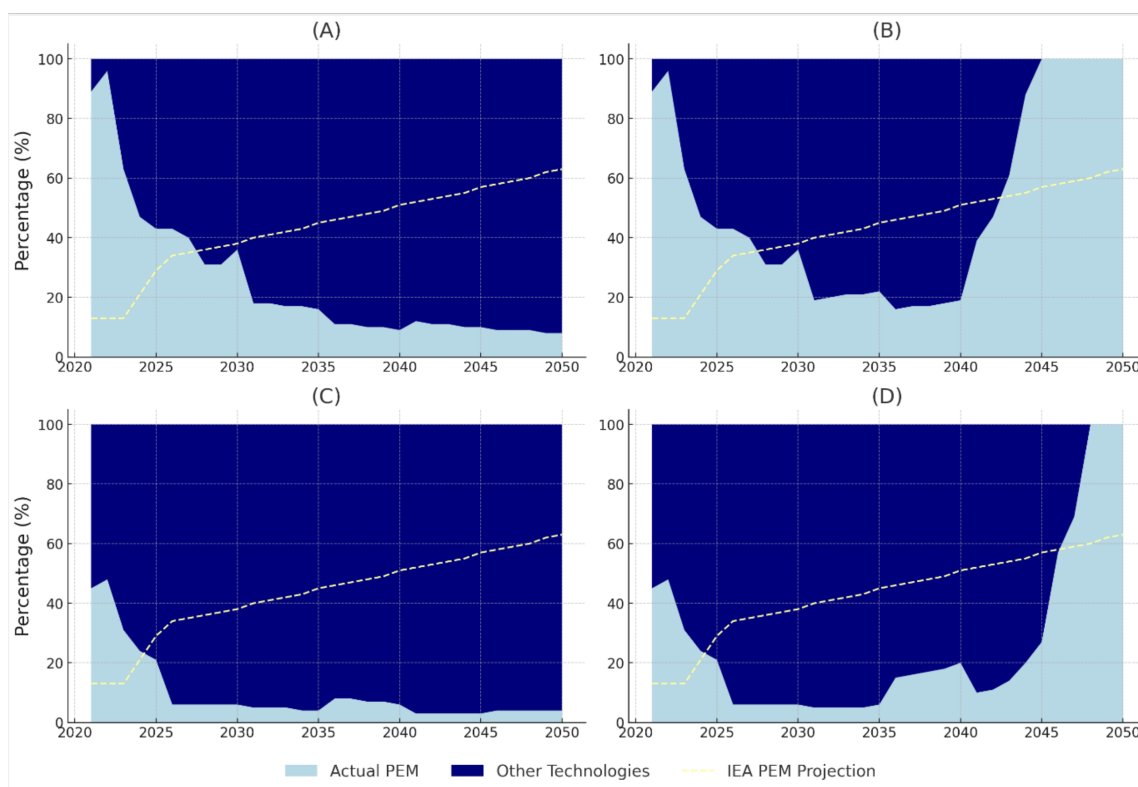


Figure 24: Projected market share of PEM electrolyzers under a 1.5 t/year iridium cap: (A) APS conservative, (B) APS advanced, (C) NZE conservative and (D) NZE advanced scenarios. Assumes 95% closed-loop recycling. Compared against IEA base case (Figure 3)

4.3 Estimating iridium loading required to meet projected PEM deployment

As the results in Section 4.2 indicate challenges in achieving expected PEM deployment with anticipated catalyst loading reductions, we reorient the MFA model to determine the required catalyst loading to meet projected PEM market share under constrained primary iridium supply of 1.5 tonnes/yr (Figure 25). To achieve the projected market share, far lower catalyst loading rates are required compared with our identified conservative and advanced scenarios based on publicly available data and projections. Achieving deployment targets aligned with the Announced Pledges (APS) and the Net Zero Emissions (NZE) scenarios require iridium loadings below 0.07 and 0.02 kg/MW respectively by 2030. Furthermore, the novel ultra-low loading scenario in the Advanced projection would fall just short, requiring >0.02 kg by 2050 instead (see Figure 25).

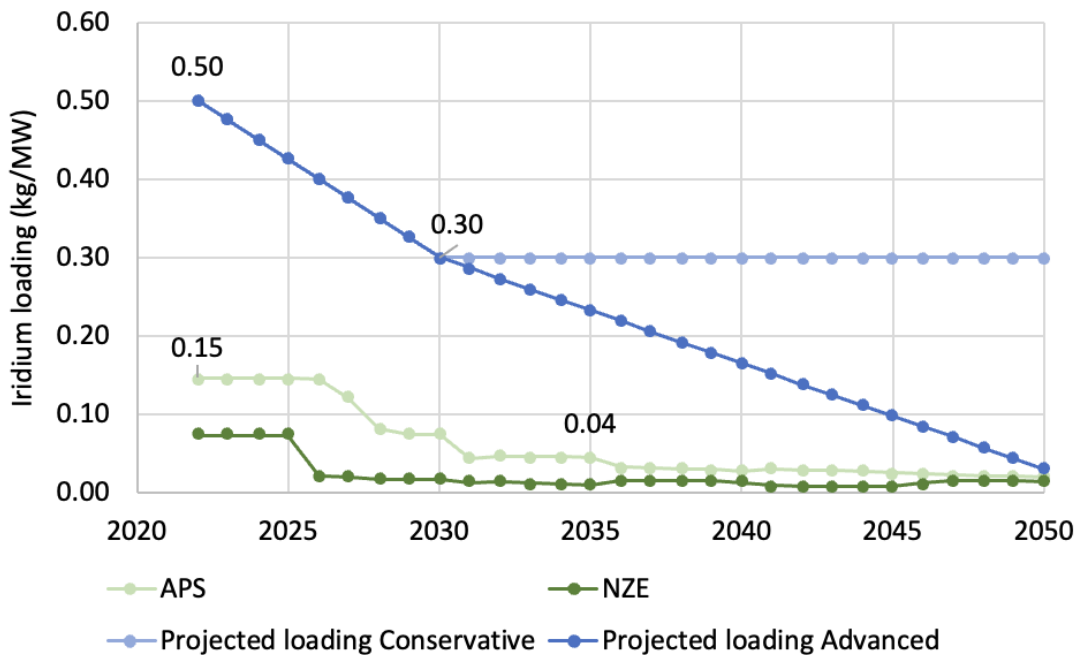


Figure 25: Required iridium loading (kg/MW) to meet APS and NZE deployment targets under a 1.5 t/year supply cap, compared to projected reductions (Figure 5)

4.4 Greenhouse gas emissions

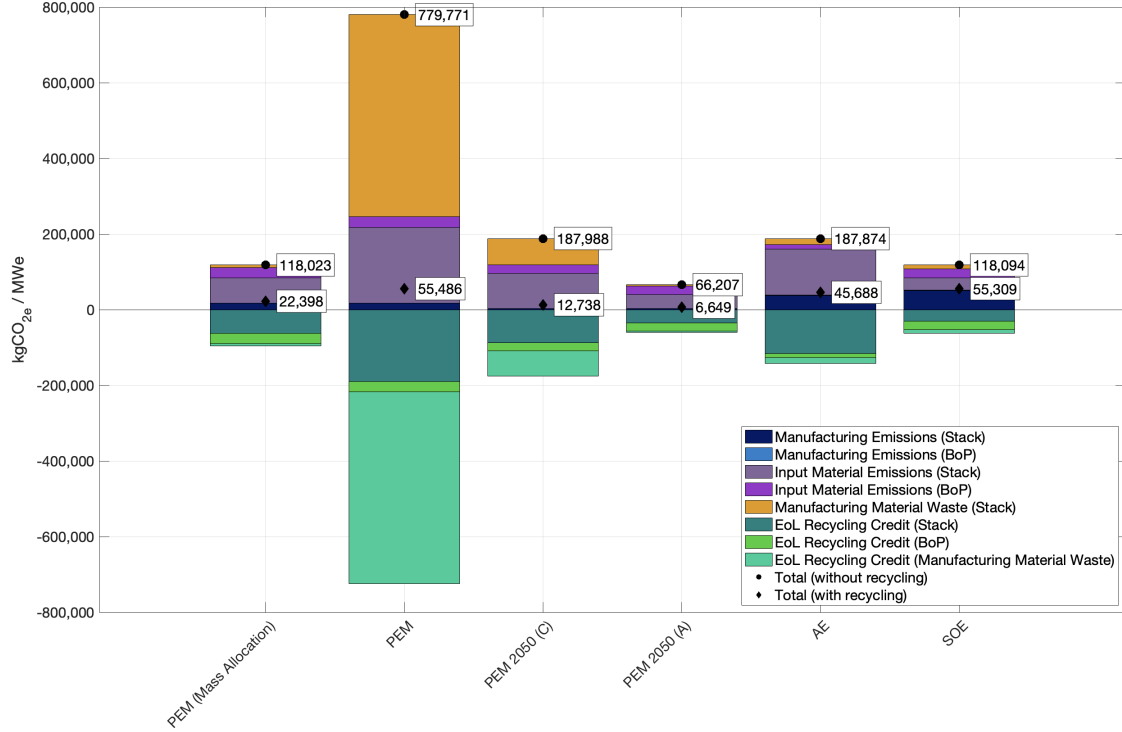


Figure 26: Lifecycle GHG emissions (2022 and 2050) for AE, SOE, and PEM electrolyzers under conservative (C) and advanced (A) scenarios, with mass vs. economic allocation for PEM

The 2022 GHG emissions of electrolyser manufacture, net of closed-loop recycling, range from 46 to 56 tCO₂eq/MWe (Figure 26). Recycling significantly reduces these emissions, particularly for PEM electrolyzers, which depend on PGMs. Without recycling, PEM manufacturing emissions would be an order of magnitude higher, making material recovery essential for scalable hydrogen deployment. PGMs have a high embodied GHG emissions intensity (33.5 tCO₂eq/kg) under economic allocation, reflecting their byproduct extraction from chromite or nickel-copper mines. In contrast, mass allocation reduces this to 0.13 tCO₂eq/kg, potentially lowering PEM manufacturing emissions to 22 tCO₂eq/MWe (Figure 26, Figure 27), though this effect is modest due to high expected recycling rates during manufacturing and at EoL. AE and SOE electrolyzers, which do not use PGMs, have lower manufacturing emissions but face other challenges, such as SOE stack degradation, which drives

higher ongoing material turnover and emissions. In Figure 28 we explore a sensitivity analysis to consider the impact of lesser PGM recycling rates finding that emissions are 23-52% greater with 75% EoL-rr and 52-118% with only a 50% EoL-rr compared to the 95% recovery baseline.

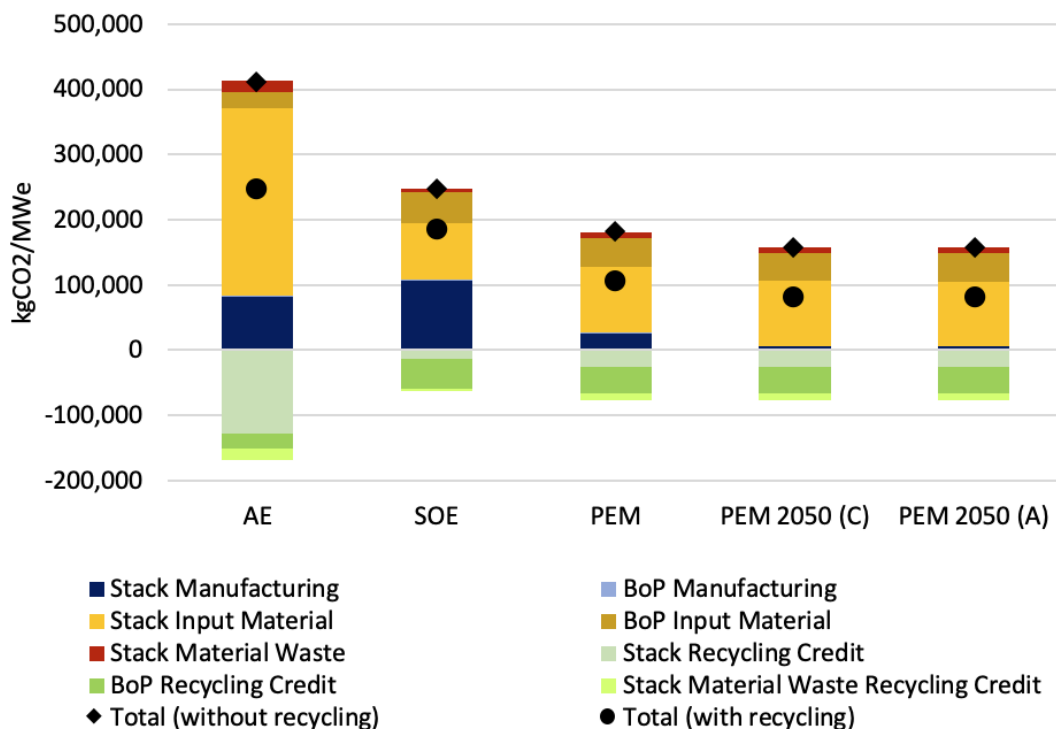


Figure 27: Sensitivity Analysis: Allocation of PGMs by mass. Total lifecycle GHG emissions currently (2022) and in the future (2050) for alkaline (AE), solid oxide (SOE) and polymer electrolyte membrane (PEM) electrolyzers under the conservative (C) and advanced (A) catalyst reduction scenarios

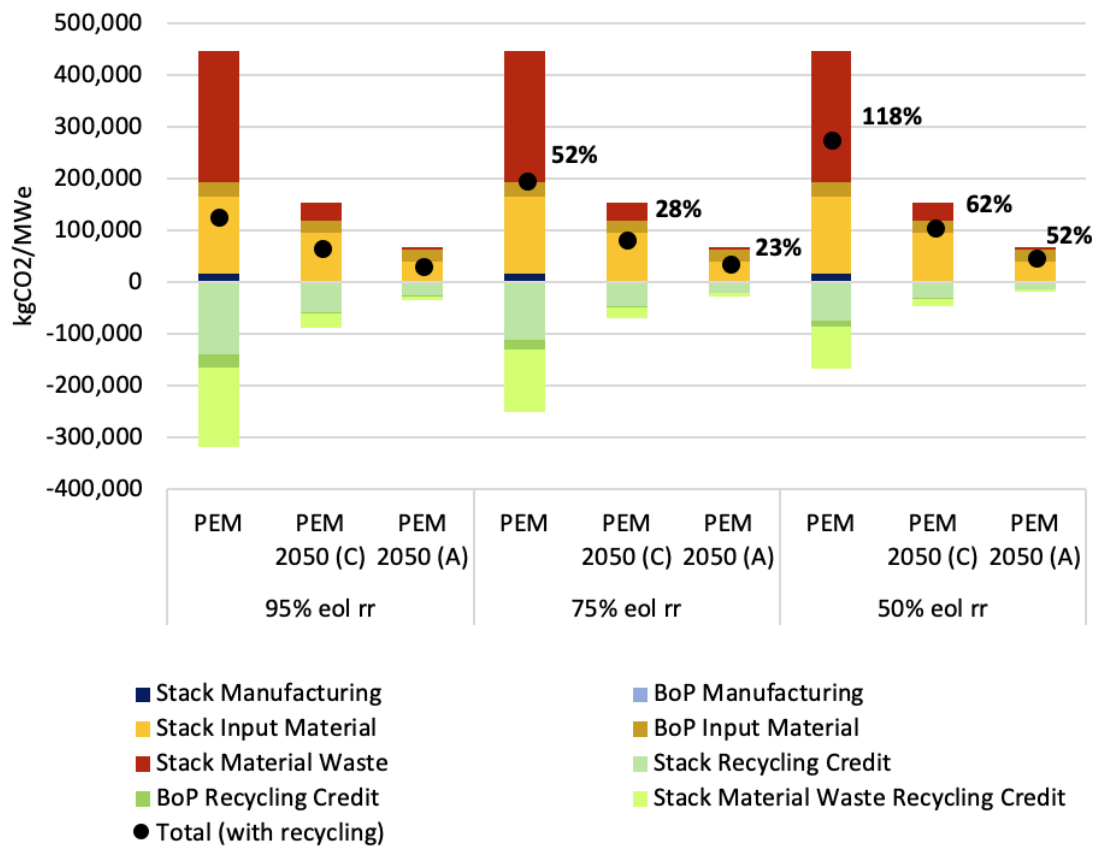


Figure 28: Sensitivity Analysis: Lifecycle GHG emissions for PEM (2022) and 2050 under conservative (C) and advanced (A) scenarios considering a tiered approach to end of life PGM recycling rates (50%, 75% and 95%)

We consider a first order estimate of future PEM electrolyser GHG emissions, which assumes reductions in PGM input in line with future catalyst loading scenarios. No other changes to the electrolyser bill of materials or manufacturing processes are considered, so these results should be interpreted only as an indication of potential future emissions reductions. In the conservative (C) scenario, reductions in catalyst material loading are expected to decrease net GHG emissions to 12.7 tCO₂eq/MWe. Under the advanced (A) scenario, this is further reduced to just 6.7 tCO₂eq/MWe. These future scenarios highlight the critical role of improving catalyst materials, making them more sustainable and competitive. By 2050, the overall GHG emissions of PEM systems could decrease by 76 – 92% largely due to advanced catalyst efficiencies that minimise material waste (Figure 26). These reductions in material use will not only lower emissions per unit of electrolyser capacity (MWe) but also influence total lifecycle emissions per kg of H_2 produced, as discussed next.

The manufacturing emissions for a PEM electrolyser (as of 2020) contribute approximately 0.775 kgCO₂/kg H_2 , while total lifecycle emissions reach 2.17 kgCO₂/kg H_2 (coupled with offshore wind, see Table 58). For AE and SOE electrolysers manufacturing emissions are 0.301 and 0.457 kgCO₂/kg H_2 respectively (Figure 29). Electricity generation is the dominant contributor to lifecycle emissions. While operational emissions can be minimised when using renewable electricity (USGS, 2022), the embodied emissions from building wind farms, solar panels, or grid infrastructure remain substantial. These electricity-related emissions make up the remainder of the total lifecycle emissions once the emissions from electrolyser manufacturing is accounted for.

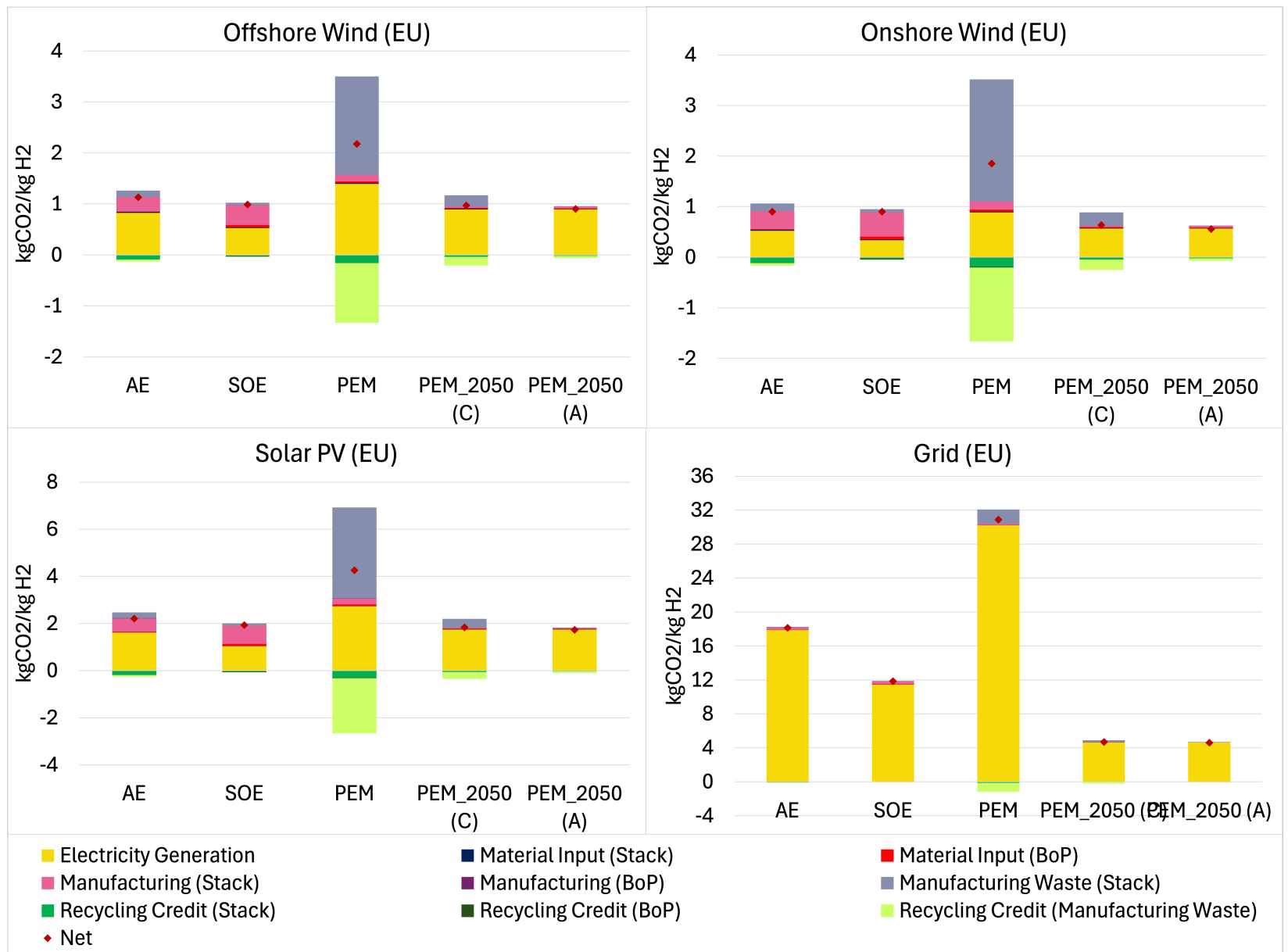


Figure 29: GHG emissions of electrolyser technologies per kg including recycling emissions credits for PEM coupled with electrical input from EU offshore and onshore wind, EU grid and China grid. Presented in $\text{kgCO}_2/\text{kg } H_2$

Table 58: GHG emissions of hydrogen electrolyser technologies by region (kgCO₂eq/kg H₂)

Technology	Europe	China	North America	India	RoW
Offshore Wind					
AE	1.13	1.13	1.13	1.13	1.13
SOE	0.99	0.99	0.99	0.99	0.99
PEM	2.17	2.17	2.17	2.17	2.17
PEM_2050 [C]	0.96	0.96	0.96	0.96	0.96
PEM_2050 [A]	0.90	0.90	0.90	0.90	0.90
Onshore Wind					
AE	0.90	0.90	0.90	0.90	0.90
SOE	0.90	0.90	0.90	0.90	0.90
PEM	1.85	1.85	1.85	1.85	1.85
PEM_2050 [C]	0.64	0.64	0.64	0.64	0.64
PEM_2050 [A]	0.56	0.56	0.56	0.56	0.56
Solar PV					
AE	2.20	2.20	2.62	2.20	2.84
SOE	1.93	1.93	2.20	1.93	2.34
PEM	4.26	4.26	4.96	4.26	5.34
PEM_2050 [C]	1.84	1.84	2.28	1.84	2.53
PEM_2050 [A]	1.73	1.73	2.17	1.73	2.42
Grid					
AE	18.14	40.36	25.50	47.82	33.10
SOE	11.84	26.06	16.55	30.84	21.42
PEM	30.89	68.44	43.32	81.05	56.17
PEM_2050 [C]	4.69	16.22	1.60	16.22	8.91
PEM_2050 [A]	4.62	16.15	1.53	16.15	8.84
LCHS	2.40	2.40	2.40	2.40	2.40
RED III	3.38	3.38	3.38	3.38	3.38

Table 59: GHG emissions of hydrogen electrolyser technologies by region, converted into (kgCO₂/MJ)

Technology	Europe	China	North America	India	RoW
Offshore Wind					
AE	0.009	0.009	0.009	0.009	0.009

Technology	Europe	China	North America	India	RoW
SOE	0.008	0.008	0.008	0.008	0.008
PEM	0.018	0.018	0.018	0.018	0.018
PEM_2050 [C]	0.008	0.008	0.008	0.008	0.008
PEM_2050 [A]	0.007	0.007	0.007	0.007	0.007
Onshore Wind					
AE	0.007	0.007	0.007	0.007	0.007
SOE	0.008	0.008	0.008	0.008	0.008
PEM	0.015	0.015	0.015	0.015	0.015
PEM_2050 [C]	0.005	0.005	0.005	0.005	0.005
PEM_2050 [A]	0.005	0.005	0.005	0.005	0.005
Solar PV					
AE	0.018	0.018	0.022	0.018	0.024
SOE	0.016	0.016	0.018	0.016	0.020
PEM	0.036	0.036	0.041	0.036	0.045
PEM_2050 [C]	0.015	0.015	0.019	0.015	0.021
PEM_2050 [A]	0.014	0.014	0.018	0.014	0.020
Grid					
AE	0.151	0.336	0.212	0.399	0.276
SOE	0.099	0.217	0.138	0.257	0.178
PEM	0.257	0.570	0.361	0.675	0.468
PEM_2050 [C]	0.039	0.135	0.013	0.135	0.074
PEM_2050 [A]	0.038	0.135	0.013	0.135	0.074

Our findings show grid-coupled total lifecycle emissions for 2020 are 18.1 kgCO₂/kgH₂ for AE, 12 kgCO₂/kgH₂ for SOE and 30.9 kgCO₂/kgH₂ for PEM (Figure 29). By 2050, PEM emissions fall to 4.7 kgCO₂/kgH₂. In contrast, (Wei et al., 2024) report 17.3, 16.2 and 19.8 kgCO₂/kg H₂ for AE, SOE and PEM respectively in 2020 and project PEM emissions of just 0.6 kgCO₂/kg H₂ by 2050. While alkaline electrolysis is comparable, the larger emission profile and more gradual decline our results stem from our explicit inclusion of GREET-based manufacturing waste streams (GREET2, 2022), which drive up PGM requirements for stack components. Because these materials carry high embedded emission factors, it is expected that the present study yields higher lifecycle emissions, especially for PEM systems.

Iridium’s embodied GHG emissions are a major contributor to the lifecycle impact of PEM electrolyzers, accounting for 38% of total manufacturing emissions (or 0.296 kgCO₂/kgH₂). However, planned reductions in iridium loading by 2050 could lower this impact to 0.003–0.205 kgCO₂/kgH₂, leading to an overall decline in PEM

manufacturing emissions. In contrast, AE and SOE electrolyzers avoid iridium use entirely, leading to lower current manufacturing emissions of 0.301 kgCO₂/kgH₂ for AE and 0.457 kgCO₂/kgH₂ for SOE. Thus, the results highlight the importance of reducing iridium in PEM systems to reduce emissions.

Despite the substantial contribution of infrastructure emissions, current regulatory frameworks primarily focus on operational emissions, excluding emissions from electrolyser manufacturing. For instance, EU RED III and UK LCHS set thresholds for hydrogen production emissions (3.38 kgCO₂eq/kgH₂ and 2.4 kgCO₂eq/kgH₂, respectively) but fail to account for embedded infrastructure emissions required to produce hydrogen at scale. For green hydrogen production from offshore wind in the UK, we estimate cradle-to-gate GHG emissions of 1.57 kgCO₂eq/kg H₂, of which electricity infrastructure (wind turbines) account for 89% of the total emission and PEM electrolyser stacks and BoP are 9% and 2% respectively (Figure 29). In 2050, this profile changes to 94% for wind and 5% for PEM stacks due to the emissions reductions seen in electrolyser stacks. This aligns with Bareiß et al. (2019), who similarly find that electricity contributes 96% of emissions with just 1% to PEM stacks in 2050 scenarios. (Bareiß et al., 2019) do not report on manufacturing waste. Again, this upstream waste inclusion could account for the higher percentage contribution of stacks to total lifecycle emissions compared to (Bareiß et al., 2019).

While H₂ production is expected to become less resource- and carbon-intensive per unit output over time through reduced catalyst loading, the sheer scale of hydrogen production will still drive significant material demand and associated emissions. Considering the potential magnitude of future electrolyser deployment, the manufacture of PEM electrolyzers alone could result in cumulative emissions of up to 1.4 Gt CO₂eq by 2050 (IEA NZE scenario, excluding recycling credit; 100 Mt CO₂eq by 2050 if crediting closed-loop recycling of manufacturing and end-of-life wastes). Excluding the impacts of infrastructure in key low carbon hydrogen policies risks being misaligned with the actual emissions burden of this fuel production pathway and therefore risks underestimating its lifecycle impacts.

4.5 Discussion

The growth of a global green hydrogen economy, driven by net-zero ambitions, will sharply increase electrolyser demand. Meeting the associated demand for critical raw materials for electrolysis technologies poses a challenge. PGMs are highly concentrated in South Africa; most countries lack domestic primary resources and have limited secondary sources, creating a risk of supply constraints and price volatility. Deep-sea mining, though explored as an alternative source, faces significant envi-

ronmental and regulatory barriers, making it unlikely to contribute significantly in the near term (Szabó, 2025). Reliance on land-based sources and efficient secondary recovery will remain essential

We find a significant material gap between the supply-side availability of iridium (current global supply at 7 t y^{-1}) and projected installed PEM electrolyser capacities, which translate into a cumulative demand of 31-126 t by 2030 and 526-1,100 t by 2050. Even under aggressive catalyst loading reductions and a 95% closed-loop recycling rate, our MFA shows net iridium demand above annual global supply until 2040 onwards (Figure 23). Secondary materials, from EoL components and manufacturing waste, can contribute to future deployment targets. However, these sources logically originate from primary sources at an earlier point in time, building up a stock of material in in-use systems. Meeting near- and medium-term projected PEM electrolyser capacity growth under iridium supply constraints will require greater reliance on alternative technologies like alkaline and solid oxide systems. Iridium production is not anticipated to rise above current levels, as it is co-produced with platinum ores and comprises only 2-4% of the overall PGM grade (Clapp et al., 2023). Recovery from other applications, such as waste electronics, could somewhat increase secondary material availability (Reller et al., 2009).

