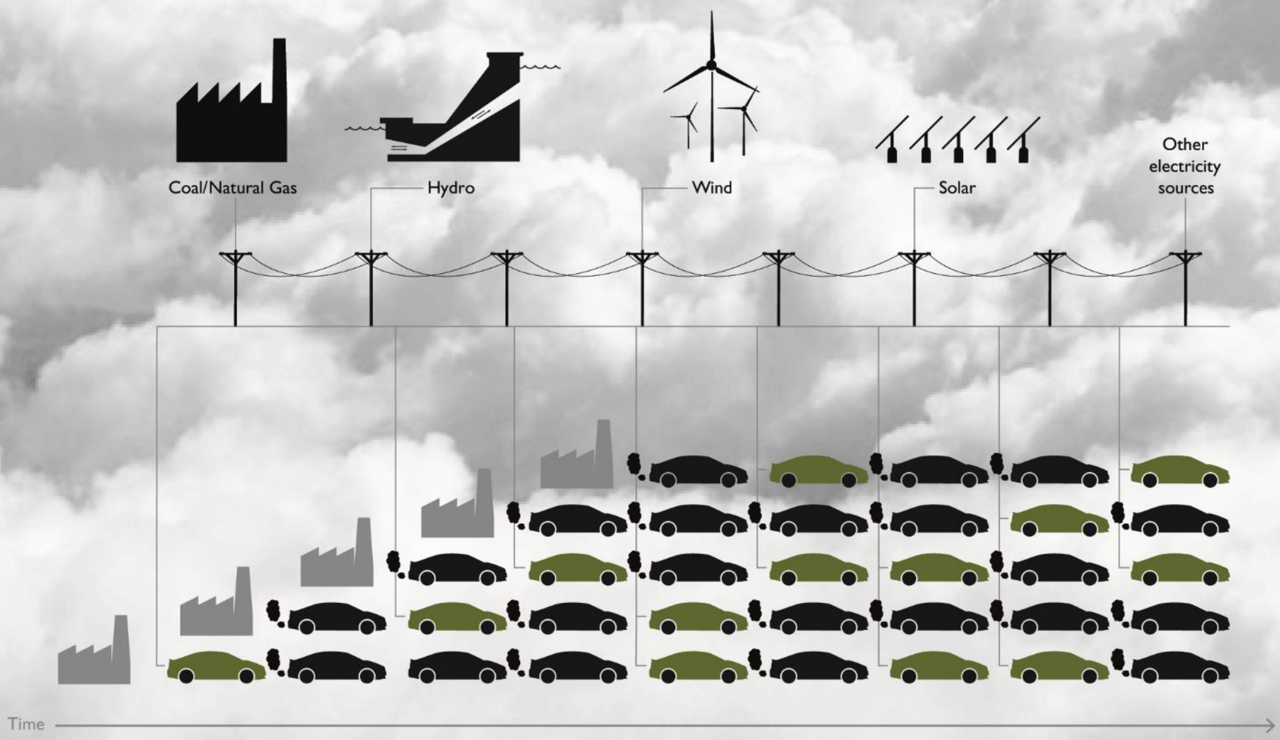

Rita Pinheiro Garcia

DYNAMIC FLEET-BASED LIFE-CYCLE ASSESSMENT

ADDRESSING ENVIRONMENTAL CONSEQUENCES OF THE INTRODUCTION OF ELECTRIC VEHICLES IN PORTUGAL

PhD thesis in Sustainable Energy Systems, supervised by Professor Fausto Miguel Cereja Seixas Freire,
presented to the Department of Mechanical Engineering, Faculty of Sciences and Technology, University of Coimbra

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ABSTRACT

Electric vehicles (EVs) have been promoted as an alternative to reduce greenhouse gas (GHG) emissions, fossil fuel dependence, and urban pollution caused by the transportation sector; however, a large scale adoption of EVs faces significant challenges. A number of studies have assessed the effects of EVs in the electricity system and the environmental impacts of different scenarios of evolution of the transportation sector. However, few studies integrating both electricity and fleet displacement effects have been performed. A dynamic fleet-based life-cycle perspective is necessary to understand the consequences and determine the extent to which the introduction of EVs in the fleet can actually reduce rather than simply shift environmental impacts of personal transport.

In this thesis, a dynamic fleet-based life-cycle framework was developed to assess the effects on environmental impacts of the introduction of EVs in a fleet. The framework combines fleet analysis and dynamic life-cycle modelling of vehicles to investigate the displacement of conventional vehicles over time, and consequential life-cycle assessment of electricity to assess the changes induced in the operation of the electricity system due to EV charging. The analysis focused on the case of introducing battery electric vehicles (BEVs) in the Portuguese light-duty fleet and on the effects on GHG emissions. A comprehensive life-cycle assessment of electricity generation and supply in Portugal was also performed to identify the main drivers of impacts, how impacts change over time, and how charging time influences BEV GHG emissions.

Reducing fleet-wide GHG emissions by displacing internal combustion engine vehicles (ICEVs) by BEVs in Portugal depends mostly on the GHG intensity of the Portuguese electricity system, on the degree of reduction in fuel consumption of new ICEVs, and on the level of penetration of BEVs. In order to achieve significant reductions compared to an increasingly more efficient ICEV fleet, a high BEV market share and electricity GHG intensity similar or lower to the current mix (485 g CO_2 eq kWh^{-1}) need to be realized. The response of the electricity system to BEV demand, regarding the changes in electricity generation by the various sources and corresponding GHG emissions, may thus determine the benefits of BEVs over conventional technologies.

Electricity GHG emissions in Portugal vary significantly between years and throughout the year. As a result of the temporal variability in electricity generation and, in particular, in the marginal supply, the time of charging can have a major influence on the GHG benefits of

BEVs in the short-term. What has been considered, in general, the most favorable charging time from the economic and operation of the electricity system perspective (off-peak hours), may not be so from an environmental standpoint. In Portugal, simply encouraging charging during the night may increase emissions from the electricity system as a result of the fossil-based marginal electricity supply (mostly coal). Therefore, charging control strategies should ensure that surplus renewable energy use by BEVs is maximized so that environmental impacts can be reduced. However, interactions with other strategies to enable renewable energy sources, such as electricity storage, may be important and should be accounted for.

When the electricity system includes significant storage of energy, for instance through pumped hydro storage (PHS), the effects of introducing BEVs go beyond the straightforward displacement of ICEVs and increase in electricity demand, to include significant indirect effects from the dynamics of storage. Such indirect effects may decrease or even offset the GHG benefits of ICEV displacement. However, the net effects on GHG emissions are very dependent on the technologies displaced both by PHS and by BEVs, so that detailed analysis is needed for any specific energy system, allowing for future technological improvements.

The dynamic fleet-based life-cycle framework developed in this thesis provides a comprehensive environmental assessment of the adoption of a new technology, because it enables explicit assessment of changes in technologies and background systems over time in a fleet perspective, as well as indirect effects related to the existing system. In particular, this framework can be used to assess the effects on environmental impacts of other electricity-using products in a fleet perspective, and of measures that improve the energy efficiency of end-use applications or that shift the use of electricity. The change-oriented approach pursued can also aid in understanding the effects of policies and strategies that enable and promote the use of electricity over other fuels.

Keywords: battery electric vehicles, electricity, fleet model, greenhouse gas emissions, indirect impacts, industrial ecology, life-cycle assessment, marginal emissions, temporal variability.

RESUMO

Os veículos elétricos (EVs) têm sido apontados como alternativa para reduzir as emissões de gases com efeito de estufa (GEE), a dependência de combustíveis fósseis e a poluição em meio urbano causadas pelo sector dos transportes, mas a sua adoção enfrenta importantes desafios. Vários estudos avaliaram os efeitos dos EVs no sistema elétrico e os impactes ambientais de diferentes cenários de evolução do setor dos transportes, mas poucos analisaram em conjunto os efeitos ambientais sobre o sistema elétrico e aqueles relativos à substituição de veículos de combustão interna (VCIs). Deste modo, para perceber as consequências e determinar se a introdução de EVs na frota reduz efetivamente os impactes ambientais associados ao transporte individual ou se apenas os transfere para outras partes do sistema é necessário adotar uma perspetiva dinâmica de ciclo de vida (CV).

Esta tese apresenta uma abordagem dinâmica de CV com base em modelos de frota para avaliar os efeitos ambientais da introdução de EVs numa frota. A abordagem combina análise de frotas e modelação dinâmica de CV de veículos, com o objetivo de avaliar os efeitos da substituição de VCIs por EVs ao longo do tempo, e avaliação consequencial de CV de sistemas elétricos, para avaliar as alterações induzidas na operação do sistema elétrico devido ao carregamento dos EVs. A análise incidiu sobre a introdução de EVs a baterias (BEVs) no parque automóvel ligeiro português e focou-se na avaliação das emissões de GEE. Foi ainda realizada uma avaliação abrangente de CV da geração de eletricidade em Portugal com o objetivo de identificar os fatores que mais contribuem para os impactes, de que forma variam os impactes ao longo do tempo e qual a influência do horário de carregamento nas emissões de GEE dos BEVs.

A redução total das emissões de GEE da frota automóvel em resultado da substituição de VCIs por BEVs em Portugal depende da intensidade de GEE do sistema elétrico português, do grau de redução no consumo de combustível dos novos VCIs e no nível de penetração de BEVs. De modo a alcançar reduções significativas em comparação com uma frota de VCIs cada vez mais eficientes, é necessário que a quota de mercado dos BEVs seja elevada e que a intensidade de GEE do sistema elétrico seja inferior ou semelhante à atual (485 g CO_2 eq kWh^{-1}). A resposta do sistema elétrico à procura dos BEVs, relativamente à variação na geração de eletricidade pelas várias fontes e correspondente variação nas emissões de GEE, determina os benefícios dos BEVs relativamente aos VCIs.

As emissões de GEE da eletricidade em Portugal variam significativamente de ano para ano e ao longo do ano. Devido à variabilidade temporal na geração de eletricidade e, em particular, na geração marginal, o horário de carregamento tem uma grande influência nas emissões de GEE causadas pelos BEVs no curto prazo. Aquele que tem sido considerado, em geral, o período mais favorável para o seu carregamento do ponto de vista da operação do sistema elétrico (horas de vazio), pode não o ser do ponto de vista da redução dos impactes ambientais. Em Portugal, incentivar o carregamento durante a noite pode resultar num aumento das emissões do sistema elétrico, uma vez que a tecnologia marginal é na maioria do tempo carvão. Deste modo, as estratégias de controlo dos carregamentos devem ser implementadas de forma a garantir a maximização da utilização da energia renovável em excesso para reduzir os impactes ambientais. No entanto, é preciso ter em conta a interação entre os BEVs e outras estratégias de utilização de energia renovável intermitente, como é o caso dos sistemas de armazenamento de eletricidade.

Quando o sistema elétrico permite o armazenamento de quantidades significativas de eletricidade, por exemplo através de barragens hidroelétricas com sistemas de bombagem, os efeitos da introdução de BEVs vão além da simples substituição de VCIs e aumento da procura por eletricidade, para incluir efeitos indiretos significativos associados à dinâmica de armazenamento. Tais efeitos podem diminuir ou mesmo anular os benefícios em termos de emissões de GEE associados à substituição de VCIs. No entanto, o efeito líquido sobre as emissões de GEE depende muito das tecnologias substituídas tanto pelos BEVs como pelos sistemas de armazenamento, pelo que é necessário efetuar uma análise detalhada para cada sistema energético, tendo em conta melhorias tecnológicas futuras.

A abordagem desenvolvida nesta tese permite avaliar de forma integrada os impactes ambientais causados pela adoção de uma nova tecnologia, uma vez que possibilita avaliar explicitamente alterações no sistema ao longo do tempo bem como efeitos indiretos. Em particular, a abordagem desenvolvida pode ser usada para avaliar os efeitos ambientais de outros produtos que consomem eletricidade numa perspetiva de frota, bem como de medidas para melhorar a eficiência energética. A abordagem consequencial implementada permite ainda ajudar a compreender as consequências de políticas e estratégias que promovam a utilização de eletricidade em substituição de outros combustíveis.

Palavras-chave: Avaliação de Ciclo de Vida, Ecologia Industrial, eletricidade, emissões de gases com efeito de estufa, emissões marginais, impactes indiretos, modelo de frota, variabilidade temporal, veículos elétricos a bateria.

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ACRONYMS

AC: Acidification.

AD: Abiotic depletion.

ALCA: Attributional life-cycle assessment

BAU: Business-as-usual

BEV: Battery electric vehicle (EV with a fully electric drivetrain)

CC: Combined cycle

CHP: Combined heat and power

CLCA: Consequential life-cycle assessment

CO_2: Carbon dioxide

DeNOx: Denitrification

DeSOx: Desulphurization

EF: Emission factor

EoL: End-of-life

EU: European Union

EUT: Eutrophication

EV: Electric vehicle

FCHEV: Fuel cell hybrid electric vehicle

FCV: Fuel cell vehicle

GHG: Greenhouse gas

GW: Global warming

HEV: Hybrid electric vehicle

ICEV: Internal combustion engine vehicle

LC: Life cycle

LCA: Life-cycle assessment

LCI: Life-cycle inventory

LDV: Light-duty vehicles

MEF: Marginal emissions factor

NG: Natural gas

NO_x: Nitrogen oxides

nREn: Non-renewable fossil energy

OD: Ozone layer depletion

OFAT: One-factor-at-a-time

PHEV: Plug-in hybrid electric vehicle

PHS: Pumped hydro storage

PO: Photochemical oxidation

PP: Power plant

PV: Photovoltaic

RES: Renewable energy sources

SO_2: Sulfur dioxide

T&D: Transmission and distribution

VKT: Vehicle kilometer travelled

WtW: Well-to-wheels

Chapter 1

INTRODUCTION

1.1 Background and motivation

The transportation sector is energy- and carbon-intensive, contributing to about 32% of the final energy consumption and 25% of the greenhouse gas (GHG) emissions in the European Union (EU28) in 2012 (European Commission 2014). In Portugal, these figures are even higher: 40% and 36%, respectively (European Commission 2014). Light-duty vehicles (LDVs) are of special concern, as they are responsible for about 15% of EU's CO_2 emissions (European Commission 2012a). The reduction of energy and GHG emissions in this sector is the goal of several current policies (e.g., EU Climate and Energy Package, U.S. Corporate Average Fuel Economy standards). Several measures to reduce energy and environmental impacts from vehicles have been proposed, which include the reduction of fuel consumption of conventional technologies (e.g., through lightweighting, downsizing, and more efficient powertrains), displacement of fossil fuels by biofuels, and development of alternative technologies (e.g., electric vehicles, fuel cell vehicles) (Leduc et al. 2010; Althaus 2012; Geyer 2016).

Recent attention has been drawn to the potential of electric vehicles (EVs) to reduce energy and environmental impacts (Althaus 2012). A number of countries have set targets for EV sales, and adopted policy measures to promote EV adoption, such as financial incentives and infrastructure deployment (IEA 2013). In Portugal, a public charging infrastructure has been deployed. This strategy, combined with the incorporation of high levels of renewable energy in Portugal's electricity mix, aimed to promote the adoption of EVs and the development of related industries. Nevertheless, the market share of EVs has been low and additional efforts are deemed to be required to boost their wide adoption.

EVs use one or more electric motors for propulsion. They can be more energy-efficient than internal combustion engine vehicles (ICEVs) and, contrary to these, do not have tailpipe emissions, a major source of air pollution in cities. They do not rely exclusively on petroleum-based fuels and can use the diversified set of energy sources used for electricity generation, including possible endogenous renewable resources, with potential impact on GHG emission reduction and the security of energy supply. Despite these advantages over conventional technologies, a large scale adoption of EVs also faces many challenges, such as: the long-term availability of mineral resources required for batteries and electronic components; the high purchase costs; the lack of charging infrastructure; the 'range anxiety' issue; and the extra burden imposed on the electricity system. By using electricity as energy carrier, EVs provide a linkage between the transportation sector and the electricity sector, which entails both risks (e.g., EVs affect the performance, efficiency and required capacity of the electric grid) and opportunities (e.g., they can assist in the integration of intermittent renewable sources in the electric grid) (Richardson 2013).

In order to assist policymaking towards GHG emission reduction, it is important to understand what the potential environmental effects of introducing EVs in a vehicle fleet are and in what conditions that introduction is environmentally beneficial. There is a growing consensus in the scientific literature that the assessment of the environmental impacts of EVs should be performed considering a life-cycle perspective (Hawkins et al. 2012; MacPherson et al. 2012; Nordelöf et al. 2014b; Batista et al. 2015). Several studies applying the life-cycle assessment (LCA) methodology have addressed the environmental impacts of EVs and compared them with those from different powertrains (e.g., McCleese and LaPuma 2002; Samaras and Meisterling 2008; Gao and Winfield 2012; Freire and Marques 2012; Hawkins et al. 2012; Marques et al. 2013; Hawkins et al. 2013; Messagie et al. 2014; Nordelöf et al. 2014; Noshadravan et al. 2015). A number of factors influencing EV life-cycle environmental impacts has been identified, namely:

- Electricity sources used for charging. Several LCA studies on EVs showed that the electricity mix used to charge the vehicles is a key aspect for the environmental performance of these vehicles and is a determinant factor in the comparison with conventional vehicles (e.g., Hawkins et al. 2012; Nordelöf et al. 2014; Noshadravan et al. 2015; Girardi et al. 2015; Garcia et al. 2015);
- Vehicle and battery manufacturing impacts, which can be twice as those from conventional technologies (e.g., GHG emissions) (Hawkins et al. 2013; Nordelöf et al. 2014b; Correia et al. 2014);
- Battery chemistry, mass, lifetime, and recharge efficiency, which influence both the manufacturing and use phase impacts of EVs (Hawkins et al. 2013; Faria et al. 2014; Marques et al. 2015b);

- Driving and charging behavior, including the use of climate control systems and other auxiliary systems, which affect energy consumption, battery lifetime, and the electricity source used for charging and corresponding environmental impacts (Faria et al. 2013; Yuksel and Michalek 2015; Rangaraju et al. 2015).

The contribution of electricity to BEV impacts is a key focus of this research, while the other aspects are addressed in less detail. A part from these factors, which contribute to the direct environmental impacts of EVs, a large scale EV adoption will entail indirect effects on the environment, such as:

- Resource criticality issues, which arise from the use in batteries and electronic components of lithium and other scarce metals with limited global supply. A large scale adoption of EVs will potentially impact reserves of these minerals and extensive recycling, which is currently poorly-developed, will probably be needed (Söderman et al. 2014).
- Technology displacement effects. The impact reduction potential of EVs is dependent on the environmental performance of the replaced technology (Miller and Keoleian 2015). EV benefits for GHG emission reduction will depend on how they compare with increasingly more energy-efficient conventional vehicles, as the introduction of EVs in the fleet is gradual and its effects will not be seen in the short term (Frischknecht and Flury 2011).
- Changes in electricity demand and therefore on the electricity system operation and configuration. A shift towards electricity in the transportation sector will place an additional stress upon the electricity system and distribution infrastructure (Hedegaard et al. 2012; Tarroja et al. 2014). On the other hand, EVs are also seen as a way of increasing renewable energy penetration, due to their potential demand response abilities (Hedegaard et al. 2012; Richardson 2013; Dallinger et al. 2013).

The assessment of this indirect effects calls for an extension of the traditional static, single-product LCA towards: (i) addressing dynamic aspects regarding the shift of technologies over time, as well as advances in material processing, technology development and changes in electricity production, and (ii) capturing the effect of scale and timing of changes, so that indirect impacts on other systems (e.g., the electricity system) can be assessed.

The absence of dynamic aspects in LCA has been pointed out as an important limitation of the methodology (Reap et al. 2008). A number of papers have discussed ways of incorporating dynamic aspects in LCA and shown their relevance for some systems and environmental impacts (e.g., Field et al. 2000; Pehnt 2006; Kendall et al. 2009; Levasseur et al. 2010; Stasinopoulos et al. 2011; Collinge et al. 2013). Among these, fleet-based life-cycle approaches have shown to be able to explicitly account for time and capture the scale of an intervention by focusing on the product fleet (i.e. the dynamic set of products in use, including the transient effects as new products replace end-of-life products in the fleet) rather than a single-product (Field et al. 2000).