Material constraints are not unique to PEM. AE technologies increasingly use ruthenium nanosheets (0.22 kg/MW (GREET2, 2022)) to enhance water splitting (Yang et al., 2022), yet global ruthenium output is only 30 tonnes/year (Loferski et al., 2018). Additionally, nickel and copper, key materials for AE have recently been added to the EU’s critical raw material list (European Parliament and Council, 2023). Furthermore, SOE relies on rare earth elements like Yttrium, Zirconia, and Lanthanum, which are costly to extract (Bai et al., 2022). Thus, future work should evaluate performance alongside the material security of all electrolysis types (Kiemel et al., 2021).

The GHG emissions associated with electrolyser manufacture become significant when put in the context of green hydrogen production. In 2022, lifecycle manufacturing emissions (net 95% closed-loop recycling) range from 46-56 $\text{tCO}_2\text{eq/MWe}$. Additionally, recycling is essential, without it, emissions would be an order of magnitude higher, particularly for PEM. For PEM systems, GHG emissions related to critical materials supply are significant, and the present study indicates that the wastage of materials in the manufacturing process is responsible for the majority of GHG emissions associated with the PEM stack. Manufacturing waste is commonly ignored in existing MFA and LCA studies of PEM systems, which is an important gap addressed by the present work, emphasising the need to consider material waste arising during manufacturing processes.

Key limitations of our model include the assumption of 95% PGM scrap recovery which may be optimistic given real-world recycling rates (Graedel et al., 2011), (Hao et al., 2019), (Johnson Matthey, 2025) and the use of a global-average wind power emission factor that masks regional electricity-grid variability. Offshore wind data from (United Nations Economic Commission for Europe (UNECE), 2022) was used for this study and as the authors note, little variation in the emissions factor for wind is seen across the globe and therefore an average was used. While we have included a tiered sensitivity analysis for PGM recycling rates (50% / 75% / 95%) to reflect conservative scenarios, these baseline assumptions still shape our demand and emissions forecasts and should be considered when interpreting our results.

In navigating the path toward achieving an alternative source of clean energy, fuelled by ambitious net-zero targets, the production of green hydrogen stands as a linchpin in the global energy transition. To meet the escalating demand for hydrogen, particularly through the widespread adoption of electrolyzers, challenges emerge in securing critical raw materials, notably PGMs. The supply-demand dynamics, especially concerning iridium, present a material gap requiring innovative solutions. In this pursuit, the recovery of PGMs from alternative sources emerges as a potential avenue to bridge the material gap. Efforts to enhance circular management of manufacturing and end-of-life wastes are crucial. To achieve such goals, the use of low or critical material free electrolysis technologies is essential. Such solutions can reduce primary material demand while also curtailing associated GHG emissions, which becomes imperative in realising a future of sustainable green hydrogen production.

4.6 Summary

This chapter shows that rapid global expansion of green hydrogen will place pressure on critical raw materials required for electrolyser technologies, particularly iridium for PEM systems. Current iridium supply is far below projected demand, with cumulative requirements reaching 31–126t by 2030 and 526–1,100t by 2050, even under aggressive catalyst loading reductions and high recycling rates. Similar material constraints affect alkaline and solid oxide electrolyzers, which rely on ruthenium, nickel, copper, and rare earth elements. Manufacturing emissions also become increasingly significant, with PEM systems especially sensitive to material waste during production. While secondary recovery and recycling can alleviate some pressure, near-term deployment will remain constrained by limited primary supply. These findings highlight the need for improved circularity, reduced material intensity, and diversification toward lower-criticality electrolysis technologies to support sustainable growth of the green hydrogen sector.

Chapter 5

Tier 2 Results: Single-vehicle Life Cycle Assessment and Life Cycle Cost Analysis of Hydrogen Vehicles (2020-2050)

This chapter presents the results from the second tier of analysis to evaluate hydrogen-powered vehicles at the vehicle level. Building on the methodological framework established in Chapter 3, it compares the environmental and economic performance of hydrogen, electric and conventional vehicles. The assessment incorporates life cycle GHG emissions and life cycle cost analysis. Sensitivity analysis explore the influence of real-world driving, the influence of charging access, fuel pricing, hydrogen transport and distribution pathways and technology learning curves to highlight the relative competitiveness of hydrogen technologies. This chapter addresses Research Question 2 and 4 by identifying key trade-offs and scenarios that determine the viability of hydrogen vehicles in future transport systems.

5.1 Environmental impact LCA results – vehicle manufacture

Figure 30 shows manufacturing emissions for each of the eight vehicle types by component. Although BEVs have no tailpipe emissions, they exhibit high emissions in the manufacturing stage (16.5t CO₂ eq. in 2020), driven by the substantial emission of li-ion battery production. However, hydrogen powered vehicles display the highest

emissions in the manufacturing stage due to hydrogen storage systems, with H_2 -ICE reaching 8.7t CO₂ eq. for this component alone. This high impact is almost entirely attributable to the extensive use of carbon fibre composites, which are widely utilised for safely containing hydrogen at 700 bar. Carbon fibre is a highly energy-intensive material to produce (Liddell et al., 2016), its dominant share – over 95% of the hydrogen storage systems emissions reflects both the large material quantity required and emissions intensity of carbon fibre manufacturing which involves energy-demanding polymerisation and carbonisation processes. We model hydrogen tanks using 100% new carbon fibre because recycled CFRP loses 35–90% of its tensile strength and cannot safely hold 700 bar (Zhu et al., 2019), (Zhang et al., 2020). Even without any recycled inputs, gradual reductions in tank mass (Table 36) cut emissions by 14–18% over time. As a result, the hydrogen storage system emerges as a key hotspot in the life cycle of hydrogen vehicles.

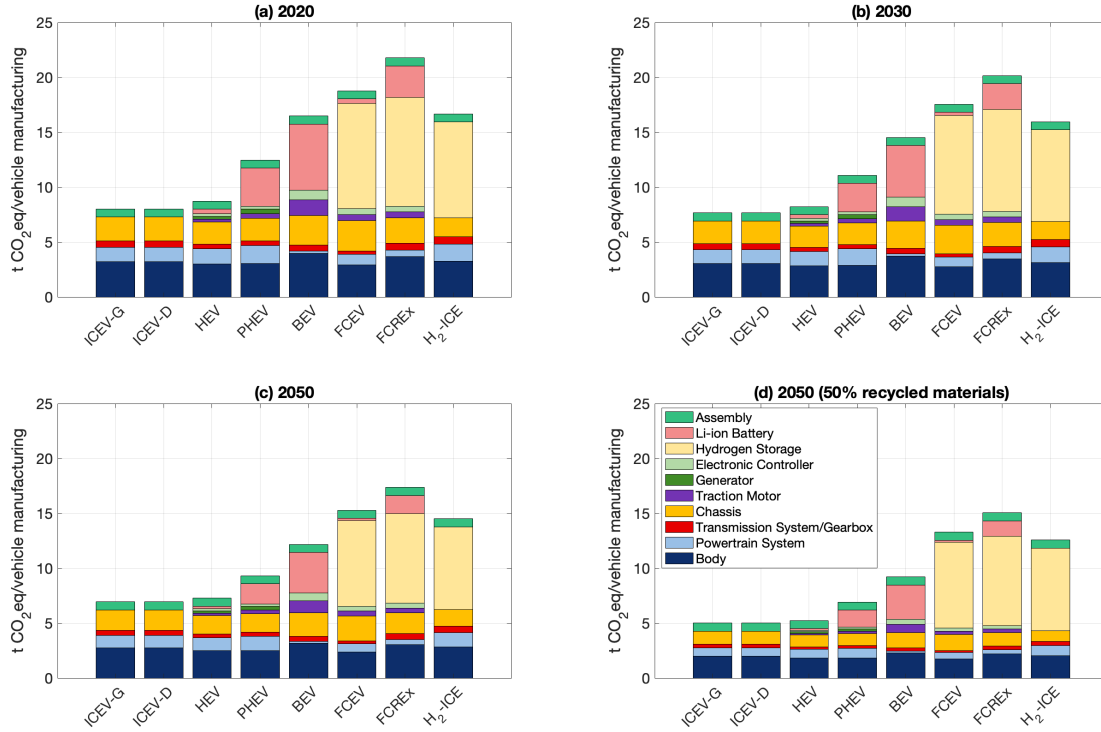


Figure 30: Life cycle emissions by vehicle production component in: (a) 2020, (b) 2030, (c) 2050 and (d) 2050 (50% recycled materials)

Over time, from 2020 to 2050, the results reveal a consistent downward trend

in manufacturing emissions across all powertrain types (Figure 30). For example, total BEV manufacturing emissions decline from 16.5 tonnes to 12.2 tonnes CO₂-eq in 2050, while FCEV emissions fall from 18.8 t to 15.3 tonnes CO₂-eq. For hydrogen vehicles, this reduction is primarily driven by a decrease in the quantity of energy-intensive materials required in the tank (carbon fibre) and fuel cells (platinum). For electric powertrains, the observed reduction stems from a combination of lower emissions intensity in battery manufacturing and decreased material demand through improved battery energy density and efficiency.

By contrast, achieving similar recycled content levels in hydrogen storage tanks remains highly uncertain. Carbon fibre-reinforced polymers, which are used in high-pressure H_2 tanks, remain technically and economically challenging to recycle. Current recovery methods, such as pyrolysis, degrade fibre quality, and the mechanical performance of recycled material is often insufficient to meet strict safety standards (Vincent and Dissanayake, 2023). As a result, no recycled content is currently used in the production of hydrogen storage systems, which are still manufactured entirely from primary materials. In addition, type IV composite tanks are expected to remain the state-of-the-art for high-pressure storage (Luthada, 2024).

In analysing emissions reductions across vehicle powertrains, the results from Figure 30A-C show varying degrees in emissions improvement from 2020 to 2050, with BEVs benefiting the most, decreasing vehicle production emissions by -26% from 2020 (Figure 30A) to 2050 (Figure 30C). FCEVs and PHEVs also see notable reductions of -19% and -25% respectively, while HEVs improve by 17% over the period. However, ICEVs remain unchanged highlighting their limited decarbonisation potential without the use of recycled materials.

We conduct an analysis for 2050 to explore the potential impact of using 50% recycled (secondary) materials across all vehicle powertrains (Figure 30D). We find the potential for further reductions from current day production emissions (Figure 30A) including PHEV (-45%), BEV (-44%), FCEVs (-29%), FCReX (-31%) and H_2 -ICE (-25%). Notably, ICEVs show reductions of -38% emphasising the environmental benefits of material recycling for conventional powertrains.

These findings underscore that BEVs, and hybrid-powertrains stand to benefit most from emissions reductions, while ICEVs remain unchanged unless material recycling is included. Hydrogen-based powertrains show comparatively lower reductions due to the continued reliance on primary material inputs for hydrogen storage. Additionally, H_2 -ICE vehicles rely on combustion-based propulsion, meaning their improvement is not as substantial as fully electrified alternatives.

5.2 Environmental impact LCA results – full life cycle

LCA results for the eight vehicle technologies from 2020 to 2050 highlight emissions across different lifecycle stages. ICEVs currently exhibit the highest total lifecycle emissions, despite having the lowest emissions associated with vehicle production. The dark blue bar summarises the vehicle-cycle results from Figure 30. The greatest contribution to lifetime emissions is the use phase, driven by fossil fuel combustion. Hybrid powertrains (HEVs) demonstrate a reduction in total emissions by 33% compared to ICEV-p or by 62% for PHEVs as their use of electric power decreases reliance on gasoline.

While technological advancements in vehicle efficiency are expected to reduce lifecycle emissions over time, the actual reductions in ICEV emissions remain marginal (Figure 31B), primarily due to assuming limited improvements to internal combustion engine technology. BEVs, while having higher production-phase emissions than ICEVs, due to lithium-ion battery manufacturing (Figure 32), they achieve the lowest lifecycle emissions of all studied vehicles, at approximately 23 tonnes CO₂, largely due to near-elimination of tailpipe emissions. However, this outcome is heavily influenced by the inclusion of recycling credits, which offset emissions in the production and end-of-life phases. Without these credits, BEV lifecycle emissions would be higher, potentially altering the comparative advantage.

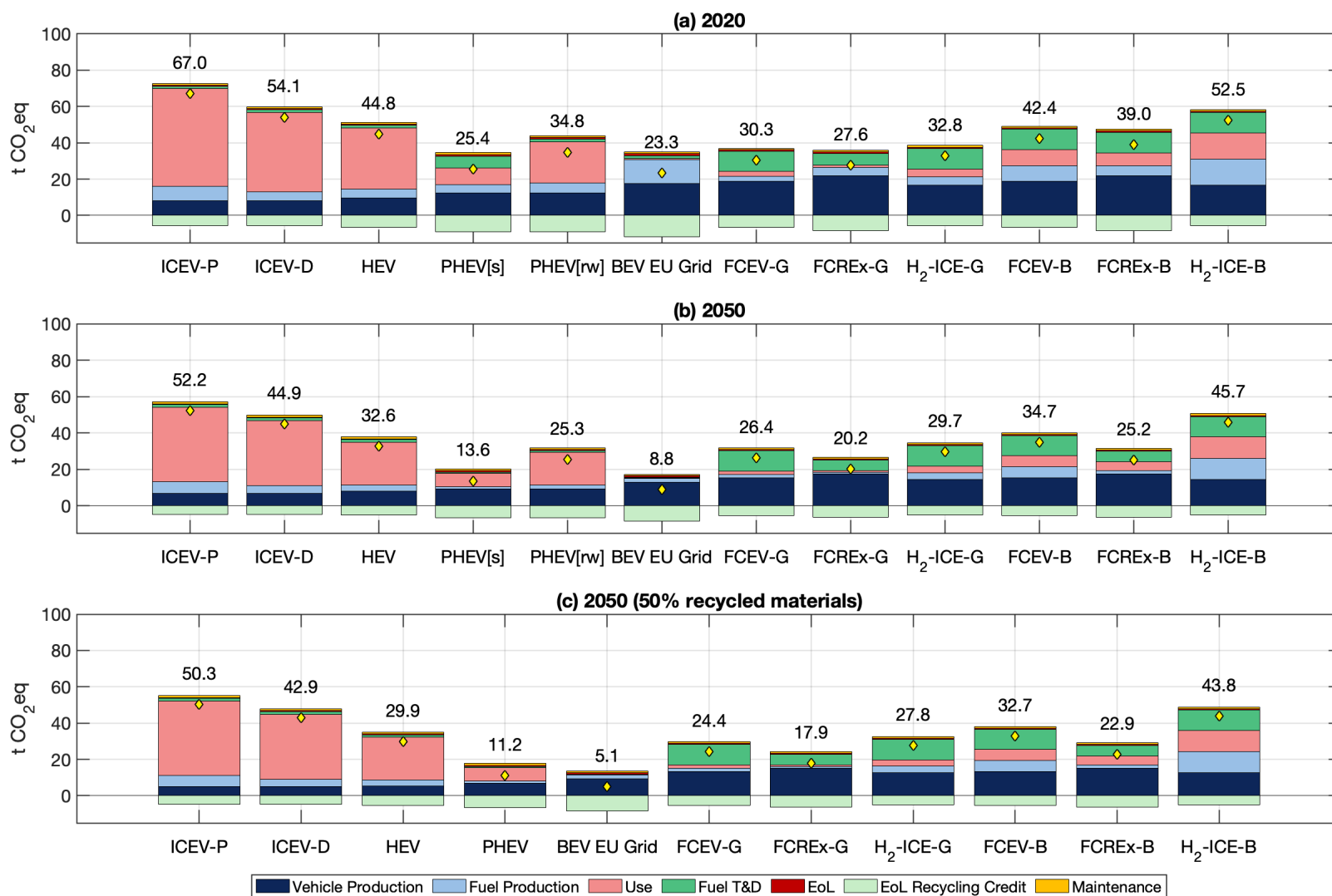


Figure 31: Total life cycle GHG emissions (tCO₂ eq.) for eight passenger vehicle technologies entering service in (a) 2020, (b) 2050, and (c) 2050 with 50% recycled material content. Stacked bars show contributions from each lifecycle phase (vehicle production, fuel production, use, fuel transport and distribution, maintenance and end-of-life). Hydrogen pathways are evaluated using green hydrogen (–G) and blue hydrogen (–B).

FCEVs and FCREx vehicles using both green (electrolysis) and blue (SMR + CCS) hydrogen benefit from lower use-phase emissions compared to ICE and hybrid vehicles, thanks to their use of low-carbon hydrogen fuel. However, production-phase emissions, particularly from hydrogen storage and fuel transportation significantly contribute to their overall emissions (Figure 31). H_2 -ICE vehicles show the highest emissions among hydrogen-powered technologies, similar to (Figure 31A), or greater than (Figure 31B) diesel when powered by blue hydrogen, mainly due to combustion inefficiencies during the use phase.

An analysis of Figure 30B and Figure 30C reveals that incorporating 50% recycled materials into vehicle manufacturing reduces start-of-life emissions (from 2050) by up to 12% for ICEVs and 17% for BEVs. However, since the use phase remains the dominant contributor to total lifecycle emissions, the overall impact of recycled content on long-term reductions is limited.

Figure 31 also highlights the impact of real-world fuel consumption on lifecycle GHG emissions from 2020 to 2050, revealing the difference between ‘*stated*’ and ‘*real-world*’ scenarios. Real-world conditions consider suboptimal charging habits, greater reliance on internal combustion engines in PHEVs, and higher fuel consumption, leading to greater emissions during the use phase (Plötz et al., 2022). The gap is most pronounced in 2050, where real-world total GHG emissions are 86% higher than manufacturer stated values, whilst being 37% higher in 2020 (Figure 31). The use phase consistently represents the largest share of life cycle emissions under real-world conditions, reinforcing the importance of tailpipe emission in GHG footprints of PHEVs. For example, in 2020, the use phase alone accounts for nearly 65% of total emissions in real-world conditions, a trend that continues in later years. Meanwhile, the contribution of other lifecycle phases remains relatively constant across both stated and real-world scenarios. However, end-of-life recycling credits, which provide emissions reductions, are insufficient to offset the GHG emission impact from real-world fuel consumption. Additionally, when considering the real-world driving of PHEVs in 2020, hydrogen powertrains fuelled by green hydrogen present a GHG saving of 6-21%. These findings emphasise the importance of incorporating real-world driving data into LCA studies to ensure accurate GHG impact assessments.

5.3 Hydrogen transport and distribution

Our study evaluates the potential emissions from hydrogen transport and distribution under different scenarios, focusing on how transport method and distance influence overall GHG emissions. The baseline scenario assumes a 100 km diesel-powered tube trailer (TT) delivery, representing a typical current approach (Wulf et al., 2018). A

sensitivity analysis was conducted to assess how increasing the hydrogen transport distance from 100 km to 400 km using diesel-powered TT affects emissions. Our results show that T&D emissions double with increased distance, rather than quadrupling, due to logistical efficiencies, consistent with findings in (GREET, 2023) (Figure 32). This highlights the need to minimise transport distances or switch to lower-emission fuels to reduce the overall lifecycle emissions of hydrogen-based vehicles. From 2030 onwards, our analysis considers two alternative supply chain scenarios aimed at reducing hydrogen T&D emissions: (1) Pipeline delivery – assumes the use of a dedicated hydrogen pipeline network, replacing diesel-powered tube trailers for long-distance transport. (2) Fuel switching for tube trailers – assumes replacing diesel with green hydrogen in existing tube trailer fleets to reduce emissions. Under these assumptions, dedicated hydrogen pipelines emerge as the lowest-emission option for delivering hydrogen to refuelling stations. When comparing a 100 km hydrogen pipeline to a 100 km diesel tube trailer, pipeline transport results in 74% lower emissions (Figure 32). This finding emphasises the environmental advantages of pipelines, especially for long-distance hydrogen transport. For tube trailer delivery, switching from diesel to renewable hydrogen results in an 85% reduction in T&D emissions (Figure 32).

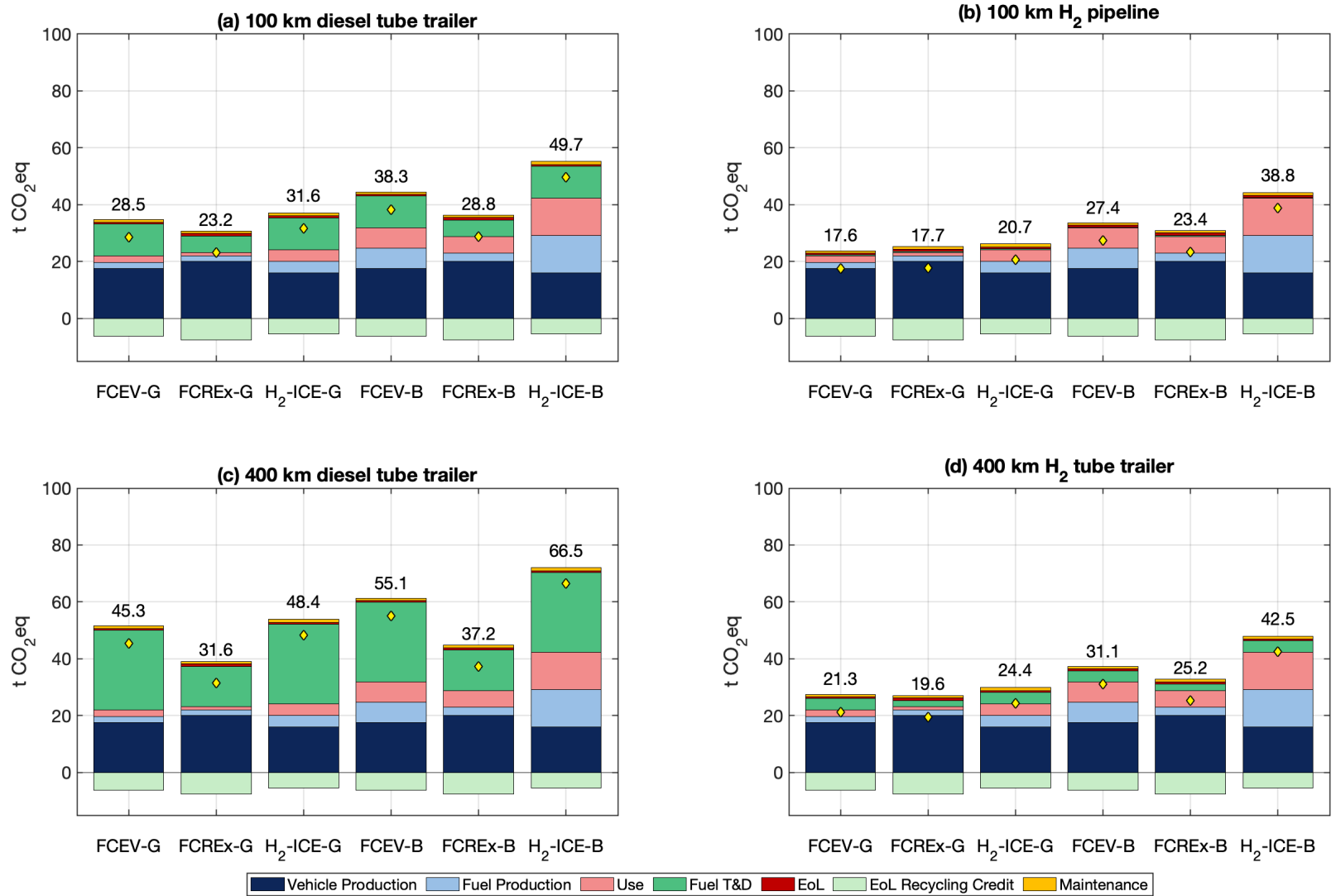


Figure 32: Impact of varying distances and modes of hydrogen T&D on lifecycle GHG emissions in 2030. (a) Base case – 100km diesel-powered TT, (b) 100km pipeline, (c) 400km diesel-powered TT and (d) 400km hydrogen-powered TT

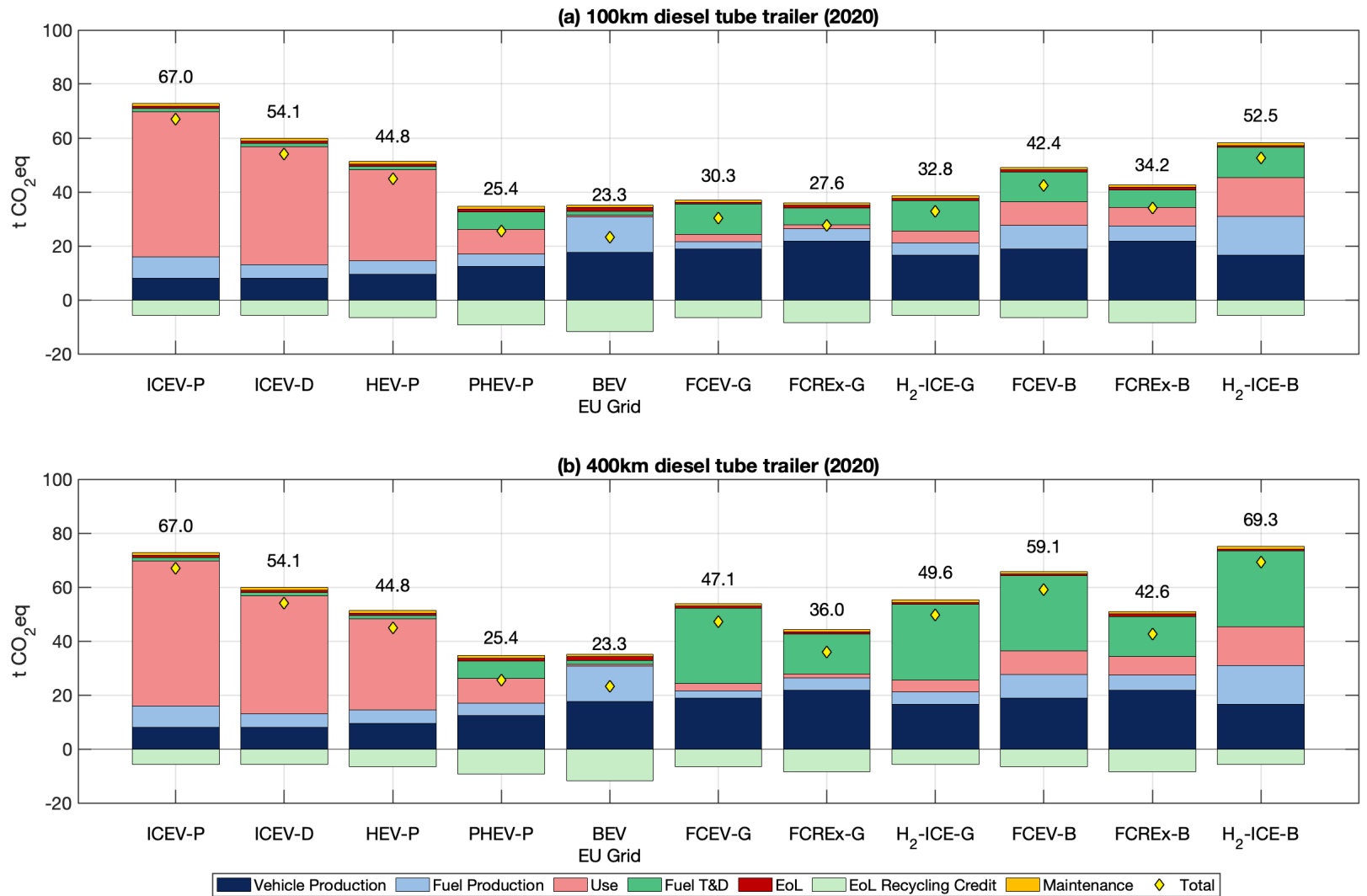


Figure 33: Sensitivity analysis: Comparison of life cycle emissions using diesel tube trailers for the delivery of H_2 fuel over 100km (top) and 400km (bottom) in 2020

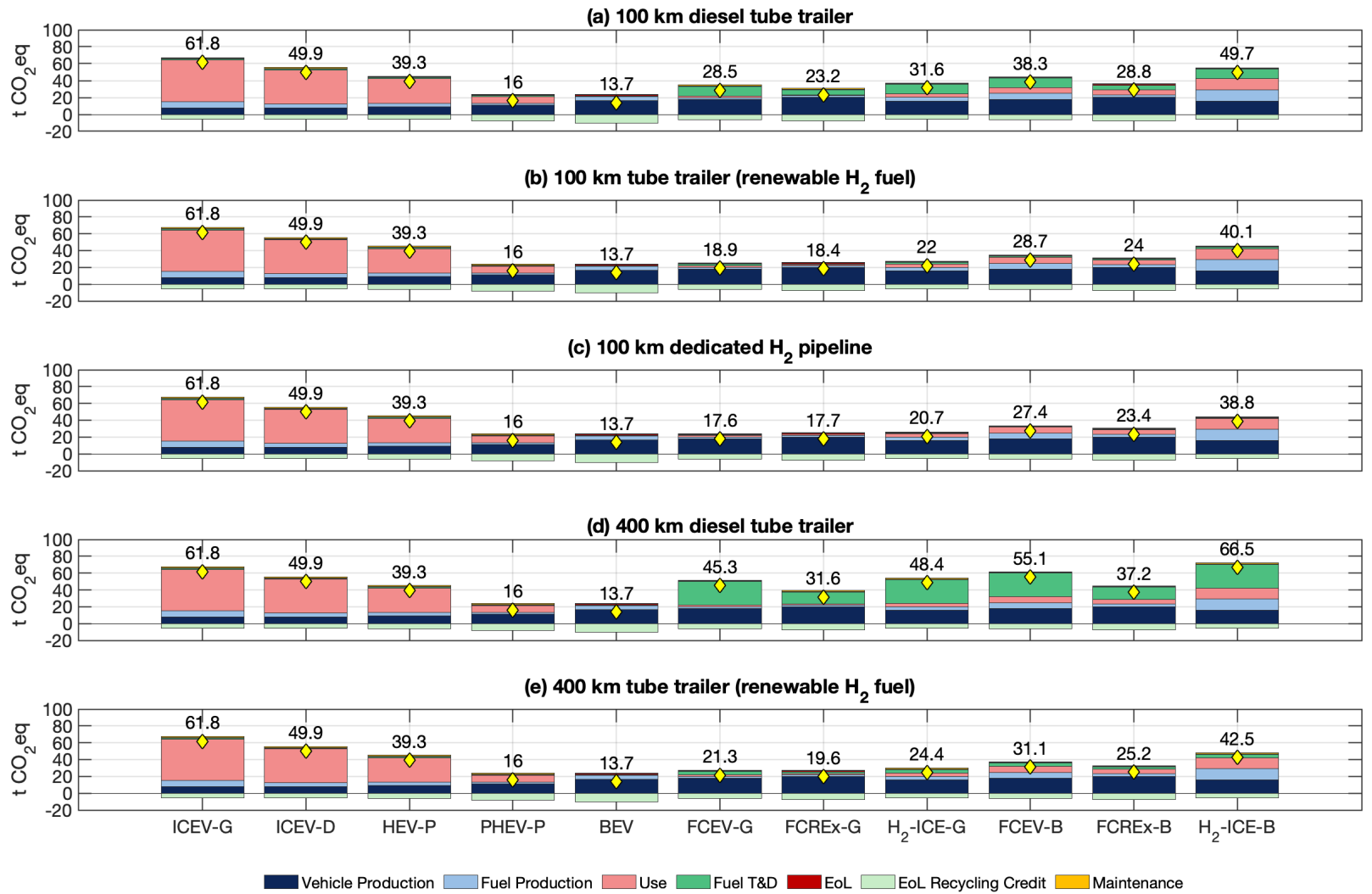


Figure 34: Comparison of life cycle emissions using (a) 100 km diesel tube trailer, (b) 100 km tube trailer using renewable hydrogen fuel, (c) 100 km dedicated H₂ pipeline, (d) 400 km diesel tube trailer and (e) 400 km hydrogen tuber trailer delivery. Analysis year 2030.