Fleet-based life-cycle approaches have been used to assess the environmental impacts (mostly energy and GHG emissions) of different scenarios of evolution of the transportation sector (e.g., Bandivadekar et al. 2008; Baptista et al. 2012; Bodek and Heywood 2008; Kromer et al. 2010; Reichmuth et al. 2013); however, few of these studies have specifically assessed displacement effects of a new technology, such as EVs, in a fleet. Most studies included EVs as one of the many options analyzed through scenario analysis, making it difficult, if not impossible, to discern what the influence of EVs alone have in the environmental impacts and what factors influence the most those impacts (EPRI 2007; Baptista et al. 2012). Even the few studies that addressed a specific technology did not address the influence on the impacts of some key issues, such as technology improvements or the electricity source, and did not take a full life-cycle approach (vehicle production and end-of-life was excluded from the assessment) (Keoleian et al. 2011; Reichmuth et al. 2013).

On the other hand, several studies have assessed the effects of EVs in the electricity system, regarding, for instance, the impact on energy and CO_2 emissions (e.g., McCarthy and Yang 2010; Camus et al. 2011; Pina et al. 2014), and the integration of renewable energy sources (RES) (Hedegaard et al. 2012; Dallinger et al. 2013). However, few integrated studies, linking both electricity impacts and fleet displacement effects have been performed (Camus et al. 2011; Hedegaard et al. 2012). Moreover, interactions between EVs and other potentially competing technologies in the grid were seldom assessed. In particular, the interaction between large storage capacity (e.g., pumped hydro storage [PHS]) and EV charging has been mostly disregarded. This effect is relevant for grid systems with large PHS capacity installed or planned to be installed in combination with RES capacity, such as the case of Portugal. The time of EV charging and its effects on environmental impacts is a current topic of research (e.g., Faria et al. 2013; Rangaraju et al. 2015), but there is still controversy as on the benefits of off-peak versus peak charging, and on whether average or marginal emissions should be assessed (Yang 2013).

Additional knowledge is thus required on the effects of a large scale adoption of EVs and on the factors influencing their environmental impacts. This thesis aims to shed light on some of these effects and on how they can be assessed through a dynamic fleet-based LCA framework.

1.2 Research questions

This PhD thesis departs from the overall question: *What are the potential environmental effects/consequences of electric vehicle adoption?* In this respect, two main classes of effects can be identified: direct or inherent effects, resulting from environmental flows associated with the production, use and end-of-life of electric vehicles; and indirect effects, resulting from changes induced in the existing system, such as displacement of conventional vehicles and changes in electricity demand. Both types of effects are addressed, but the main focus is on indirect effects. In particular, the main objective of this thesis is to investigate the life-cycle GHG emission reduction potential of EVs displacing ICEVs in a light-duty vehicle fleet and address aspects of the interaction between the transportation and the electricity sectors in the short- and long-term, within a dynamic fleet-based LCA framework.

An applications-driven approach is followed. The analysis is focused on the case of introducing battery electric vehicles (BEVs) in the Portuguese light-duty fleet as the technological and geographical scope of this research. Portugal was chosen as a case study for its favorable conditions for EV deployment (charging network in place, policy incentives for buying EVs, and high share of renewable energy sources installed and planned to be installed in the electricity system). The analysis is centered on BEVs, rather than including hybrid technologies, which can be seen, in a simplified way and in the particular scope of this thesis, as an intermediate state between conventional and electric vehicles. By having a more detailed focus on a specific technology, it is possible to perform a more comprehensive analysis on the underlying factors contributing to the environmental effects of a technology shift. The analysis of the environmental impacts of introducing BEVs in the Portuguese fleet presented in this thesis provides an example of how potential effects of the deployment of a new technology can be analyzed and assessed within a dynamic fleet-based LCA framework.

This research aims to contribute to answer the general question asked at the beginning of this section, with emphasis on the assessment of two main indirect effects of electric vehicles adoption: (i) the displacement of conventional vehicles in the feet; and (ii) the changes induced in the electricity system due to EV charging. However, primarily to assess the effects of EVs in the electricity system it is necessary to have a comprehensive understanding of the specific electricity system under study. Life-cycle inventories and models of electricity generation, transmission and distribution were implemented in order to provide a comprehensive picture of the environmental impacts of electricity supply for the specific case of Portugal.

Although it is recognized that resource criticality issues are an important topic of research in the context of EV impacts, this effect is beyond the scope of this thesis primarily because it requires a global scale and this thesis has a narrower geographical scope. The approach developed in this thesis may be used to assess a large set of environmental impacts; however, due to the complexity of the models developed, most of the results presented focus on greenhouse gas (GHG) emissions for the sake of simplicity. The exception was the assessment of annual electricity impacts, which comprised several impact categories, as the models developed allowed its assessment in a straightforward way.

Deriving from the general question and the research needs identified, five research questions were formulated and are presented in Table 1.1 along with the specific objectives to be pursued.

Table 1.1 Research questions and specific objectives.

Research questions	Specific objectives	Chapters
1. *What are the conditions under which the displacement of conventional vehicles by EVs in the Portuguese light-duty fleet reduces transportation GHG emissions?*	(i) To develop a parameterized dynamic fleet-based life-cycle model of the Portuguese light-duty fleet; (ii) To develop scenarios exploring options to reduce light-duty vehicle GHG emissions; (iii) To assess fleet-wide GHG emissions over time for each scenario and analyze the influence of different model parameters on the results.	3
2. What are the life-cycle environmental impacts attributed to electricity generation and supply in Portugal in the last decade and how do they vary between years and throughout the year? How does this variability affect the environmental impacts of an EV as a function of the time of charging?	(i) To develop life-cycle models of the main electricity generation technologies available in Portugal as well as the Portuguese transmission and distribution grid infrastructure; (ii) To assess life-cycle environmental impacts of the Portuguese annual electricity mix from 2003 to 2014 and examine how the recent changes in the technology portfolio affected the environmental performance of the electricity generated and supplied in Portugal; (iii) To characterize the temporal variability in the GHG emissions of the Portuguese electricity mix in the last years and to assess GHG emissions of an EV charged at different hours of the day; (iv) To provide a comprehensive understanding of the environmental impacts of generating and supplying electricity in Portugal, including the main drivers of impacts and how they change over time.	4
3. What are the potential effects of EV deployment in the GHG emissions of the Portuguese electricity system in the short-term?	(i) To determine the short-term marginal technology affected by a change in electricity demand using regression techniques; (ii) To assess the effect of hour of the day and load in the marginal supply and the implications of EV charging for overall GHG emissions.	5
4. How does the addition of EVs to a grid with a large storage capacity influence the GHG emissions of the electricity system?	(i) To investigate the interactions between EVs and pumped hydro storage (PHS) by comparing the changes in GHG emissions due to the introduction of EVs for different scenarios in Portugal. (ii) To shed light on the interaction between competing strategies to reduce renewable energy curtailment, such as EVs and PHS, and its effects in the electricity GHG emissions.	5
5.What insights does one gain by applying an attributional and a consequential LCA approach to the same electricity system?	(i) To compare the results of an attributional LCA of an EV as a function of the time of charging with those from a consequential LCA of the same system; (ii) To assess the influence of electricity storage on average and marginal electricity GHG emissions; (iii) To identify the main methodological differences and differences in scope, and its implications for the interpretation of results.	4 and 5

1.3 Contribution

This thesis contributes to the environmental assessment of the introduction of electric vehicles in a fleet, in particular by:

1. Demonstrating how potential indirect effects of the deployment of electric vehicles can be analyzed and assessed within a dynamic fleet-based life-cycle framework;
2. Shedding light on potential effects of the introduction of electric vehicles in Portugal;
3. Increasing knowledge on the environmental impacts of the electricity system in Portugal, including the main drivers of impacts and how they change over time;
4. Providing insights on the application of attributional and consequential LCA to electricity systems.

Most of the research in this PhD thesis is based on the following core articles published or under review in ISI-indexed journals (abstracts and keywords of the articles are presented in Appendix I):

1. **Garcia, R.**, Gregory, J., Freire, F. (2015). Dynamic fleet-based life-cycle greenhouse gas assessment of the introduction of electric vehicles in the Portuguese light-duty fleet. *The International Journal of Life Cycle Assessment* 20(9):1287-1299. http://dx.doi.org/10.1007/s11367-015-0921-8

   JCR® impact factor (2014): 3.988

2. **Garcia, R.**, Marques, P., Freire, F. (2014). Life-cycle assessment of electricity in Portugal. *Applied Energy* 134:563-572. http://dx.doi.org/10.1016/j.apenergy.2014.08.067

   JCR® impact factor (2014): 5.613

3. **Garcia, R.**, Freire, F., Clift R. (2015). Effects on greenhouse gas emissions of introducing electric vehicles into an electricity system with large storage capacity. (submitted)
4. **Garcia, R.**, Freire, F. (2015). Fleet-based life-cycle approaches: a review focusing on energy and environmental impacts of vehicles. (submitted)

This PhD research also contributed to the following articles:

5. Domingues, A.R., Marques, P., **Garcia, R.**, Freire, F., Dias, L. (2015). Applying Multi-Criteria Decision Analysis to the Life-Cycle Assessment of Vehicles. *Journal of Cleaner Production* 107:749-759.
   http://dx.doi.org/10.1016/j.jclepro.2015.05.086
   JCR® impact factor (2014): 3.844
6. Faria, R., Marques, P., **Garcia, R.**, Moura, P., Freire, F., Delgado, J., Almeida, A. (2014). Primary and Secondary Use of Electric Mobility Batteries from a Life Cycle Perspective. *Journal of Power Sources* 262:169-177.
   http://dx.doi.org/10.1016/j.jpowsour.2014.03.092
   JCR® impact factor (2014): 6.217
7. **Garcia, R.**, Freire, F. (2014). Carbon footprint of particleboard: a comparison between ISO/TS 14067, GHG Protocol, PAS 2050 and Climate Declaration. *Journal of Cleaner Production* 66:199-209.
   http://dx.doi.org/10.1016/j.jclepro.2013.11.073
   JCR® impact factor (2014): 3.844
8. Rangaraju, S., **Garcia, R.**, De Vroey, L., Marques, P., Messagie, M., Freire, F., Van Mierlo, J. (2015). Key parameters influencing the results of life cycle assessment of battery electric vehicle. (in final preparation for submission to an ISI-indexed journal)
9. Marques, P., **Garcia, R.**, Kulay, L., Freire, F. (2015). Comparative life-cycle assessment of $LiMn_2O_4$ and $LiFePO_4$ batteries for electric vehicles. (in final preparation for submission to an ISI-indexed journal)

In addition, more than ten articles related to this PhD research were published in conference proceedings with scientific refereeing. The full list of publications is presented in Appendix II.

1.4 Thesis outline

This thesis is composed of six chapters, including this introductory chapter, and is structured as follows (see Fig. 1.1 for an overview of how chapters are interlinked):

Chapter 2 reviews the state-of-the-art on the two major topics this research will contribute to: fleet-based life-cycle approaches and life-cycle assessment of electricity. Firstly, a critical review of the recent literature addressing **fleet-based LCA approaches**, including an overview of the modelling approach, its main applications, with focus on the assessment of pathways of evolution of the transportation system, is provided and research needs are identified. Secondly, the **application of LCA to electricity systems** is reviewed, highlighting the differences between attributional and consequential modeling approaches.

Chapter 3 starts with a description of the **dynamic fleet-based life-cycle model** of the Portuguese light-duty fleet developed, including the life-cycle parameters, data sources, and main assumptions. The main features of the model as well as its main limitations are discussed. The fleet-wide GHG emissions of displacing ICEVs by EVs across different scenarios are analyzed. Finally, a sensitivity analysis is performed to assess the influence of different parameters in the results and ranking of scenarios. The dynamic fleet-based LCA implemented provides the scale and timing for assessing the effects of EV load in the power grid, addressed in Chapter 5.

Chapter 4 presents a **comprehensive LCA of the electricity system in Portugal**, including the assessment of annual environmental impacts from 2003-2014, and hourly GHG emissions from 2012-2014. The influence of the time of charging of BEVs in GHG emissions is also evaluated form an attributional perspective. The models developed provide the ground for the assessment of the effects of BEVs in the Portuguese electricity system presented in Chapter 5.

Chapter 5 analyses the combined effects of BEV adoption on the electricity system and on the displacement of ICEVs in the fleet. A **dynamic fleet-based LCA framework** is implemented, which combines the dynamic fleet-based life-cycle model developed in Chapter 3 and **consequential LCA of electricity**, building on the LC models of electricity generation developed in Chapter 4. Firstly, a consequential life-cycle model of the Portuguese electricity system is implemented to assess the effects of an increase in electricity demand in the short-term. This model is then applied to the introduction of BEVs and results are presented for a range of scenarios highlighting the potential short-term effects of BEVs. Secondly, the **interactions between BEV and pumped hydro storage** in the electricity system are explored and the changes in GHG emissions for different scenarios in Portugal up to 2025 are compared.

Chapter 6 summarizes the key findings with respect to the main research questions of this thesis and provides recommendations for further research.

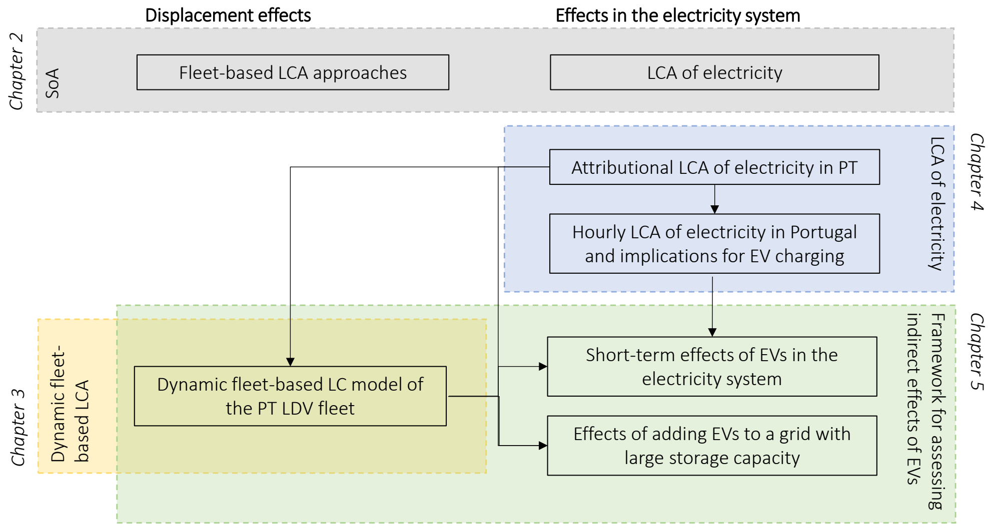

Fig. 1.1 Thesis overview.

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Chapter 2

STATE-OF-THE-ART

Abstract This chapter reviews the state-of-the-art on the two major topics this research will contribute to: Section 2.1 reviews fleet-based life-cycle approaches, including an overview of the modelling approach and its main applications, with focus on the assessment of pathways of evolution of the transportation system, and the identification of research gaps; Section 2.2 reviews the application of LCA to electricity systems, highlighting the main differences between attributional and consequential modeling approaches.

2.1 Fleet-based life-cycle approaches[1]

2.1.1 Introduction

Alternative vehicle technologies are being promoted as a way of reducing energy consumption and environmental impacts in the transportation sector. There is a growing consensus in the scientific literature that the assessment of the environmental impacts of these alternative technologies should be performed considering a life-cycle perspective (Hawkins et al. 2012; MacPherson et al. 2012; Nordelöf et al. 2014b). This avoids, for example, that technologies that have no tailpipe emissions, such as battery electric vehicles (BEVs), are presented as having no impacts. In fact, if upstream processes such as fossil-based electricity generation are included in the assessment, BEVs can have higher LC impacts than conventional technologies (i.e. internal combustion engine vehicles [ICEVs]) (Hawkins et al. 2013; Nordelöf et al. 2014b). Moreover, excluding vehicle production and end-of-life of the assessment ignores important sources of impacts (Hawkins et al. 2013; Nordelöf et al. 2014b).

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[1] Significant portions of this section appear in: Garcia R., Freire F. (2015). Fleet-based life-cycle approaches: a review focusing on energy and environmental impacts of vehicles (submitted).

The life-cycle assessment (LCA) methodology is often used to assess and compare the environmental impacts of vehicle technologies, but most studies are based on a static analysis of single vehicles (e.g., McCleese and LaPuma 2002; Samaras and Meisterling 2008; Gao and Winfield 2012; Freire and Marques 2012; Hawkins et al. 2012; Marques et al. 2013; Hawkins et al. 2013; Messagie et al. 2014; Nordelöf et al. 2014; Noshadravan et al. 2015) not addressing advances in material processing technology development, and changes in background processes (Nordelöf et al. 2014). When the goal is to assess new or changing technologies, transient effects may be important and cannot be captured with such static, single-product analysis (Field et al. 2000). Moreover, if the goal is to explore solutions able to reduce overall environmental impacts (e.g., to meet medium/long-term policy targets), both scale and timing of adoption may influence the results (Hillman and Sandén 2008; Stasinopoulos et al. 2011).

The absence of dynamic aspects in LCA has been pointed out as one main limitation of the methodology (Reap et al. 2008). A number of papers have discussed ways of incorporating dynamic aspects in LCA and shown their relevance for some systems and environmental impacts (e.g., Pehnt 2006; Kendall et al. 2009; Levasseur et al. 2010; Collinge et al. 2013). Two main focal points can be distinguished: impact assessment and system modeling. The temporal distribution of emissions is not generally taken into account in LCA: emissions occurring during the whole life cycle are aggregated into a single emission and their potential impacts on the environment are assessed as if they occurred at the same time (Levasseur et al. 2008). However, for some substances, their environmental impact is determined both by the timing of their release and the rate at which they decay or are removed from the environment, which highlights the importance of emission timing in the life-cycle impact assessment phase (Levine et al. 2008). Recently, this issue has received much attention, namely regarding time-dependent greenhouse gas (GHG) emission impact (e.g., Levasseur et al. 2012; Garcia and Freire 2014), and a number of approaches has been presented (O'Hare et al. 2009; Kendall et al. 2009; Levasseur et al. 2010; Cherubini et al. 2011; Kendall 2012). In this research, dynamic impact assessment is beyond the scope of the assessment and the focus is on the dynamics involving the technological system.

The consideration of dynamic aspects in system modeling and life cycle inventory phases has been investigated in a number of studies (e.g., Kandelaars and Bergh 1997; McLaren et al. 2000; Field et al. 2000; Pehnt 2006; Fischer and Pflieger 2007; Levine et al. 2007;

Stasinopoulos et al. 2011). Dynamic life cycle modeling is important when assessing future developments and the related environmental impacts. Dynamical aspects that can be relevant for such assessments include changes in boundary conditions over time, such as legal regulations, technical innovation, changes in technology or changes in used materials or upstream processes (Fischer and Pflieger 2007).

The assessment of the effects of deploying a new technology, such as EVs, in the environmental impacts requires an approach that is able to capture these dynamic aspects, as well as indirect displacement effects, as new products replace old products. Furthermore, it should be able to capture the scale and timing of the intervention, so that indirect impacts on other system (e.g., the electricity system) can be assessed. Within the dynamic modeling approaches to LCA developed so far, fleet-based LCA, originally presented by Field et al. (2000), has the potential to meet all these prerequisites.

This sections aims at performing a critical review of the recent literature addressing fleet-based LCA approaches, by providing: (i) an overview of the modelling approach; (ii) its main applications, with focus on the assessment of pathways of evolution of the transportation system; and (iii) an analysis of the key aspects underlying energy and environmental impacts of vehicle fleets (focusing on electrification pathways).

2.1.2 Fleet-based life-cycle assessment

Life-cycle assessment (LCA) is a methodology to systematically assess the environmental impacts directly and indirectly associated with a product system throughout its entire life cycle, from the 'cradle' (i.e. raw material extraction), to the 'grave' (i.e. final waste disposal). Because of its holistic approach, LCA is able to shed light on potential trade-offs between different environmental impacts and between different parts of the life cycle. LCA is one of the key tools of Industrial Ecology (Graedel 1996) and has been widely used to support policies and performance-based regulation (e.g., biofuels policies in the EU and EUA), as well as consumer-based information, such as carbon footprint standards (e.g., PAS 2050, GHG Protocol Product Standard, ISO 14067:2013) and environmental product declarations (e.g., ISO 14025:2006, EN 15804:2012). The LCA methodology is well-described elsewhere (e.g., Guinée et al. 2002; Baumann and Tillman 2004; Rebitzer et al. 2004).