5.4 Economic impact lifecycle cost results

In terms of future projections for ICEVs the costs of these vehicles are not expected to decrease significantly over time. However, future gasoline and diesel price projections used in this study are based on the assumption that fuel prices increase driven by carbon taxation, emissions trading schemes, and broader policy efforts to incentivise low-emission alternatives.

Currently, hydrogen vehicles exhibit the highest overall costs among all vehicle technologies due to the high price of electrolytic (green) hydrogen, which was priced at approximately €11/kg H_2 in 2020 and is projected to decline to €5/kg by 2050 under conservative estimates or €4/kg under more aggressive scenarios (International Council on Clean Transportation (ICCT), 2022), (Houchins and James, 2019), (National Renewable Energy Laboratory (NREL), 2022)). However, when fuelled with blue hydrogen – which benefits from lower production costs – hydrogen vehicles are already cost-competitive with ICEVs (Figure 35A). As green hydrogen becomes more affordable, the total LCC of all hydrogen vehicles is projected to fall below ICEVs from 2030 onwards.

By 2050, the expected reductions in both hydrogen vehicle component costs and hydrogen fuel prices enable FCREx vehicles to emerge as an economically comparable technology to hybrid vehicles. On the other hand, H_2 -ICE vehicles consistently remain the most expensive among the three hydrogen vehicle technologies. This higher cost is attributed to the greater hydrogen fuel consumption required by H_2 -ICE vehicles due to their lower efficiency compared to FCEVs and FCREx vehicles.

Figure 35 illustrates the impact of taxes and subsidies on the total life cycle cost of vehicle powertrains in 2020 and 2050, providing a direct comparison to the cost structures presented in Figure 35. Taxes significantly increase the cost of every powertrain. ZEVs benefit from road tax and registration fee exemption, in conjunction with subsidies in 2020 equating to a €10,000 saving compared to ICEVs over the lifecycle. However, the increased cost of ZEVs lead to a greater associated VAT cost which largely offsets these savings. From 2035 onwards, it is expected that ZEVs will be required to pay a registration fee of €2,000, in addition, road tax exemptions and purchase subsidies will also have ended.

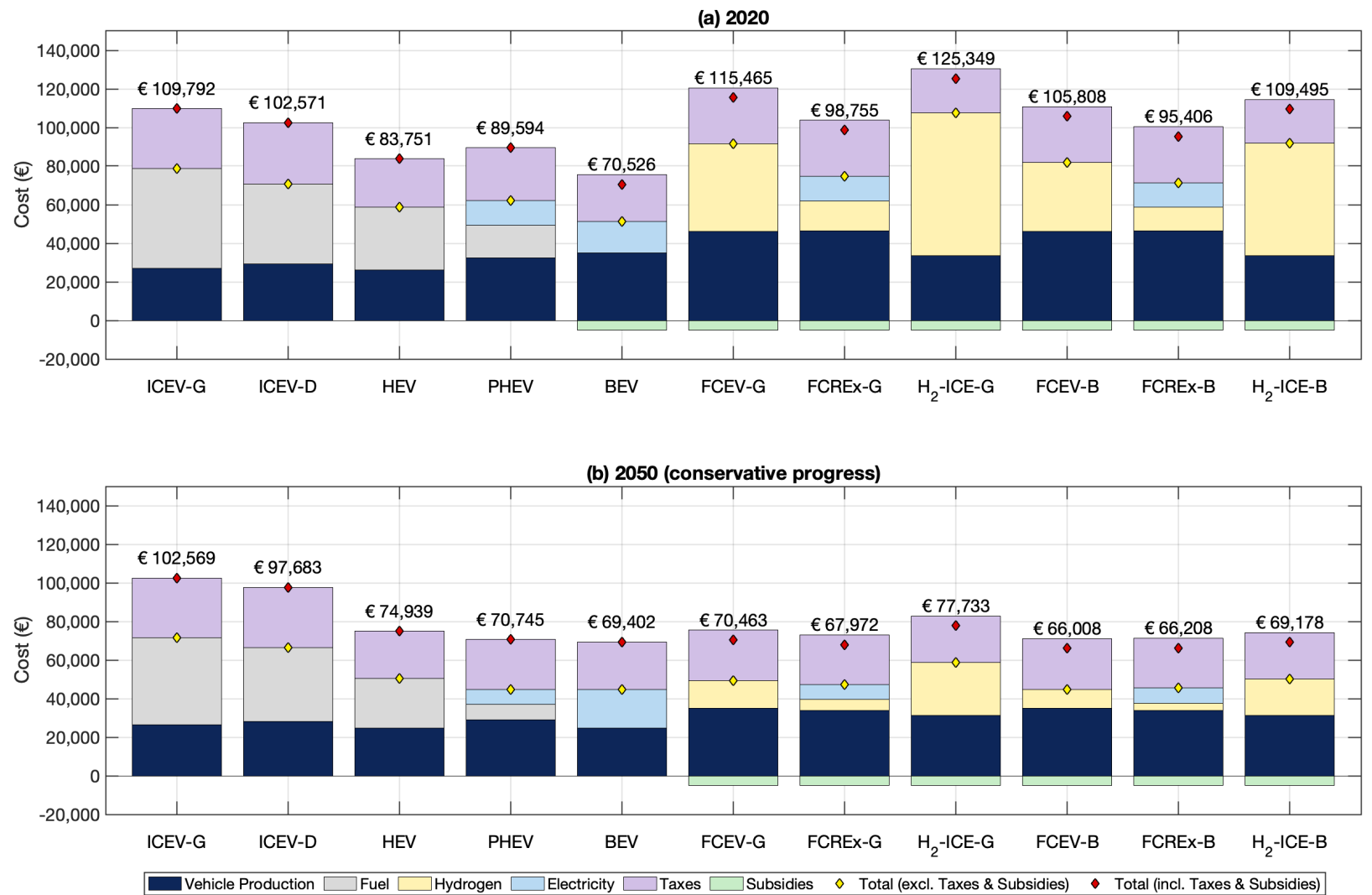


Figure 35: Total life cycle cost of vehicle powertrains in (a) 2020 and (b) 2050 conservative fuel and vehicle price progress, including the impact of taxes, subsidies and policy levers

5.4.1 Cost per mile Analysis

Figure 35 illustrates the projected cost per mile for various vehicle powertrains from 2020 to 2050. In 2020, hydrogen-powered vehicles, particularly FCEVs and H_2 -ICE, exhibit the highest cost per mile, ranging from €0.63 to €0.68. In contrast, ICEV costs remain relatively stable over time, with minor fluctuations due to variations in fuel prices. Notably, fuel prices are projected to increase by 2050, driven by global oil market trends and environmental regulations (Table 40).

Hybrid electric vehicles demonstrate moderate cost efficiency, with costs declining from €0.46 per mile in 2020 to €0.41 by 2050 due to technological advancements and improved fuel efficiency. Similarly, PHEVs follow a comparable trajectory, benefiting from advancements in battery technology and reductions in electricity and fuel costs.

BEVs are the most cost-effective powertrain from the outset, with an initial cost of €0.35 per mile in 2020. This cost advantage is primarily attributed to the lower price of electricity (€0.06 per mile in 2020) compared to other fuel sources. Over time, further reductions occur, reaching €0.26–0.28 per mile by 2050, primarily driven by declining battery prices.

Among hydrogen powertrains, FCREx vehicles present the lowest cost per mile (€0.52 in 2020), benefiting from partial reliance on electricity, which offsets the high cost of hydrogen fuel. This cost continues to decline, reaching €0.39 per mile by 2050, comparable to PHEVs. Likewise, FCEVs experience a significant cost reduction to €0.41 per mile in 2050, aligning with HEV costs. These findings suggest a narrowing cost gap between battery electric and hydrogen-powered vehicles, indicating that by 2050, hydrogen vehicles could offer higher range capabilities at a competitive cost. However, these cost reductions depend on technology improvements and the large-scale deployment of hydrogen production. It is worth noting that in 2035, the cost per mile for ZEVs is slightly higher than in 2030 because of the anticipated phase-out of government subsidies.

The model shows that H_2 -ICE vehicles have the highest cost per mile (€0.68 in 2020). While costs decline to €0.45 per mile by 2050 - primarily due to reductions in hydrogen fuel costs and improved vehicle fuel efficiency - H_2 -ICE remains more expensive than other ZEVs. Nevertheless, in the short to medium term (2025–2030), H_2 -ICE could offer a competitive alternative to conventional gasoline vehicles, with moderately lower costs by 2030. H_2 -ICE could not only provide immediate emissions reductions at a comparable cost per mile but also drive investments in hydrogen production, storage, and refuelling infrastructure. As a transitional technology, H_2 -ICE could stimulate demand, encouraging the expansion of hydrogen refuelling stations that would ultimately benefit more cost-effective hydrogen powertrains such as FCEVs and FCREx vehicles.

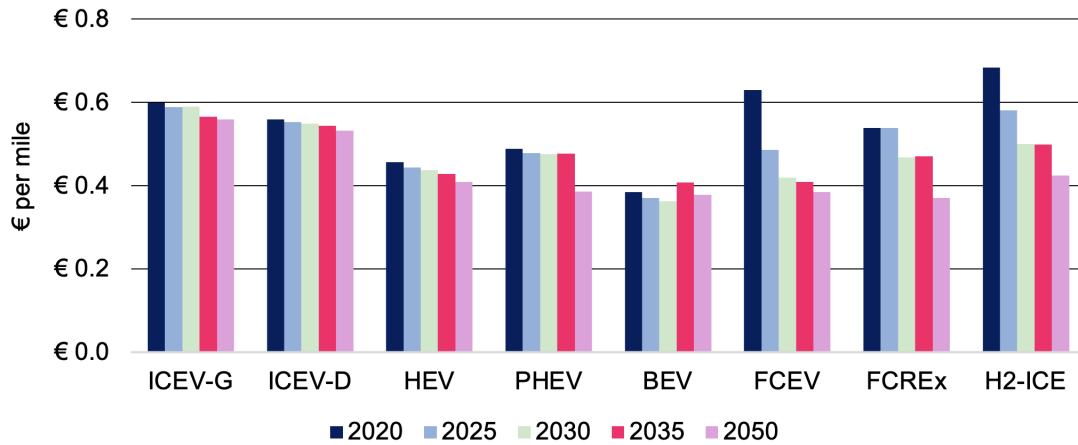


Figure 36: Cost of powertrains per mile (€/mile) from 2020 to 2050

5.4.2 Impact of electricity costs of lifecycle cost results

A sensitivity analysis was conducted to evaluate the impact of charging behaviours on the overall cost-effectiveness of BEVs. Three scenarios are considered: (1) primarily charging at home, (2) frequent but not exclusive home charging, and (3) absence of home charging infrastructure. The findings indicate that, by 2050, BEV owners without access to home charging - particularly urban residents without private driveways - may find hydrogen-powered drivetrains either cost-equivalent (FCEV) or more cost-effective (FCREx) alternatives Figure 37 and Figure 38. However, BEVs remain the most economical option when predominantly charged at home, especially during off-peak hours.

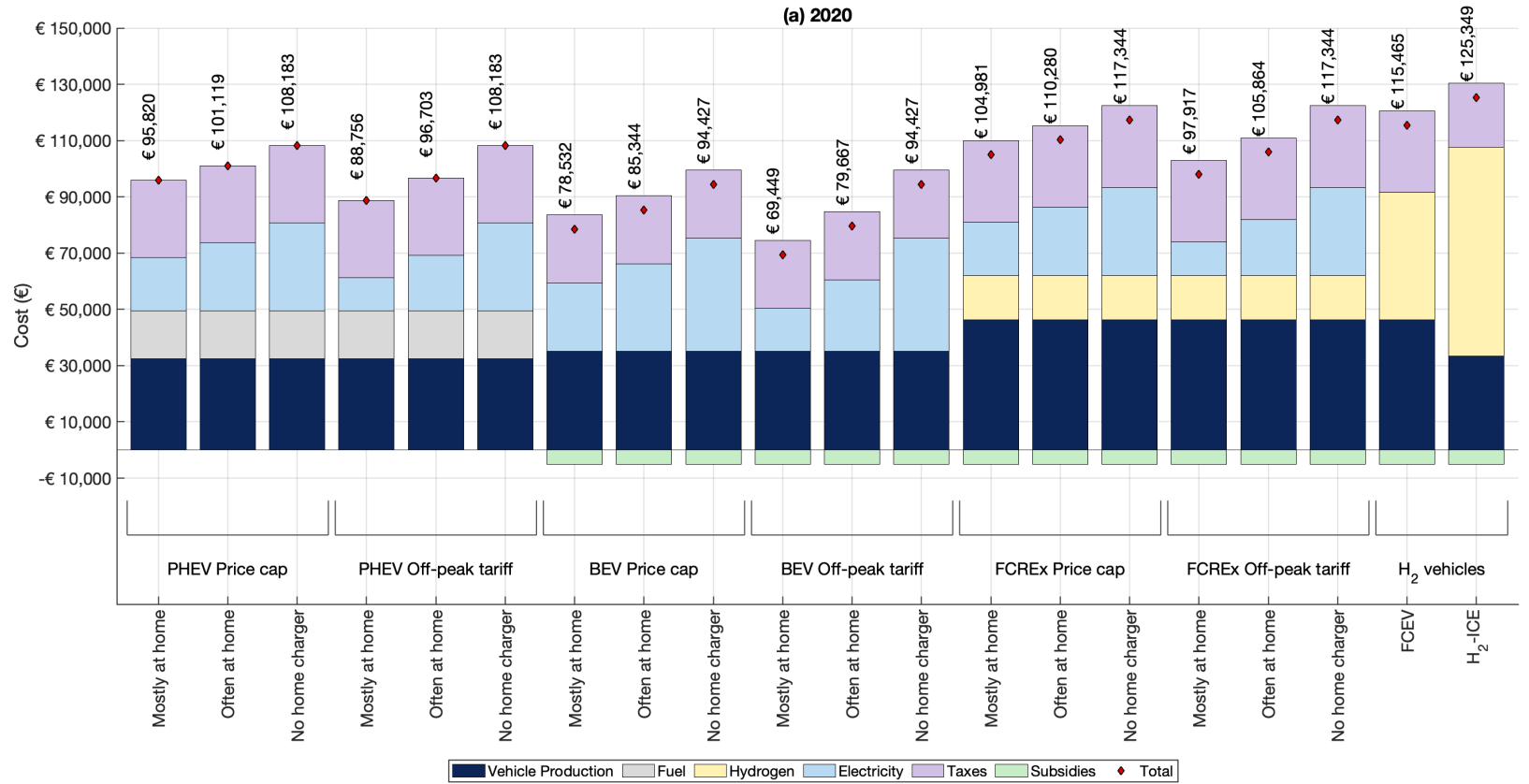


Figure 37: Sensitivity analysis: The impact on varying electricity costs on the LCC of BEV vehicles and comparison with H_2 and Conventional vehicles in 2020. Note: the cost of average European electricity in 2020 is €0.29 (conservative/aggressive), €0.30 (price cap) and €0.10 off-peak. Hydrogen costs displayed here are from green production methods.

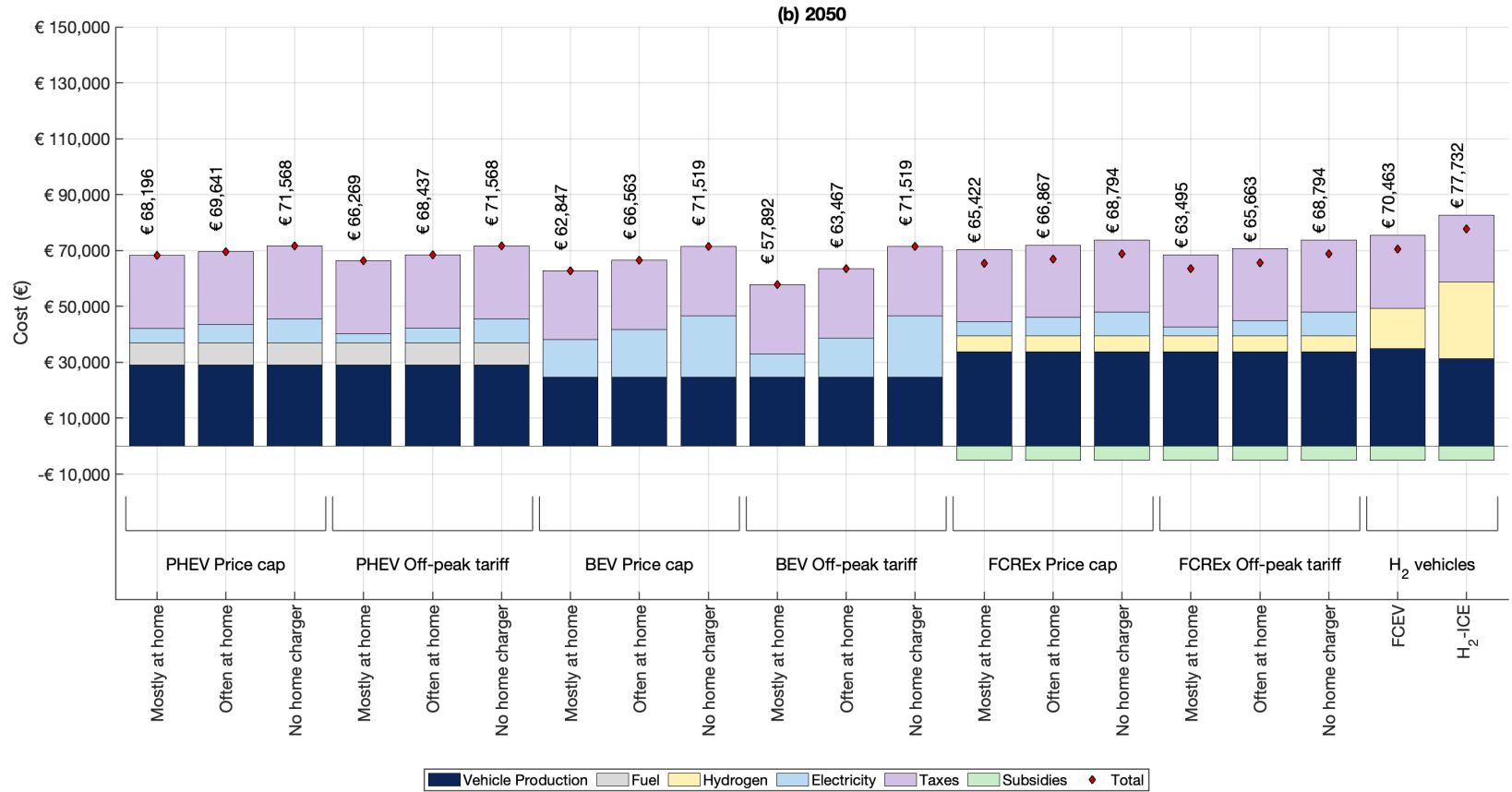


Figure 38: Sensitivity analysis: The impact on varying electricity costs on the LCC of BEV vehicles and comparison with H_2 and Conventional vehicles in 2050. Note: the cost of average European electricity in 2050 is €0.35 (conservative), €0.38 (aggressive) and is fixed for price cap and off-peak. Hydrogen costs displayed here are from green production methods.

The subsequent analysis displays the impact of electricity pricing on BEV cost-effectiveness. Charging primarily at home, particularly using off-peak tariffs, leads to the lowest lifetime costs, reinforcing the importance of accessible and affordable home charging infrastructure. Conversely, reliance on public or workplace charging leads to higher costs, narrowing the economic advantage of BEVs compared to hydrogen vehicles. Future reductions in BEV operating costs are expected to be driven by declining battery and electricity prices and improvements in battery efficiency. Hydrogen powertrains, while currently less cost-effective, are projected to become increasingly competitive, particularly in scenarios where BEV home charging is impractical. The projected decrease in hydrogen fuel prices, coupled with advancements in fuel cell efficiency, suggests that by 2050, FCEVs and FCREx vehicles could offer a viable alternative for consumers with limited charging access. Moreover, H_2 -ICE vehicles, despite remaining more expensive than BEVs, may serve as a transitional technology in advance of lower-cost fuel cell drivetrains, facilitating the development of hydrogen infrastructure and accelerating market adoption.

5.5 Abatement costs

Figure 39 presents the abatement costs (€ per ton CO₂) for BEVs and hydrogen-powered powertrains compared to internal combustion engine vehicles using gasoline (ICEV-g). These costs are calculated based on total cost differences and the respective CO₂ reductions achieved over each technology's lifetime. ICEV-g reflects vehicle efficiency and emissions for the modelled year rather than a fixed 2020 baseline or zero-cost reference.

In 2020, BEVs is the most cost-effective option for reducing emissions, with a negative abatement cost of €59 per ton CO₂ avoided. This cost-effectiveness is primarily attributed to lower operational expenses, particularly the lower cost of electricity compared to gasoline. However, charging strategies significantly influence these savings. Charging off-peak at home yields the greatest cost benefits (€60), whereas reliance on public or workplace charging results in higher costs (Figure 39A). Among hydrogen powertrains, FCREx vehicles and H_2 -ICE powered by blue hydrogen are the only technologies to show negative abatement costs in 2020; €10-16 per ton CO₂ avoided for FCREx, and a slightly negative cost of -€0.27 per ton CO₂ for H_2 ICE. Despite the relatively high cost of hydrogen fuel overall, FCREx vehicles benefit economically from their ability to operate on electricity for 50% of their usage, reducing fuel expenditure and improving cost-effectiveness. In contrast, FCEVs and H_2 -ICE vehicles powered by green hydrogen show positive abatement costs of €20 and €31 per ton CO₂, respectively, indicating that while these technologies contribute to emis-

sions reductions, their economic viability is currently constrained by the high cost of electrolytic hydrogen.

By 2050, abatement costs decline considerably across all technologies, reflecting substantial cost reductions and enhanced efficiency. Notably, green hydrogen fuelled H_2 -ICE vehicles, which exhibited the highest abatement cost in 2020, see their cost drop to -€139 per ton CO₂ in an advanced scenario, largely due to projected decreases in green hydrogen fuel prices (Figure 39C) and lower vehicle production costs. This underscores the potential of H_2 -ICE powertrains as a viable low-carbon alternative, provided further improvements in the scale-up and cost reduction of hydrogen fuel.

Although BEVs consistently achieve the lowest lifecycle emissions across scenarios, abatement cost modelling favours hydrogen technologies in the future, particularly green hydrogen fuelled H_2 -ICE vehicles. This is largely due to the projected reduction of hydrogen fuel and vehicle costs, compounded by discounting mechanisms that make long-term investments cost-effective in net-present value. In summary, our findings show that the economic of hydrogen vehicles improve as they mature. The declining abatement costs from 2020 to 2050 emphasise the role of technology improvements and economies of scale in making hydrogen-powered vehicles financially viable to contribute to large-scale transport sector deployment.

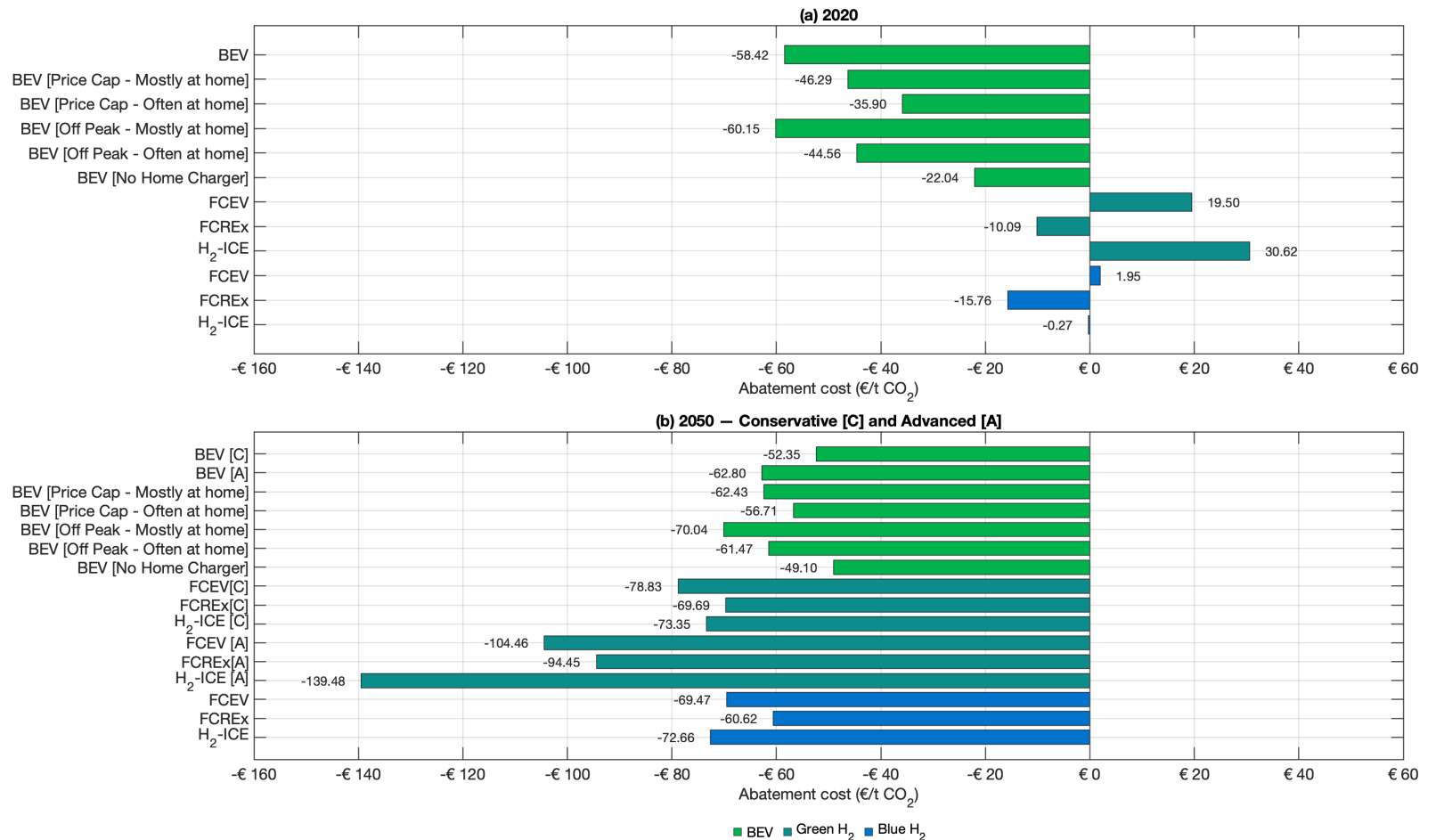


Figure 39: Abatement cost of alternative vehicle powertrains relative to a gasoline internal combustion engine vehicle (ICEV-g), expressed in €/ton CO₂. Panels show results for 2020 (a) and 2050 (b), with Conservative [C] and Advanced [A] technology scenarios respectively. Negative abatement costs indicate that the alternative powertrain delivers emissions reductions at a net cost saving compared to ICEV-g; therefore, larger negative values represent greater cost-effective decarbonisation potential. Positive values indicate higher cost per tonne of CO₂ abated.

5.6 Discussion

The results of this study provide critical insights into the environmental and economic performance of various vehicle technologies from 2020 to 2050. Our comparative LCA shows the differences in emissions across the lifecycle stages of vehicles. Similarly, the LCC analysis highlights the evolving cost-competitiveness of these technologies over time, reflecting the impact of technological advancements, fuel price fluctuations, and policy interventions.