LCA is traditionally a product-centered approach, i.e. most LCA studies estimate the life-cycle environmental impacts of a single product. Most of these studies also assume that the model parameters are constant. This means that the interactions between elements of the product system are excluded and only a snapshot of the system behavior is presented (Stasinopoulos et al. 2011). Therefore, the model may be valid only over a short period of time or even for a single point in time. Alternatively, Field et al. (2000) proposed a new fleet-based approach that circumvents these issues by considering the product fleet (i.e. the dynamic set of products in use, including the transient effects as new products replace end-of-life products in the fleet) rather than a single-product (Fig. 2.1).

Fleet-based life-cycle assessment is a modeling approach that combines the LCA methodology with a fleet model that describes the stocks and flows associated with a class of products over time. It implies a different approach to the functional unit and system boundary, since it takes into account "the set of units in service" (i.e. the stock), rather than a single unit or a functional unit, and explicitly introduces the notion of time, by integrating in the life-cycle model the dynamics associated with substituting older products by new products in the fleet (or the product stock). Instead of capturing a snapshot in time, it is able to account for changes over time in resource flows and environmental impacts, which is an important feature when assessing the environmental impacts of a technology transition. The life-cycle inventories obtained are different from traditional LCI, as process flows change over time.

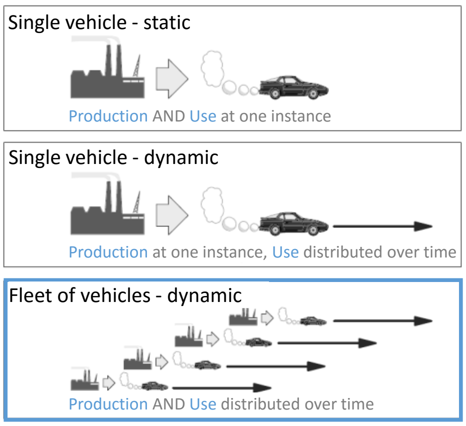

Fig. 2.1 Comparison between single-product and fleet-based LCA approaches (based on Kirchain 2002).

When assessing the environmental impacts of the introduction of a new technology, several dynamic aspects need to be considered: (i) the substitution of an old technology by a new one does not occur immediately but rather is distributed along time; (ii) because the new technology does not instantaneously gain 100% market share, production of older products may continue for some time; (iii) even if production ceases, the old technology does not abruptly disappear from use, especially if it has a long service life (Field et al. 2000; Levine et al. 2008). As a result, no steady-state condition exist: the number of products manufactured, in use and being disposed of change rapidly and in complex ways (Field et al. 2000; Levine et al. 2008). Moreover, technological improvements may continue to occur, probably at a higher rate for the new technology, but also for the old technology, as long as its market share is significant, and background processes are also likely to change. All these aspects can be captured by a dynamic fleet-based LCA, although the latter two were not originally addressed in the Field et al. (2000) article.

Comparisons between a baseline product and its alternative can be made either through an *ab initio* scenario (Fig. 2.2a), in which two separate fleets, one using the baseline product and the other using the alternative product, are assumed to grow at the same rate to a steady-state size; or a *displacement* scenario (Fig. 2.2b), which considers that the fleet of baseline products already in use is gradually displaced by the new product fleet (Field et al. 2000). The latter is more appropriate to assess the effects of introducing electric vehicles in the existing light-duty vehicle fleet.

2.1.3 Applications

2.1.3.1 Product comparison

Most fleet-based LCA studies focused on vehicle or vehicle components, and aimed at comparing the use of lightweight materials, such as aluminum versus steel in vehicle manufacturing (Cáceres, 2009; Das, 2000, 2005; Field et al., 2000; Stasinopoulos et al., 2011). Field et al. (2000) used two models of product fleet growth (exponential and logistics) and considered both *ab initio* and *displacement* scenarios to compare the CO_2 emissions of steel-intensive and aluminum-intensive vehicles. Although using a merely illustrative example, they found that it would take 10 years until the introduction of aluminum-intensive vehicles would result in a reduction of total CO_2 emissions, when considering the fleet, against 6.5 years if a conventional crossover analysis (i.e. comparing two products) would be performed. Cáceres (2009) built on Field et al. (2000) and considered also the mass efficiency of material substitution. Das (2000) presented an *ab initio* scenario and estimated both energy consumption and CO_2 emissions of aluminum versus conventional steel and ultralight steel car body-in-white, while Das (2005), using a similar approach, estimated energy consumption of an automotive liftgate inner. More recently, Stasinopoulos et al. (2011) combined LCA and system dynamics to compare the LC energy consumption of car bodies-in-white made from steel and aluminum. They used computation methods similar to Field et al. (2000) and Das (2000), but their model allowed for growth in vehicle size (similar to Das, 2000) and gradual adoption of the alternative product. In general, fleet-based studies suggest that it takes longer for the higher energy intensive production of aluminum components to be offset by fuel savings during the use stage, compared to product-based studies.

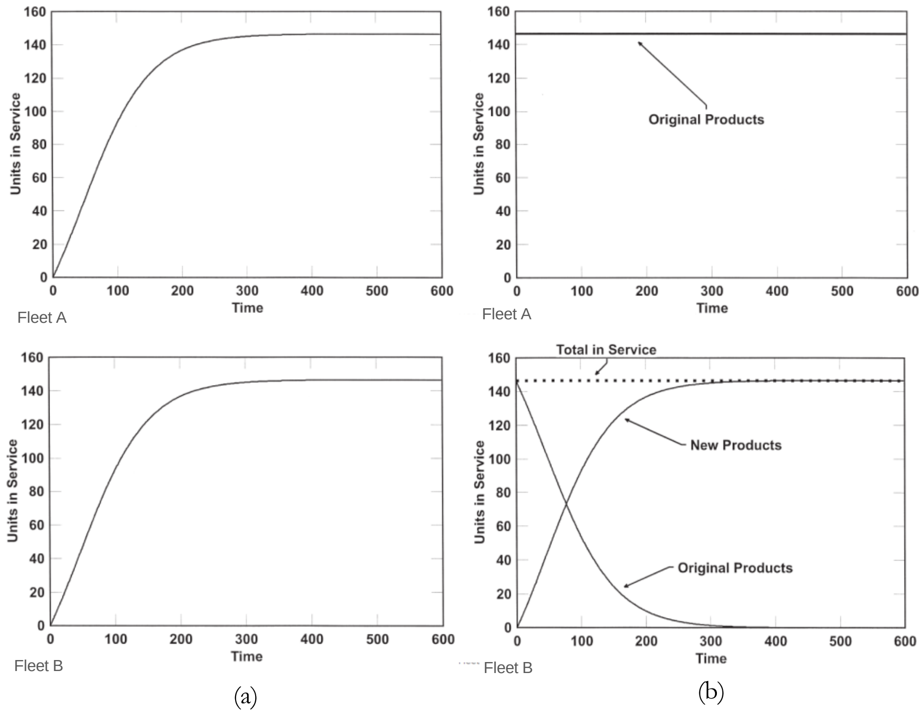

Fig. 2.2 *Ab initio* product scenario (a) and displacement scenario (b) (source: Field et al. 2000).[2]

[2] © Copyright 2001 by the Massachusetts Institute of Technology and Yale University.

2.1.3.2 Optimization of product service life

Other approaches that use a fleet-based LCA approach include the work of Kim et al. (2004; 2006) and De Kleine et al. (2011). They integrated fleet-LCA with dynamic programming to analyze the optimal service life of products and the effects of technology turnover on environmental performance. They analyzed tradeoffs between the fact that extending the service life of an existing product avoids the additional resource consumption and environmental impacts associated with the production of the new product and that, at the same time, the replacement of older, inefficient products with newer, more efficient ones reduces environmental impacts during the use phase. Kim et al. (2004) explored optimal fleet conversion policies based on mid-sized internal combustion engine vehicles in the USA, by modeling the lifetime emission profiles as functions of accumulated mileage. A similar approach was used to assess the optimal replacement policy for refrigerators (Kim et al. 2006) and air conditioners (De Kleine et al. 2011).

2.1.3.3 Assessment of impacts of products on a social scale

Yokota et al. (2003) also presented a fleet-based approach in which LCA and Population Balance Model were integrated to quantitatively assess the total environmental impacts induced by the product population of air conditioners in Japan over time. The model was found to be useful to set targets of product performance and in policymaking.

2.1.3.4 Modeling recycling processes

Fleet-based LCA can also be used to explore the generation of scrap material from the system and the implications on recycling, as it can track the flow and accumulation of materials over time (Field et al. 2000; Cheah 2010; Stasinopoulos et al. 2011). Considering the fleet rather than a single product as the unit of analysis simplifies the assessment of recycling processes as it allows for estimating the emergence and availability of scrap material and account for time dependencies in the rate of recovery and usage of the products, and avoids the simplifying assumptions required in a product-centered approach (Field et al. 2000). For instance, Stasinopoulos et al. (2011) used a fleet-based LCA within a system dynamics framework to compare the life-cycle energy consumption of steel and aluminum car bodies-in-white, incorporating two dynamic processes: the flow of car bodies-in-white into and out of the fleet, and the recycling of aluminum from end-of-life car bodies-in-white back into new car body-in-white production. They found that product-centered approaches underestimate the long-term energy benefits of aluminum components by not accounting for changes in the availability of recycled aluminum.

2.1.3.5 Environmental assessment of scenarios of evolution of the transportation sector

Fleet models coupled with life-cycle-based approaches have been used to assess the environmental impacts of alternative pathways of evolution of the transportation sector. These studies typically involve a scenario analysis and, in general, can be divided into two groups as regards its main objective: (i) those that assess the overall reduction in the environmental impacts achieved by implementing different technology/fuel pathways – *what if* analysis (e.g., Bandivadekar et al. 2008; Baptista et al. 2012; Bodek and Heywood 2008; Kromer et al. 2010; Reichmuth et al. 2013); and (ii) those that explore pathways that allow achieving certain policy targets (e.g., emission reduction; fuel economy improvements) – *backcasting* analysis (Cheah and Heywood 2011; Melaina and Webster 2011; Singh and Strømman 2013) – and its underlying uncertainty (Bastani et al. 2012b; Bastani et al. 2012c; Bastani et al. 2012a).

Although that is not always the main goal of the reviewed studies, *what if* analyses are more directed to the assessment of the effect of introducing alternative technologies in the fleet, because they allow to assess the overall environmental impacts resulting from different levels of penetration of a technology in different conditions and compute the change in emissions relative to a baseline scenario. *Backcasting* studies, on the other hand, aim at exploring the magnitude, combinations and timings of the changes required to meet the policy target and what are the implications and limitations of those pathways (Cheah and Heywood 2011; Melaina and Webster 2011). The introduction of alternative vehicle technologies can be one of the options analyzed in these studies (Cheah and Heywood 2011; Melaina and Webster 2011; Singh and Strømman 2013), but the main goal is to assess what the level of penetration of the new technology must be and not the effect of the decision of its deployment – the main focus of this research.

2.1.4 Fleet-based approaches addressing energy and environmental impacts of vehicle fleets

An overview of selected studies using a fleet-based LC approach to assess environmental impacts of light-duty vehicle fleets is presented in Table 2.1. The studies were selected based on three criteria: i) a fleet-based approach is used, ii) a life-cycle perspective (here understood in a broader sense, i.e. the studies selected assess more than just tailpipe emissions) is taken, and iii) electric vehicles (PHEVs and/or BEVs) are within the set of vehicle alternatives addressed in the scenarios of a *what if* analysis.

Most studies were performed for the USA or regions within the USA (EPRI 2007; Bandivadekar et al. 2008; Plotkin and Singh 2009; Kromer et al. 2010; Keoleian et al. 2011; Reichmuth et al. 2013) and the time scope addressed varied between 20 and 45 years. In general, the role of EVs in the reduction of the LC impacts of a vehicle fleet over time was only one of the many options assessed. A wide range of technologies was usually addressed, although some studies focused the analysis on PHEVs (EPRI 2007; Keoleian et al. 2011), but none specifically on BEVs. Baptista et al. (2012) analysis also considered heavy-duty vehicles. All studies assessed both energy consumption and GHG emissions (except Palencia et al. [2012], which assessed CO_2 emissions only), while some also accounted for other tailpipe emissions (EPRI 2007; Baptista 2011; Keoleian et al. 2011). A well-to-wheels perspective (i.e. excluding vehicle manufacturing and end-of-life) is the most common in the literature, and only a few studies have taken a full life-cycle approach (Bandivadekar et al. 2008; Baptista 2011).

Key aspects were addressed through scenario analysis, such as: alternative vehicle penetration rates (e.g., EPRI 2007; Bandivadekar et al. 2008; Plotkin and Singh 2009; Baptista 2011; Keoleian et al. 2011; Palencia et al. 2012); electricity grid evolution over time (e.g., EPRI 2007; Keoleian et al. 2011); technology improvements (e.g., Bandivadekar et al. 2008; Baptista 2011; Reichmuth et al. 2013); charging profile (e.g., Keoleian et al. 2011); and the effect of policies and prices (e.g., Plotkin and Singh 2009). Table 2.2 presents a more in depth analysis of key aspects of the selected studies, which are addressed in the following sections.

2.1.4.1 *Electricity modelling*

Some studies assumed a fixed electricity mix throughout the analysis (Bandivadekar et al. 2008; Plotkin and Singh 2009; Palencia et al. 2012), or performed a sensitivity analysis considering single technology scenarios (Reichmuth et al. 2013), or scenarios of grid decarbonization (Kromer et al. 2010; Baptista et al. 2012). Only few studies assessed the effect of the different electricity scenarios in the results (Kromer et al. 2010; Reichmuth et al. 2013) or modelled the electricity system as part of the fleet assessment (EPRI 2007; Keoleian et al. 2011), despite the fact that electricity generation is an important aspect of the environmental assessment of EVs (Samaras and Meisterling 2008; Frischknecht and Flury 2011; Freire and Marques 2012; Hawkins et al. 2013; Nordelöf et al. 2014a). EPRI (2007) and Keoleian et al. (2011) used an electric power capacity factor dispatch model to simulate retirement of existing generation capacity and additions of new capacity and to simulate how capacity is dispatched, considering different scenarios of evolution of the electricity system and different levels of penetration of PHEVs.

These studies showed that the electricity generation source had a large impact on the GHG emissions and increasing renewable energy penetration significantly reduced overall emissions (Keoleian et al. 2011; Reichmuth et al. 2013). The studies that addressed the effects of electricity emissions in the results used marginal emissions to assess impacts of electricity charging from EVs (except Kromer et al. [2010]), while all others assumed that EVs were charged with the average electricity mix.

2.1.4.2 Charging profile

The effect of charging time or the temporal variability in emissions was generally disregarded, with studies ignoring the charging profile of EVs (Bandivadekar et al. 2008; Kromer et al. 2010; Palencia et al. 2012; Baptista et al. 2012) or assuming a fixed profile (usually nighttime) (EPRI 2007; Plotkin and Singh 2009; Reichmuth et al. 2013). Only one study assessed different EV charging profiles (Keoleian et al. 2011) and found that these only modestly influenced GHG emissions (between 3.5% reduction and 1.6% increase in 2030 compare to the baseline charging), but suggested that the effect would increase as battery size increases (they only assessed PHEVs).

2.1.4.3 Fuel economy improvements and vehicle weight reduction

Fuel economy improvements in new vehicles was generally taken into account, but some studies only considered one scenario (EPRI 2007; Keoleian et al. 2011; Baptista et al. 2012) or did not consider improvements over time in alternative technologies (Keoleian et al. 2011; Reichmuth et al. 2013). Other studies addressed different scenarios of fuel economy improvements (Bandivadekar et al. 2008; Plotkin and Singh 2009; Kromer et al. 2010; Palencia et al. 2012; Reichmuth et al. 2013) and also assessed the effect of this parameter in the results (Kromer et al. 2010; Palencia et al. 2012; Reichmuth et al. 2013). Weight reduction (e.g., lightweighting) was implicitly considered in the fuel economy scenarios in Bandivadekar et al. (2008), Plotkin and Singh (2009), Kromer et al. (2010), and Palencia et al. (2012), and its effect was accounted for in the vehicle production impacts in Bandivadekar et al. (2008) and Palencia et al. (2012).

Reducing fuel consumption is one of the key ways to reduce fleet GHG emissions, but it needs to be combined with other measures, such as high penetration of alternative powertrains and biofuels, to bring about significant reductions (Bandivadekar et al. 2008; Kromer et al. 2010; Reichmuth et al. 2013). Reichmuth et al. (2013) assessed the influence of improving the fuel economy of gasoline ICEVs in the GHG emissions of the U.S. fleet until 2050 and found that increasing fuel economy reduces GHG emissions, but cannot achieve the target emission level unless efficiency reaches very high levels. Moreover, emissions eventually increase as vehicle growth surpass efficiency improvements. Kromer et al. (2010) found that efficiency improvements through weight reduction have higher effect in an ICEV-dominated fleet, than in a fleet dominated by highly efficient hybrid vehicles. Palencia et al. (2012) showed that vehicle weight reduction in BEVs only slightly reduced WtW emissions, due to the already high efficiency of BEVs.

2.1.4.4 Other aspects

Other key aspects addressed by these studies include biofuel use (Bandivadekar et al. 2008; Kromer et al. 2010; Reichmuth et al. 2013), and demand-side interventions, such as decrease in vehicle fleet size and travel (Bandivadekar et al. 2008; Kromer et al. 2010). Kromer et al. (2010) and Reichmuth et al. (2013) found that increasing biofuel use would contribute to long-term GHG emission reduction, but, due to resource availability constraints, this measure needs to be combined with others aiming at decreasing fleet energy demand, such as increasing vehicle fuel economy or introducing more efficient hybrid vehicles. However, these studies neglect land use change, which is an important and controversial issue in the LCA of biofuels (Malça and Freire 2012; Castanheira and Freire 2013; Castanheira et al. 2014).

Regarding demand-side interventions, Bandivadekar et al. (2008) showed that reducing fleet growth and travel demand have the potential to reduce fuel use by 19% and, if combined with high improvements in vehicle fuel economy, could achieve similar reductions to a scenario with high penetration of advanced vehicles (39%). The authors note that, as this measure affects all vehicles in the fleet, emission reduction happens sooner, resulting in higher cumulative emission reduction during the period of analysis. However, the contribution of demand-side reductions declines as the share of advanced technologies increases (Kromer et al. 2010).

2.1.4.5 Impact reduction potential of electric vehicles

Emission reductions are very dependent on the underlying assumptions of the study. Some studies only presented aggregated results for each scenario considered, which usually comprised a large number of assumptions, making it difficult to discern the main drivers of emission reduction (EPRI 2007; Baptista et al. 2012). Others, on the other hand, disaggregate results by main contributors, such as electricity mix, efficiency improvements, weight reduction, allowing a more in depth assessment.

In general, the latter studies show that the introduction of alternative technologies have the potential to significantly reduce fleet GHG emissions (Bandivadekar et al. 2008; Kromer et al. 2010; Keoleian et al. 2011; Reichmuth et al. 2013). Bandivadekar et al. (2008) reported that up to 31% reduction in fuel use and 24% in GHG emissions (of 40% and 35% of total fuel use and emission reduction potential, respectively) could be achieved with the introduction of alternative technologies in the U.S. fleet in 2035. Reichmuth et al. (2013) showed that in 2050, EVs could reduce emissions up to 61%, from a total of 77% reduction potential. Kromer et al. (2010) demonstrated that EVs could increase the reduction potential in 2050 relatively to 1990 levels by 29-42%. Keoleian et al. (2011) estimated that introducing PHEVs in California could reduce fleet GHG emissions by 0.4-10.9% in 2030.

2.1.5 Concluding remarks

Fleet-based life-cycle approaches have been applied with different purposes: to model recycling processes (Field et al. 2000; Stasinopoulos et al. 2011), to compare "products that are 'dirty' to make and 'clean' to use with products that are 'clean' to make and 'dirty' to use" (Field et al. 2000), to optimize product service life (Kim et al. 2004; Kim et al. 2006), and to assess the impacts of products in a social-scale (Yokota et al. 2003). When the purpose is to assess new or changing products or technologies, the results of a fleet-based LCA are generally different from a product-centered approach (Field et al. 2000; Stasinopoulos et al. 2011), as the number of products manufactured, in use and being disposed of changes rapidly and in complex ways over time and the overall effects of this dynamic behavior, captured in a fleet-based approach, are not accurately described by simple linear combinations of single-product life cycles (Field et al. 2000). The issue of evaluating the impacts of introducing alternative vehicle technologies is thus appropriately addressed by a fleet-based LCA approach. Such an approach provides a more comprehensive environmental assessment of the adoption of a new technology, because it enables explicit assessment of changes in technologies and background systems over time and provides the scale and timing for assessing other indirect impacts, such as the effects of displacing an older technology or the effects of changing the energy pathway.

Several studies have combined fleet models with life-cycle approaches to assess scenarios of evolution of the light-duty transportation sector, with emphasis on the U.S. fleet. Table 2.3 compares the reviewed articles and places this research into context. Most of the studies do not include all stages of the life cycle, frequently disregarding vehicle production and end-of-life impacts. Only few aimed at assessing in particular the effect of introducing a new technology in the fleet GHG emissions (the majority assessed alternative vehicle penetration as one options in many to reduce emissions). Several key aspects were identified and included in the scenario development, such as: fleet penetration rate; electricity source; fuel economy improvements; vehicle weight reduction. Nevertheless, the different studies dealt with these aspects with different degrees of comprehensiveness and their effects on the results were only occasionally assessed. The integration of all these aspects in the analysis of the potential of EVs to reduce fleet LC impacts has not been fully explored. This research aims to fill this gap by investigating the fleet-wide environmental benefits of displacing internal combustion engine vehicles (ICEVs) by BEVs, taking into account the full life cycle, and assessing the influence of vehicle weight reduction, fuel consumption reduction, electricity sources, fleet and travel demand growth rate, while considering changes in vehicle composition, battery weight and improvements in materials GHG intensity over time. A major focus of this research is on the life-cycle modeling of the electricity system and the interactions with BEV introduction. The next section reviews the application of LCA to electricity systems, focusing on both attributional and consequential approaches.

Table 2.1 Overview of selected studies using a fleet-based LC approach to assess environmental impacts of vehicle fleets.

Reference	Vehicle technologies	System boundary	Temporal scope	Geographical scope	Environmental impacts	Scenario variables
EPRI (2007)	ICEV, HEV, PHEV	Well-to-wheels	2010-2050	U.S.	GHG emissions, air quality impacts.	PHEV penetration; GHG intensity of the electricity sector.
Bandivadekar et al. (2008)	ICEV, HEV, PHEV, BEV, FCV	Full life cycle	2010-2035	U.S., Europe	Fuel consumption, GHG emissions.	Powertrain efficiency improvements; advanced technology penetration; fuel mix (e.g. by including biofuels); vehicle weight.
Plotkin and Singh (2009)	ICEV, HEV, PHEV, BEV, FCV	Well-to-wheels	2005-2050	U.S.	Oil use, GHG emissions.	Vehicle costs; fuel prices, government subsidies; and others.
Baptista et al. (2012)	ICEV, HEV, PHEV, BEV, FCHEV	Full life cycle	2010-2050	Portugal	Energy, CO_2 emissions, tailpipe emissions (HC, CO, PM, NO_x).	VKT, advanced technology penetration, energy source (biofuels, electricity generation mix).
Keoleian et al. (2011)	ICEV, PHEV	Well-to-wheels	2010-2030	Michigan, U.S.	Energy, GHG emissions, criteria air pollutant emissions (CO, Pb, NO_x, PM10, VOC, SO_x).	PHEV penetration; charging behaviors; future grid mixes.
Palencia et al. (2012)	ICEV, CNG, BEV, FCHEV	Well-to-wheels[a]	2010-2050	Colombia	Energy, CO_2 emissions.	Powertrain efficiency improvements; lightweighting.
Reichmuth et al. (2013)	IECV, PHEV, BEV, FCHEV	Well-to-wheels	2010-2050	U.S.	Petroleum consumption, GHG emissions.	ICEVs efficiency improvements; alternative powertrains penetration; electricity mix; hydrogen sources; biofuels utilization.
Kromer et al. (2010)	ICEV, HEV, PHEV	Well-to-wheels	2010-2050	U.S.	Petroleum consumption, GHG emissions.	Penetration of alternative technologies; vehicle weight reduction; biofuel feedstock; transportation demand.

[a] This study addresses vehicle production and end-of-life, but does not present results for CO_2 emissions regarding these life-cycle stages.

Table 2.2 Key aspects of the selected life-cycle fleet-based studies.

Reference	Electricity sector modeling	Charging profile	Fuel economy improvements	Vehicle weight reduction	Reduction potential	Key findings
EPRI (2007)	Marginal emissions. Three scenarios for the total GHG emissions intensity of the electric sector; The electricity sector model NESSIE was used to model the three scenarios.	Fixed charging profile (74% between10 PM and 6 AM and 26% between 6 AM and 10 PM).	Fuel economy of ICEVs and HEVs improve 0.5% per year; PHEV with same fuel economy as HEVs when in conventional mode.	Not considered.	3.4-10.3 Pg CO_2 eq (2012-2050); 163-612 Tg CO_2 eq (in 2050), depending on the PHEV penetration rate and scenario of evolution of the electricity system.	GHG emissions are reduced significantly across all scenarios.
Bandivadekar et al. (2008)	Average US grid mix with small changes in future scenarios.	No charging profile scenarios.	Different scenarios of fuel economy improvements.	Implicitly included in fuel economy scenarios.	Up to 40% reduction in fuel use and 35% in GHG emissions Up to 31% reduction in fuel use and 23% reduction in GHG emissions due to the introduction of alternative technologies.	Substantial potential to reduce fleet fuel use and GHG emissions exist. Reducing fuel consumption, weight reduction, high market share of advanced powertrains need to be realized.
Plotkin and Singh (2009)	Fixed electricity mix comprised of non-renewable sources only.	Nighttime charging.	Different scenarios of fuel economy improvements.	Implicitly included in fuel economy scenarios.	Up to more than 40% reduction in oil use; 13-47% reduction in GHG emissions (2050).	Advanced vehicle technologies will need a combination of factors to succeed: high oil prices; significant reductions in technology costs; and strong economic incentives for their purchase.

Reference	Electricity sector modeling	Charging profile	Fuel economy improvements	Vehicle weight reduction	Reduction potential	Key findings
Baptista et al. (2012)	Two scenarios for average annual electricity mix evolution: 60% and 91% RES in 2050.	No charging profile scenarios.	Fixed.	Not considered.	2-66% reduction in energy use; 7-73% reduction in GHG emissions (2010-2050); 4-29% reduction in energy use; 10-33% reduction in GHG emissions (2050).	Alternative vehicle technologies can help to lower impacts, but different deployments of alternative technologies may lead to similar impacts.
Keoleian et al. (2011)	Considers both average and marginal electricity generation and four electric grid scenarios, varying the amount of renewable generation added, the amount of nuclear capacity added and the number of retirements to existing generation assets.	Eight scenarios developed by varying charging timing, charging infrastructure and battery size.	Fixed (for ICEVs only).	Not considered.	0.4-10.9% reduction in GHG emissions (0.4-11 Tg CO_2 eq) (2030); 2-34 km^3 gasoline (2010-2030).	Introduction of PHEVs reduces GHG emissions and gasoline consumption; charging scenarios only modestly affected GHG emissions; increasing RES penetration and retiring old coal PP significantly reduced emissions.
Palencia et al. (2012)	Current Colombian mix.	No charging profile scenarios.	Conventional and lighweighting scenario.	Conventional and lightweighting scenario.	Up to 245 PJ; 19 Tg CO_2 (2050).	Switching to electric powertrains has larger impact than lightweighting on energy consumption and CO_2 emissions; Slow stock turnover and fleet size increment prevent larger reductions.

Reference	Electricity sector modeling	Charging profile	Fuel economy improvements	Vehicle weight reduction	Reduction potential	Key findings
Reichmuth et al. (2013)	Marginal emissions based on the Oak Ridge Competitive Electricity Dispatch Model (22% coal and 78% natural gas). Constant over time as default. Sensitivity analysis considering a linear transition from the default marginal mix to a single electricity source.	Nighttime charging.	Three scenarios for gasoline fuel economy improvements (2016 CAFE standards - 28 mpg; 2025 CAFE standards - 44 mpg, and 60 mpg in 2025).	Not considered.	23-77% reduction in GHG emissions (2010-2050); 44-61% due to EVs.	Efficiency improvements according to CAFE standards and alternative technologies operated with current electricity mix and hydrogen production processes alone will not reach the long term reduction target. A combination of efficiency improvements, biofuels and low-GHG fueled alternative technologies is necessary.
Kromer et al. (2010)	Two scenarios: base case and low-carbon grid (50% non-GHG emitting sources; 15% natural gas; and 35% coal).	No charging profile scenarios.	Two scenarios: base case and improvement due to additional weight reduction.	Included in efficiency improvement baseline scenario (20%). Additional scenario considering 35% weight reduction.	-10-65% reduction in GHG emissions (1990-2050); EVs increase reduction potential in 29-42%.	Changes to vehicle technologies comprise the higher reductions, namely regarding fuel efficiency, weight reduction, and high penetration of PHEVs. Improvements to the electric grid had only a small impact, due to the low penetration of PHEVs and low electric-range considered. The contribution of demand-side reductions declines as the share of advanced technologies increase.

Table 2.3 Comparison between reviewed literature regarding key aspects.

Reference	**System boundary**		**Alternative technologies**			**Electricity modeling**					**Fuel economy improvements**	**Vehicle weight reduction**	**Material production efficiency**
	Full LC	***WtW***	***BEV***	***PHEV***	***Others***	***Avg.***	***Marg.***	***Fixed***	***Scenarios***	***Modeling***			
EPRI (2007)		*		*			**		*	*	*		
Bandivadekar et al. (2008)	*		**	**	**	*		*			*	*	*
Plotkin and Singh (2009)		*	*	*	*	*		*			*	*	
Baptista et al. (2012)	*		*	*	*	*			*		*		*
Keoleian et al. (2011)		*		**		**	**		*	*	*		
Palencia et al. (2012)		*	*		*	*		*			**	**	
Reichmuth et al. (2013)		*	**	**	**		*		**		**		
Kromer et al. (2010)		*		**		*			**		**	**	
This research	*****		******			******	******		******	*****	******	******	*****

* included in the model; ** included in the model and influence on the results assessed.

2.2 Life-cycle assessment of electricity

2.2.1 Introduction

Electricity generation is considered to be one of the most complex systems to address in LCA (Curran et al. 2005). The electricity system is composed by a large set of power plants (including baseload, intermediate and peaking plants), and transmission and distribution infrastructures that generate and distribute electricity to consumers. Since electricity can hardly be stored, generation and consumption must be matched in real-time. The dynamic operation of the grid makes the mix of power plants and the emissions associated with electricity generation vary over time as demand and supply changes. Due to this complexity, linking a given supplier and corresponding emissions to a specific load is virtually impossible in most electricity systems, thus complicating the assessment of the environmental impacts of electricity generation and supply (Weber et al. 2010).

Two different main perspectives on how to perform LCA exist: attributional and consequential. Attributional LCA (hereinafter ALCA) aims at describing the environmentally relevant physical flows of a past, current or potential product system (Ekvall and Weidema 2004; Curran et al. 2005) and can be used to assess the environmental impacts of a product at a given point in time. When assessing electricity systems, ALCA can be used to assign the emissions of a generation mix to each consumption point in a specific timeframe, resulting in average emissions per kWh of electricity generated or consumed. Consequential LCA (hereinafter CLCA), on the other hand, describes how environmentally relevant physical flows would have been or will be changed in response to possible past, present or future decisions (Ekvall and Weidema 2004; Ekvall et al. 2005; Finnveden et al. 2009). CLCA can be used to assess how emissions from the electricity system change in response to a change in electricity demand, taking into account structural and operational changes to the system. Before exploring the application of ALCA and CLCA to electricity systems, a discussion on selected topics regarding the main differences between both approaches, which are considered relevant in the context of this thesis, is provided. A summary of the differences between ALCA and CLCA can be found in Brandão et al. (2014, Table S1) and a more complete review of the topic in Earles and Halog (2011) and Zamagni et al. (2012).

2.2.2 Attributional and consequential approaches

Most LCA studies published so far are attributional in nature. In contrast, the number of CLCA studies has only recently gained momentum (Zamagni et al. 2012), encompassing applications such as biofuels (e.g., Lemoine et al. 2010; Reinhard and Zah 2011), energy (e.g., Eriksson et al. 2007; Pehnt et al. 2008; Rehl et al. 2012), and food products (e.g., Dalgaard et al. 2007; Thomassen et al. 2008).

The debate on the usefulness and applicability of ALCA and CLCA is still ongoing (Weidema et al. 2009; Finnveden et al. 2009; Zamagni et al. 2012; Anex and Lifset 2014). The opinions are diverging (as recently demonstrated by the several "Letters to the Editor" of the Journal of Industrial Ecology about Plevin et al. [2014] article), but many authors consider both approaches to be relevant depending on the question at stake (e.g., Zamagni et al. 2012; Brandão et al. 2014; Hertwich 2014; Suh and Yang 2014). ALCA is typically used for hotspot identification, product declarations and for generic consumer information (Tillman 2000). On the other hand, CLCA is deemed more appropriate for decision-making and policy development (Tillman 2000; Plevin et al. 2014; Brandão et al. 2014), because it addresses indirect effects not captured in ALCA, such as substitution and rebound effects (Brandão et al. 2014). Both ALCA and CLCA can be used to assess impacts of product systems in the past (retrospective analysis), present and future (prospective analysis) (Curran et al. 2005).

Ekvall et al. (2005) argues for the complementarity in the information provided by both CLCA and ALCA. Brandão et al. (2014) and Guinée (2016) also suggest that CLCA and ALCA may have a complementary role in policy: CLCA in policy development, and ALCA in policy implementation and monitoring, and guidance of consumer choices. Although it appears to be consensual that the consequential model is the most appropriate for decision-makers, who are concerned with making choices, it is also suggested that a decision-maker has first to identify the major contributors, which is an attributional problem (Heijungs et al. 2007). This discussion leads to what is still an open topic of research in LCA: how to identify the type of questions that are more appropriately answered by ALCA and CLCA (Zamagni et al. 2012), or simply how to choose the suitable model (or combination of models) to the given question (Suh and Yang 2014).

The different approaches of ALCA and CLCA are reflected in several methodological choices (Tillman 2000). The definition of the system boundary, namely the selection of processes to include in the assessment, is one of them (Weidema 2003; Zamagni et al. 2012). ALCA includes all relevant flows from raw material extraction to waste management (cradle-to-grave system). On the other hand, the system boundary in CLCA is defined to include only the activities contributing to the environmental consequences of a change (marginal or affected processes), regardless of whether these are within or outside the cradle-to-grave system of the product investigated (Tillman 2000; Finnveden et al. 2009). There are, however, still open questions about which type of process(es) and marginal effects should be included in CLCA and how to identify them (Zamagni et al. 2012). In some CLCA studies, one single marginal supplier is identified (Schmidt and Weidema 2008), whereas in others, economic models are used to project market-mediated effects (Kløverpris et al. 2008; Dandres et al. 2012). The role of scenario modeling in enhancing the value of the study and reducing the uncertainty in the choice of the marginal supply should also be explored (Brandão et al. 2014).

Other differences between ALCA and CLCA include the type of data used and how co-products are handled. ALCA typically utilizes average data for each unit process within the life-cycle. Average data represent the average environmental burdens of producing a unit of the product in the system. On the other hand, CLCA uses marginal data for the purpose of assessing the consequences of a change in the life-cycle (Ekvall and Weidema 2004). Marginal data represent the effects of a small change in the output of products from a system on the environmental burdens of the system. The handling of co-products in CLCA is performed by using system expansion, therefore avoiding allocation (Weidema 2003). In ALCA, allocation of impacts is often the method applied (Thomassen et al. 2008); however, system expansion may also be used, contrary to what some authors advocate (e.g., Weidema 2014). In fact, system expansion is the recommended method to solve multifunctionality issues since the first ISO standards for LCA (ISO 1998). The major difference between ALCA and CLCA approaches to system expansion lies on the type of data used (average versus marginal) (Finnveden et al. 2009) and on the identification of an actual displacement (ALCA) as opposed to a theoretical displacement (CLCA) (Brandão et al. 2014).

Although it is possible to find several ALCA studies on electricity (e.g., Weisser 2007; Varun and Prakash 2009; Santoyo-Castelazo et al. 2011; Mallia and Lewis 2012), the number of studies that systematically applied CLCA to electricity systems is much lower (e.g., Mathiesen et al. 2009; Pehnt et al. 2008; Lund et al. 2010). The next section reviews the application of ALCA and CLCA to electricity systems, focusing on the geographical and temporal issues of ALCA and on the type of questions addressed and the approaches used to identify the marginal process(es) or technology(ies) in CLCA.

2.2.3 Attributional LCA applied to electricity systems

ALCA has been widely applied to assess the environmental performance of electricity systems, including LCAs of single electricity generation technologies (e.g., Odeh and Cockerill 2008; Pehnt et al. 2008; Lenzen 2008; Martínez et al. 2010; Nishimura et al. 2010; Desideri et al. 2012; Sastre et al. 2014; Yang and Chen 2014; Thakur et al. 2014), electricity transmission and distribution (e.g., Harrison et al. 2010; Bumby et al. 2010; Jones and McManus 2010; Jorge et al. 2011a; Jorge et al. 2011b; Jorge and Hertwich 2013; Turconi et al. 2013), and country or region electricity mixes (e.g., Weber et al. 2010; Ou et al. 2011; Santoyo-Castelazo et al. 2011; Mallia and Lewis 2012; Garcia et al. 2014).

Electricity consumption is often a key driver of environmental impacts, be it in the production phase or in the use phase of products and services. Two main dimensions are important when assessing the environmental impacts of electricity: temporal and spatial. Several studies have assessed the emissions associated with electricity supply and demand in different regions and times. Geographical and temporal aspects associated with the ALCA of electricity are discussed in the following section.

2.2.3.1 Geographical and temporal aspects of the electricity mix

Geographical scope

Geographical variations in electricity generation occur at the energy source (e.g., availability of renewables, origin and type of fossil fuels), and technology (e.g., efficiency, capacity factor) levels (Weber et al. 2010). The choice of the geographical boundary (i.e. whether a smaller or larger region is selected for the electricity generation mix) is thus an important issue in ALCA studies. The use of large boundaries tends to mask the heterogeneity between regions. For instance, Weber et al. (2010) assessed CO_2, SO_2, and NO_x emissions of electricity in the U.S. considering three distinct levels of spatial disaggregation of the grid system and found significant differences in the results obtained. Colett et al. (2015) developed a new method for allocating GHG emissions from electricity to consumers in the U.S. and applied it to the primary aluminum industry. They found GHG emission factors that were significantly different than other studies using different regional mixes and inter-regional trading assumptions.

Most databases rely on aggregate national statistics to model electricity generation and supply. Using these figures may introduce additional uncertainty in the assessment of impacts from electricity supply because they may not represent the physical boundaries of the grid system. For instance, in Portugal, data used to model electricity generation in ecoinvent 3.0 database relies on the International Energy Agency (IEA) database (IEA 2011), which aggregates electricity data from continental Portugal and the islands (Madeira and the Azores), whose grid systems are physically disconnected and have quite different compositions. Moreover, considering or disregarding electricity trade between regions (imports and exports) may also have an important effect in the environmental impacts of electricity consumption (for instance, in Norway, Switzerland, Slovakia, and Austria) (Soimakallio and Saikku 2012). Depending on the goal of the study, using the smallest region from a system operation and data availability perspectives (Yang 2013) and including electricity imports and exports (Soimakallio and Saikku 2012) may be recommended.

Temporal scope

When assessing impacts from electricity consumption in ALCA studies, a common assumption is to use average national (or regional) statistics to calculate emissions factors for electricity generation (Weber et al. 2010). However, using annual average mix figures in ALCA may not be the most appropriate production mix in many cases. The annual national mix can vary significantly from year to year due, for instance, to changes in electricity demand, technology portfolio, hydro availability, and net imports (Soimakallio et al. 2011). For example, in Portugal, GHG emissions from the annual electricity mix varied 26% between 2012 and 2014 (see Section 3.1). Therefore, the reliability and applicability of the results of LCA studies to reflect the situation for other years can be diminished if a single GHG emission factor is used. This is particularly important for products systems that have a long service life, such as vehicles (10-15 years), or buildings (e.g., 50 years).

Moreover, when using annual figures, the variability within the year is lost. The difference between annual and shorter periods may be highly relevant when assessing emissions from processes that do not consume electricity continually throughout the year (i.e., that use electricity mainly or exclusively during certain hours of the day, such as a company operating during daytime, or that use electricity during a particular time of the year, like air-conditioning in the summer) (Soimakallio et al. 2011), or that can vary their time of consumption, such as EVs (Faria et al. 2014; Rangaraju et al. 2015). However, only a few LCA studies have addressed the temporal variability in electricity. Messagie et al. (2014b) calculated GHG emissions of electricity generation in Belgium per kWh for each hour in 2011; however, demonstrating the effect of considering hourly emissions in LCA results was not the aim of the research. Spork et al. (2015) developed a method to calculate hourly electricity GHG emissions based on real-time data for Spain. They showed that the use of hourly emission factors can significantly improve the accuracy of the GHG emissions that are attributed to a company's electricity consumption (for a company operating during the day, about 5-9% difference in hourly emissions compared to the annual value in 2012 was found).