The LCA results demonstrate that ICEVs currently have the highest total life-cycle emissions, primarily due to the combustion of fossil fuels during the use phase (Figure 31). Despite lower emissions in the production phase, their heavy reliance on non-renewable energy sources leads to a substantial environmental burden. One of the key findings of this study is that compared to ICEVs, electrified vehicles including hybrids, range-extended vehicles and fully electric models, show a shift in the dominant source of life cycle emissions from the use phase to materials and manufacturing. Additionally, the assumptions underlying real-world driving behaviours, recycling credits and infrastructure choices are critical in shaping the relative environmental performance of different vehicle technologies (Figure 31).

BEVs, despite having higher production-phase emissions due to battery manufacturing, still have the lowest overall lifecycle emissions. This is mainly attributed to the near elimination of tailpipe emissions during the use phase - ultimately, as the electricity grid decarbonises, so too does the emissions impact of BEVs. Hydrogen-powered vehicles, particularly those using green hydrogen, present a promising alternative with lower use-phase emissions. However, their environmental impact is highly influenced by upstream emissions, especially in the manufacture of hydrogen storage and H_2 fuel transportation. The sensitivity analysis shown in Figure 30 on the use of secondary materials illustrates the potential to mitigate these impacts, particularly in hydrogen vehicles, where a reduction in manufacturing emissions is observed when the share of recyclates is increased.

The analysis of hydrogen transport and distribution scenarios further highlights the importance of infrastructure choices in minimising the lifecycle emissions of hydrogen vehicles. The base case scenario, involving diesel-powered tube trailers, indicates significant emissions, particularly over longer transport distances. However, the adoption of dedicated hydrogen pipelines or the switch to lower-emission fuels, such as renewable hydrogen, can markedly reduce T&D emissions (Figure 32). That said, pipelines require very large, continuous flow rates thus limiting the economic viability to mature, large-scale markets (International Energy Agency (IEA), 2019), (Khan et al., 2021). This finding emphasises the need for strategic investments in

hydrogen infrastructure to enhance the environmental viability of hydrogen-powered vehicles.

In this study, we assume current biofuel blending for gasoline and diesel and do not model future drop-in fuels (European Court of Auditors, 2023). Fixing the fuel mix lets us isolate hydrogen powertrain impacts and costs. Although out of scope here, renewable drop-in fuels can cut ICEV upstream GHGs but carry a price premium over conventional fuels (Brown et al., 2020). Future LCA and LCC studies should include biofuels to evaluate liquid-fuel decarbonisation alongside hydrogen.

The LCC analysis reveals the dynamic nature of vehicle cost competitiveness over time. At present, green hydrogen powered vehicles exhibit the highest overall costs (Figure 35), driven by expensive hydrogen fuel and vehicle components (fuel cells and H_2 storage tanks). Interestingly, FCREx vehicles would already be cost-competitive with ICEVs if deployed today. This is largely because their hybrid operation (utilising 50% electricity) substantially offsets the cost burden of hydrogen fuel. Similarly, all hydrogen powertrains fuelled by blue hydrogen are currently cost-competitive with gasoline ICEVs. Blue hydrogen is expected to play a critical role in meeting hydrogen demand during the transition phase, particularly as green hydrogen production scales up (Noussan et al., 2021). However, its use raises concerns about continued reliance on fossil fuel infrastructure and the long-term sustainability of SMR. For example, the environmental performance of blue hydrogen is highly sensitive to upstream methane leakage and the efficiency of CCS systems (Bauer et al., 2022), (van der Spek et al., 2022), (Howarth and Jacobson, 2021), (International Energy Agency (IEA), 2022b).

Projections indicate that the cost of green hydrogen is expected to fall considerably, making hydrogen vehicles more competitive by 2050. In comparison, BEVs have the lowest lifecycle costs through the study period, benefiting from declining battery costs and improved energy efficiency - favouring BEVs as the dominant low-cost option. However, sensitivity analysis on BEV charging habits highlights the influence of infrastructure availability on cost-effectiveness. For residents without access to home charging, hydrogen vehicles may present a more viable option in the future, especially if the costs of hydrogen production decrease as expected. However, the limited availability of hydrogen refuelling infrastructure currently poses a significant barrier to widespread adoption. In contrast, BEVs are likely to remain the most economically viable option for suburban and rural users who have access to home or workplace charging, further reducing their operational costs and enhancing their overall cost competitiveness.

Beyond cost dynamics, understanding vehicle survival rates offers further nuance to the long-term viability of zero-emission technologies. This study assumes a lifespan of 15 years across all ZEVs as there is limited data to differentiate lifetimes for

hydrogen vehicles as the first vehicle deployment are yet to reach their end-of-life. For BEVs, researchers have found that lifetimes now match ICEV counterparts (University of Birmingham, 2025). However, it is worth noting that survival rates may differ between BEVs and hydrogen powertrains with factors such as technology learning curves and maintenance ease influence these rates. Improved access to primary data would allow future research to explore these differences more deeply, to understand how the variability in survival rates could impact the long-term environmental and economic viability of each vehicle.

Despite BEVs exhibiting the lowest lifecycle emissions and cost advantages in early projections, long-term abatement cost models (Figure 39) are shown to favour hydrogen vehicles, particularly H_2 -ICE due to anticipated reductions in fuel and vehicle prices. Discounting future costs within these models magnifies economic benefits of delayed investments, making hydrogen powertrains attractive for large-scale decarbonisation in 2050 scenarios. This divergence between environmental and economic optimality underscores the need for multi-metric evaluation frameworks when comparing low-carbon transport technologies.

Considering the analysis of powertrains on a cost per mile basis (€), H_2 -ICE could position itself as a competitive technology in the medium-term (2030) having moderately lower costs than a conventional vehicle, competitive with PHEVs, whilst offering emissions reductions. H_2 -ICE benefit from existing ICEV manufacturing infrastructure that could be repurposed for H_2 -ICE production with minimal investment, trained mechanics who could service ICE parts and a familiarity to consumers compared to FCEVs. These factors could ease the transition to hydrogen technologies whilst stimulating the demand for hydrogen fuel in the automotive sector.

Future research should consider incorporating the total cost of ownership (TCO) as an additional evaluation metric, given the evolving market conditions and increasing importance of depreciation rates in consumer decision-making. As recent studies have highlighted (Titcomb, 2024), the depreciation rates for BEVs may be higher than anticipated, which could significantly impact their overall economic appeal. For example, Autotrader (Titcomb, 2024) finds that EVs lose >50% of their value in just three years, for ICEV this is currently 35-40%. On average, people in the UK own their cars for 5 years, meaning that while the first owner may face significant depreciation, the next buyer stands to benefit from a reduced purchase price. Thus, giving an imbalance in the TCO of first and subsequent EV owners. Future implementation of taxes and subsidies are unexpected. Whilst this study follows the current policy direction of little subsidies for ZEVs, future political decisions could provide more favourable conditions to reduce the total LCC of BEV or H_2 vehicles. However, due to the unexpected nature of these economic instruments they have not

been incorporated into the modelling of this study.

It is also important to recognise that this study has a Europe-centric focus, so caution should be taken when applying its findings to other regions as our cost and environmental assumptions may not reflect market realities elsewhere. Notably, Europe’s electricity grid is already heavily decarbonised, whereas the average grid in Asia is twice as carbon-intensive (Ember, 2024); therefore, GHG reduction and abatement costs may not be as high in other regions. However, there are key insights from this study that can be generalised. For example, the high cost of producing green hydrogen across most regions. While specific cost drivers such as renewable electricity prices, electrolyser costs, and financing conditions vary globally, green hydrogen is still significantly more expensive than fossil-derived alternatives, making cost a central consideration (Sun and Elgowainy, 2019), (International Council on Clean Transportation (ICCT), 2022). As a result, the economic viability of hydrogen vehicles powered by green hydrogen depends on ongoing cost reductions, technological advancements, and supportive policy frameworks.

5.7 Summary

Our research findings provide a comprehensive analysis of the environmental and economic performance of hydrogen and electrified vehicles, projecting futures to 2050. We find a complex interplay between technology improvements, infrastructure developments and policy interventions in shaping the low-carbon vehicle transition. While BEVs currently offer the low-cost, low-emission technology option, there are scenarios where hydrogen powertrains are more affordable. Notably, the sensitivity analysis of charging/refuelling access highlights the importance of infrastructure planning in ensuring equitable access to low-carbon transport. Future research incorporating the TCO, considering key factors such as depreciation and evolving policy landscapes are important to refine these projections and inform effective decarbonisation strategies.

Chapter 6

Tier 3 Results: UK Light-Duty Vehicle Fleet Modelling

This chapter extends the previous analysis carried out in Chapters 4 and 5 to the national scale, modelling the deployment of hydrogen technologies across the UK light-duty vehicle fleet. Using a dynamic stock model and scenario-based approach, it simulates fleet turnover, energy demand, lifecycle emissions, and economic costs from 2025 to 2050 under varying hydrogen adoption pathways. The scenarios reflect different policy ambitions, infrastructure development rates and technology mixes, allowing for comparative evaluation of environmental and economic outcomes. This chapter addresses Research Question 3 and 4 by quantifying the system-wide implications of hydrogen integration in UK transport decarbonisation.

6.1 Annual and cumulative fleet-wide greenhouse gas emissions of the UK LDV fleet

Fleet-wide life cycle GHG emissions from the UK LDV fleet are projected to decline substantially between 2025 and 2050 across all scenarios, driven by increased electrification, the gradual rollout of hydrogen vehicles and the progressive decarbonisation of upstream supply chains (Figure 40 to 43). Despite these overall reductions, the magnitude and sources of emissions differ depending on the technology adoption pathway.

For the discussion of results we focus on two key scenarios for comparison (Figure 40 and Figure 42). In the Base scenario (Figure 40), which assumes limited hydrogen deployment (5% fleet share by 2050), cumulative life cycle emissions over

the period are estimated at approximately 1.03 GtCO₂e. In earlier years, operational emissions dominate, largely arising from ICEV tailpipe emissions. These emissions decline sharply after 2035 as BEVs become more prevalent (due to the introduction of the ZEV mandate) and the continued decarbonisation of the UK electricity grid (see Figure 7). By 2050, emissions from vehicle manufacturing and end-of-life processes become increasingly significant, reflecting a burden shift from fuel production onto the production of vehicle components, namely battery, fuel cell and hydrogen storage tank materials and production methods as discussed later.

In contrast, the Major H_2 scenario (Figure 42), which assumes a 30% hydrogen vehicle share by 2050, results in cumulative emissions of 1.09 GtCO₂e – a 6% increase compared to the Base scenario. This increase is mainly driven by the higher emissions from hydrogen-related components. Cumulative fuel cell and hydrogen storage tank production emissions increase to 73 MtCO₂e, a 420% increase over the 14 MtCO₂e seen in the Base case (Figure 44), reflecting the higher use of high-pressure CFRP tanks and platinum-group metals. Likewise, hydrogen production emissions reach 24 MtCO₂e, compared to just 4 MtCO₂e in the Base scenario (Figure 44). The increase is closely linked to ongoing reliance on yellow hydrogen, particularly during the early stages of infrastructure deployment.

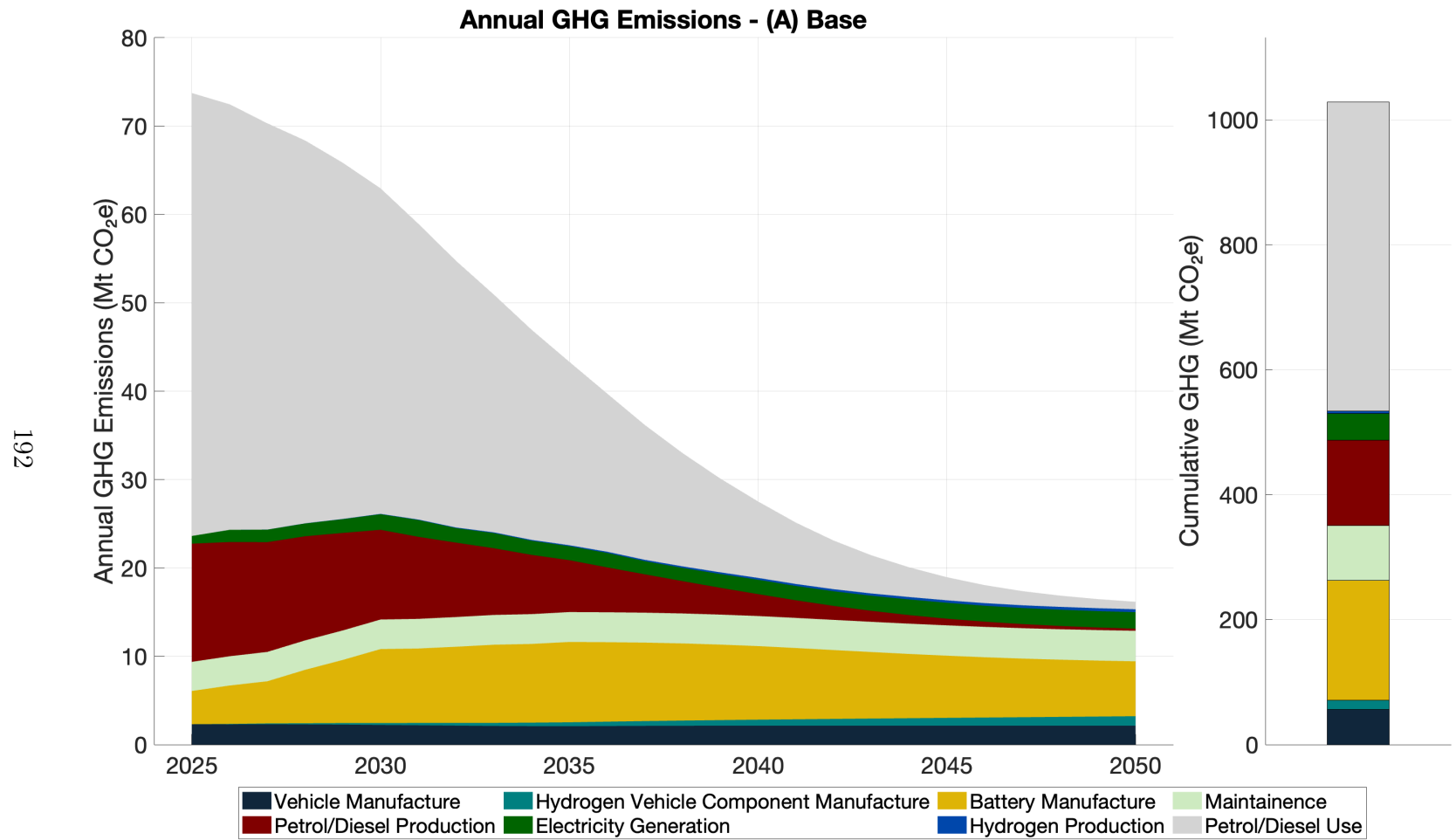


Figure 40: Fleet-wide life cycle GHG emissions for the Base scenario (a) scenario

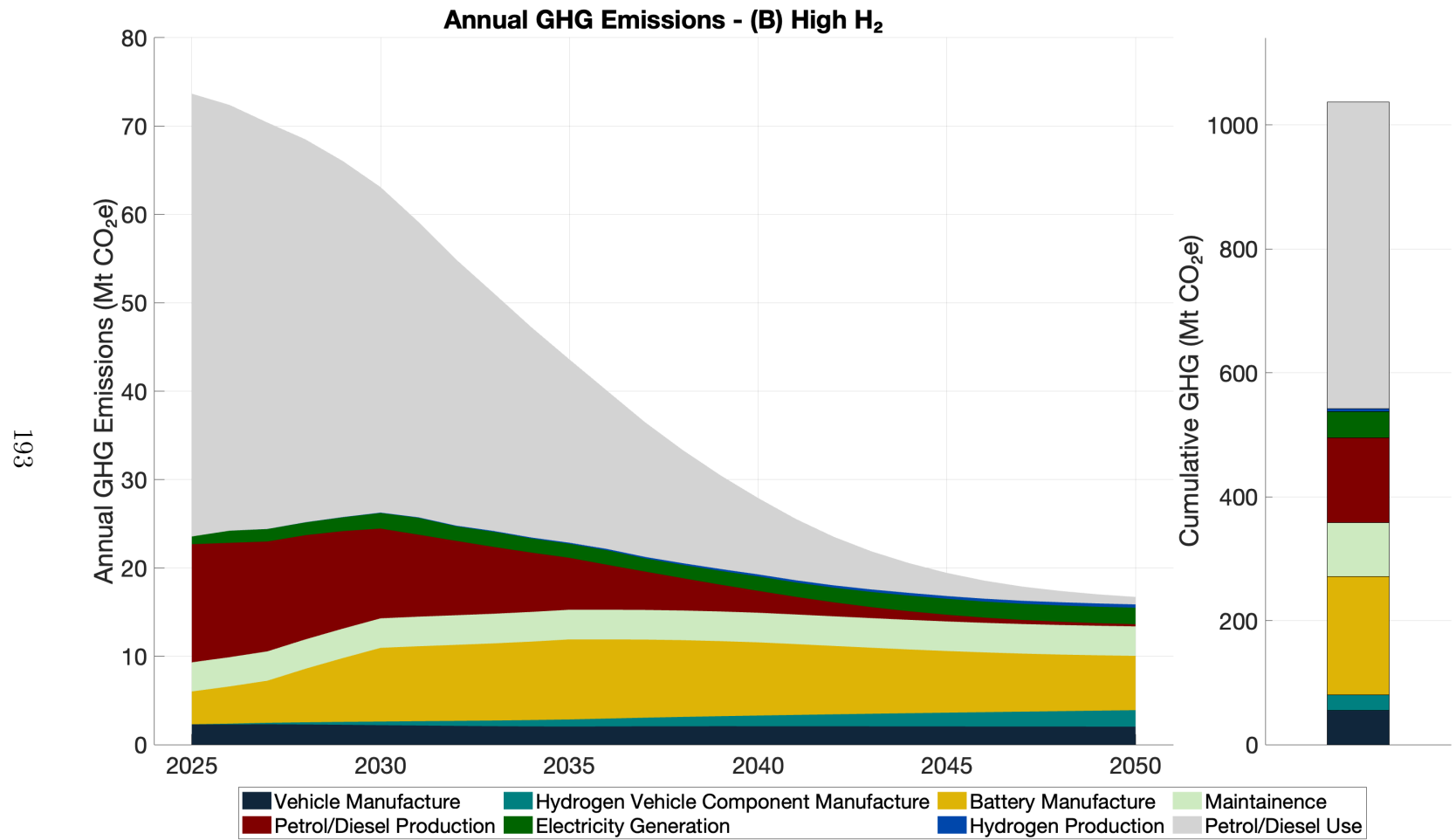


Figure 41: Fleet-wide life cycle GHG emissions for the High H_2 scenario (b) scenario

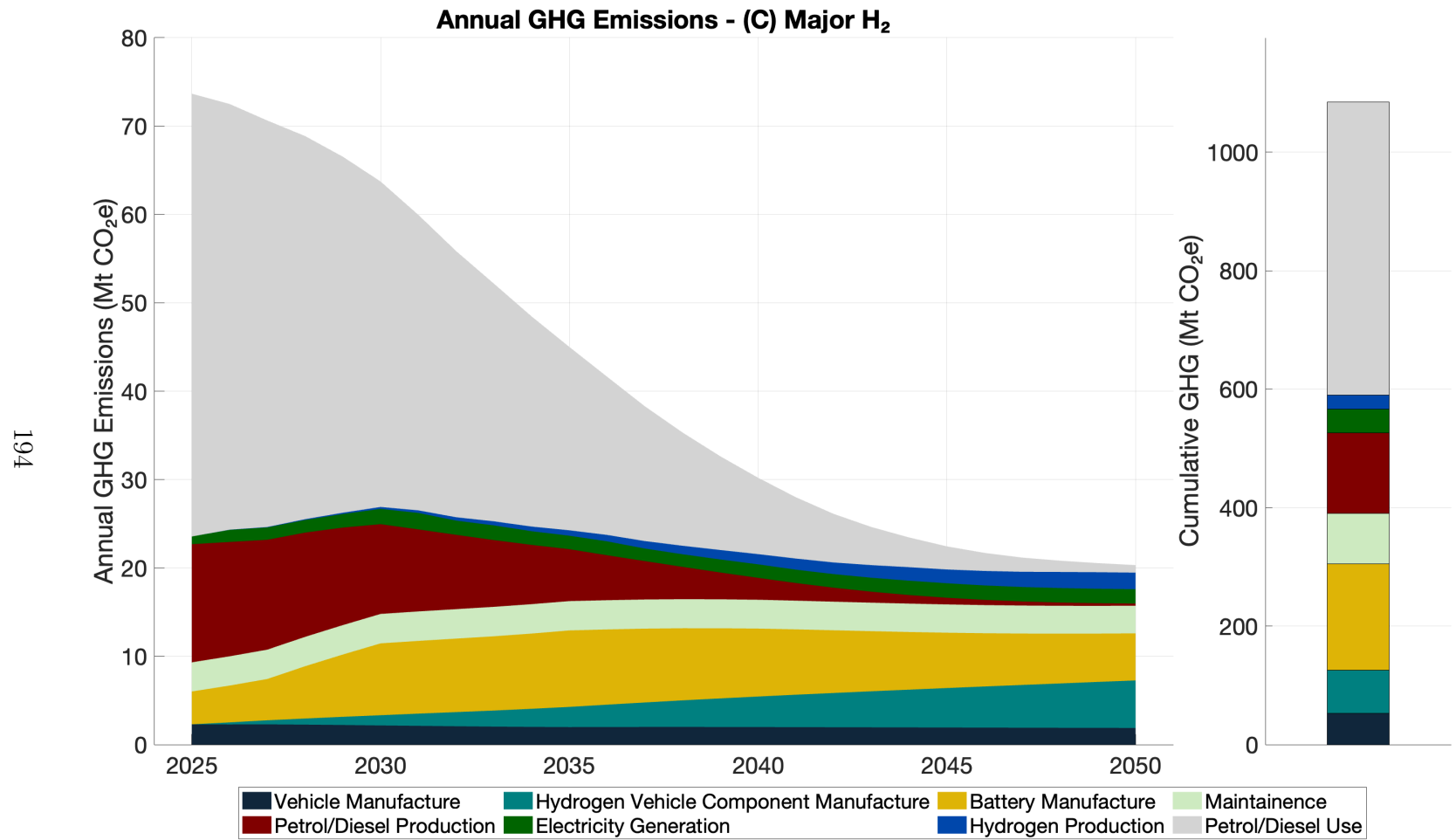


Figure 42: Fleet-wide life cycle GHG emissions for the Major H₂ scenario (c) scenario

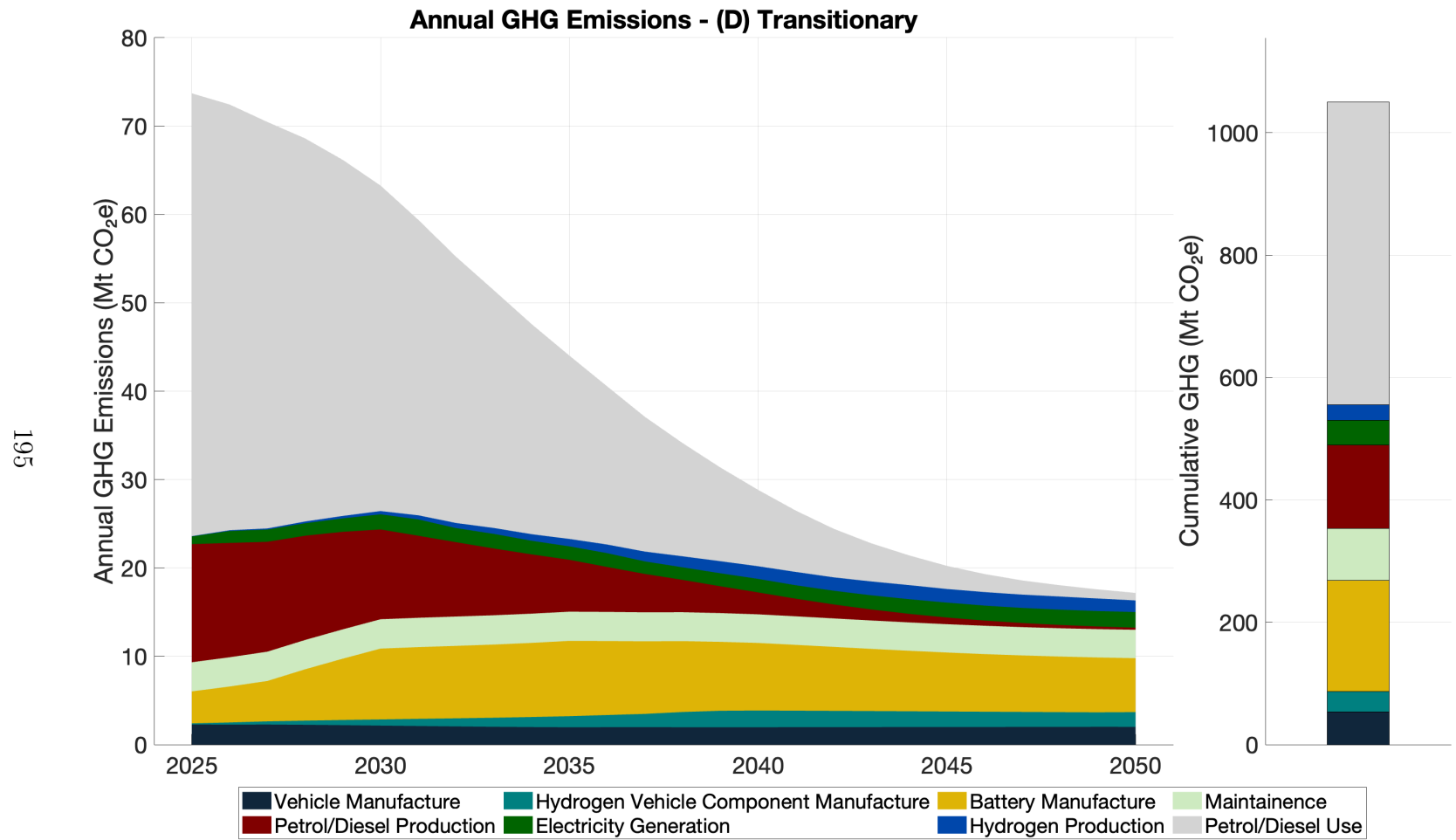


Figure 43: Fleet-wide life cycle GHG emissions for the Transitionary scenario (d) scenario

Across both scenarios, tailpipe emissions (targeted directly by the government’s ZEV mandate) decline as ICEVs are gradually phased out and replaced by zero-emission BEVs and hydrogen-powertrains, which produce no direct CO₂ emissions during use. This results in a significant reduction in operational GHG emissions from the use phase. However, while ZEVs eliminate tailpipe emissions, they introduce a new source of operational emissions upstream: electricity and hydrogen production. This analysis highlights the growing importance of upstream hydrogen production as a contributor to overall emissions. Building upon previous assessments by (Davies et al., 2024), this study explicitly captures these emissions, providing new insight into the GHG implications of hydrogen mobility. In the Major H_2 scenario, hydrogen production accounts for approximately 2% of life cycle emissions by 2050.

Vehicle manufacturing emissions remain relatively stable over time, but as tailpipe emissions fall, they will make up a large share of life cycle emissions by 2050. In both scenarios, cumulative vehicle production emissions (including raw materials and components) are 56-58 MtCO_{2e} (Figure 44). However, as expected in the Major H_2 scenario, the additional requirement for hydrogen-specific components (fuel cells and high-pressure CFRP tanks) increases emissions, raising the total cumulative vehicle production-related emissions (vehicle production plus battery and hydrogen components) from 263 MtCO_{2e} in the Base scenario to 310 MtCO_{2e} (Figure 44). This rise is partly offset by a decline in battery-related emissions, from 192 MtCO_{2e} to 179 MtCO_{2e}, reflecting a lower BEV share. Nonetheless, it is important to note that the decrease in battery emissions does not fully counter the significant increase in emissions from hydrogen-specific components, as hydrogen vehicles still need smaller batteries for auxiliary power and energy buffering, or as part of FCREx configurations.

Overall, while hydrogen vehicles eliminate tailpipe emissions, their full life cycle GHG impacts remain highly sensitive to both the carbon intensity of upstream hydrogen fuel production and the material demands of vehicle technologies. These findings underline the need for parallel strategies to decarbonise hydrogen supply chains, improve material efficiency, and enhance the circularity of critical components to fully realise the environmental benefits of hydrogen in the UK LDV sector.