2.2.4 Consequential LCA applied to electricity systems

Table 2.4 presents examples of consequential-based LCA studies applied to electricity systems. These studies were selected based on two criteria: (i) the study is change-oriented; and (ii) electricity is an important factor for the results. The questions addressed in CLCA studies of electricity systems are generally related to the assessment of the change in emissions due to a change in electricity demand (e.g., increase in electricity consumption due to EVs) (Lund et al. 2010) or in the availability of a certain fuel or technology to generate electricity (Pehnt et al. 2008; Mathiesen et al. 2009). The change in emissions due to a change in electricity demand or supply depends on which plant is providing the power, i.e. the marginal technology. Since the environmental impacts of electricity generation depend on a mix of technologies, which is highly variable through the day (peak versus off-peak) and time of year (seasonal differences), the identification of the marginal technology is not obvious (Lund et al. 2010; Soimakallio et al. 2011; Siler-Evans et al. 2012; Zivin et al. 2014). Moreover, different types of marginal effects can be included and various methods have been proposed to identify them, as discussed in the following section.

Table 2.4 Consequential-based electricity case-studies.

Reference	Application	Question(s) addressed/purpose of the study	Consequence(s)/Change(s) addressed	Temporal scope	Approach to determine marginal supply
Mathiesen et al. (2009)	Increase in waste incineration	(i) To study the uncertainties and simplifications involved in identifying the marginal energy technology in CLCA studies: (ii) to assess the net change in electricity generation of the technologies affected by an increase in waste incineration	Affected technologies (electricity and heat) due to a 10% increase in waste incineration	Long-term	**Scenario analysis** (use of different future energy scenarios); **Energy system analysis** (use of the EnergyPLAN model)
Lund et al. (2010)	Consumption of marginal electricity in Denmark	To determine the long-term yearly average marginal electricity production and calculate the corresponding environmental impacts	Marginal production changes due to an increase in electricity demand in the Danish energy system in 2030.	Long-term	**Energy system analysis**: identification of the "long-term yearly average marginal (YAM) technology", which depends on the marginal capacities (long-term effects) and the marginal supply (short-term effects), using the EnergyPLAN model (Lund, 2007) to identify the hourly affected technologies
McCarthy (2009)	Operation of advanced vehicles (PHEVs, BEVs, FCVs) in California	To determine the marginal electricity mix and GHG emissions associated with operating advanced vehicles (PHEVs, BEVs, FCVs)	Changes in the operation of the California grid in response to added vehicle and fuel-related electricity demand	Short-term and long-term	**Merit order-based approach:** Electricity-dispatch model of California
Pehnt et al. (2008)	Increase in offshore wind capacity in Germany	To determine the environmental impacts resulting from the introduction of extra offshore wind capacity in the German power system until 2020	Change in operation of conventional electricity system due to an increase in wind offshore capacity	Long-term	Detailed bottom-up LCA model coupled with a stochastic European electricity market model

Reference	Application	Question(s) addressed/purpose of the study	Consequence(s)/Change(s) addressed	Temporal scope	Approach to determine marginal supply
Hawkes (2010)	Micro-CHP and air source heat pumps	(i) To calculate marginal CO_2 emission factors (MEF) for Great Britain (no LCA approach); (ii) To assess the contribution of commissioning and decommissioning of power plants; (iii) To project MEFs for the next 5 to 15 year and assess uncertainties; (iv) To apply the methodology to micro-CHP and air source heat pumps case-studies.	CO_2 reduction resulting from demand-side interventions (micro-CHP and air source heat pumps)	Short-term	**Empirical approach**: regression of historic data (calculates linear regression coefficients of change in the system CO_2 rate versus change in total system demand)
Siler-Evans et al. (2012)	Demand-side interventions (efficiency measures in lighting)	To calculate MEFs for CO_2, NO_x, and SO_2, and the correspondent share of coal-, gas-, and oil-fired generators, for the U.S. electricity system (no LCA approach)	Avoided emissions resulting from supply- and demand-side interventions	Short-term	**Empirical approach:** regression of historic data (based on Hawkes [2010])
Zivin et al. (2014)	PHEV charging	To estimate hour-of-day marginal emission rates for CO_2, NO_x, and SO_2, for the U.S. electricity system, accounting for electricity trade within regions (no LCA approach)	Change in emissions resulting from an increase in electricity demand due to the charging of plug-in electric vehicles	Short-term	**Empirical approach:** regression of historic data (regression of hourly emissions at grid interconnection level on hourly electricity consumption for subsets of regions within the U.S., taking into account the generation mix within the interconnected electricity markets and the shifting load profiles through the day)
Eriksson et al. (2007)	District heating (waste incineration, biomass combustion, natural gas combustion)	(i) To compare the environmental consequences of district-heat production from waste and competing fuels (biomass, natural gas) in Sweden; (ii) to test a combination of dynamic energy system modeling and LCA for a decision making purpose.	Scenario analysis: two options for energy recovery (combined heat and power (CHP) or heat only), two alternatives for external, marginal electricity generation (fossil lean or intense), and two alternatives for the alternative waste management (landfill disposal or material recovery).	Short-term	**Dynamic optimizing model (NELSON)** used to identify the marginal technologies for electricity production. Electricity scenario independently developed outside the LCA study.

Reference	Application	Question(s) addressed/purpose of the study	Consequence(s)/Change(s) addressed	Temporal scope	Approach to determine marginal supply
Hawkes (2014)	Heat pumps	(i) To develop a methodology for estimating long-term marginal emission factors that takes account of structural and operational effects in the electricity system; (ii) to investigate the long-run marginal emissions factor associated with the electrification of heating in Britain.	The response of the model in terms of capacity addition to serve the additional heat pump demand, and the change in electricity output from the system as a result of those changes.	Long-term	Optimization model based on TIMES
Raichur et al. (2015)	Operation of EVs	(i) To develop a modeling approach for estimating emissions from nuclear, coal, natural gas, and hydropower generators in reaction to short-term changes to electricity demand, taking into account a set of operating constraints; (ii) to demonstrate the use of model to estimate marginal CO_2 emissions associated with the electricity consumption during the use of EVs	Change in electricity emissions due to an increase in demand by EVs.	Short-term	Dispatch model including operating constraints

2.2.4.1 Identification of the marginal electricity supply

Temporal scope

The response of the system to a change in demand will vary between the short- and the long-term. In the short-term, the system will respond by changing the utilization of the existing production capacity in existing power plants (Weidema et al. 1999). The analysis focuses on which plants will increase generation to meet the additional electricity demand, which can vary significantly in time. In the long-term, changes in electricity demand will likely influence the timing and nature of new power plant investments and/or the decommissioning of old plants (Curran et al. 2005; Soimakallio et al. 2011). The long-term effects involve changes in the production capacity and/or technology, but can also lead to effects on its operation (Soimakallio et al. 2011; Yang 2013).

Constrained and unconstrained technologies

Only power plants that can change their operation or capacity in response to a change in demand (i.e. unconstrained technologies) can be (part of) the marginal electricity supply. If the production capacity of a technology is fixed, it cannot be the long-term marginal supply; if its production volume is fixed, it cannot be the short-term marginal supply. Technologies can be constrained by: natural constraints, political constraints, quotas, emission limits, or the demand for co-products (Ekvall and Weidema 2004; Yang 2013).

In the long-term, the new capacity installed is likely to be the one which satisfies the new load curve at the lowest long-term costs (Curran et al. 2005). However, investment decisions on new capacity are affected by several factors concerning the market evolution (e.g., investment costs, fuel prices) and policy measures regulating emissions (Soimakallio et al. 2011). Nevertheless, the long-term marginal supply will not necessarily be fully produced at such capacity, but will likely involve a mix of different technologies (Lund et al. 2010).

In the short-term, the marginal supply is likely the technology with the highest variable cost operating at the time of the change in demand. If the increase in demand is higher than the operation capacity of that plant, another plant is brought online. The marginal supply is typically a dispatchable fossil-based generator, but can vary significantly over time (between day and night or winter and summer) and between regions (Lund et al. 2010; Siler-Evans et al. 2012). In the U.S., coal is the dominant marginal fuel source in some regions and natural gas in others, but oil-fired power plants can also supply marginal electricity (Siler-Evans et al. 2012).

A response to a change in demand might involve the use of hydro power. On an hourly basis, hydro reservoir plants are often dispatched to meet daily peaks rather than baseload, which means they could be on the margin; however, in general, on an annual basis, hydro may be considered an energy-constrained resource, as only a fixed amount of water is available annually (Curran et al. 2005; McCarthy and Yang 2010). If pumped hydro storage is in place, as is the case of most hydro reservoir plants in Portugal, there is a higher degree of flexibility in the use of water to generate electricity, but in general, the system tends to maximize total production over a long period not effecting overall emissions (Dotzauer 2010).

Due to its low operation cost, nuclear power plants are usually used for baseload and are rarely on the (short-term) margin. Solar and wind power plants rarely alter generation as a result of additional demand, given their lack of load-following ability and low variable cost. Solar and wind are thus constrained technologies in the short-term (i.e. their output will be fully utilized irrespective of the additional demand) (Yang 2013). However, in the long-term, renewables may be included in the marginal supply. For instance, wind capacity expansion could be enabled by increasing the penetration of EVs, as their charging flexibility may improve the economics of wind investments (Yang 2013).

Review of approaches

Different approaches have been presented regarding how to identify the marginal electricity supply, depending on the type of marginal effects considered. The first efforts to identify affected technologies were based on a heuristic approach (e.g., Weidema et al. 1999; Weidema 2003; Ekvall and Weidema 2004), which can be generically applied to any system. This five-step procedure to identify the marginal technology, originally presented by Weidema et al. (1999), encompasses: (i) the definition of the time horizon; (ii) the identification of the competing products in the markets affected; (iii) the identification of the general market trend; (iv) the identification of the technologies on the market capable of responding to changes in demand; and, finally, (v) the identification of the marginal technology. Weidema et al. (1999) and Weidema (2003) provided several examples to demonstrate the application of the procedure to various markets, including agricultural, metals, energy, forest-based, and plastics. Nevertheless, the application of this procedure in

CLCA studies of energy systems was found to be non-systematic and inconsistent and the simplifications involved increased the uncertainty in the identification of the marginal technology (Mathiesen et al. 2009; Zamagni et al. 2012). Moreover, Mathiesen et al. (2009) and Lund et al. (2010) pointed out that the application of this procedure in electricity systems ignores the dynamics of the system, since it is assumed that the marginal supply will be fully produced by one long-term marginal technology, identified by comparing the costs of different technologies.

Traditionally, the marginal electricity production technology in CLCA has been assumed to be either coal or natural gas (Weidema 2003). However, by using dynamic optimization models (Eriksson et al. 2007; Pehnt et al. 2008) and energy system analysis simulation tools (Mathiesen et al. 2009; Lund et al. 2010), other authors showed that the marginal source of electricity is a mixture of different technologies using different fuels resulting from the constraints and dynamics of the electricity system. In particular, Lund et al. (2010) proposed an approach to identify the "long-term yearly average marginal (YAM) technology", which depends on the marginal capacities (long-term effects) and the marginal supply (short-term effects), by using the EnergyPLAN model (Lund 2007) to identify the hourly affected technologies. McCarthy (2009) developed an electricity-dispatch simulation tool for California for both the short-term (EDGE-CA model) and the long-term energy scenarios (LEDGE-CA model) to identify marginal power generation and GHG emissions associated with operating advanced vehicles (PHEVs, BEVs, FCVs).

These merit order-based approaches (i.e. models that assume that the order of dispatch is defined by the cost of operation of each generator) can be used to identify the technologies on the margin and estimate annual, monthly or time-of-day marginal emission factors for the short-term and long-term (McCarthy 2009; Mathiesen et al. 2009; Hawkes 2010; Lund et al. 2010; Hawkes 2014). Nevertheless, they do not capture other factors influencing dispatch, such as logistics of plant operation, transmission constraints, plant availability, etc. (Hawkes 2010; Graff Zivin et al. 2014). Raichur et al. (2015) developed an approach to estimate emissions from electricity generation in reaction to short-term changes in demand taking into account a number of operation constraints and found that these were important to achieve good estimates of the system behavior and associated emissions.

Other authors suggested the use of empirical approaches to the estimation of marginal emission factors (MEFs) based on regression of historical data which implicitly account for operation constraints (Hawkes 2010; Siler-Evans et al. 2012; Graff Zivin et al. 2014). Hawkes (2010) developed a method for estimating marginal emission factors for Great Britain for use in determining the CO_2 reduction performance of demand-side interventions. The author calculated linear regression coefficients of change in the system CO_2 emission rate versus the change in total system demand (Hawkes 2010). Based on this approach, Siler-Evans et al. (2012) calculated MEFs for CO_2, NO_x and SO_2 for the U.S. and further estimated the share of marginal generation from coal-, gas-, and oil-fired generators. Zivin et al. (2014) took Siler-Evans et al. (2012) calculations further by accounting for the effects of electricity trade within U.S. regions.

The empirical approaches to derive MEFs, though applied in studies not LCA related, were shown to be valid for determining the short-term marginal technologies, without requiring sophisticated simulation models (Hawkes 2010; Siler-Evans et al. 2012; Graff Zivin et al. 2014). Nevertheless, since CLCA studies are generally used for decision-support related to decisions that have long-term implications, the long-term marginal supply needs to be assessed. A methodology to project MEFs into the future taking into account both structural and operations effects in the electricity system was developed by Hawkes (2014). He used a TIMES-based cost optimization model of the Great Britain electricity system to simulate commissioning and decommissioning of power plants in response to a change in demand (large scale installation of heat pumps) against a baseline scenario. Long-term MEFs were calculated by dividing the change in emissions over the entire time horizon (2010-2050) by the change in demand over the same period. He found that the long-term MEF was lower than the short-term MEF in Britain.

The estimation of the long-term marginal effects involve many uncertainties (Mathiesen et al. 2009; Itten et al. 2012; Zamagni et al. 2012). Scenario modeling could provide an important role in modelling multiple product-related futures regarding technology development, new investments and others, and in minimizing uncertainty from the choice of the marginal supply (Mathiesen et al. 2009; Zamagni et al. 2012; Brandão et al. 2014).

2.2.5 Concluding remarks

The assessment of the environmental impacts associated with electricity generation and consumption can be performed using both ALCA and CLCA, depending on the goal and scope of the study. Nonetheless, the application of each approach presents some challenges. For ALCA, choosing the appropriate mix from a geographical and temporal perspective is a key issue that may significantly influence results. For a product with a relatively long service life and a variable electricity-consumption profile, like EVs, an approach with a temporal resolution that is able to capture the coincidence between electricity use and generation (Lund et al. 2010; Hawkes 2014) requires the use of high-resolution electricity data not always available. From a geographical perspective, depending on the goal of the study, it may be recommended that the smallest region from a system operation and data availability viewpoints is selected (Yang 2013) and that electricity imports and exports are included (Soimakallio and Saikku 2012).

Regarding CLCA, a number of approaches to model the marginal technology(ies) in electricity systems has been presented in the literature. While some were applied in a LCA context (Pehnt et al. 2008; e.g.,, Mathiesen et al. 2009; Lund et al. 2010), others were developed to evaluate the benefits of demand-side interventions and account for direct emissions only (Hawkes 2010; Siler-Evans et al. 2012; Graff Zivin et al. 2014). Different effects require different methods: for assessing short-term effects, changes to the operation of the current electricity system need to be assessed, and operational constraints should be accounted for (Hawkes 2010; Raichur et al. 2015). For the assessment of long-term consequences, both operational and structural changes need to be taken into account (Lund et al. 2010; Hawkes 2014). Nevertheless, a systematic approach to consequential modeling of electricity systems has not been developed yet. The role of scenario modeling in that context is also a topic for which further research is needed (Zamagni et al. 2012).

Chapter 3

DYNAMIC FLEET-BASED LIFE-CYCLE ASSESSMENT OF THE PORTUGUESE LIGHT-DUTY FLEET[3]

Abstract The adoption of a new technology entails changes to the existing system. One of the effects of EV adoption is the displacement of conventional technologies, which is addressed in this chapter through dynamic fleet-based LCA. A dynamic fleet-based life-cycle model of the Portuguese light-duty fleet was developed and is described in Section 3.1. Scenarios of evolution of the light-duty fleet in Portugal are developed and assessed, in Section 3.2, and the parameters that influence the most the results identified through sensitivity analysis. The fleet-based LCA implemented provides the scale and timing for assessing other indirect impacts, such as the effects of EV load in the power grid, addressed in Chapter 5.

3.1 Dynamic fleet-based life-cycle modeling of the Portuguese light-duty fleet

3.1.1 Introduction

A dynamic fleet-based life-cycle (LC) model of the Portuguese light-duty fleet was developed, following the principles of fleet-based LCA highlighted in Section 2.1.2. The main dynamic features of the model include: (i) the flow of vehicles in and out of the fleet; (ii) changes in vehicle technologies over time (fuel/electricity consumption, vehicle and battery weight, distance traveled); and (iii) changes in background processes over time (electricity generation, material production energy intensity). The model spans from 1995 to 2030 and allows for the assessment of the effects on environmental impacts (with focus on GHG emissions) of displacing internal combustion engine vehicles (ICEVs) by electric vehicles in Portugal within a range of scenarios. This section describes the dynamic fleet-

[3] Significant portions of this chapter appear in: Garcia R., Gregory J., Freire F. (2015). Dynamic fleet-based life-cycle greenhouse gas emissions of the introduction of electric vehicles in the Portuguese light-duty fleet. *Int J Life Cycle Assess* 20(9): 1287-1299. http://dx.doi.org/10.1007/s11367-015-0921-8

based life-cycle model, including the LC parameters, data sources and main assumptions, and highlights the main features of the model as well as its main limitations.

3.1.2 Dynamic fleet-based life-cycle model

3.1.2.1 Model overview

A dynamic fleet-based life-cycle (LC) model was developed to assess fleet-wide LC greenhouse gas (GHG) emissions over time, from 1995 to 2030. The model integrates: (i) a vehicle stock sub-model of the Portuguese light-duty fleet; and (ii) dynamic life-cycle sub-models of three vehicle technologies (gasoline ICEV, diesel ICEV and BEV). Fleet-wide impacts in each year are a combination of the impacts of single vehicles and the number of vehicles in the fleet across all ages and technologies. Fig. 3.1 shows an overview of the model, including the main inputs and outputs.

The vehicle stock sub-model estimates the annual stock of vehicles by technology, the age of vehicles in the fleet, and the number of vehicles, by age, that leave the fleet every year, from 1995 up to 2030. The dynamic LC sub-models were developed for three vehicle technologies: diesel ICEV (~67% market share in 2010), gasoline ICEV (~33%), and battery EV (BEV) (0.01%). The vehicle LC was divided into three main stages: (i) production, (ii) use and (iii) end-of-life, and modeled the emissions from these stages as functions of vehicle age and model year, which makes these LC models dynamic. The vehicle manufacturing stage includes raw material acquisition, transportation, and processing, as well as parts and components manufacturing and vehicle assembly. The use stage accounts for vehicle operation (tailpipe and tire abrasion emissions) and maintenance, as well as fuel, and electricity production and distribution. The end-of-life stage accounts for vehicle and battery dismantling, recycling, and disposal of components. Road infrastructure, refueling stations for ICEVs, and charging points for EVs were excluded from the assessment, as their contribution to the impacts is deemed to be minor (Lucas et al. 2012).