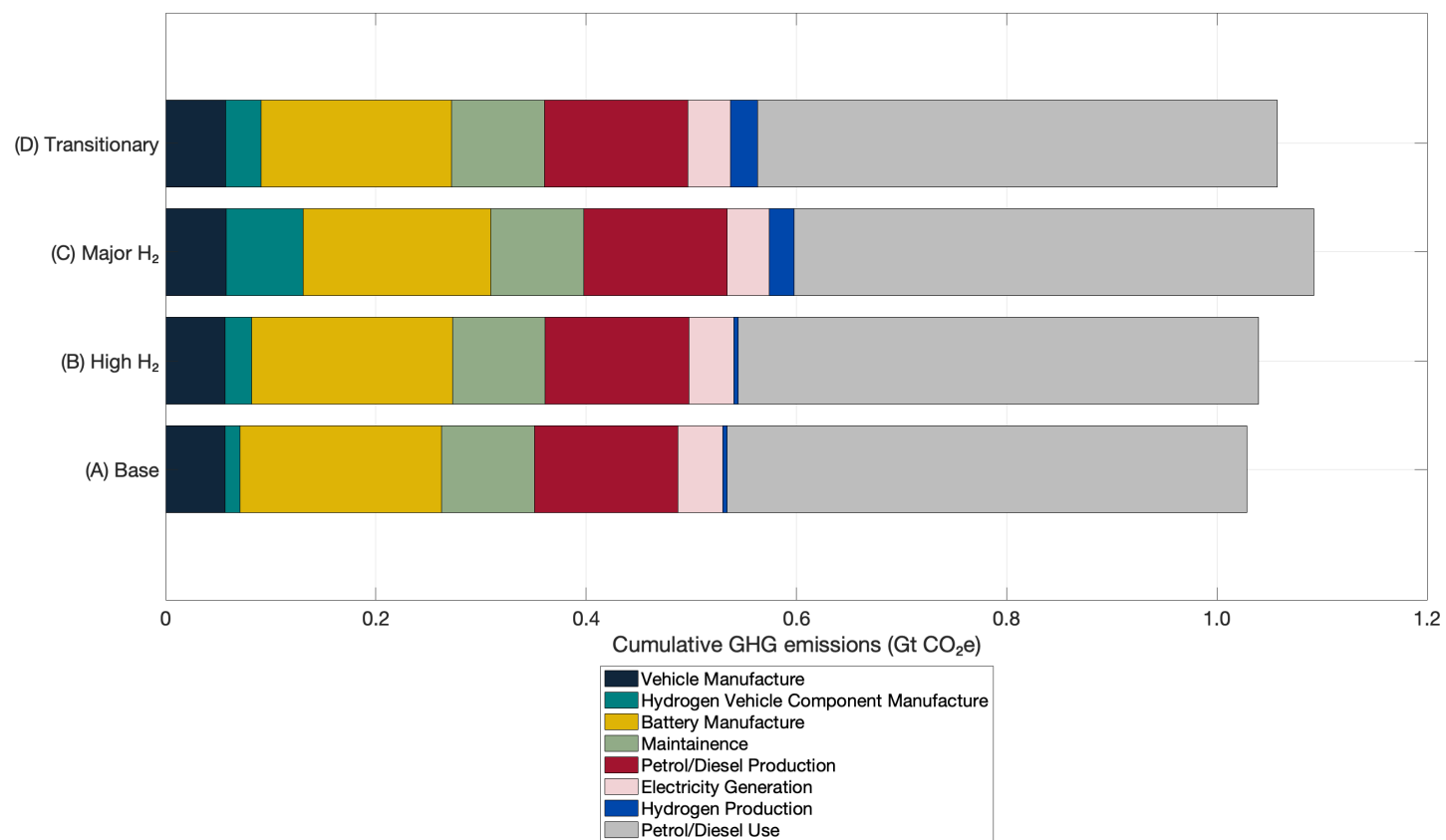


Figure 44: Cumulative life cycle GHG emissions (2025-2050) for the UK LDV fleet across all four hydrogen adoption scenarios

6.2 Annual and cumulative life cycle costs of the UK LDV fleet

This section presents a detailed comparison of the life cycle costs of the UK LDV fleet under two decarbonisation scenarios: a Base scenario dominated by battery electric vehicles, and a Major H_2 scenario featuring substantial hydrogen vehicle deployment. The analysis covers the period from 2025 to 2050, with costs presented both cumulatively and on an annual basis.

As shown in Figures 45 to 48, the Base scenario yields a total cumulative LCC (production costs only) of approximately £1.97 trillion, whereas the Major H_2 scenario shows an 11% increase to £2.18 trillion. When including fuel and electricity costs, the Base scenario totals £3.17 trillion, which increases by 8% to £3.43 trillion in the Major scenario (Figure 49). This cost difference is mainly driven by the higher material and manufacturing costs associated with a greater deployment of hydrogen vehicle technologies. The Major H_2 scenario, for example, experiences particularly steep increases in vehicle production costs due to the introduction of larger hydrogen vehicles such as FCEV SUVs, which require more substantial structural components and higher-cost powertrain systems compared to smaller BEVs (Figure 18).

In 2025, the annual LCC of the LDV fleet is approximately £128 billion (£104 billion excluding use-phase costs) with the dominant cost categories being vehicle production (£53 billion), vehicle maintenance (£31 billion) and petrol/diesel use (£22 billion). Costs associated with hydrogen technologies (fuel cells and tanks) are negligible at this stage, reflecting their minimal deployment in the Base scenario. By 2030, battery production costs peak at £14 billion in 2030 due to increasing EV uptake and high initial battery costs. However, the cost of batteries is projected to improve over time leading to a reduction in battery production costs of £10 billion in 2050 despite a continued rise in market demand due to battery manufacturing efficiency and technological maturity.

In 2030, hydrogen component costs stay low, making up less than 1% of the total LCC. By 2050, while fuel cell and H_2 tank costs remain relatively modest compared to other categories, they increase to £675 million overall due to a small but growing hydrogen fleet. Overall, the Base scenario shows a steady decline in total annual costs from the late 2020s onwards, driven by technological learning, cost reductions in batteries, and decreasing reliance on fossil fuels. Despite modest increases in hydrogen-related expenditures, the scenario remains mainly driven by battery-electric pathways.

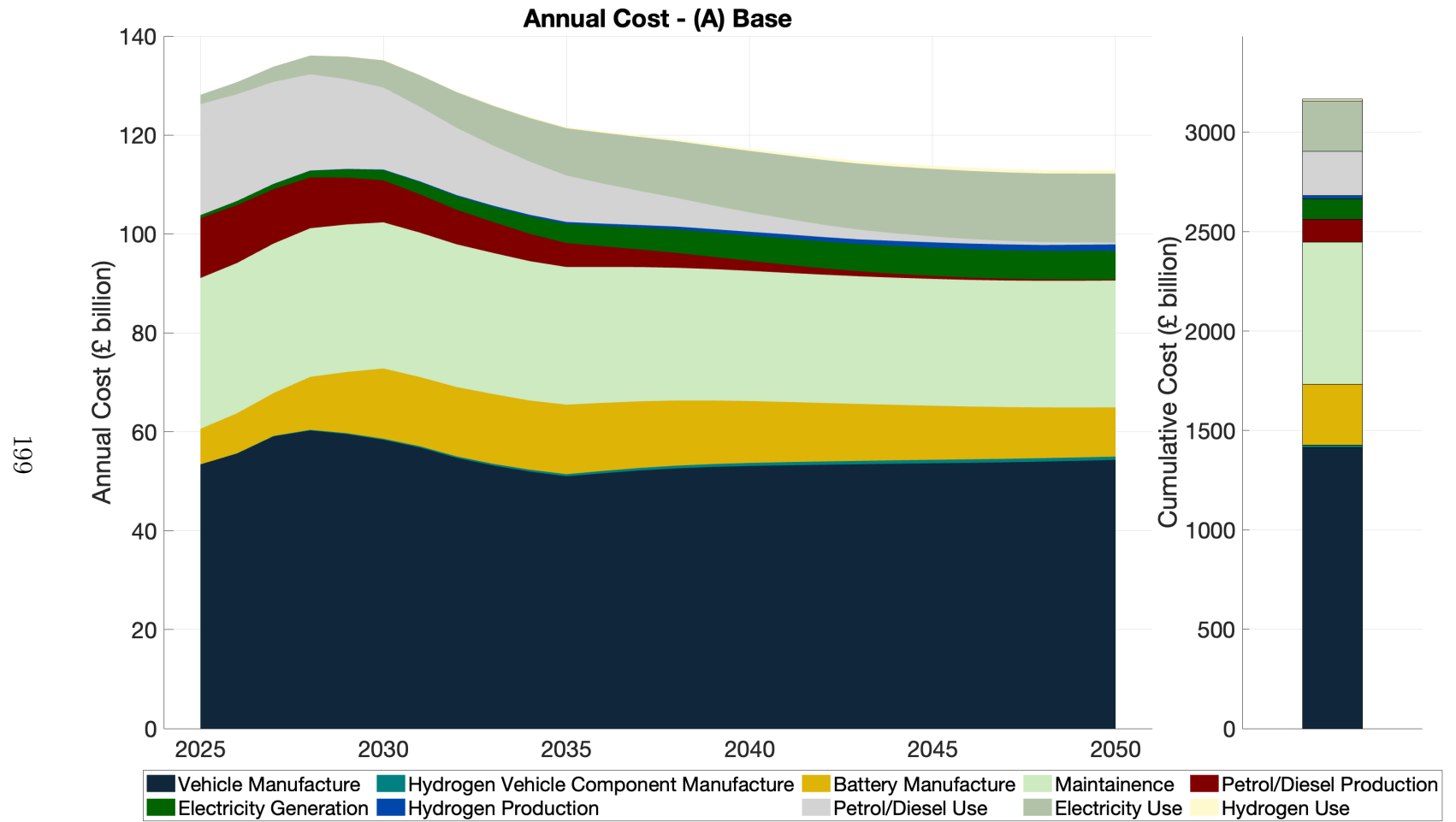


Figure 45: Annualised life cycle costs (£) associated with the UK LDV fleet from 2025 to 2050 in Base (a) scenario

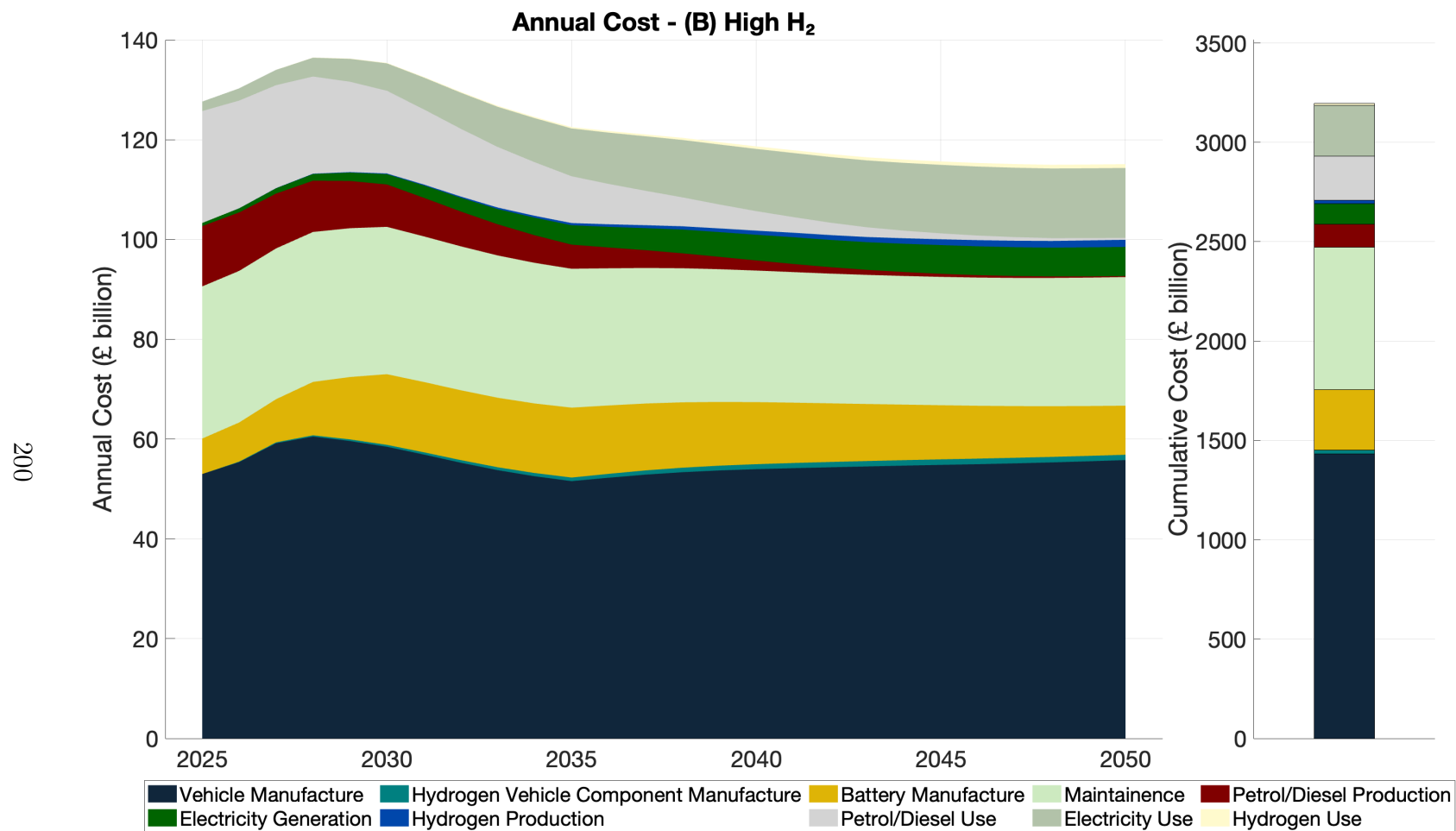


Figure 46: Annualised life cycle costs (£) associated with the UK LDV fleet from 2025 to 2050 in High H_2 (b) scenario

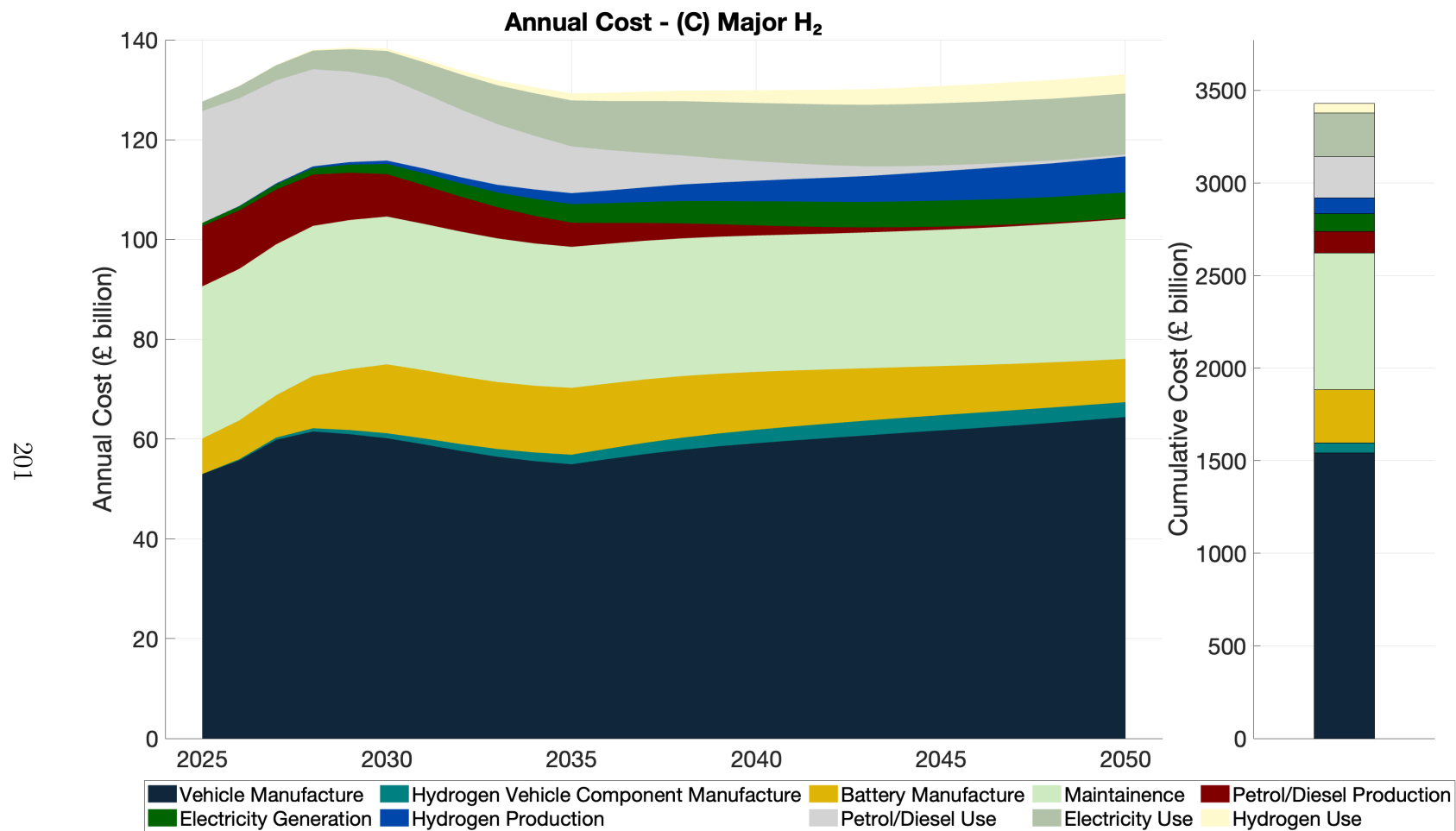


Figure 47: Annualised life cycle costs (£) associated with the UK LDV fleet from 2025 to 2050 in Major H_2 (c) scenario

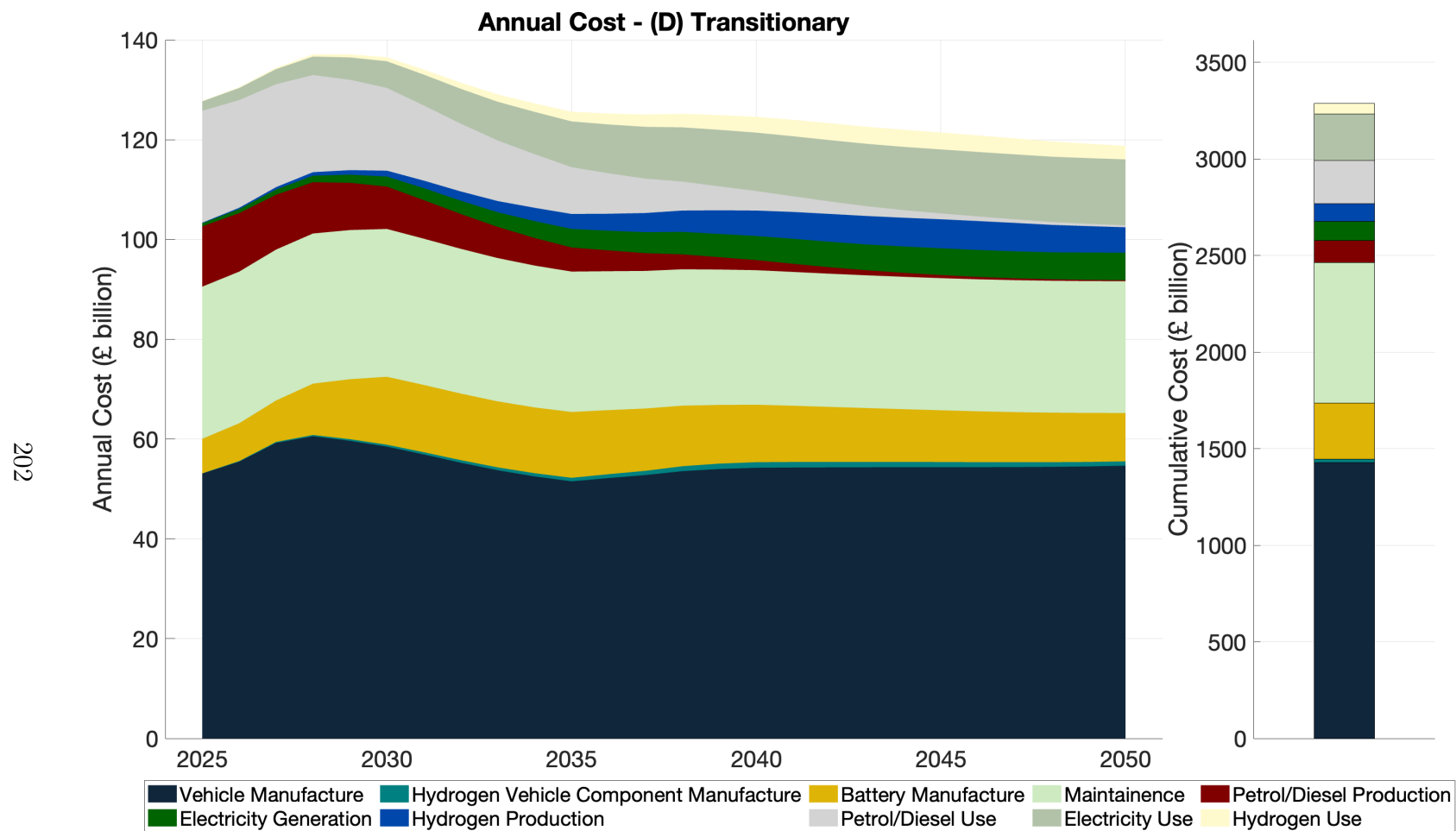


Figure 48: Annualised life cycle costs (£) associated with the UK LDV fleet from 2025 to 2050 in Transitional (d) scenario

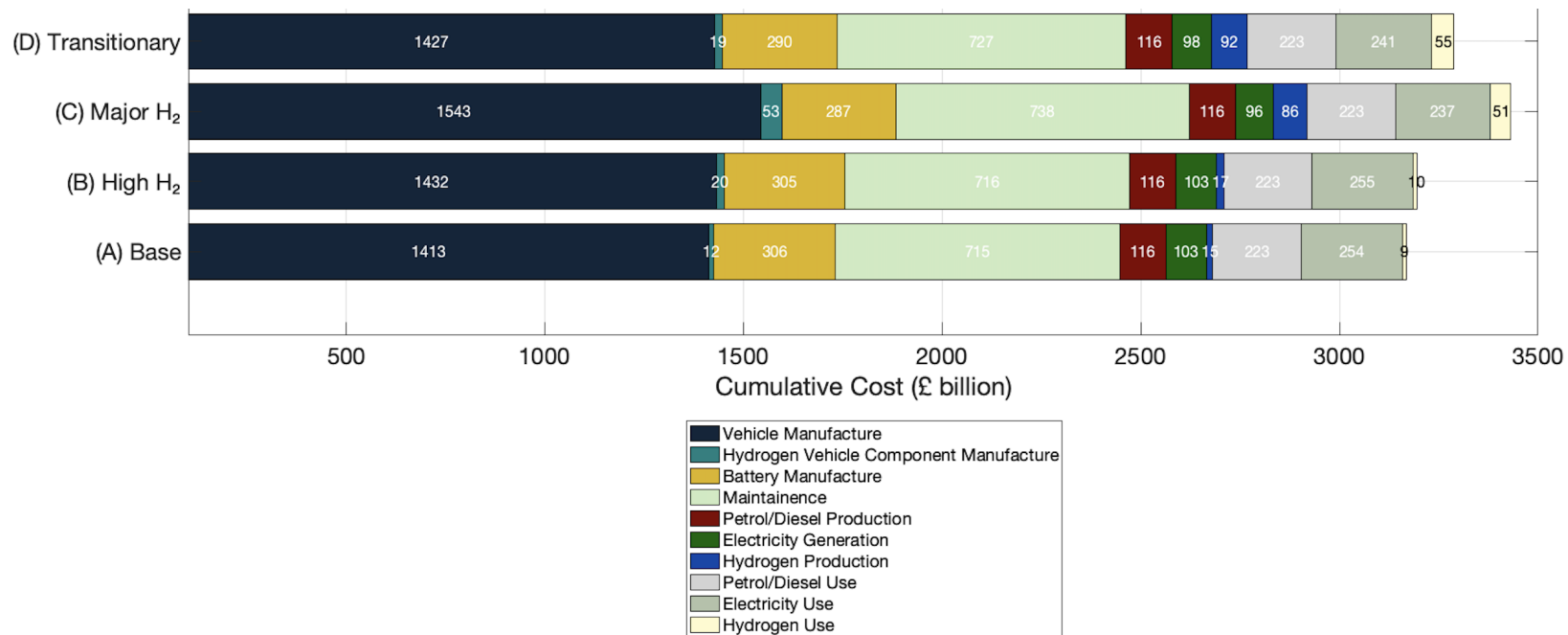


Figure 49: Cumulative life cycle cost (2025-2050) for the UK LDV fleet across all four hydrogen adoption scenarios

6.3 Improved hydrogen fuel cell and storage tank production

To evaluate the impact of future manufacturing advancements on hydrogen vehicles, a sensitivity analysis was carried out on hydrogen fuel cell and storage tank production (Figures 50 to 57). This is based on the assumption that costs and emissions associated with these components will decrease over time due to improvements in manufacturing, such as increased use of renewable energy and reduced use of carbon-intensive materials like PGMs (National Renewable Energy Laboratory, 2018), (National Renewable Energy Laboratory, 2021), (Advanced Propulsion Centre UK, 2020).

Under the Major H_2 scenario, cumulative life cycle costs for fuel cell and tank production drop from £53 billion to £29 billion - a 45% reduction - when improved manufacturing is factored in. In the Base scenario, the same improvement leads to a decrease from £12 billion to £7 billion (42% reduction). These reductions are especially significant given the relatively small share of hydrogen vehicles in the Base scenario, highlighting the disproportionate cost burden of early-stage hydrogen technologies. Quantifying cumulative GHG emissions also sees a notable decline. In the Major H_2 scenario, fuel cell and tank emissions fall from 73 MtCO_{2e} to 49 MtCO_{2e}, a 33% reduction, while the Base scenario sees a drop from 14 MtCO_{2e} to 10 MtCO_{2e} (29% reduction).

Despite these improvements, the overall system-level impact remains modest. Total cumulative costs in the Major H_2 scenario decrease by £24 B, or 0.5%, while GHG emissions fall by 23 MtCO_{2e}, or 2.1%. This reflects the relatively small share of fuel cell and tank production in the total life cycle footprint.

Overall, although improved manufacturing of hydrogen components doesn't significantly change the cost balance between hydrogen mobility and electrification, it's one way to reduce the economic and environmental impact of deploying hydrogen vehicles. This could be backed up by other key measures, such as cutting hydrogen production costs, increasing low-carbon deployment, providing subsidies, and taking advantage of economies of scale. Together, these measures could make hydrogen technologies more cost-competitive and accelerate their integration into a decarbonised transport system.