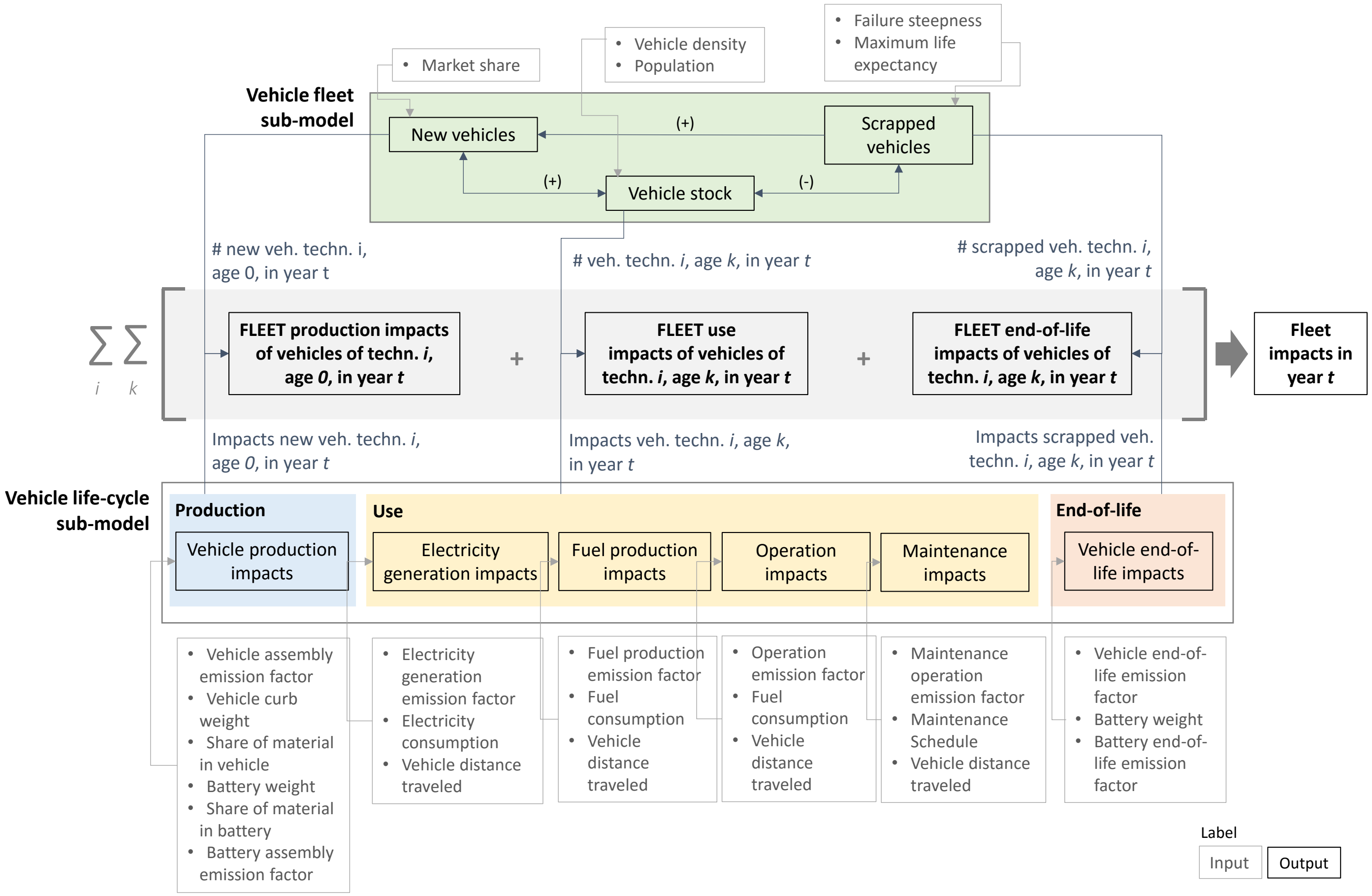

Fig. 3.1 Dynamic fleet-based life-cycle model overview. *i*: technology (gasoline, diesel, BEV); *k*: vehicle age; *t*: calendar year. Positive causal link + the two variables change in the same direction; negative causal link − the two variables change in opposite directions.

The vehicle stock sub-model is based on the U.S. passenger vehicle fleet model developed by Bandivadekar et al. (2008) and improved by Cheah (2010), and was adapted to the Portuguese context following the work of Moura (2009). The model was further developed to include different vehicle technologies, i.e. electrical engines (BEVs), in addition to internal combustion engines (gasoline and diesel), and was parameterized for the specific analysis undertaken.

3.1.2.2 Vehicle stock sub-model

The vehicle stock sub-model tracks the number of vehicles in use in the Portuguese light-duty fleet, by technology (i) and age (k), from 1995 to 2030 (t). LDVs up to 25 years old and three technology types – gasoline ICEV (g), diesel ICEV (d), and BEV (e) – were considered (note that BEVs only started to be sold in Portugal in 2010). Details about the model equations, parameters, and data source can be found in Table 3.1 and Fig. 3.2.

The total fleet turnover is expressed as the number of vehicles in the fleet in the previous year subtracted by the number of scrapped vehicles and adding the number of new vehicles entering the stock. The total vehicle stock was calculated by multiplying the vehicle density (i.e. the number of vehicles per 1000 inhabitants) by the population in each year. The vehicle density for Portugal was estimated by calibrating a logistic curve based on vehicle data from ACAP (2011) and demographic data from PORDATA (2011), for the time period between 1974 and 2010 (r^2=0.998). Population projections were obtained from INE (2009). Fig. 3.3 shows the estimated vehicle stock over time. The number of LDVs being driven in Portugal currently exceeds 4.5 million, 3 times more than in 1990. Vehicle density increased from about 163 to 422 vehicles per 1000 inhabitants in the same period.

Table 3.1 Vehicle stock sub-model parameters and data sources. Dashed cells indicate parameters that are subject to sensitivity analysis. n.a.: not applicable; Y: yes; N: no; *i*: vehicle technology (*g*: gasoline ICEV; *d*: diesel ICEV; *e*: BEV); *k*: vehicle age; *t*: calendar year.

Variable		Parameters		Units	Historical data source ($t \leq 2010$)	Projected data source ($t > 2010$)	Time dependence	Baseline value (i) *g*	*d*	*e*
N(*i,0,t*)	**New vehicles**									
		q(*i,t*)	market share	%	ACAP (1999; 2003; 2005; 2011)	Calculated based on assumption	Y [a]	33	67	0
S(*i,k,t*)	**Scrapped vehicles**									
		λ(*t*)	failure steepness	n.a.	Moura (2009)	Assumption	Y	See Fig. A-2		
		μ(*t*)	maximum life expectancy	years	Moura (2009)	Assumption	Y	See Fig. A-2		
		p(*k,t*)	probability of surviving	n.a.	Calculated (intermediate parameter)	Calculated (intermediate parameter)	Y	See Fig. A-2		
F(*i,k,t*)	**Vehicle stock**									
		d(*t*)[b]	vehicle density	Vehicles per 1000 inhabitants	Calculated (intermediate parameter)	Calculated (intermediate parameter)	Y	See Fig. A-1		
		θ	vehicle density multiplier	n.a.	n.a.	Assumption	N	1		
		n(*t*)	population	inhabitants	PORDATA (2011)	INE (2009)	Y	See Fig.7 in INE (2009)		

[a] Constant from 2010 for baseline scenario.

[b] d(*t*) follows a logistic curve calibrated based on vehicle data from ACAP (2011) and demographic data from PORDATA (2011).

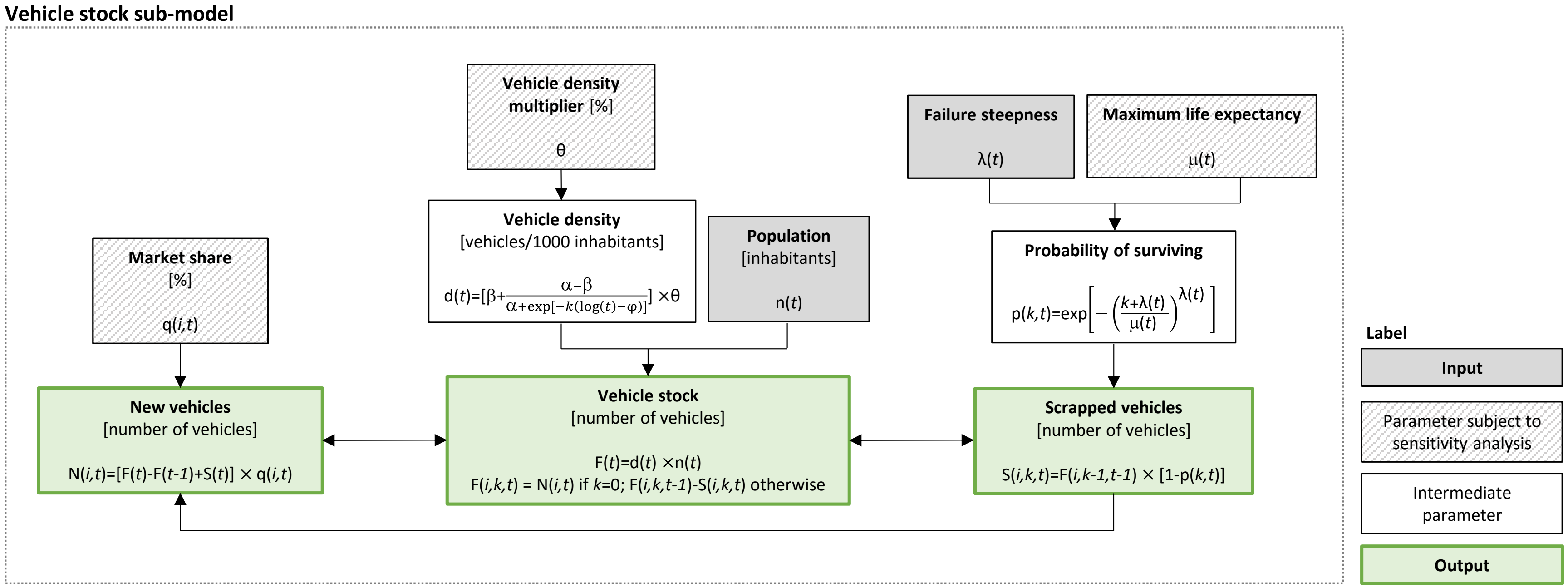

Fig. 3.2 Vehicle stock sub-model description (*i*: vehicle technology; *k*: vehicle age; *t*: calendar year). All parameters are defined in Table 3.1. The vehicle density is described by a logistic curve, which was calibrated using vehicle and demographic data for Portugal for the period between 1974 and 2010 (r^2=0.998) (see Fig. A-1, in Appendix III); the probability of surviving is described by a modified Weibull distribution, calibrated for Portugal conditions based on Moura (2009) (see Fig. A-2, in Appendix III).

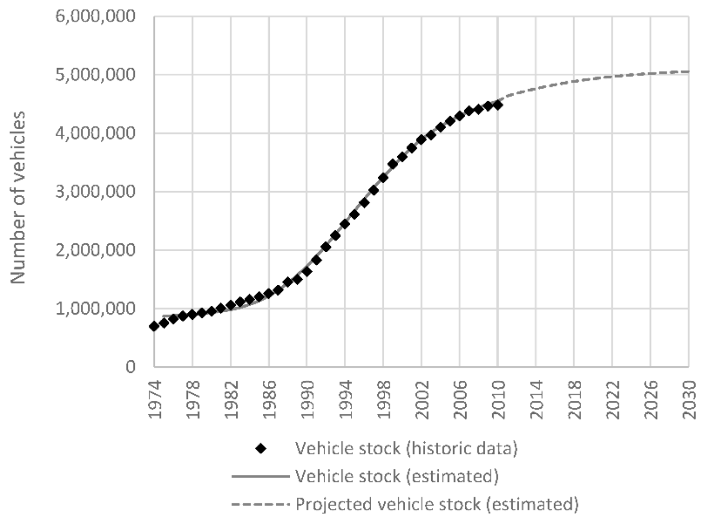

Fig. 3.3 Portuguese light-duty vehicle stock.

Vehicle scrappage was estimated by using a modified Weibull distribution, which characterizes the survival rate of vehicles in the fleet as a function of vehicle age. The calibration of the survival curve for Portuguese conditions done by Moura (2009) for model years 1995, 2000, and 2005 was used. The same survival curve from 2005 onwards and a similar curve for all vehicle types were assumed. The simulation started with the characterization of the Portuguese vehicle fleet composition (age and technology distribution) in 1995 used in Ceuster et al. (2007) and depicted in Table A-1, in Appendix III. Total vehicle sales were derived from the accumulated vehicle stock. The number of new vehicles of each technology was calculated by multiplying the total sales by its market share. Over 2005 to 2010, about 220,000 to 235,000 new vehicles entered the fleet each year, while 115,000 to 195,000 older vehicles were retired annually.

3.1.2.3 Dynamic life-cycle sub-model

Dynamic life-cycle sub-models for the three vehicle technologies (diesel ICEV, gasoline ICEV, and BEV) were developed, taking into account vehicle production, use and end-of-life. Table 3.2 presents the model parameters, and data sources. More details on the modeling of the different life-cycle stages is presented next and additional information can be found in Appendix III.

Table 3.2 Vehicle life-cycle sub-model parameters and data sources. Dashed cells indicate parameters that are subject to sensitivity analysis. n.a.: not applicable; Y: yes; N: no; i: vehicle technology (g: gasoline ICEV; d: diesel ICEV; e: BEV); k: vehicle age; t: calendar year; a: material; m: maintenance operation.

Variable	Parameters		Units	Historical data source ($t \leq 2010$)	Projected data source ($t > 2010$)	Time dependence	Baseline value (i)		
							g	***d***	***e***
$\mathbf{I_p(\mathit{i,0,t})}$	**Vehicle production impacts**								
$\mathbf{I_{av}}$	**Vehicle assembly impacts**								
	e_v	vehicle assembly emission factor	kg CO_2 eq $vehicle^{-1}$	Keoleian et al. (2012)	Keoleian et al. (2012)	N	938		
$\mathbf{I_v(\mathit{i,0,t})}$	**Vehicle manufacturing impacts**								
	$w_v(i,0,t)$	vehicle curb weight	kg $vehicle^{-1}$	European Commission (2012)	Calculated (intermediate parameter)	Y	See Fig. A-3		
	ν	vehicle curb weight reduction rate	%	n.a.	Bandivadekar et al. (2008)	N	0		
	$r_v(i,0,t,a)$[a]	share of material in vehicle	%	Cheah (2010)	Cheah (2010)	Y	See Fig. 6-2 (a) and Fig. 6-3 in Cheah (2010)		
	$e_m(t,a)$[b]	material production emission factor	kg CO_2 eq kg^{-1} material	Keoleian et al. (2012); Cheah (2010)	Keoleian et al. (2012); Cheah (2010)	Y	See data sources		
$\mathbf{I_b(\mathit{i,0,t})}$	**Battery manufacturing impacts**								
	$w_b(i,0,t)$[c]	battery weight	kg	Faria et al. (2014)	USABC (2014)	Y	n.a.	n.a.	See Table A-2
	$r_b(i,0,t,a)$	share of material in battery	%	Dunn et al. (2012)	Dunn et al. (2012)	N	n.a.	n.a.	See Table 2 in Dunn et al. (2012)
	ω	battery weight reduction rate	%	n.a.	Assumption	N	n.a.	n.a.	0
$\mathbf{I_{ab}}$	**Battery assembly impacts**								
	e_b	battery assembly emission factor	kg CO_2 eq $battery^{-1}$	Dunn et al. (2012)	Dunn et al. (2012)	N	n.a.	n.a.	0.457

Variable	Parameters		Units	Historical data source ($t \leq 2010$)	Projected data source ($t > 2010$)	Time dependence	Baseline value (i)		
							g	***d***	***e***
$I_u(i,k,t)$	**Vehicle use impacts**								
$I_e(i,k,t)$	**Electricity generation impacts**								
	$e_e(t)$	electricity generation emission factor	kg CO_2 eq kWh^{-1}	Garcia et al. (2014)	Calculated based on assumption	Y		See Fig. A-9	
	$c_e(i,k,t)$	electricity consumption	kWh km^{-1}	Faria et al. (2014)	Calculated (intermediate parameter)	Y	n.a.	n.a.	See Fig. A-5
	$\varepsilon(i)$	electricity consumption reduction rate	%	n.a.	Assumption	N	n.a.	n.a.	0
$I_f(i,k,t)$	**Fuel production impacts**								
	$e_f(i)$	fuel production emission factor	kg CO_2 eq kg^{-1} fuel	Jungbluth (2007)	Jungbluth (2007)	N	0.729	0.523	n.a.
	ι	fuel production emission factor rate of change	%	n.a.	Assumption	N	0	0	n.a.
	$c_f(i,k,t)$[d]	fuel consumption	kg km^{-1}	European Commission (2012)	Calculated (intermediate parameter)	Y	See Fig. A-4		n.a.
	$\varphi(i)$	fuel consumption reduction rate	%	n.a.	Assumption	N	0	0	n.a.
	$j(i,k,t)$	vehicle distance travelled	km	Calculated (intermediate parameter)	Calculated (intermediate parameter)	Y			
	$y(i,0,t)$[e]	first-year vehicle distance travelled	km	t=2005: Azevedo (2007); other years calculated	Calculated (intermediate parameter)	Y		See Table A-4	
	$\varrho(i,t)$	first-year vehicle distance travelled growth rate	%	Calculated based on assumption	Calculated based on assumption	Y		See Table A-5	
	$x(i,k)$	indexed-mileage	n.a.	Calculated based on data from Azevedo (2007)	Same as $t \leq 2010$	Y		See Fig. A-6	
	$\zeta(i)$	n.d.[f]	n.a.	Calculated based on data from Azevedo (2007)	Same as $t \leq 2010$	N	-0.313	-0.32	-0.313
	$\sigma(i)$	n.d.[g]	n.a.	Calculated based on data from Azevedo (2007)	Same as $t \leq 2010$	N	1.4173	1.3623	1.4173

Variable	Parameters		Units	Historical data source ($t \leq 2010$)	Projected data source ($t > 2010$)	Time dependence	Baseline value (i)		
							g	*d*	*e*
$I_o(i,k,t)$	**Operation impacts**								
	$e_o(i)$	operation emission factor	kg CO_2 eq kg^{-1} fuel	Moura (2009)	Moura (2009)	N	3.1856	3.1375	n.a.
$I_m(i,k,t)$	**Maintenance impacts**								
	$e_m(i,m)$	maintenance operation emission factor	kg CO_2 eq $operation^{-1}$		See Table A-3				
	$m(i,k,t,m)$	maintenance schedule	n.a.		See Table A-3				
$I_l(i,k,t)$	**End-of-life impacts**								
I_{lv}	**Vehicle end-of-life impacts**								
	e_{lv}	vehicle end-of-life emission factor	kg CO_2 eq $vehicle^{-1}$	Keoleian et al. (2012)	Keoleian et al. (2012)	N		278	
$I_{lb}(i,k,t)$	**Battery end-of-life impacts**								
	e_{lb}	battery end-of-life emission factor	kg CO_2 eq kg^{-1} battery	Faria et al. (2014)	Faria et al. (2014)	N	n.a.	n.a.	390

[a] The main changes are related to the substitution of cast iron and conventional steel by lightweight materials such as high-strength steel, aluminum, and plastics.
[b] Emission factors for t=2000 from Keoleian et al. (2012), evolution according to Cheah (2010).
[c] Assuming Li-ion battery pack energy density increases from 80 Wh kg^{-1} today (24 kWh capacity) to 235 Wh kg^{-1} in 2020 (45 kWh capacity), and constant thereafter, according to USABC (2014).
[d] Since fuel consumption in real-world conditions is considerably higher than measured in test-cycles, mainly due to the use of energy consuming devices such as air conditioners, a 17% increase in real-world consumption factors compared with test-cycle figures was assumed, according to Nemry et al. (2008). Density of gasoline 0.748 kg L^{-1}; Density of diesel: 0.837 kg L^{-1}.
[e] First yr mileage by powertrain (gasoline and diesel) for 2005 (Azevedo, 2007); since BEVs are about 70% more energy efficient than gasoline ICEVs, a higher VKT was assumed in order to account for the expected rebound effect, in line with Silva (2011).
[f] Slope of indexed mileage curve.
[g] Constant parameter of indexed mileage curve.

Vehicle production

Fleet environmental impacts of vehicle manufacturing in each year were determined by the sum of manufacturing impacts of all new vehicles entering the fleet. The environmental burdens of vehicle manufacturing include vehicle materials and assembly burdens. Vehicle and battery material burdens are proportional to vehicle curb weight and battery weight, which varies with model year. Because impacts from vehicle production are accounted for in the year the vehicles are produced, they are independent of vehicle service life. Fig. 3.4 details the calculation of vehicle production impacts. Additional information about vehicle and battery weight data can be found in Appendix III.

Material composition of ICEVs was assumed to change over time according to Cheah (2010). The main changes are related to the substitution of cast iron and conventional steel by lightweight materials such as high-strength steel, aluminum, and plastics. Material composition of BEVs and batteries was assumed constant. Iron, steel, aluminum, and magnesium material production (i.e. extraction and processing) was assumed to become more energy-efficient and less GHG intensive over time (evolution according to Cheah 2010). Regarding other materials, energy use and GHG emissions was assumed constant over time. Energy intensity and GHG emissions from 1995-1999 were assumed equal to 2000.

Vehicle use

The use of the vehicle includes both vehicle and fuel life cycles. Use stage burdens are a function of vehicle distance travelled, fuel consumption, and emission factors. The use stage fleet impacts in each year result from the sum of use-related impacts from all vehicles in the fleet. These include impacts from fuel production and distribution, electricity generation and distribution, vehicle operation, and maintenance (see Fig. 3.5 for more details about the calculation of vehicle use impacts).

Environmental impacts of fuel production and distribution include resource extraction, initial conversion of petroleum, transport of petroleum, fuel production, and distribution of gasoline and diesel. GHG emissions from gasoline and diesel production were obtained from Jungbluth (2007) and assumed constant over time, primarily due to a lack of information on how these emissions would evolve. A sensitivity analysis to assess the effect of a change in the fuel supply chain performance over time on the overall fleet GHG emissions was performed.

Environmental impacts of electricity generation and distribution include extraction, processing and transport of fuels, operation of power plants, construction and decommissioning of power plants, waste management, transmission and distribution (T&D) grid infrastructure, and T&D grid losses. GHG emissions from electricity generation and supply in Portugal were obtained from Garcia et al. (2014) (Chapter 4). The average of the emission factors in 2003-2012 (485 g CO_2 eq kWh^{-1}) was used as a constant value up to 2030, in order to account for the variability between years. Variations of this emission factor were assessed in the sensitivity analysis. In this assessment it was assumed that EVs did not influence grid emissions, that is, an attributional approach is used. The potential indirect effects of EV charging in the grid are investigated in Chapter 5.

Environmental impacts of vehicle operation (combustion phase) include direct tailpipe and tire abrasion emissions. The operation emission factor is assumed constant and estimated based on the carbon content of the fuel as being fully oxidized into CO_2 (Moura 2009). Environmental burdens from maintenance are a function of the cumulative distance traveled. Maintenance operations are performed according to Table A-3, in Appendix III. It was assumed that fuel and electricity consumption remain constant over the life of the vehicle, because there is little evidence that the effect of vehicle deterioration and defective maintenance on fuel consumption can be generalized to the vehicle population (Austin and Ross 2001). More details about vehicle fuel consumption assumptions can be found in Appendix III.

The distance travelled by a vehicle varies depending on a number of factors, such as vehicle age (due to deterioration, reduced reliability, and shifting of primary to secondary car usage [Kim 2003; Moura 2009]), technology (diesel vehicles tend to be driven more than gasoline vehicles), and utilization purpose. Annual vehicle distance traveled estimations were based on vehicle inspection data for Portugal for 2005 from Azevedo (2007), and Azevedo and Cardoso (2009). Different vehicle distance traveled profiles for gasoline and diesel ICEVs were estimated. For BEVs the same profile as gasoline ICEVs was assumed; however, since BEVs are about 70% more energy efficient than gasoline ICEVs, a higher distance travelled was assumed in order to account for the expected rebound effect, in line with Silva (2011). More details about vehicle distance traveled assumptions can be found in Appendix III.

Vehicle end-of-life

Fleet environmental impacts of vehicle end-of-life in each year were determined by the sum of end-of-life impacts of all scrapped vehicles leaving the fleet. The environmental burdens of vehicle end-of-life include the dismantling of the vehicle and the battery. The energy use of materials that are recycled and later used in a vehicle are taken into account in the burdens for each specific material. Fig. 3.6 details the calculation of vehicle end-of-life impacts.

3.1.3 Concluding remarks

The dynamic fleet-based LC model developed allows for the assessment of the environmental impacts of displacing internal combustion engine vehicles (ICEVs) by EVs over time, taking into account the full life cycle. Because of the parameterization undertaken, it enables the assessment of the effect of different model parameters in the results, such as vehicle weight reduction, fuel consumption reduction, electricity sources, and fleet and travel demand growth rate, while considering changes in vehicle composition, battery weight, and material production over time. Although it was originally set to assess GHG emissions, it allows for estimating other environmental impacts, provided that emissions factors for those impact categories are inputted in the model.

The focus of this research is on the dynamics involving the technological system, and thus dynamic impact assessment is not addressed here. The dynamic life-cycle inventory model developed can, nevertheless, be used, with little modifications, to assess GHG emission impacts using impact characterization functions, such as the ones developed by Kendall (2012).

An average vehicle was used to represent each technology in the assessment without disaggregating by engine size or other metric. Although different vehicle sizes may have considerably different environmental impacts, such disaggregation would significantly increase the computational and data gathering effort. With such disaggregated data it would be possible to assess trade-offs regarding the displacement of smaller versus larger vehicles, but such detailed assessment is out of the scope of this thesis.

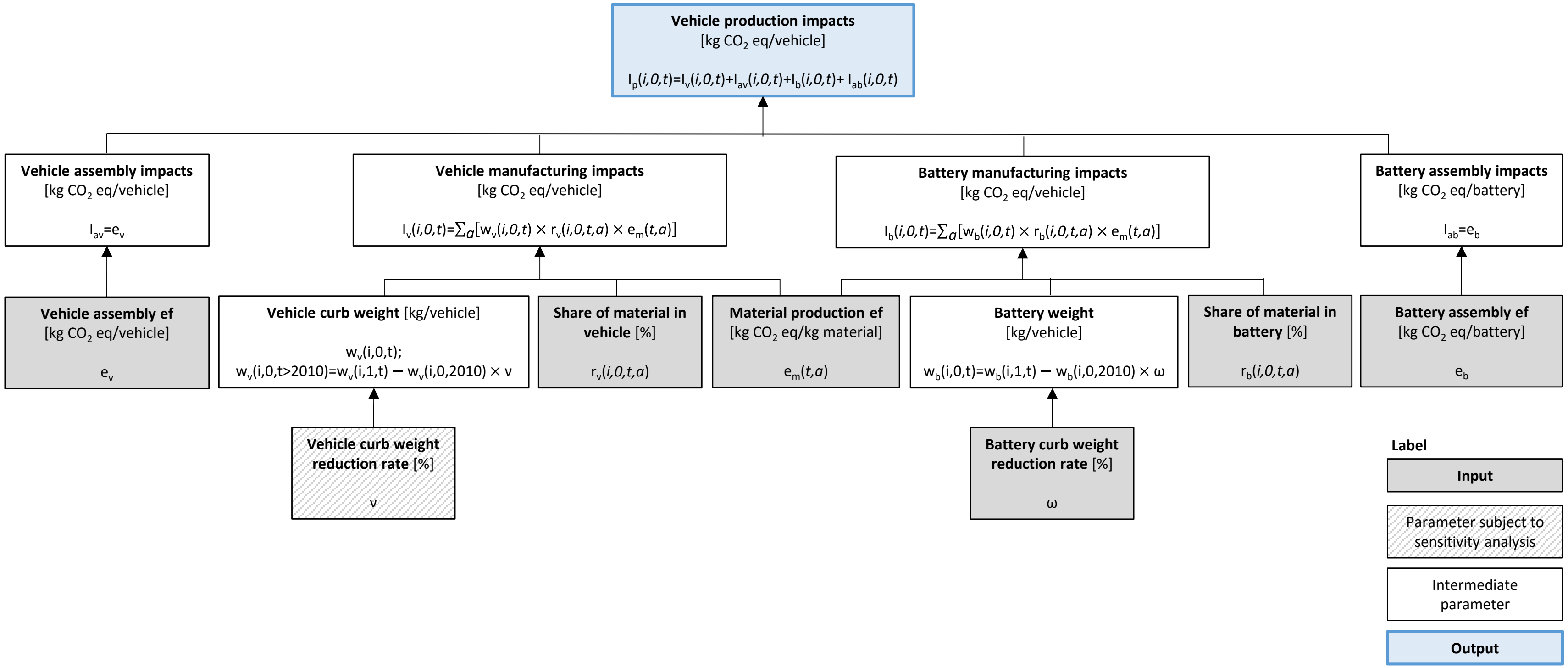

Fig. 3.4 Calculation of vehicle production impacts (*i*: vehicle technology; *k*: vehicle age; *t*: calendar year; *a*: material). All parameters are defined in Table 3.2. ef: emission factor.

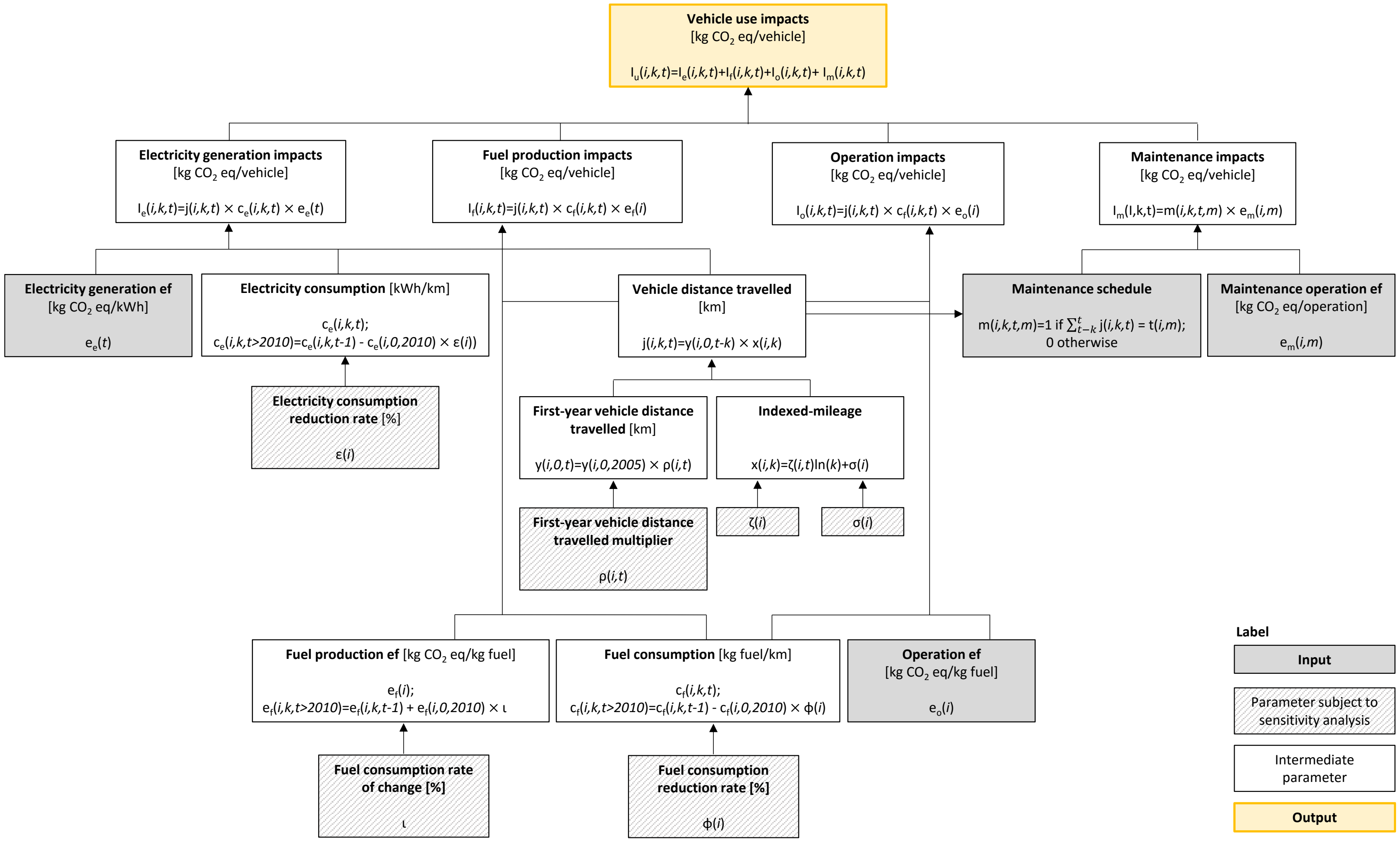

Fig. 3.5 Calculation of vehicle use impacts (*i*: vehicle technology; *k*: vehicle age; *t*: calendar year; *m*: maintenance operation). All parameters are defined in Table 3.2. ef: emission factor.

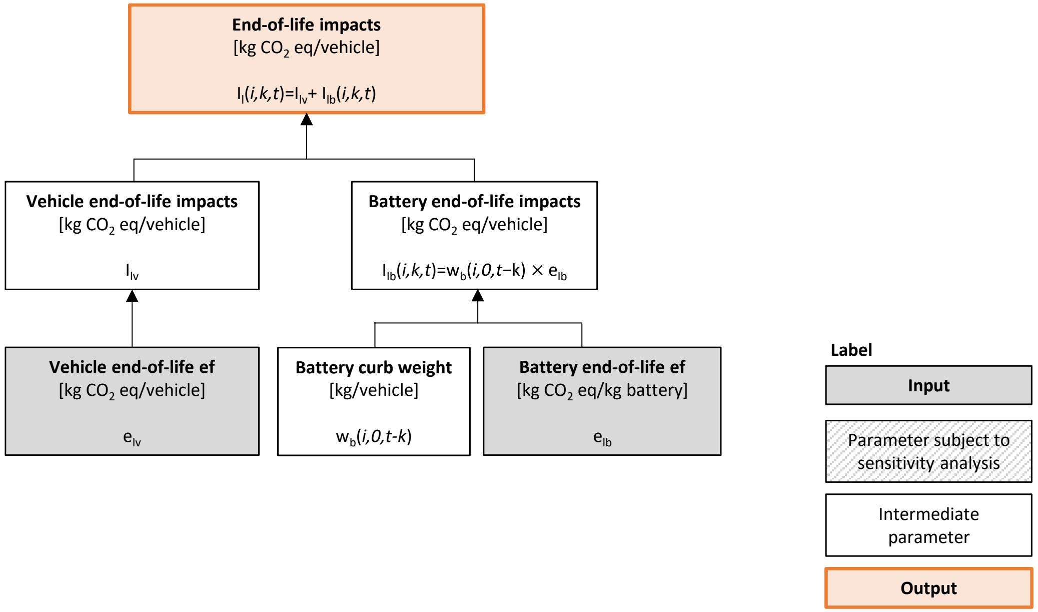

Fig. 3.6 Calculation of vehicle end-of-life impacts (*i*: vehicle technology; *k*: vehicle age; *t*: calendar year). All parameters are defined in Table 3.2. ef: emission factor.

3.2 GHG emissions of the introduction of electric vehicles in the Portuguese light-duty vehicle fleet[4]

3.2.1 Introduction

The fleet-wide environmental benefits of displacing internal combustion engine vehicles (ICEVs) by EVs across different scenarios is investigated in this section, using the dynamic fleet-based LC model presented in Section 2.1. The analysis takes into account the increasing fuel consumption reduction of ICEVs and the necessary reductions in the electricity mix impacts, within different fleet penetration scenarios, fleet and distance travelled growth rates, and changes in vehicle weight and composition and battery technologies over time. In particular, the aim of this section is to assess whether displacing ICEVs by EVs in the Portuguese light-duty fleet is environmentally beneficial, taking into account the dynamic behavior of the fleet. It also aims to identify the conditions under which this displacement is beneficial. The range of conditions were defined by a set of parameters: electrical grid intensity, EV fleet penetration, and reduction in ICEV fuel consumption.

3.2.2 Scenarios of electric vehicles penetration and technology improvements

Options for reducing LDV GHG emissions include adoption of alternative powertrains, such as BEVs, and technology improvements, such as vehicle lightweighting and efficiency improvements. Possible combinations of these options were explored by constructing four scenarios: (i) Business-as-usual (BAU), in which ICEVs continue to dominate the fleet (constant diesel/gasoline ICEV market share), but no new vehicle technology improvements occur; (ii) ICEV improve, characterized by improvements on fuel consumption of new ICEVs to meet EU targets and vehicle lightweighting; (iii) BEV dominate, in which the emphasis is on the aggressive introduction of BEVs in the fleet, reaching 100% of vehicle sales in 2030, and no improvements in ICEVs take place; and (iv) Combined, which associates BEV aggressive penetration and ICEV improvements.

[4] Significant portions of this section appear in: Garcia R., Gregory J., Freire F. (2015). Dynamic fleet-based life-cycle greenhouse gas emissions of the introduction of electric vehicles in the Portuguese light-duty fleet. Int J Life Cycle Assess 20(9): 1287-1299. http://dx.doi.org/10.1007/s11367-015-0921-8

Characterization of each scenario is shown in Table 3.3, in contrast with the 2010 fleet. It should be noted that, although all parameters are kept constant in 2010-2030 in the BAU scenario, the fleet size and composition do not remain constant due to the dynamic evolution of the fleet.

Table 3.3 Scenario description.

Scenarios	Market share by powertrain (q)			Technology improvements		
	Gasoline ICEV	Diesel ICEV	BEV	Vehicle weight reduction rate per year (ν)	Fuel consumption reduction rate per year (ϕ)	BEV battery weight reduction rate per year (ω)
2010	33%	67%	0.01%	--	--	--
2030						
Business-as-usual	33%	67%	0%	0%	0%	--
ICEV improve	30%	70%	0%	0.8%	2.5%	--
BEV dominate	0%	0%	100%	0.8%	--	1.9%
Combined	0%	0%	100%	0.8%	2.5%	1.9%

A rapid shift from gasoline to diesel ICEVs has recently occurred in Portugal. In 1995 only 10% of all LDV sold were diesel-powered, compared with 68% in 2010. Market share of BEVs was only 0.01% in 2010. The BAU scenario assumes the same market share as 2010 and the ICEV improve scenario that 30% of new vehicles are gasoline ICEVs and 70% diesel ICEVs, following recent trends, as depicted in Fig. 3.7A. The BEV dominate and Combined scenarios represent rapid penetration of BEVs assuming that its market share reaches 100% in 2030, following an S-shaped curve (Fig. 3.7B). In the BAU and ICEV improve scenarios, 42% of the fleet in 2020 is gasoline ICEVs and 58% is diesel ICEVs, and only in 2030 does the sales fraction match the fleet composition, as shown in Fig. 3.7C. The BEV dominate and Combined scenarios lead to a fleet similar to the ICEV scenarios in 2020 (60% diesel ICEVs, 36% gasoline ICEVs and 4% BEVs), and composed of 36% diesel ICEVs, 16% gasoline ICEVs and 48% BEVs in 2030, as depicted in Fig. 3.7D.

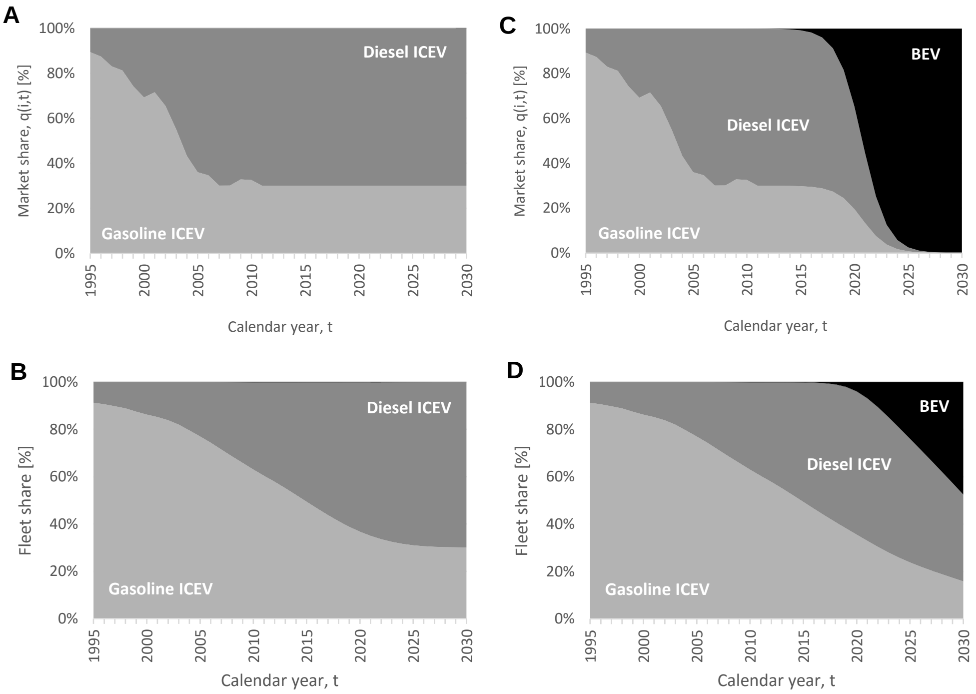

Fig. 3.7 Market share and fleet share of vehicle technologies in the Portuguese light-duty fleet for the *Business-as-usual (BAU)/ICEV improve* (A and B, respectively) and *BEV dominate/Combined* (C and D, respectively) scenarios in 1995-2030. 1995-2010 data were retrieved from ACAP (2011).