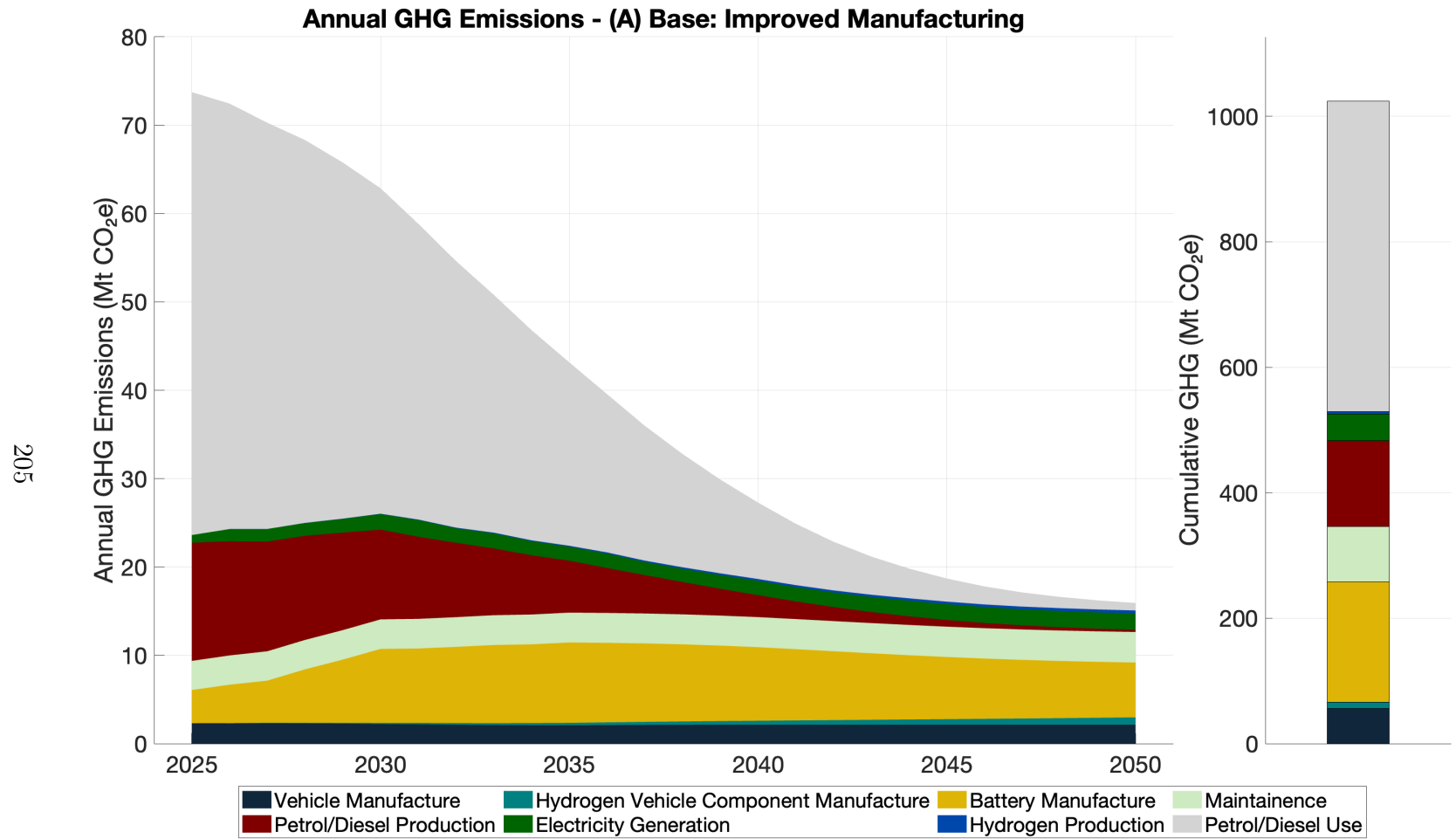


Figure 50: Annualised life cycle GHG emissions associated with the UK LDV fleet from 2025 to 2050 in Base (a) scenario, sensitivity analysis for improved manufacturing

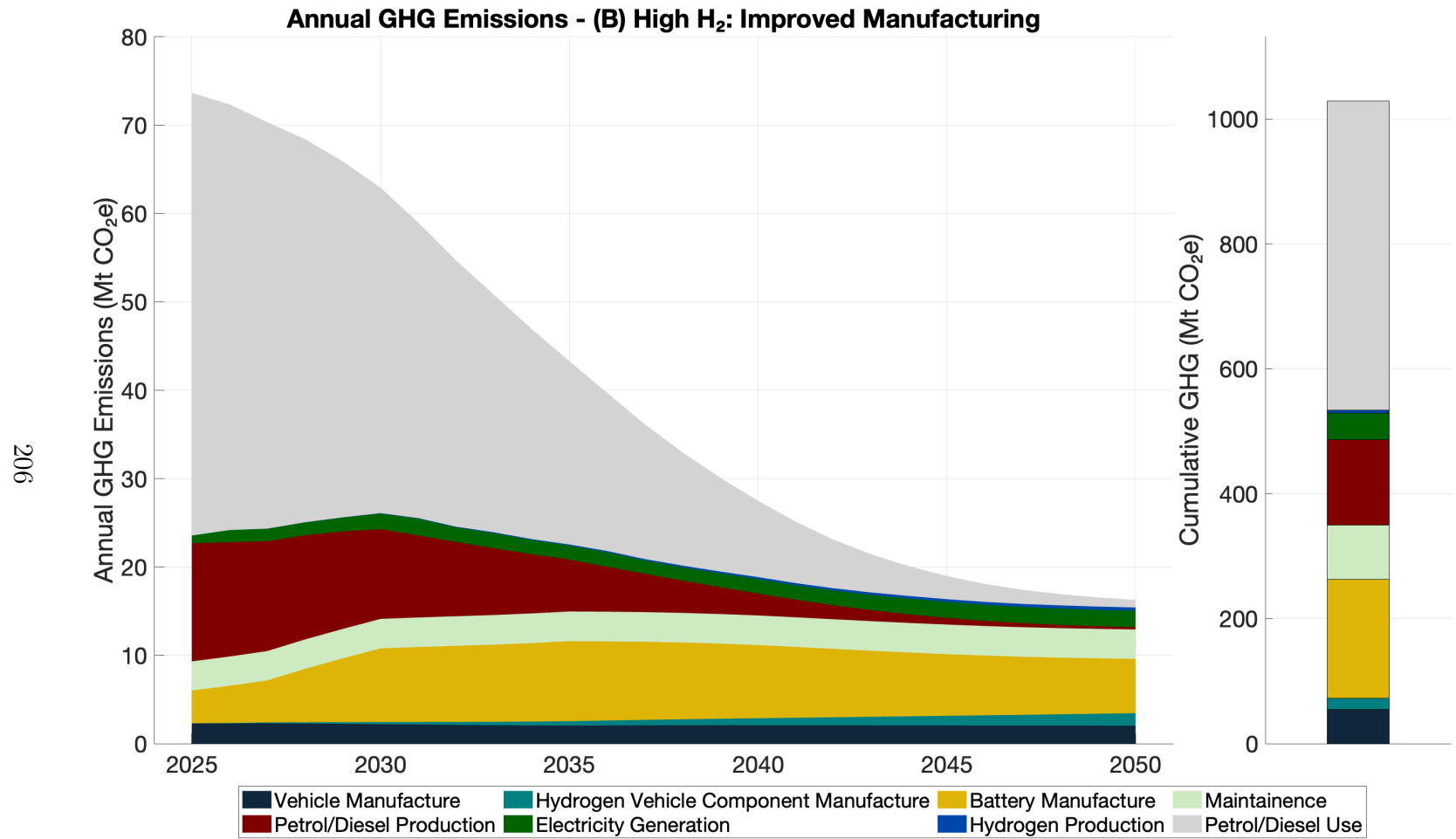


Figure 51: Annualised life cycle GHG emissions associated with the UK LDV fleet from 2025 to 2050 in High H₂ (b) scenario, sensitivity analysis for improved manufacturing

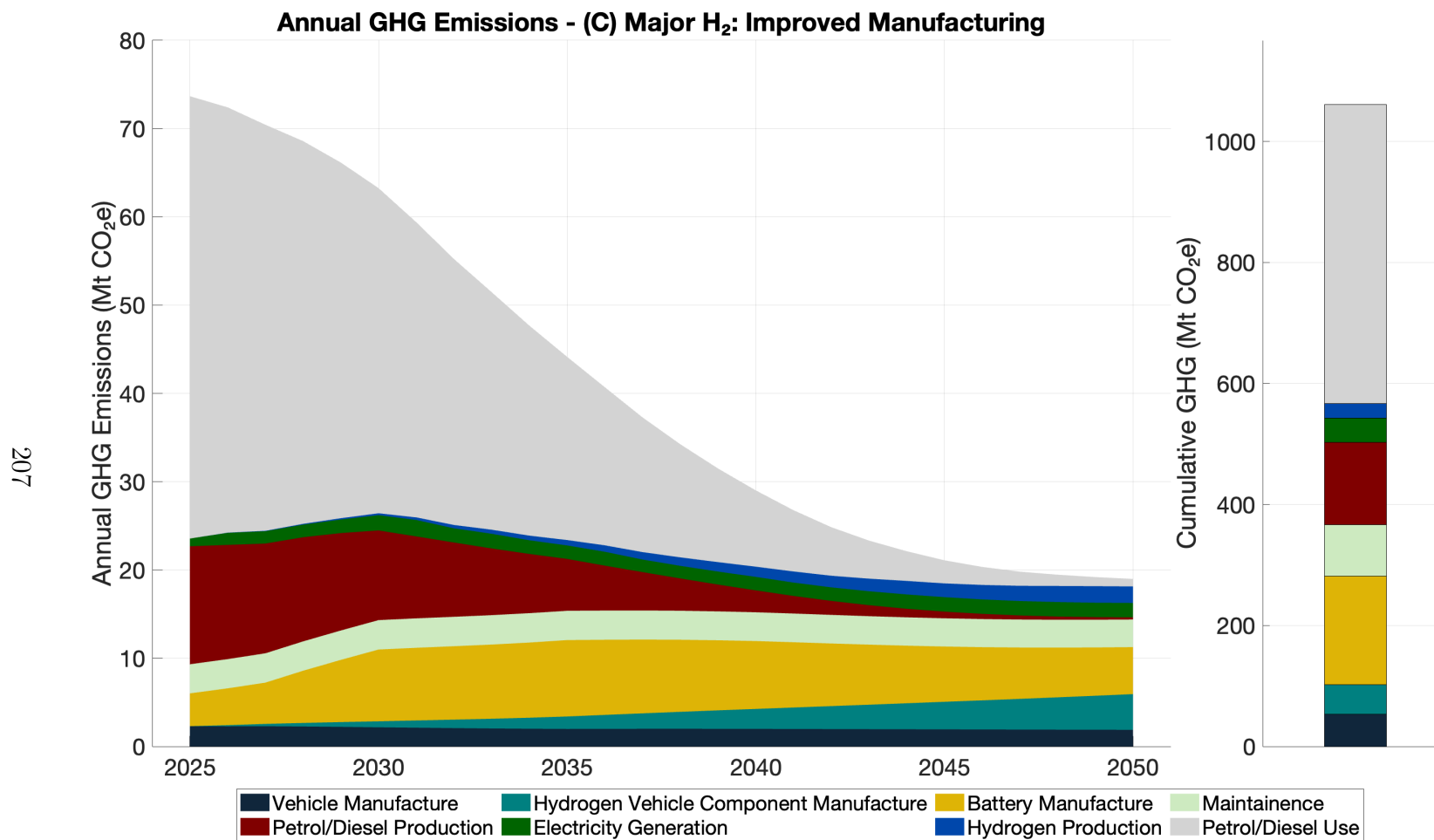


Figure 52: Annualised life cycle GHG emissions associated with the UK LDV fleet from 2025 to 2050 in Major H₂(c) scenario, sensitivity analysis for improved manufacturing

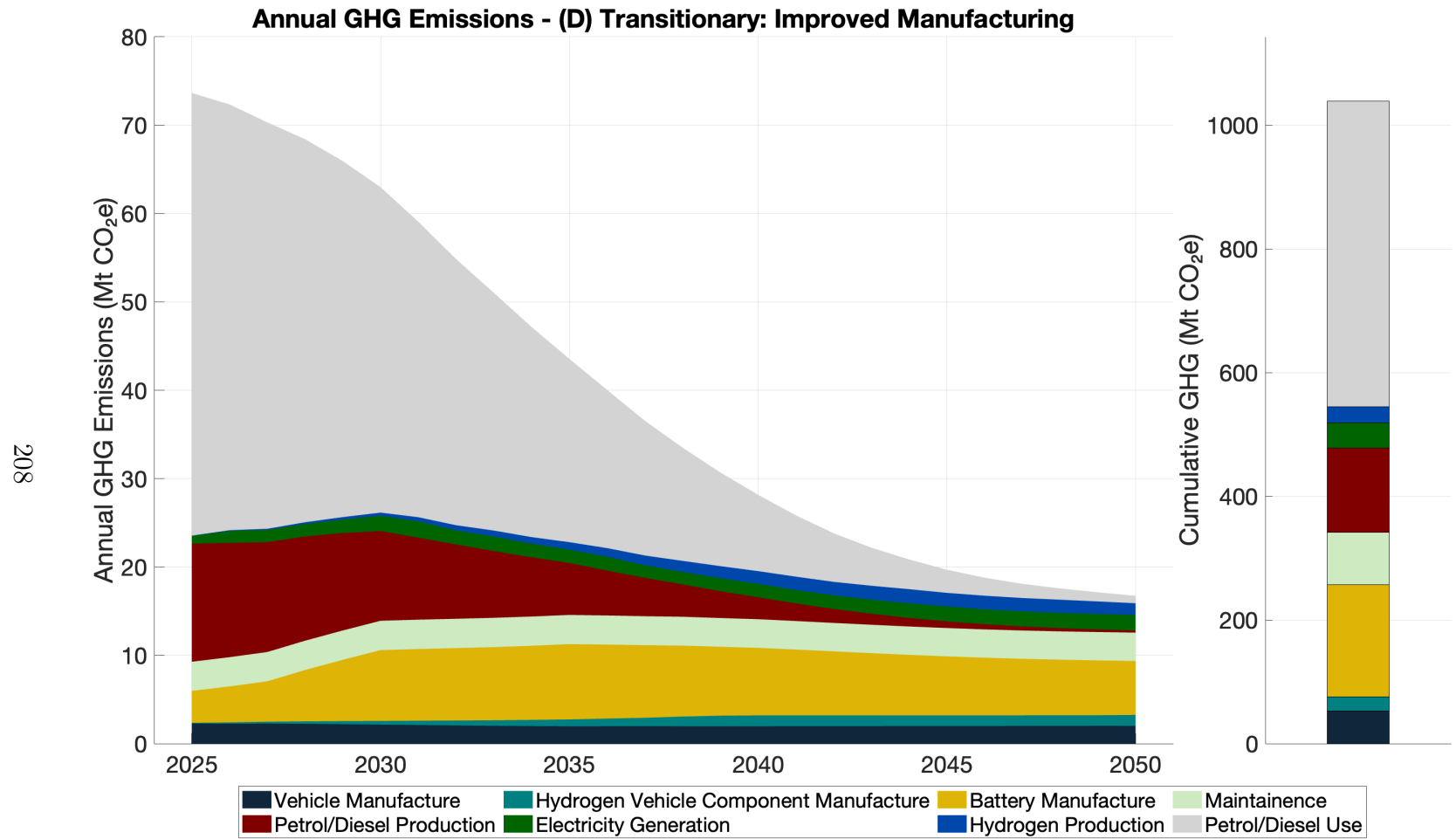


Figure 53: Annualised life cycle GHG emissions associated with the UK LDV fleet from 2025 to 2050 in Transitionary (d) scenario, sensitivity analysis for improved manufacturing

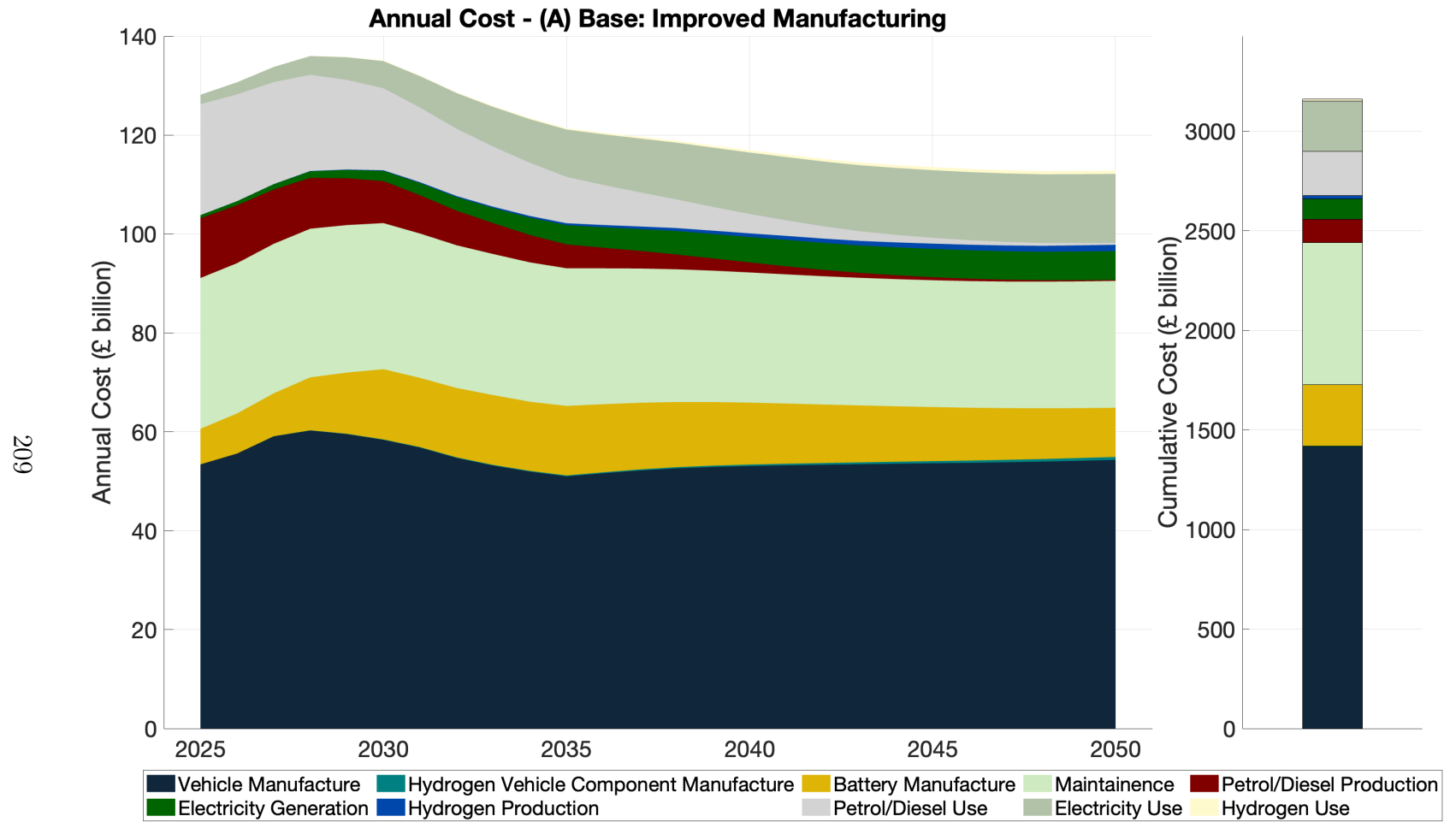


Figure 54: Annualised life cycle costs (£) associated with the UK LDV fleet from 2025 to 2050 in Base (a) scenario, sensitivity analysis for improved manufacturing

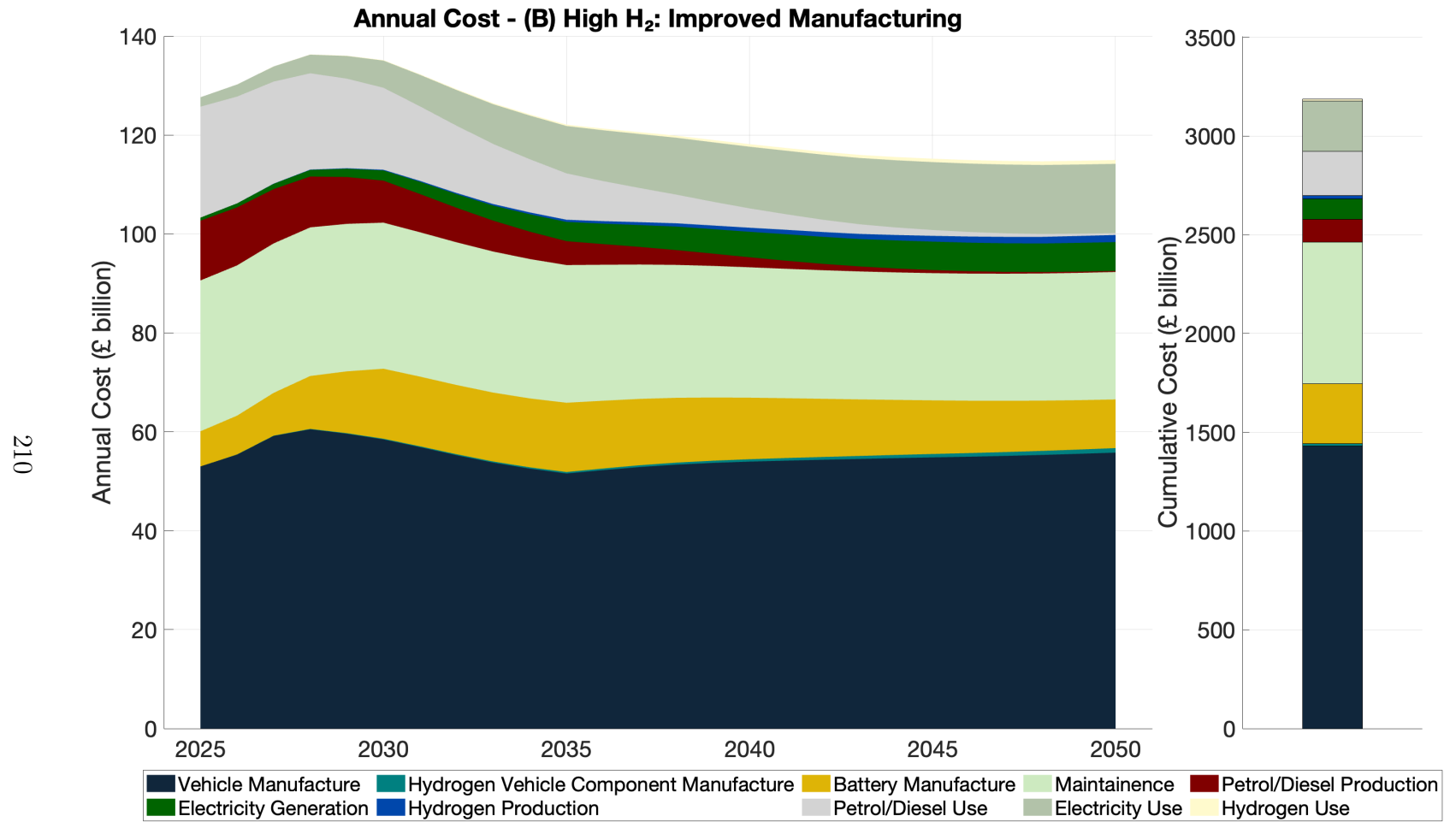


Figure 55: Annualised life cycle costs (£) associated with the UK LDV fleet from 2025 to 2050 in High H_2 (b) scenario, sensitivity analysis for improved manufacturing

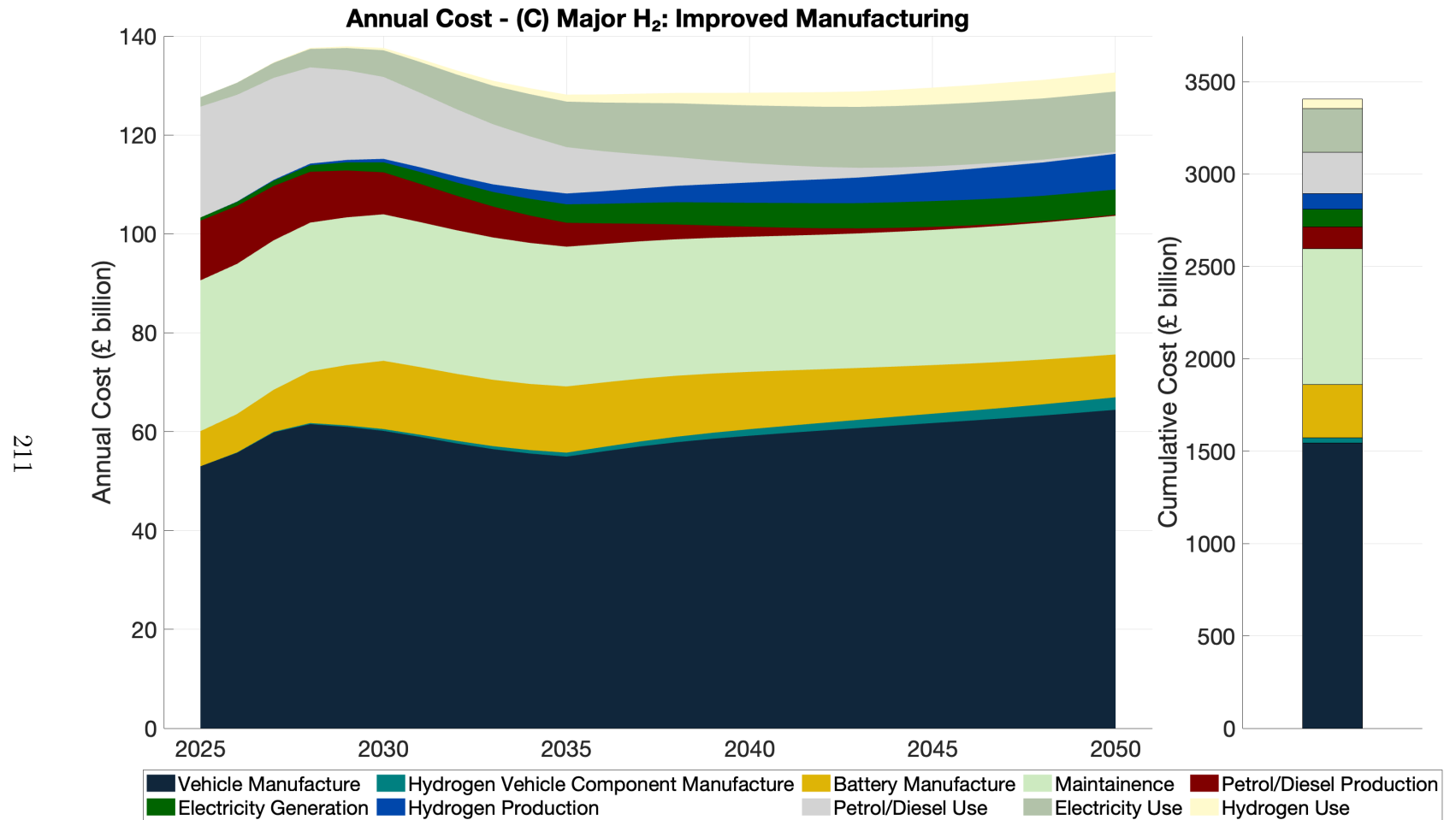


Figure 56: Annualised life cycle costs (£) associated with the UK LDV fleet from 2025 to 2050 in Major $H_2(c)$ scenario, sensitivity analysis for improved manufacturing

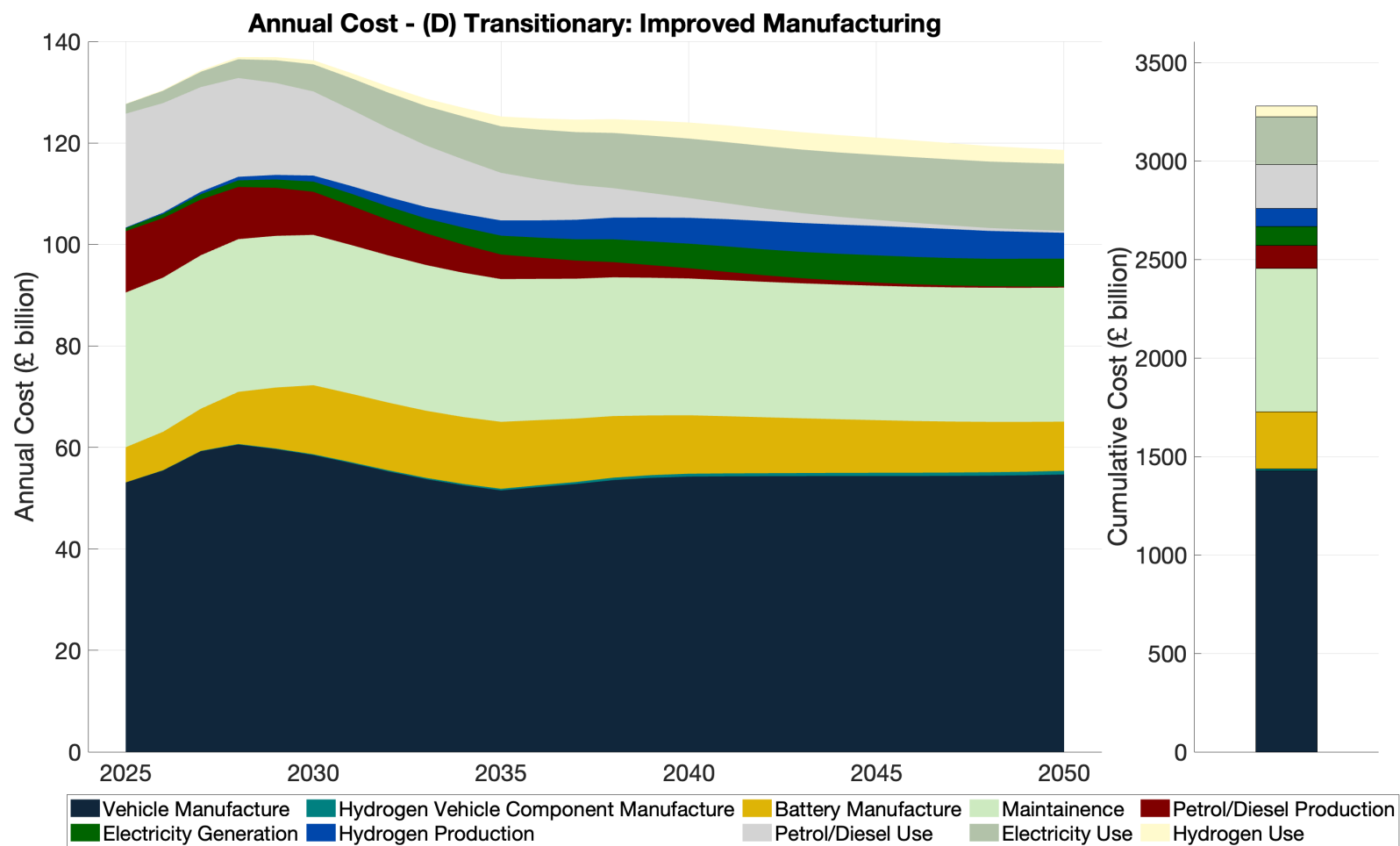


Figure 57: Annualised life cycle costs (£) associated with the UK LDV fleet from 2025 to 2050 in Transitional (d) scenario, sensitivity analysis for improved manufacturing

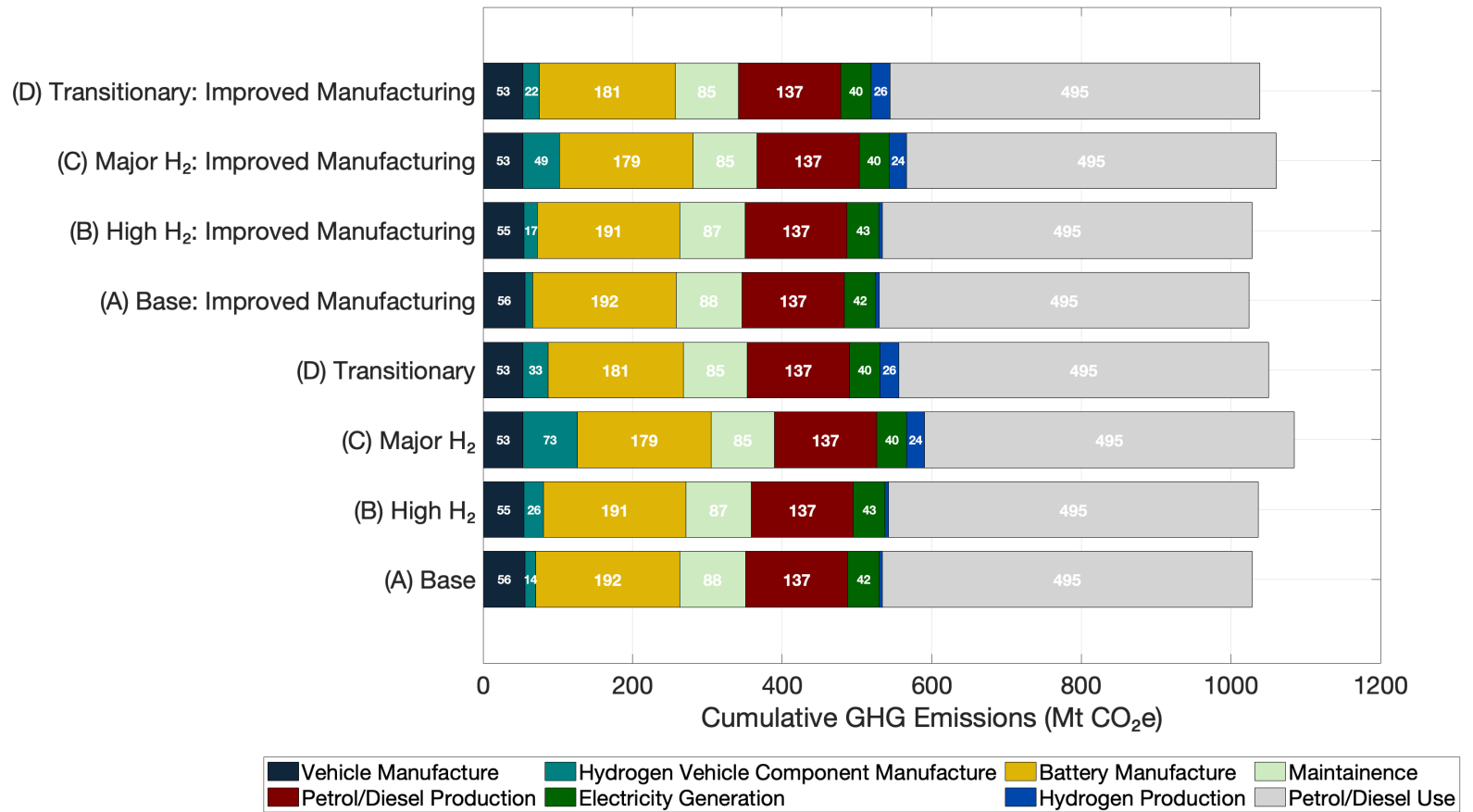


Figure 58: Cumulative GHG emissions across the study period including sensitivity analysis for improved manufacturing

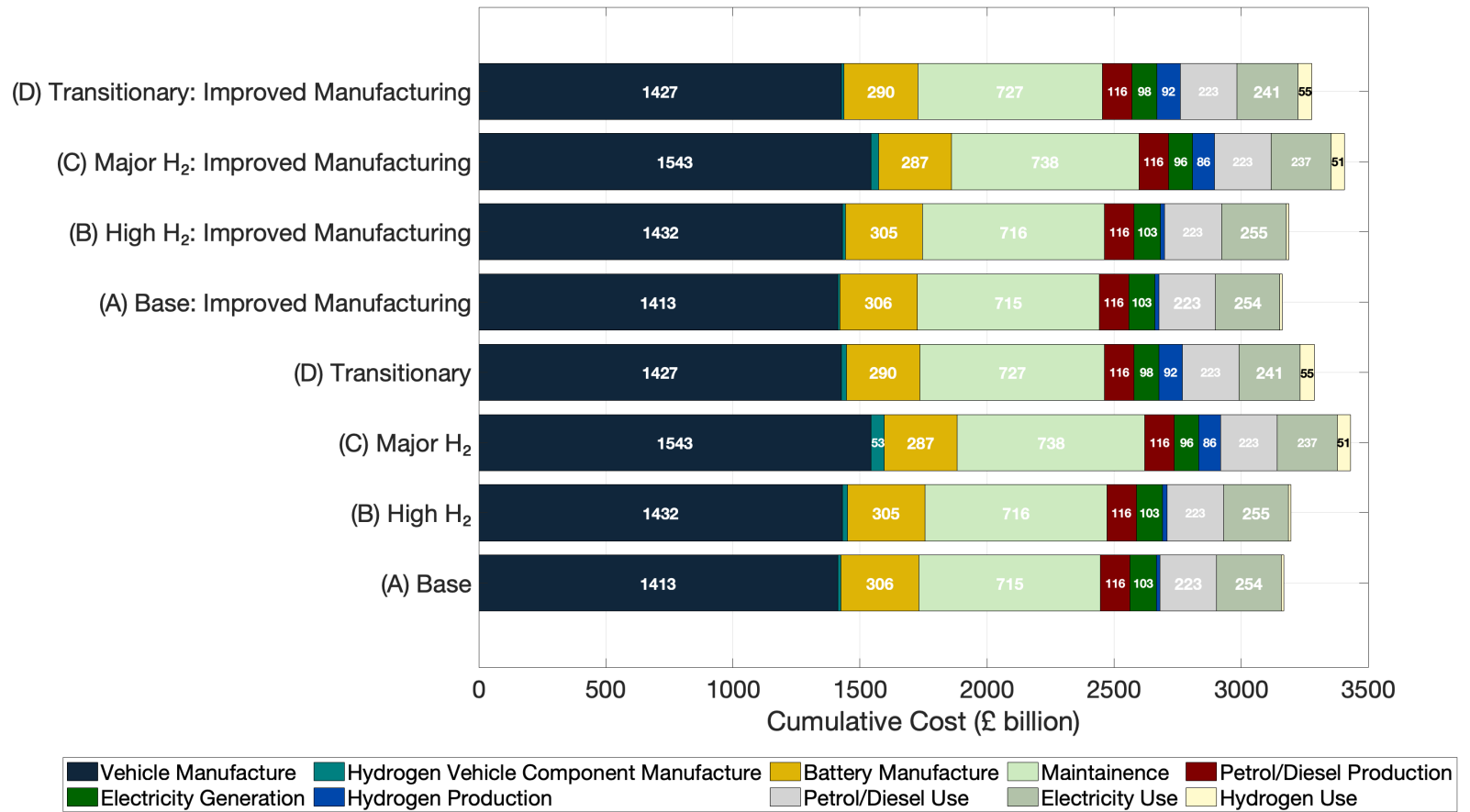


Figure 59: Cumulative costs across the study period including sensitivity analysis for improved manufacturing

6.4 Charging price variability in cost modelling

A key finding from the cost analysis is that decarbonising the UK fleet under the Major H_2 scenario (a 30% market share of hydrogen vehicles by 2050) would cost approximately £259 billion more than the BEV-dominant Base scenario. While this represents a notable difference, in cumulative terms this is just an 8% increase in total costs. This indicates that changes in electricity pricing or charging habits could potentially eliminate the cost gap. To explore this, a sensitivity analysis was conducted to determine the break-even electricity price at which the Base scenario would match the cost of the Major H_2 pathway. Holding all other parameters constant, the analysis reveals that if electricity prices rise to £0.38/kWh, the cumulative cost of the Base scenario converges with that of the Major H_2 scenario. This is an important consideration as this break-even threshold is not just theoretical, but as discussed in Section 5.4, public charging rates in the UK already exceed this price. For instance, Gridserve’s (GRIDSERVE, 2025) charging network currently lists prices ranging from £0.49 to £0.79/kWh depending on location and charger type. This suggests that in situations where home charging is limited or unavailable, BEV fleet costs could approach or even surpass those of hydrogen pathways. Further evidence is provided by the cumulative cost modelling across different charging behaviour scenarios, shown in Table 60. In the Base scenario with *no home charger*, electricity costs rise to £759 billion, and total cumulative costs reach £3.67 trillion - substantially higher than the Major H_2 scenario (£3.43 trillion). Even in the *often at home* charging case, total costs (£3.47 trillion) exceed those of the hydrogen pathway. Only in the *mostly at home* scenario does the Base case become more cost-effective (£3.32 trillion), underscoring the critical role of charging access in shaping fleet cost outcomes. This analysis reframes the debate: rather than asking whether hydrogen is “too expensive”, we instead consider under what conditions hydrogen pathways become competitive with fleet-wide electrification? Electricity pricing, charging infrastructure, and consumer utilisation all play a decisive role, and many of these variables already fall within the range of current market conditions. This sensitivity analysis, centred on price parity thresholds, highlights how small shifts in key assumptions can alter the relative performance of BEV and hydrogen pathways. It underscores the significance of incorporating behavioural and infrastructural factors into lifecycle cost modelling, particularly when assessing long-term, fleet decarbonisation strategies.

Table 60: Cumulative electricity and total lifecycle costs under different charging scenarios

Scenario	Electricity Cost (£ Billion)	Total Lifecycle Cost (£ Trillion)
Base Case	254	3.17
Major Hydrogen (30% H ₂ by 2050)	237	3.43
Base – No Home Charging	759	3.67
Base – Often Charging at Home	555	3.47
Base – Mostly Charging at Home	406	3.32

6.5 Annual and cumulative energy demand

Figures 60 to 63 illustrate the evolution of energy demand across four UK LDV transition scenarios from 2025-2050, highlighting both shared trajectories and divergences. All pathways show a substantial decline in annual energy demand, falling from approximately 800 PJ in 2025 to 300-350 PJ by 2050 - a reduction of 56-63%. This decrease in energy demand reflects the shift from fossil fuel vehicles to ZEVs. Figure 63 (Transitional scenario) highlights an important relationship between hydrogen adoption pathways and total energy demand. Despite featuring only moderate hydrogen penetration (7% by 2050), Transitional results in the highest cumulative energy demand (13,430 PJ), surpassing the Major H₂ scenario (Figure 62), which deploys hydrogen more extensively. The Transitional scenario has a greater penetration of hydrogen internal combustion engine vehicles, which operate at lower efficiencies (25-30%) compared to battery electric vehicles (85-90%) and fuel cells (45-55%). The findings emphasise that not only the quantity of hydrogen adoption, but the type of hydrogen vehicle technology, plays a key role in shaping long-term energy demand. In contrast, the Base scenario (Figure 60), characterised by extensive BEV uptake, yields the lowest cumulative energy demand (12,701 PJ), reflecting the well-to-wheel efficiency of electric drivetrains. The 729 PJ difference between Base and Transitional translates to 29 PJ per annum. To contextualise this, it represents approximately 2% of the UK's annual road transport energy demand ((Department for Energy Security & Net Zero (DESNZ), 2024), see Figure 65). While modest in percentage terms, this margin arises from drivetrain efficiency differences within the LDV fleet, reinforcing the importance of technology selection in shaping national energy outcomes.

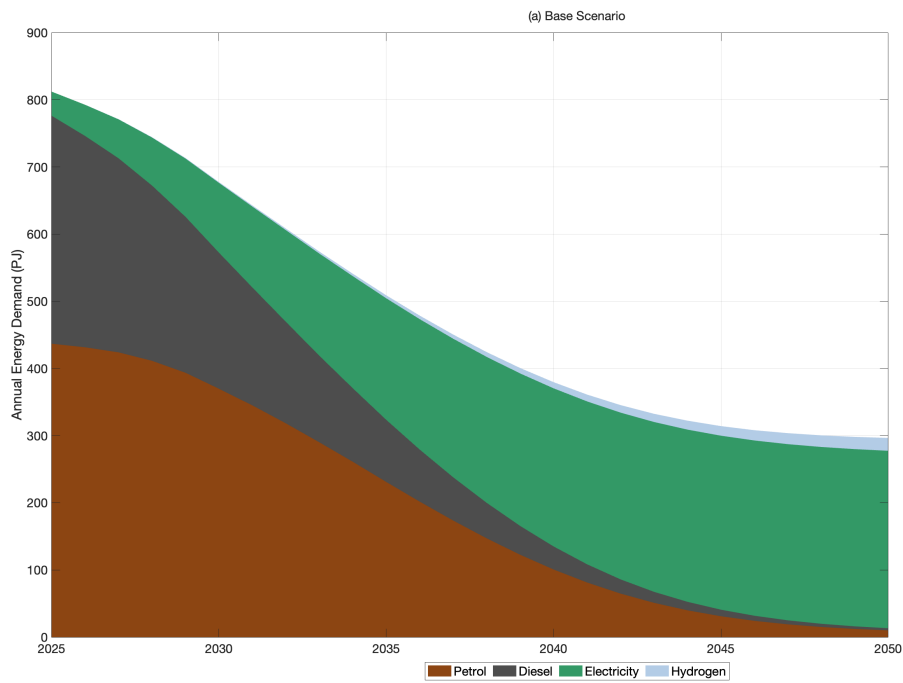


Figure 60: Annual energy demand for the UK LDV fleet to 2050, in the Base scenario (a)

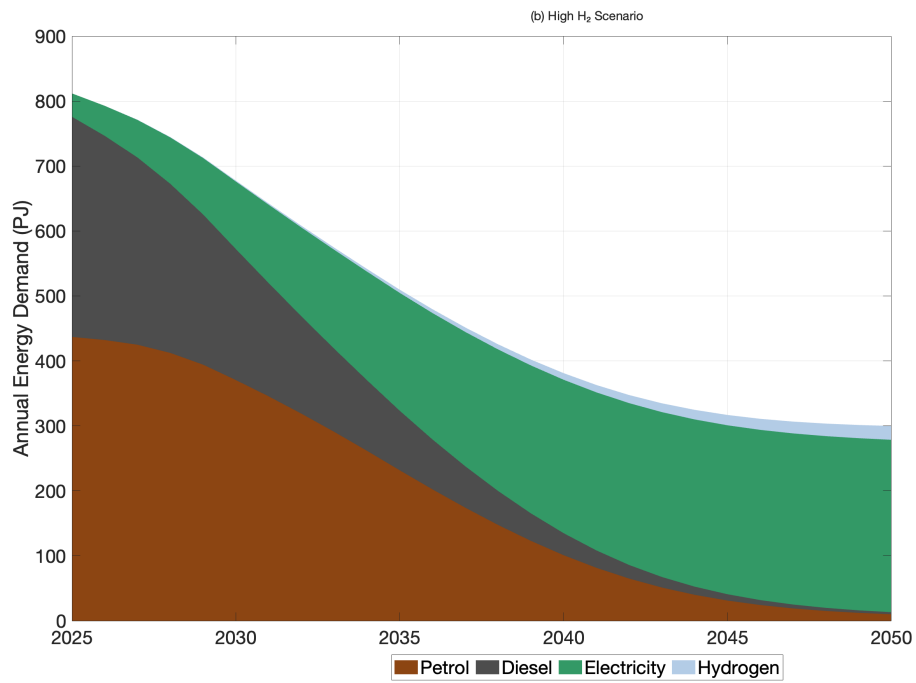


Figure 61: Annual energy demand for the UK LDV fleet to 2050, in the High H_2 scenario (b)

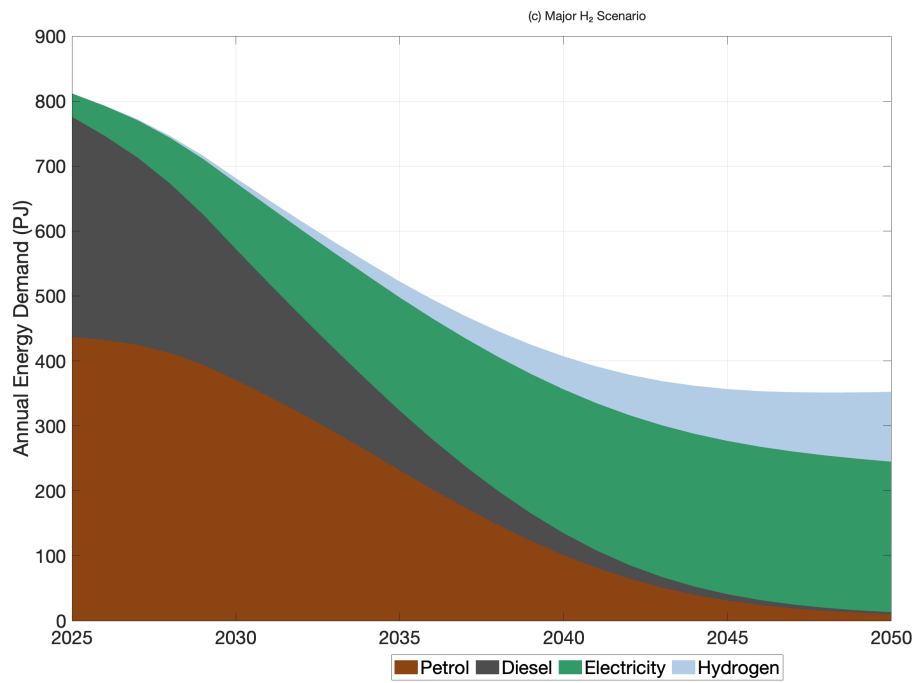


Figure 62: Annual energy demand for the UK LDV fleet to 2050, in the Major H_2 scenario (c)

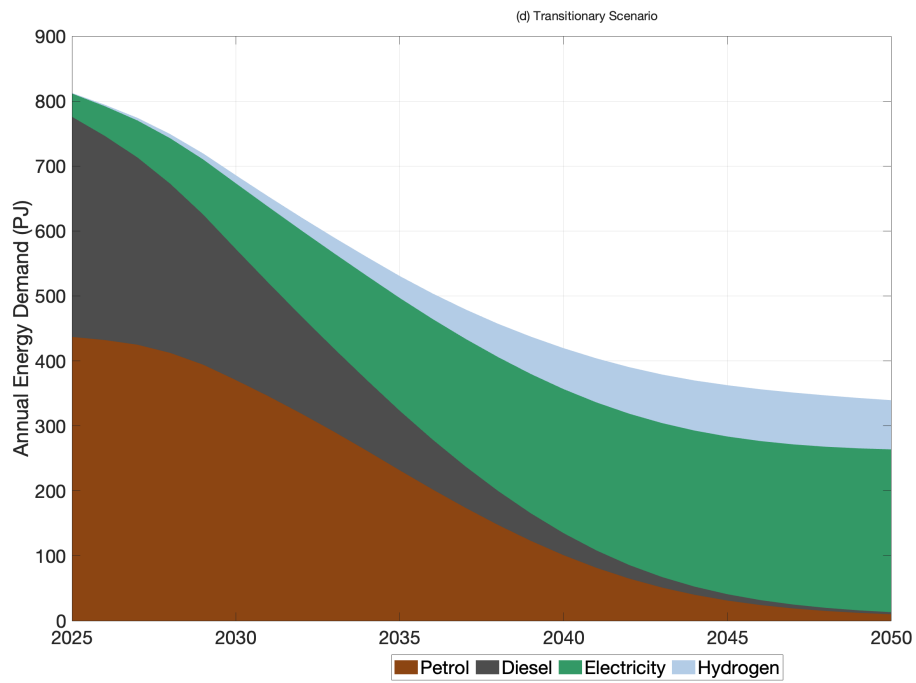


Figure 63: Annual energy demand for the UK LDV fleet to 2050, in the Transitional scenario (d)

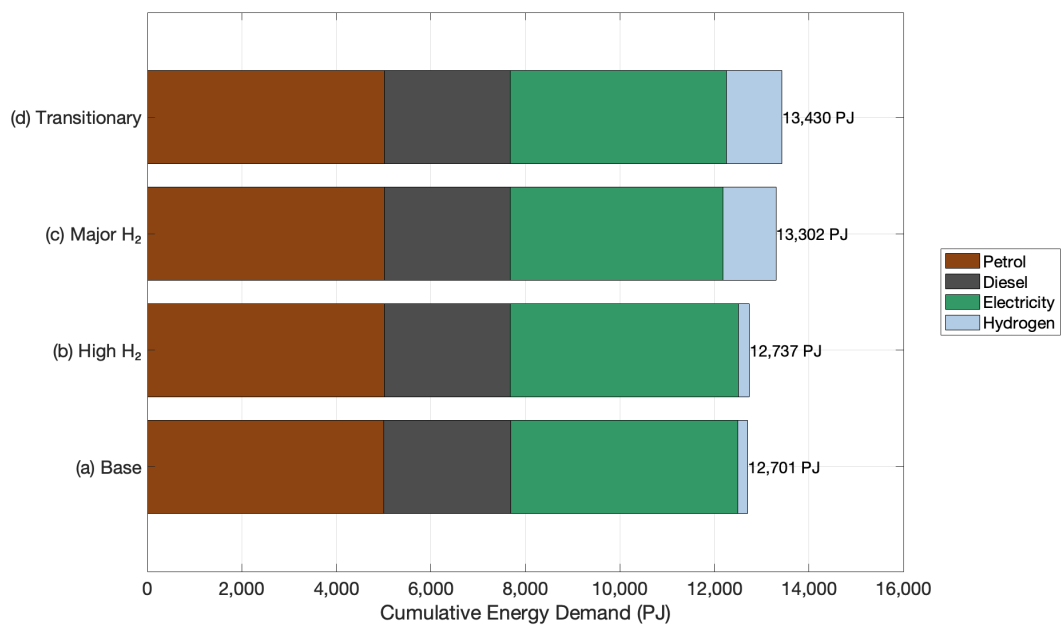


Figure 64: Cumulative energy demand for the UK LDV fleet, projected to 2050

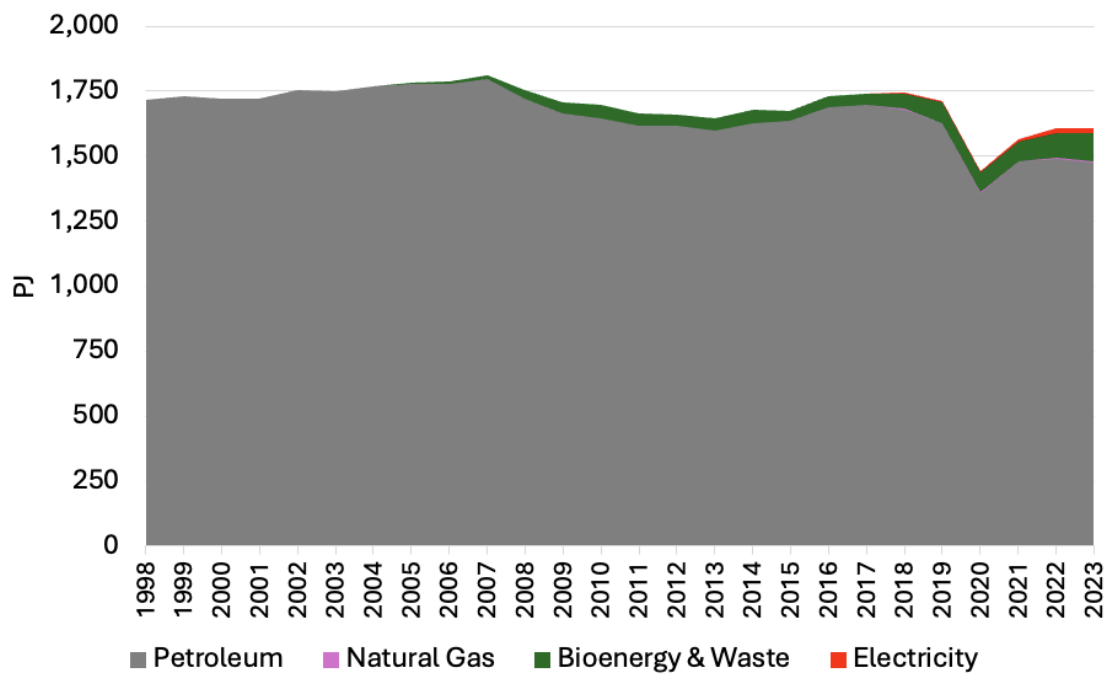


Figure 65: Final Energy Consumption for road transport in the UK (1998-2023) converted from ktoe to PJ. Sourced from (Department for Energy Security & Net Zero (DESNZ), 2024)

6.6 Results

Overall, the four scenarios analysed in this study demonstrate a broad spectrum of potential outcomes for the UK LDV fleet, reflecting varying levels of hydrogen deployment and electrification. Key findings include: Base Scenario - This scenario, with a limited hydrogen vehicle share of just 5% by 2050, achieves the lowest cumulative GHG emissions at approximately 1.03 GtCO₂e. This scenario is heavily reliant on BEVs and exhibits the lowest overall life cycle costs, estimated at £3.17 trillion over the 25 years (£1.97 trillion for vehicle production costs only). The cost efficiency is primarily driven by the declining cost of batteries and reduced fuel expenditures as fossil fuel use decreases significantly. Major H_2 Scenario: With a 30% hydrogen share by 2050, this scenario presents a more hydrogen-intensive pathway, resulting in cumulative emissions of approximately 1.09 GtCO₂e – a 6% increase over the Base case. Cumulative life cycle costs are also significantly higher, reaching £3.43 trillion (£2.18 trillion production costs only), representing a 8% increase in total costs and 11% increase in production costs compared to the Base scenario. This cost difference is primarily driven by the higher material and manufacturing costs associated with hydrogen vehicle technologies, including fuel cell and hydrogen storage tank production costs of £53 billion (compared to £12 billion in the Base scenario), hydrogen production costs of £86 billion (versus £15 billion in the Base scenario), and increased vehicle production costs due to the deployment of larger hydrogen vehicles such as SUVs. Intermediate Scenarios: The High H_2 and Transitional scenarios provide insights into intermediate pathways with moderate hydrogen deployment. The High H_2 scenario achieves cumulative costs of £3.20 trillion (£1.99 trillion production costs) with emissions of 1.04 GtCO₂e, positioned closely to the Base case. The Transitional scenario, characterised by more substantial hydrogen internal combustion engine adoption results in total costs of £3.29 trillion (£2.04 trillion production costs) with emissions of 1.06 GtCO₂e, reflecting the challenges of maintaining dual technology pathways while building hydrogen production capacity. Manufacturing Sensitivity Analysis: The sensitivity analysis examining improved hydrogen manufacturing demonstrates significant component-level cost reductions of 42-45% and emission reductions of 29-33% for fuel cells and storage tanks. However, these improvements translate to modest system-level impacts, with total cost reductions of 0.5% and emission reductions of 2.1% in the Major H_2 scenario, reflecting the relatively small share of these components in the overall life cycle footprint. These findings highlight the economic challenges of deploying hydrogen while stressing the importance of upstream supply chain decarbonisation. The hydrogen-led scenarios, although providing routes for deep decarbonisation, come at a cost premium com-

pared to the BEV-dominated Base case. This premium mainly stems from higher costs of hydrogen production infrastructure, fuel cell systems, and the material demands of hydrogen storage technologies, as well as continued dependence on yellow (grid-connected electrolysis) hydrogen during early deployment stages. The results underline that achieving the climate benefits of hydrogen mobility requires parallel strategies to decarbonise hydrogen supply chains, improve material efficiency, and increase the circularity of critical components.

6.7 Discussion

The findings from this study reveal tensions between the theoretical potential of hydrogen mobility and its practical realisation within the UK’s decarbonisation strategy. By expanding the UK-FLAME framework to include hydrogen-specific components and supply chains, this research offers a detailed, fleet-scale assessment of hydrogen vehicle deployment scenarios in the UK context. The results reveal important insights that emphasise the complexity of hydrogen’s role in the transport decarbonisation system. One of the most notable findings is the ongoing economic penalty linked to hydrogen deployment across all scenarios examined. The Major H_2 scenario shows a 8% rise in total life cycle costs (£3.43 trillion versus £3.17 trillion) and an 11% increase in production costs (£2.18 trillion versus £1.97 trillion) compared to the Base scenario. The £53 billion cumulative cost for fuel cell and tank production in the Major H_2 scenario, compared to just £12 billion in the Base scenario, represents a 340% increase for components serving only 30% of the fleet. This disproportionate cost burden reflects the current immaturity of hydrogen vehicle manufacturing processes and the reliance on expensive materials such as platinum-group metals and carbon fibre reinforced polymers. More critically, hydrogen production costs reach £86 billion in the Major H_2 scenario versus £15 billion in the Base case (a 473% increase), highlighting how upstream supply chain costs dominate the hydrogen value proposition. These findings are consistent with recent techno-economic assessments highlighting hydrogen production and distribution as primary cost drivers in the total cost of ownership for fuel cell vehicles (as shown in Section 5 and noted by (Robinius et al., 2018), (Samsun et al., 2022)). Our results, however, go beyond individual vehicle analysis by illustrating how these costs scale at the fleet level, showing that even modest rates of hydrogen adoption can place an economic burden on the transport system. When compared to battery electric vehicle deployment, scenarios with greater hydrogen adoption exhibit higher cumulative GHG emissions despite both technologies eliminating tailpipe emissions. The Major H_2 scenario results in 1.09 GtCO₂e compared to 1.03 GtCO₂e in the Base scenario

- a 6% increase showing the differential environmental performance of zero-emission vehicle technologies. This finding reveals that while both pathways eliminate operational emissions, hydrogen vehicles impose higher upstream environmental burdens through manufacturing and supply chain processes compared to battery electric alternatives.

The analysis identifies two primary mechanisms driving this emissions penalty. First, hydrogen-specific component manufacturing generates 73 MtCO₂e in the Major H_2 scenario versus 14 MtCO₂e in the Base case, reflecting the carbon intensity of platinum extraction, CFRP production, and fuel cell manufacturing processes. Second, hydrogen production emissions increase to 24 MtCO₂e compared to 4 MtCO₂e in the Base scenario, reflecting the substantially higher volumes of hydrogen fuel required to serve 30% versus 5% fleet penetration under the same (National Energy System Operator, 2024) Hydrogen Evolution production mix.

‘*Burden-shifting*’ from tailpipe emissions being the primary driver of emissions to manufacturing has implications for policy frameworks focused on tailpipe emissions. As Figure 40 shows, the UK’s ZEV mandate effectively addresses direct operational emissions, reducing these emissions to a negligible amount (0.85 MtCO₂e) by 2050 but fails to account for the upstream environmental impacts of different zero-emission technologies. Our results suggest that policies promoting hydrogen vehicles without parallel requirements for low-carbon hydrogen production may inadvertently increase total transport system emissions. This finding resonates with recent life cycle studies of hydrogen systems that emphasise the critical importance of production pathway selection (e.g., (Bieker, 2021)). The dominance of upstream emissions also reveals the vulnerability of hydrogen vehicle environmental performance to supply chain decarbonisation trajectories. While our analysis assumes the (National Energy System Operator, 2024) Hydrogen Evolution scenario with gradual transition toward low-carbon electrolysis, delays in renewable energy deployment could worsen the emissions profile of hydrogen mobility as grid-connected electrolysis’ environmental profile is dominated by the ‘*greenness*’ of the electricity input, Chapter 4 and (Duke et al., 2025), (Bareiß et al., 2019). The energy demand analysis shows an efficiency penalty linked to H_2 -ICE adoption. Specifically, the Transitional scenario, which deploys only 7% hydrogen vehicles by 2050, results in a higher cumulative energy demand (13,430 PJ) than the Major H_2 scenario (13,302 PJ), despite the latter having a much larger hydrogen share (30% deployment). This unexpected result is caused by the technological composition of the hydrogen fleet - most notably, the early deployment of H_2 -ICE vehicles in the Transitional pathway. Because both scenarios follow the same hydrogen production development, the efficiency paradox is due not to upstream sourcing but to drivetrain choice. The early use of H_2 -ICE

vehicles in the Transitional scenario creates a period of higher energy intensity, emphasising the importance of powertrain efficiency in influencing overall system energy outcomes. H_2 -ICE vehicles operate at well-to-wheel efficiencies of 25-30% compared to 30-35% for conventional petrol/diesel ICEVs, 45-55% for FCEVs and 85-90% for BEVs. As a result, the Transitional scenario's reliance on H_2 -ICE as a bridge technology introduces a period of elevated energy intensity, which persists even as more efficient powertrains are gradually introduced. This finding raises the consideration of technology pathway selection in hydrogen deployment strategies. While H_2 -ICE vehicles offer the advantage of zero tailpipe emissions, they impose greater upstream energy penalties. Thus, decarbonisation strategies should not only consider emissions profiles but also system-level energy efficiency. Although all scenarios achieve transport decarbonisation by 2050, the chosen pathway affects cumulative emissions and energy demand. Electrification provides the greatest reduction in both metrics, but other system-level factors should be taken into account that are outside the scope of this analysis. For instance, scenarios dominated by BEVs risk worsening electricity demand peaks (Li and Jenn, 2024), whereas hydrogen can offer grid balancing opportunities (Ghirardi et al., 2023). Furthermore, hydrogen refuelling infrastructure requires fewer centralised stations compared to the numerous distributed EV charging points needed for full electrification, which could reduce spatial and logistical burdens (Crönert and Minner, 2021).

The sensitivity analysis evaluating improved manufacturing processes uncovers a fundamental limitation in component-focused innovation strategies. Despite achieving significant reductions in fuel cell and tank production costs (42-45%) and emissions (29-33%), these gains result in marginal system-level benefits, with total cost reductions of just 0.5% and emission reductions of 2.1% in the Major H_2 scenario. Its limited system-level impact shows that manufacturing emissions and costs make up a relatively small part of the wider hydrogen vehicle life cycle. The analysis indicates that investing in manufacturing efficiency, while beneficial for component competitiveness, cannot solve the fundamental cost and emissions issues facing hydrogen mobility. In specific scenarios, particularly those with limited access to home charging, the cost advantage of electrification is negated. As shown in Table 60, the Base scenario with no home charging results in higher cumulative costs than the Major H_2 pathway. This highlights a key finding: the economic viability of electrification pathways is highly sensitive to assumptions of charging infrastructure and user behaviour. The sensitivity analysis shows that increases in electricity prices ($>£0.38/\text{kWh}$) or consumer charging patterns like those choosing to charge at rapid public chargers, or without access to home charging, can push BEV-dominated cost pathways into parity with hydrogen alternatives. Thus, reinforcing the importance of

viewing lifecycle cost modelling at both the vehicle level and the broader system context. While manufacturing improvements offer limited cumulative reductions in the hydrogen pathway, strategic interventions in electricity pricing and charging logistics may offer more substantial leverage in shaping the cost-effectiveness of decarbonisation strategies.

These findings offer useful insights for UK decarbonisation policy, particularly in relation to public investment priorities. The ongoing costs and emissions penalties associated with hydrogen deployment indicate that widespread subsidies for hydrogen vehicles may not be the most efficient way to use limited decarbonisation resources. Instead, the analysis supports targeted deployment in specific applications where hydrogen's operational advantages (e.g., quick refuelling and extended range) present clear value that outweighs the economic and environmental costs.

By taking into account temporal changes in supply chain emissions, manufacturing processes, and technology performance, the UK- H_2 -FLAME framework offers a more realistic evaluation of hydrogen mobility impacts than traditional static LCA approaches. The inclusion of multiple hydrogen production pathways and their evolution over time addresses a key limitation in previous studies that assumed fixed fuel supply conditions. However, several limitations should be acknowledged. Firstly, end-of-life costs were excluded from the fleet model. While increasingly relevant, especially for battery recycling in BEVs, their inclusion was beyond the scope of this study due to data limitations in early-stage hydrogen vehicle technologies. Unlike BEVs, which follow established regulations like the EU Battery Directive and Extended Producer Responsibility, hydrogen systems lack comparable end-of-life pathways or cost benchmarks. Also, the analysis assumes constant recycling rates for critical materials, particularly the 5% rate for CFRP components, which may underestimate the potential for circular economy approaches to reduce environmental impacts. Furthermore, relying on (National Energy System Operator, 2024) Hydrogen Evolution scenario for energy supply mix projections introduces uncertainty regarding the actual trajectory of hydrogen production decarbonisation. Additionally, focusing on light-duty vehicles may not capture the full system benefits that could emerge from integrated hydrogen deployment across all transport modes. The cost analysis, while comprehensive in scope, relies on current technology cost trajectories that may not fully account for breakthrough innovations, economies of scale or unexpected technological developments. Finally, the treatment of maintenance costs as static across scenarios may also underestimate the potential for learning effects in hydrogen vehicle servicing and repair. The findings from this study highlight several important areas for future research. First, when data becomes available the model should be updated to include end-of-life costs to better reflect the regula-

tory landscape that requires battery recycling, particularly as similar calls are being made for fuel cells and hydrogen storage tanks (International Platinum Group Metals Association, 2025). Also, integrating hydrogen vehicle deployment with broader energy system modelling could uncover system-level interactions not captured in this transport-focused analysis. Second, expanding this framework to include heavy-duty vehicles, shipping, and aviation could offer a more comprehensive assessment of hydrogen’s role in overall UK transport decarbonisation.

6.8 Summary

Across the four vehicle decarbonisation pathways modelled, a BEV-dominated fleet delivers the lowest cumulative emissions and system costs, while greater hydrogen deployment consistently increases both due to higher upstream manufacturing burdens, hydrogen production emissions, and lower drivetrain efficiency - particularly in scenarios relying on H_2 -ICE vehicles. Although improvements in fuel cell and storage tank manufacturing reduce component-level costs and emissions, their influence on total system outcomes is minimal. The analysis also highlights the sensitivity of electrification pathways to charging access and electricity pricing, with limited home-charging availability capable of eroding BEV cost advantages. Overall, the findings show that while all scenarios achieve near-zero operational emissions by 2050, the choice of technology pathway shapes cumulative impacts, with electrification offering the most cost-effective route and hydrogen requiring targeted, supply-chain-aligned deployment to avoid unintended increases in system-level emissions.

Chapter 7

Conclusions and Future Work

7.1 Summary

This chapter concludes the study by highlighting the key findings in relation to the research questions, aims, and objectives of the thesis. It also outlines the value and contributions of this study to the wider context of transport decarbonisation before reviewing limitations and providing recommendations for future work. Hydrogen-powered mobility offers a potential pathway for decarbonising the transport sector, especially in areas where BEVs face operational limitations. The introduction of hydrogen vehicles is currently led by FCEVs and is projected to be driven by increasing volumes of PEM electrolysis-produced hydrogen. However, its sustainability depends not only on tailpipe emissions but also on upstream supply chain impacts, including material requirements, production emissions and end-of-life recovery challenges. Therefore, it is important to quantitatively understand the sustainability concerns of hydrogen vehicles (and their fuel) from a life cycle perspective. This could be achieved by considering the evolution of technology, changes in market share of electrolysis technologies, the transition towards more renewable energy in the electricity mix, and the recycling technologies that provide a secondary source of materials for hydrogen vehicle and electrolyser manufacture. This thesis examines the sustainability of hydrogen-powered mobility across multiple scales, from global supply chains to national fleet deployment, using an integrated dynamic life cycle assessment, life cycle cost, and material flow analysis framework. Through a three-tiered analytical approach covering PEM electrolyser production (Chapter 4), vehicle-level technology comparisons (Chapter 5), and UK fleet-scale deployment scenarios (Chapter 6), this research addresses key questions about the environmental, resource, and economic implications of hydrogen vehicles in transport decarbonisation. It focuses on four

key questions: (1) What are the life cycle GHG emissions and material requirements of PEM electrolyzers, and how do these evolve over time? (2) How do emerging hydrogen vehicle technologies compare to battery electric and conventional powertrains in terms of life cycle GHG emissions and life cycle cost? (3) What are the fleet-level GHG and cost implications of hydrogen vehicle deployment in the UK through to 2050? (4) To what extent can circular economy strategies (such as material recovery and recycling) reduce GHG and material impacts of PEM and hydrogen vehicle production? Chapter 3 presents the integrated LCA and MFA framework used to assess the environmental and resource implications of PEM electrolyser production. The foreground system features component-level inventories for stack and balance-of-plant materials, assumptions on catalyst loading for iridium and platinum, and recycling rates informed by industry reports and academic literature. Background data sources include estimates of iridium supply from the US Geological Survey (USGS), projections of the electricity mix from the International Energy Agency (IEA), and emissions factors from Ecoinvent 3.7. The model is applied across scenarios from 2020 to 2050 in Chapter 4 to evaluate future material demand and emissions trajectories under varying assumptions about technology development, recycling uptake, and electricity decarbonisation. Chapter 5 applies a combined LCA and LCC framework to compare hydrogen vehicle technologies with BEVs and ICEVs. The analysis includes FCEVs, H_2 -ICE, and FCREx systems. Foreground data for vehicle components, fuel cell stacks, hydrogen tanks, and battery packs are drawn from manufacturer specifications, GREET model inventories, and an extensive literature review. Hydrogen production pathways include centralised electrolysis and steam methane reforming, with transport and distribution modelled via pipeline and road (tube-trailer) delivery over 100 or 400 km. Cost assumptions reflect projected learning rates for fuel cell systems, batteries, and vehicle components, as well as fuel and electricity prices, with sensitivity analysis applied to key economic parameters. Developments in policy and financial incentives are also included based on stated policies. Chapter 6 extends the LCA-LCC framework to fleet-scale deployment scenarios in the UK from the present day (2025) through to 2050. The fleet model builds upon previous study by Milovanoff et al. and Davies et al. to simulate four future pathways with varying hydrogen penetration: Base (BEV-dominant, 5% hydrogen fleet by 2050), High H_2 (15% hydrogen fleet by 2050), Transitional (mixed BEV and hydrogen fleets, H_2 -ICE used a transitional technology to FCEV and FCREx uptake), and Major H_2 (30% hydrogen fleet by 2050). Vehicle stock projections are based on UK government transport forecasts, with technology adoption curves adjusted to match hydrogen deployment projections. We include the costs of vehicle purchase, fuel production and distribution, vehicle manufacturing, and operational

costs (e.g., refuelling, maintenance, and taxes). We also calculate fleet emissions using data inputs relating to hydrogen production emissions factors, incorporating upstream supply chain impacts from Chapters 4 and 5.

7.2 Key Findings

(1) What are the life cycle GHG emissions and material requirements of PEM electrolyzers, and how do these evolve over time? Regarding research question 1, the analysis in Chapter 4 reveals a ‘*supply gap*’ between global iridium supply (7 tonnes annually and 500 tonnes global reserves) and projected PEM electrolyser demand. Even with aggressive catalyst loading reductions and 95% closed-loop recycling, our findings show that cumulative iridium demand will reach 31-126 tonnes by 2030 and 526-1,100 tonnes by 2050, far exceeding current supply capacity. Net iridium demand will exceed annual global supply until the 2040s, creating substantial material security risks for green hydrogen production using PEM electrolysis. The development of PEM manufacturing over time shows both improvements and persistent challenges in terms of GHG emissions. When using renewable electricity for an energy input source, manufacturing waste is the dominant source of GHG emissions for PEM systems, contributing 46-56 tCO₂eq/MWe in 2022. While technological improvements and increased recycling rates are projected to reduce emissions intensity over time, the absolute scale of projected electrolyser deployment means total manufacturing emissions will continue growing substantially. Without recycling, emissions would be an order of magnitude higher throughout the study period, emphasising the critical and increasing importance of circular economy approaches as deployment scales. The study demonstrates that manufacturing waste, commonly ignored in existing LCA studies, represents a significant and growing gap in current sustainability assessments of hydrogen production technologies. These findings demonstrate a critical materials gap that threatens the GW-scale deployment of PEM electrolyzers. In this work, we find that even under optimistic assumptions for reductions in PGMs used for catalysis, the projected demand for iridium exceeds current supply with platinum also facing constraints in low-circularity scenarios. This is particularly important in the context that primary production is unlikely to scale up, thus material availability is dependent on end-of-life recovery. The analysis also quantifies the catalyst loading trajectories that would be required to meet IEA deployment forecasts – 0.02-0.07 kg/MW by 2030 and >0.02 kg/MW in 2050 – revealing that without major advances in material efficiency or the development of PGM-free catalysts, PEM technology as a major production technology cannot deliver on global green hydrogen ambitions. Our findings highlight the urgent need to accelerate the development of ultra-low or

PGM-free catalysts.

(2) How do emerging hydrogen vehicle technologies compare to battery electric and conventional powertrains in terms of life cycle GHG emissions and life cycle cost? The analysis in Chapter 5 shows a fundamental shift in emission sources from use-phase (conventional vehicles) to manufacturing and upstream supply chains (electrified vehicles). For hydrogen vehicles, upstream emissions from hydrogen production, transport, and distribution significantly influence overall environmental performance. Infrastructure choices, notably the transition from diesel-powered transport to dedicated pipelines, can markedly reduce these upstream burdens. BEVs currently have the lowest lifecycle emissions, dropping from 23 tCO₂ eq in 2020 to 5 tCO₂ eq in 2050, majorly outperforming conventional ICEVs (67 tCO₂ eq to 50 tCO₂ eq in 2050). Among hydrogen tech, FCReX leads at 28 tCO₂ eq in 2020, improving to 18 in 2050, followed by FCEVs 30-42 tCO₂ eq to 24-33 t CO₂ eq and H₂-ICE (33-53 t CO₂ eq to 27-44 tCO₂ eq). On a life cycle cost basis, FCReX vehicles, if deployed today, could demonstrate immediate cost-competitiveness with conventional ICEVs despite the current high cost of hydrogen fuel due to their hybrid operation utilising 50% electricity. On the other hand, FCEVs currently face higher costs due to the high price associated with manufacturing fuel cells and hydrogen storage systems, without savings attributed to the reduced hydrogen consumption as seen in FCReX vehicles. However, projections indicate substantial cost reductions for hydrogen fuel and component costs by 2050, making FCEVs increasingly competitive. H₂-ICE vehicles consistently show the highest LCC among hydrogen vehicles despite having the lowest vehicle production costs. This is due to H₂-ICE being less efficient than fuel cell alternatives (and thus greater consumption of hydrogen per km). Quantifying this on a cost per mile basis, we find that BEVs demonstrate clear economic advantages at €0.35 per mile, compared to €0.60 for conventional vehicles or €0.52-0.68 for hydrogen technologies. By 2050, BEVs will maintain an economic advantage at €0.26-0.28 per mile, while hydrogen technologies do achieve some cost reductions, for example: FCReX: €0.39, FCEVs: €0.41, H₂-ICE: €0.45 per mile. However, despite hydrogen vehicles having a cost premium compared to EVs, operational challenges, access to charging points, or consumer range anxieties may lead to a willingness to pay a premium. However, our sensitivity analysis shows that this price premium may be offset in specific use cases. For example, for users without access to home charging, hydrogen vehicles become more cost-effective than BEVs due to the high cost of EV public charging. This challenges the assumption that BEVs are universally the lowest-cost-zero-emission option and highlights the operational constraints, infrastructure access and consumer preferences may drive demand for hydrogen vehicles despite their higher purchase costs. (3) What are the

fleet-level GHG and cost implications of hydrogen vehicle deployment in the UK through to 2050? To answer research question 3, Chapter 6 carries out a fleet-level analysis to quantify the effectiveness of hydrogen in reducing emissions and economic cost compared to electrification-dominated scenarios. The Major H_2 scenario (30% hydrogen fleet penetration) demonstrates economic and environmental penalties: total lifecycle costs increases by 6% from £4.65 trillion to £4.94 trillion (£290 billion additional cost), while cumulative GHG emissions also rise by 6% from 1.03 CO₂e to 1.09 GtCO₂ (60 MtCO₂e additional emissions) compared to the Base scenario. These higher costs stem primarily from hydrogen production costs escalating from £24 billion in the Base scenario to £135 billion in Major H_2 (£111 billion increase) and a greater demand for expensive vehicle components requiring PGMs and carbon fibre reinforced polymers. Fuel cell and tank production costs are £53 billion in the Major H_2 scenario compared to £12 billion in the Base case - a 340% increase for components serving just 30% of the fleet, highlighting the cost burden of mass hydrogen LDV deployment. Although both pathways achieve transport decarbonisation by 2050, the chosen technological route influences cumulative impacts. Early deployment of H_2 -ICE vehicles results in efficiency penalties, with the Transitional scenario showing higher energy demand despite reduced hydrogen deployment. This highlights the importance of powertrain choice in deployment strategies. Finally, the analysis demonstrates that manufacturing-focused improvements, while beneficial for component cost-competitiveness, cannot address fundamental system-level cost and emissions challenges. A 42-45% reduction in fuel cell production costs translates to only 0.5% total system cost reduction, indicating that upstream supply chain issues dominate hydrogen vehicle economics. (4) To what extent can circular economy strategies reduce GHG and material impacts of PEM and hydrogen vehicle production? As outlined in Chapter 4, circular economy strategies are essential for improving the sustainability of hydrogen technologies. Manufacturing waste has emerged as a crucial, yet often overlooked, source of both material loss and emissions. This thesis finds that material wastage during PEM manufacturing processes accounts for the majority of GHG emissions linked to the PEM stack. This finding challenges existing LCA studies, which do not typically include manufacturing losses in their assessments. Thus, highlighting a methodological gap in current LCAs and underscores the need for improved manufacturing efficiency alongside end-of-life recovery. For PEM electrolyzers, achieving high recovery rates – up to the theoretical maximum of 95% – is not only critical for reducing life cycle GHG emissions, but also for mitigating critical material demand pressures. With a 95% recovery rate for PGMs, manufacturing emissions are currently quantified to be 56 tCO₂eq/MWe. Without recycling, emissions increase by an order of magnitude to 780 tCO₂eq/MWe,

highlighting the significant environmental impact of linear material use. However, environmental implications extend beyond emissions. Given the limited global supply of platinum and iridium, as outlined in response to research question 1, recovery is also important for closing the ‘material gap’. Without effective recycling infrastructure and efficient use of these materials, the cumulative demand for iridium for PEM deployment will far exceed available global reserves and threaten the scalability of green hydrogen production. In this regard, recycling is not just a level for emissions reductions, but also a requirement for material sustainability and long-term technology viability. By 2050, due to reduced catalyst loading and, more importantly, a decrease in PGM waste during manufacturing, lifecycle GHG emissions decrease to 7-13 tCO₂ (recycling) or 66- 188 tco₂ (without recycling). Nonetheless, even with a vigorous 95% closed-loop recycling system, the analysis shows that demand for iridium will still exceed its annual global supply until 2040. This demonstrates that recycling alone cannot address the near-term material constraints. At the vehicle level, increasing the amount of recycled materials can cut down the GHG emissions related to manufacturing. In particular, reducing the amount of energy-intensive PGMs and CFRP materials used in hydrogen vehicle components. However, the fleet level analysis presented in Chapter 6 shows that despite achieving reductions in fuel cell and tank production emissions (29-33%) and costs (42-45%) through improved manufacturing and material reductions, component gains translate to a slight cumulative emission reduction of 2.1% and cost reduction of 0.5% in the Major H_2 scenario. This disconnect between component-level and system-level impacts highlights that manufacturing emissions and costs represent a relatively small fraction of the total hydrogen vehicle lifecycle. Instead, the dominant factor is the upstream hydrogen production emissions and costs. Therefore, circular economy strategies focused on vehicle components alone will not address fundamental system-level reductions. Effective sustainability for hydrogen technologies demands a combination of efficient recycling processes with the development of alternative technologies, such as PGM-free electrolyzers, and improvements in material efficiency.

7.3 Dynamic LCA Conclusions

A key methodological contribution of this thesis is the use of a dynamic, time-evolving life cycle modelling framework, in contrast to the predominantly static approaches used in the hydrogen technology (production and vehicle) LCA literature. Static LCA studies typically provide a snapshot assessment under fixed assumptions, often representing a single year or steady-state system. In these approaches, background inputs such as the grid electricity mix, hydrogen production pathways and technology

inventories are commonly treated as constant. While static LCA is valuable for benchmarking technologies under clearly defined conditions, it is limited in its ability to evaluate transition pathways that unfold over decades.

The framework developed in this thesis differs in three important ways. First, it explicitly accounts for temporal change in background energy systems and fuel supply chains, including the decarbonisation trajectory of electricity and the evolution of hydrogen production mixes. Second, it captures technological change through scenario-based modelling of future improvements in manufacturing processes, material intensities, and technology performance. Third, it scales impacts from components and vehicles to national deployment by modelling fleet composition and adoption pathways through to 2050, enabling cumulative impacts to be quantified rather than only per-vehicle impacts.

These differences affect the interpretation of hydrogen mobility. Under a conventional static LCA framing, hydrogen vehicles can appear environmentally competitive when assessed under assumed low-carbon hydrogen supply conditions. However, the dynamic modelling in this thesis shows that cumulative impacts during transitional phases can dominate system outcomes. In particular, early-stage hydrogen deployment occurs under partially decarbonised energy systems and immature manufacturing supply chains, meaning that upstream emissions and costs accumulate over time even if long-term tailpipe emissions are eliminated. This helps explain why scenarios with higher hydrogen penetration can exhibit higher cumulative GHG emissions than BEV-dominant pathways, despite both being *"zero tailpipe emission"* options.

Similarly, static single-vehicle LCA and LCC assessments can mask system-level scaling effects. By simulating national uptake, this thesis demonstrates that hydrogen production and component manufacturing costs scale strongly with fleet penetration and can impose disproportionate system-level economic burdens relative to their share of the fleet. These scaling dynamics are not visible in static assessments but are central to policy evaluation, since they determine the aggregate cost and emissions implications of deployment targets.

Finally, the dynamic modelling framework reveals pathway dependence that static assessments cannot capture. The higher cumulative energy demand in the Transitional scenario is driven by early deployment of H_2 -ICE vehicles, demonstrating that the timing of technology deployment influence long-term outcomes. A static comparison of drivetrain efficiencies would not reflect this compounding effect over decades of fleet turnover.

Overall, this comparison indicates that static LCA remains useful for technology benchmarking, but dynamic fleet-level modelling is better suited for evaluating na-

tional decarbonisation strategies where technologies, energy systems and adoption pathways evolve over time. The conclusions of this thesis therefore extend beyond identifying which technologies can be low-carbon in principle, toward evaluating the conditions and transition trajectories under which hydrogen mobility contributes to (or undermines) cumulative emissions reduction and system affordability. The specific methodological and empirical contributions enabled by this dynamic approach are summarised in Table 61.

Table 61: Delivery of Thesis Objectives and Contribution to Knowledge

Thesis Objective	What This Thesis Delivered	Contribution to Knowledge
(1) Quantify critical material demands of green hydrogen production	Integrated material flow analysis with environmental impact assessment for PEM electrolysis systems	Expands existing cradle-to-gate system boundaries to dynamic, cradle-to-grave assessments for PEM electrolysis to include quantified environmental burdens of critical materials
(2) Evaluate the GHG environmental impact of green hydrogen production	Full cradle-to-grave LCA including end-of-life modelling for PEM electrolyzers, fuel cells, and hydrogen storage tanks	Addresses lack of EoL data in hydrogen LCAs and enables more complete lifecycle emissions accounting
(3) Explore advancements and future evolution of hydrogen technologies	Introduced dynamic modelling elements reflecting technological improvement and future deployment scenarios	Moves hydrogen LCA beyond static inventories toward forward-looking assessments
(4) Compare the environmental and economic viability of hydrogen vehicles, including emerging systems (FCREx, H_2 -ICE)	Comparative cradle-to-grave LCA and lifecycle cost analysis of FCEV, FCREx, and H_2 -ICE within a unified framework	To my knowledge, this work provides one of the first integrated LCA-LCC assessments of emerging hydrogen vehicle configurations

Thesis Objective	What This Thesis Delivered	Contribution to Knowledge
(5) Assess cradle-to-grave fleet-level environmental and resource impacts of hydrogen vehicle deployment	Developed a dynamic fleet-level LCA model incorporating national fleet composition, infrastructure scenarios, and real-world driving data	Advances vehicle LCA from single-vehicle static analysis to system-level, policy-relevant modelling

7.4 Limitations

Study location and regional applicability: While work on PEM electrolysis takes a global approach to green hydrogen production, the vehicle and fleet studies focus on Europe and the UK. This study’s European focus limits its direct applicability to other regions with different energy systems, economic conditions, and policy frameworks. The findings may not reflect market realities in regions with more carbon-intensive electricity grids or different cost structures for hydrogen production and vehicle deployment. However, key insights on the high cost of green hydrogen production, the importance of supply chain decarbonisation, and the challenges of material constraints are likely to have broader applicability across different regional contexts. Expanding the estimation of vehicle and fleet costs and emissions in other geographical regions can be easily adapted and expanded using the methods and models developed. This would increase the accuracy of the results and make them more applicable to a wider audience. Including country-specific government grants and electricity grids, for example, could add a new level of detail to both the analysis and discussions.

Non-financial costs: Although lifecycle cost covers the entire financial costs associated with owning and operating a vehicle or fleet, the work could be improved by expanding the scope of study to quantify the non-financial costs associated with hydrogen vehicle deployment, today and in the future. For example, comparing productivity losses due to the charging times of EVs or changes to operating routes (and thus mileage) based on the location of projected hydrogen refuelling stations or EV charging point developments.

Limited data availability: LCA relies on accurate and comprehensive data to evaluate the environmental impacts at each stage of a product’s life cycle. However, data availability can be limited, especially for emerging hydrogen technologies. For several hydrogen vehicle types, because their market penetration is low or they are not yet in operation, inventory data is difficult to collect or not readily available in public life cycle inventory databases like Ecoinvent or GREET. To address this, in cases where primary data were unavailable, assumptions were informed by literature and expert consultation, using proxy data from technologies with similar performance and material profiles. For example, this was the case when constructing a life cycle inventory for H_2 -ICE and FCREx vehicles. Nonetheless, it should be acknowledged that such assumptions could influence the accuracy of results, and if available in the future, primary data - which is more representative - should be utilised.

Technological Uncertainty: Long-term projections inherently carry uncertainty regarding novel innovations. There may be unexpected technological develop-

ments or changes in cost trajectories that may not be captured in current learning curve models. To address this, the analysis considers multiple scenarios to give a broader overview of potential futures, considering the uncertainty involved in technological development.

7.5 Future Work

This section discusses the recommendations for future work in the following studied areas: life cycle impact categories, hydrogen technology development, PGM recovery and system integration.

Lifecycle impact categories. This thesis evaluated the environmental impact of hydrogen mobility by analysing the greenhouse gas emissions, material demand, and life cycle costs across vehicles and the UK fleet. The assessment included temporal variability in hydrogen production pathways, including the decarbonisation of electricity and evolving material requirements for PEM electrolyzers and hydrogen vehicle components. The analysis showed that upstream supply chain choices – particularly the carbon intensity of hydrogen production and sourcing of critical materials strongly influence lifecycle outcomes. Future research could explore additional impact categories beyond GHG emissions, material demand and cost and incorporate other sustainability indicators such as water use, land occupation, eutrophication, acidification, etc. These indicators are important because they capture environmental effects that are not reflected in carbon or material metrics alone. Including a broader set of lifecycle impact categories would enable a more comprehensive assessment of hydrogen production and mobility systems.

Hydrogen technology development. This thesis encompassed three primary hydrogen production technologies: AE, PEM, and SOE, as well as one fuel cell technology, PEMFC. Future advancements in electrolysis technologies may include a new technology, anion exchange membranes (similar to PEM but without the use of critical materials), and other fuel cell types that could be used in vehicles, such as direct methanol fuel cells (DMFCs) and alkaline fuel cells (AFCs). Although solid oxide fuel cells (SOFCs), phosphoric acid fuel cells (PAFCs), and molten carbonate fuel cells (MCFCs) are established fuel cell types, they are mainly used for large-scale stationary power, rather than in vehicles. As a result, future work could focus on alternative or lower TRL systems, as novel production methods and fuel cells may become more attractive as the technology progresses. Further research could assess the environmental and cost implications of breakthrough technologies when data becomes available.

PGM recovery. Future research could investigate the availability of PGMs,

particularly iridium and platinum, from secondary sources such as spent automotive catalysts and urban mines. These materials represent a potentially significant reservoir of critical metals that could support the scaling of PEM electrolyzers and fuel cell systems without relying solely on primary extraction. Automotive catalytic converters, which historically used platinum, palladium, and rhodium, are expected to generate increasing volumes of recoverable PGMs as ICEVs are phased out. Urban mining - recovering metals from end-of-life electronics, industrial scrap, and consumer goods - may also offer viable pathways for supplementing PGM supply. Quantifying the recovery potential, purity, and timing of these flows would help assess whether they can meaningfully offset future demand, particularly during the 2025–2040 electrolyzer deployment window. Integrating these sources into dynamic material flow models could improve projections of resource availability and inform recycling infrastructure planning and technology development for low-impact PGM recovery.

System integration. Future research could look beyond LDV vehicle analysis to examine hydrogen’s role within wider energy system transitions. Incorporating hydrogen mobility into comprehensive whole-system energy models would allow for assessment across sectors not captured in transport-focused studies. Multi-sector deployment studies could identify opportunities for infrastructure sharing, such as co-located refuelling hubs for transport and industrial users that improve utilisation or reduce system-level costs. By embedding hydrogen mobility into a broader energy system context, future work could better quantify its contribution to national decarbonisation goals, identify optimal deployment pathways, and support integrated planning for infrastructure and investment.

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