3.2.3 Output metrics

Fleet-wide impacts up to 2030 were assessed using two metrics: (i) Total fleet life-cycle (LC) GHG emissions (in Mton CO_2 eq); and (ii) Fleet LC GHG emissions per km (in g CO_2 eq km^{-1}). The first metric addresses the societal concern of reducing global GHG emissions. The second metric addresses the viewpoint of the policies that aim at reducing GHG emissions from LDVs by targeting specific emissions (e.g., per km) of new vehicles or fleets of new vehicles. Examples of these policies are the European Union (EU) legislation, which set binding emission targets for new vehicle fleets, and the Corporate Average Fuel Economy (CAFE) standards in the United States (US), which aims at improving the fuel economy of new vehicles sold in the US, indirectly reducing their specific GHG emissions. GHG emissions impact was assessed using the IPCC 2007 method (IPCC 2007).

3.2.4 Results and discussion

3.2.4.1 Model baseline

Fig. 3.8 shows the LC GHG emissions evolution for each scenario. Total LC GHG emissions of the fleet increased from 1995 to 2010 due to an increase in fleet size. Nevertheless, a reduction of impacts per km occurred, resulting from the rapid increase in market share of diesel ICEVs as well as a reduction in the fuel consumption of new gasoline ICEVs. Total LC GHG emissions of the fleet are expected to continue to increase until 2017 in the *BAU* scenario, due to the combined effect of fleet size and vehicle distance travelled growth. As the fleet size and distance travelled stabilize, a 4% reduction compared to 2010 is observed. This occurs because new vehicles entering the fleet are replacing older, higher-emitter vehicles. Even though the new vehicles are not improving over time, the overall fleet emissions are improved by the elimination of the older vehicles. Until 2025, reducing fuel consumption of new ICEVs (*ICEV improve*) has a larger effect on the LC GHG emissions than the introduction of BEVs in the fleet (*BEV dominate*), since it takes time for BEV share in the fleet to become significant. Nevertheless, a slightly higher reduction in 2010-2030 GHG emissions is obtained in this scenario (34%) than in the *ICEV improve* scenario (30%). The *Combined* scenario leads to an extra 5% reduction (39%). LC GHG emissions per km continue to decrease for all scenarios (except the *BAU*, in which it stabilizes around 2025). A steeper reduction occurs in the *Combined* scenario (40% decrease), while the *ICEV improve* (37%) reaches a slightly higher reduction in 2030 than the *BEV dominate* scenario (34%).

The shape of the curves and the ranking of scenarios obtained for *Total fleet GHG emissions* and *GHG emission per km* analysis differ. While the *Total fleet GHG emissions* assessment shows that GHG emissions from the Portuguese LDV fleet have been increasing and only after 2017 will start to decrease, the *GHG emissions per km* analysis shows a reduction tendency along time. This means that, although the emissions of an average km travelled in the fleet have been decreasing, mainly because gasoline ICEVs have been replaced by diesel ICEVs, the absolute emissions from the fleet have increased, as a result of the increase in the number of vehicles and distance travelled. This effect cannot be captured by the *per km* analysis. On the other hand, the ranking of scenarios in the *Total GHG emission* analysis is very dependent on the number of km travelled by the fleet, which changes according to the scenario (a higher share of diesel ICEVs results in a higher total distance travelled). In the *BEV dominate* and *Combined* scenarios the total distance travelled by the fleet in 2030 is about 10% lower than in the *BAU* and *ICEV improve* scenarios. This effect is discussed in the sensitivity analysis (Section 3.2.4.3).

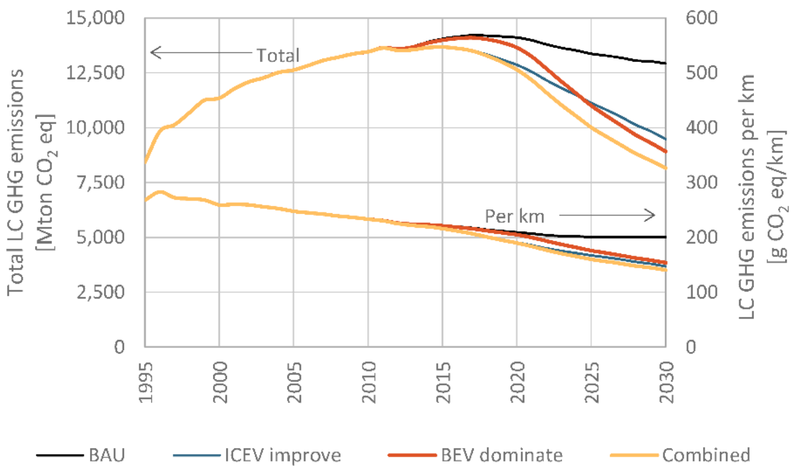

Fig. 3.8 Total life-cycle (LC) GHG emissions of the fleet (left axis) and LC GHG emissions per km (right axis) for the *Business-as-usual (BAU)*, *ICEV improve*, *BEV dominate* and *Combined* scenarios from 1995 to 2030.

3.2.4.2 Contribution analysis

Fig. 3.9 shows the contribution of the life-cycle stages to the fleet LC GHG emissions in 2010, 2020 and 2030 for the four scenarios. The category "Vehicle production, maintenance, and EoL" includes materials production, vehicle assembly, maintenance, and end-of-life (EoL) impacts. In 2010-2020, the contribution of each stage varies little between scenarios (1-2%) and the operation stage accounts for most of the fleet impacts (72-74%). This trend continues in both *BAU* and *ICEV improve* scenarios in 2030. In the *BEV dominate* and *Combined* scenarios, there is a shift of impacts from the fuel production to the electricity generation stage and, to a smaller extent, to the vehicle production, maintenance and EoL stages, in 2030. In absolute terms, vehicle operation impacts are reduced by 56-63% compared to the *BAU* scenario, but indirect impacts (which include fuel production, electricity generation, and vehicle production, maintenance and EoL) increase by 36-39%.

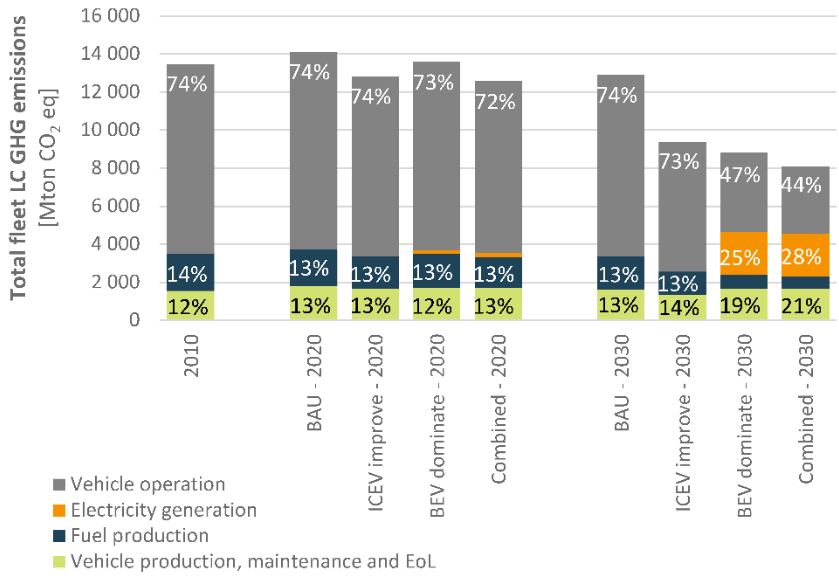

Fig. 3.9 Contribution of the life-cycle stages to the fleet LC GHG emissions in each scenario in 2010, 2020 and 2030.

3.2.4.3 Sensitivity analysis

A sensitivity analysis was performed to assess the influence of different model parameters on the *Total fleet LC GHG emissions* and *GHG emissions per km* in each scenario. The one-factor-at-a-time (OFAT) method was used and the parameters listed in Table 3.4 were varied between their lower and upper bounds. The rationale behind the choice of the lower and upper bounds for each parameter is presented in the Appendix III. Detailed results of the sensitivity analysis are presented in Figs. A-10 to A-13 (*Total fleet LC GHG emissions*) and A-14 to A-17 (*Fleet LC GHG emissions per km*) in Appendix III. Table 3.5 shows how the ranking between scenarios changes in the sensitivity analysis.

The sensitivity analysis for *Total fleet LC GHG emissions* in 2020 shows that the diesel ICEV indexed mileage (x(*d,k*)) and the vehicle density multiplier (θ) have the largest influence in the results in all scenarios (variations of 9-17%, as shown in Fig. A-11 in Appendix III). Nevertheless, varying these parameters does not change the ranking of the scenarios. On the other hand, although a change in the diesel ICEV fuel consumption reduction rate (φ(*d*)) leads to no more than 10% variation in the fleet GHG emissions in 2020, if its value is close to its higher bound, the *BEV dominate* scenario becomes better than the *ICEV improve* scenario. The other parameters do not significantly affect the total fleet GHG emissions in all scenarios (less than 8% variation). Regarding 2030 results, although the diesel ICEV indexed mileage (x(*d,k*)) and the vehicle density multiplier (θ) continue to have a high influence in the results (11-19%), the electricity generation emission factor ($e_e(t)$) is the parameter with higher influence in the *BEV dominate* and *Combined* scenarios (24-35% variation) and the diesel ICEV fuel consumption reduction rate ($x(d,k)$) in the *BAU* and *ICEV improve* scenarios (up to 32% variation). Nevertheless, varying these parameters does not change the ranking of the scenarios, except for the electricity generation emission factor, which, at its higher bound, makes the *ICEV improve* scenario better than the *BEV dominate* (-19%) and *Combined* (-13%) scenarios. When BEV distance travelled is increased to match ICEV distance travelled (BEV first-year vehicle distance travelled, y(e,0,2010), upper bound), the *ICEV improve* scenario becomes slightly better than the *BEV dominate* scenario (<0.5%). The scrappage rate (described by the maximum life expectancy, $\mu(t)$) has a higher influence in the *BAU* scenario (up to 14% variation) than on the other scenarios (less than 7%). Changing the fuel production emission factor (ι) has a higher influence in the ICEV scenarios (up to 11% variation) than in the BEV scenarios (up to 5%), as expected. All parameters increase their influence in the results as time passes; except the gasoline ICEV indexed mileage ($\varphi(g)$), which decreases, and the maximum life expectancy ($\mu(t)$), which varies.

When the *LC GHG emissions per km* perspective is examined, the sensitivity analysis shows little influence by all parameters (less than 10% change) in 2020. In 2030, the diesel ICEV fuel consumption reduction rate ($\varphi(d)$) and the electricity generation emission factor ($e_e(t)$) are the most influential parameters (up to 35% change), similar to the *Total fleet LC GHG emissions* perspective. Keeping the other parameters constant, if diesel ICEV fuel consumption reduces enough, the *Combined* scenario may no longer be better than the *ICEV improve* scenario and the *BAU* scenario becomes slightly better than the *BEV dominate* scenario. This is because the diesel ICEV fuel consumption positively affects a higher number of vehicles in the *BAU* and *ICEV improve* scenarios (due to a higher diesel ICEV market share). Moreover, if the electricity generation emission factor increases significantly, the *ICEV improve* scenario becomes better than the *Combined* scenario and the *BEV dominate* becomes the scenario with higher impacts. The *BEV dominate* scenario becomes slightly better than the *ICEV improve* scenario if the BEV first-year distance travelled (y(e,0,2010)) or the fuel production emission factor (ι) approach the upper bound (1.3 and 0.3%, respectively).

Results for *Total fleet LC GHG emissions* are sensitive to more parameters than *GHG emissions per km*. In particular, they are more sensitive to those parameters that affect the fleet dynamic, such as those that change the vehicle stock (vehicle density multiplier, θ), the scrappage rate (maximum life expectancy, $\mu(t)$), and the activity level of the fleet (indexed mileage of diesel vehicles, $x(d,k)$, which have higher distance traveled per vehicle). When impacts per km are analyzed, the fleet size and turnover do not significantly affect the results. Only parameters influencing the operation (fuel consumption, $\varphi(t)$, and electricity emission factor, $e_e(t)$) play a role in the impacts per km. For example, an increase in the fleet size (illustrated in Fig. 3.10 by assuming the upper bound figure for the vehicle density multiplier, θ) amplifies the total fleet impacts in all scenarios (resulting in a lower reduction of impacts in 2010-2030, and, in the *BAU* scenario, even an increase in the 2010-2030 impacts), without affecting the performance per km. On the other hand, increasing the distance travelled by BEVs decreases *GHG emissions per km* in the *BEV dominate* and *Combined* scenarios, while increasing the *Total fleet LC GHG emissions.* To achieve an effective reduction of the fleet GHG emissions, focus should be given not only on reducing operation impacts but also on reducing the fleet size and activity, and that can only be assessed through a *Total fleet LC GHG emissions* analysis.

Table 3.4 Parameters for sensitivity analysis. *d*: diesel ICEV; *g*: gasoline ICEV; *e*: BEV; *k*: vehicle age (in years).

<table>
<tr><th rowspan="2">Parameter</th><th rowspan="2">Lower bound</th><th colspan="4">Scenario baseline</th><th rowspan="2">Upper bound</th></tr>
<tr><th>Business-as-usual</th><th>ICEV improve</th><th>BEV dominate</th><th>Combined</th></tr>
<tr><td>Fuel consumption reduction rate (gasoline ICEV), $\varphi(g)$ [% $year^{-1}$]</td><td>0</td><td>0</td><td>2.5</td><td>0</td><td>2.5</td><td>4.0</td></tr>
<tr><td>Fuel consumption reduction rate (diesel ICEV), $\varphi(d)$ [% $year^{-1}$]</td><td>0</td><td>0</td><td>2.5</td><td>0</td><td>2.5</td><td>4.0</td></tr>
<tr><td>Electricity consumption reduction rate (BEV), $\varepsilon(e)$ [% $year^{-1}$]</td><td>0</td><td colspan="4">1.25</td><td>2.5</td></tr>
<tr><td>Electricity generation emission factor, $e_e(t)$ [g CO_2 eq kWh^{-1}]</td><td>20</td><td colspan="4">485</td><td>1100</td></tr>
<tr><td>Vehicle curb weight reduction rate, ν [% $year^{-1}$]</td><td>0</td><td>0</td><td>0.8</td><td>0.8</td><td>0.8</td><td>1.75</td></tr>
<tr><td>Indexed mileage (gasoline ICEV), $x(g,k)$</td><td>$-0.4\ln(k)+1.42$</td><td colspan="4">$-0.313\ln(k)+1.4173$</td><td>$-0.2\ln(k)+1.27$</td></tr>
<tr><td>Indexed mileage (diesel ICEV), $x(d,k)$</td><td>$-0.4\ln(k)+1.42$</td><td colspan="4">$-0.33\ln(k)+1.3623$</td><td>$-0.2\ln(k)+1.27$</td></tr>
<tr><td>Indexed mileage (BEV), $x(e,k)$</td><td>$-0.4\ln(k)+1.42$</td><td colspan="4">$-0.313\ln(k)+1.4173$</td><td>$-0.2\ln(k)+1.27$</td></tr>
<tr><td>Vehicle density multiplier, θ [% $year^{-1}$]</td><td>-1.5</td><td colspan="4">1</td><td>3</td></tr>
<tr><td>Maximum life expectancy, $\mu(t)$ [years]</td><td>30</td><td colspan="4">35</td><td>40</td></tr>
<tr><td>First-year vehicle distance travelled (BEV), y(e,0,2010) [km]</td><td>10500</td><td colspan="4">13929</td><td>17500</td></tr>
<tr><td>Fuel production emission factor rate of change, ι [% $year^{-1}$]</td><td>-0.5</td><td colspan="4">0</td><td>0.7</td></tr>
</table>

Table 3.5 Ranking of scenarios according to the results of the sensitivity analysis (1 – higher impact; 4 – lower impact). Highlighted cells represent a change in the ranking of scenarios compared to the baseline. Cell shading indicates the difference (Δ) in impacts between those cells, according to the legend. Parameters are defined in Table 3.4. All parameters were analyzed, but only those whose ranking changed are presented here.

	LC GHG emissions per km								***Total fleet LC GHG emissions***							
	Upper bound				**Lower bound**				**Upper bound**				**Lower bound**			
	Business-as-usual	*ICEV improve*	*BEV dominate*	*Combined*	*Business-as-usual*	*ICEV improve*	*BEV dominate*	*Combined*	*Business-as-usual*	*ICEV improve*	*BEV dominate*	*Combined*	*Business-as-usual*	*ICEV improve*	*BEV dominate*	*Combined*
2015																
Baseline	1	3	2	4	1	3	2	4	1	3	2	4	1	3	2	4
$\varphi(g)$	1	3	2	4	1	3	2	4	1	3	2	4	1	3	2	4
$\varphi(d)$	1	3	2	4	1	**4**	2	**3**	1	3	2	4	1	3	2	4
ε	1	3	2	4	1	3	2	4	1	3	2	4	1	3	2	4
$e_c(l)$	1	**4**	2	**3**	1	3	2	4	1	**4**	2	**3**	1	3	2	4
$y(e,0,2010)$	1	3	2	4	1	**4**	2	**3**	1	3	2	4	1	3	2	4
ι	1	3	2	4	1	3	2	4	1	3	2	4	1	3	2	4
2020																
Baseline	1	3	2	4	1	3	2	4	1	3	2	4	1	3	2	4
$\varphi(g)$	1	3	2	4	1	3	2	4	1	3	2	4	1	3	2	4
$\varphi(d)$	1	3	2	4	1	3	2	4	1	**2**	**3**	4	1	3	2	4
ε	1	3	2	4	1	3	2	4	1	3	2	4	1	3	2	4
$e_c(l)$	1	**4**	2	**3**	1	3	2	4	1	**4**	2	**3**	1	3	2	4
$y(e,0,2010)$	1	3	2	4	1	3	2	4	1	3	2	4	1	3	2	4
ι	1	3	2	4	1	3	2	4	1	3	2	4	1	3	2	4
2025																
Baseline	1	3	2	4	1	3	2	4	1	**2**	**3**	4	1	**2**	**3**	4
$\varphi(g)$	1	3	2	4	1	3	2	4	1	2	3	4	1	2	3	4
$\varphi(d)$	1	**2**	**3**	4	1	3	2	4	1	2	3	4	1	2	3	4
ε	1	3	2	4	1	3	2	4	1	**3**	**2**	4	1	2	3	4
$e_c(l)$	1	**4**	2	**3**	1	**2**	**3**	4	1	**4**	**2**	**3**	1	2	3	4
$y(e,0,2010)$	1	**2**	**3**	4	1	3	2	4	1	**3**	**2**	4	1	2	3	4
ι	1	3	2	4	1	3	2	4	1	2	3	4	1	2	3	4

	LC GHG emissions per km								*Total fleet LC GHG emissions*							
	Upper bound				**Lower bound**				**Upper bound**				**Lower bound**			
	Business-as-usual	*ICEV improve*	*BEV dominate*	*Combined*	*Business-as-usual*	*ICEV improve*	*BEV dominate*	*Combined*	*Business-as-usual*	*ICEV improve*	*BEV dominate*	*Combined*	*Business-as-usual*	*ICEV improve*	*BEV dominate*	*Combined*
2030																
Baseline	1	3	2	4	1	3	2	4	1	2	3	4	1	2	3	4
$\varphi(g)$	1	**2**	**3**	4	1	3	2	4	1	2	3	4	1	2	3	4
$\varphi(d)$	1	**2**	**3**	4	**2**	**4**	**1**	**3**	1	2	3	4	1	**3**	**2**	4
ε	1	**4**	2	**3**	1	**2**	**3**	4	1	2	3	4	1	2	3	4
$e_c(t)$	**2**	**4**	**1**	**3**	1	**2**	**3**	4	1	**4**	**2**	**3**	1	2	3	4
$y(e,0,2010)$	1	**2**	**3**	**4**	1	3	2	4	1	**3**	**2**	4	1	2	3	4
ι	1	**2**	**3**	4	1	3	2	4	1	2	3	4	1	2	3	4

Legend: $\Delta<1\%$ | $1\%\leq\Delta<5\%$ | $5\%\leq\Delta<10\%$ | $10\%\leq\Delta$

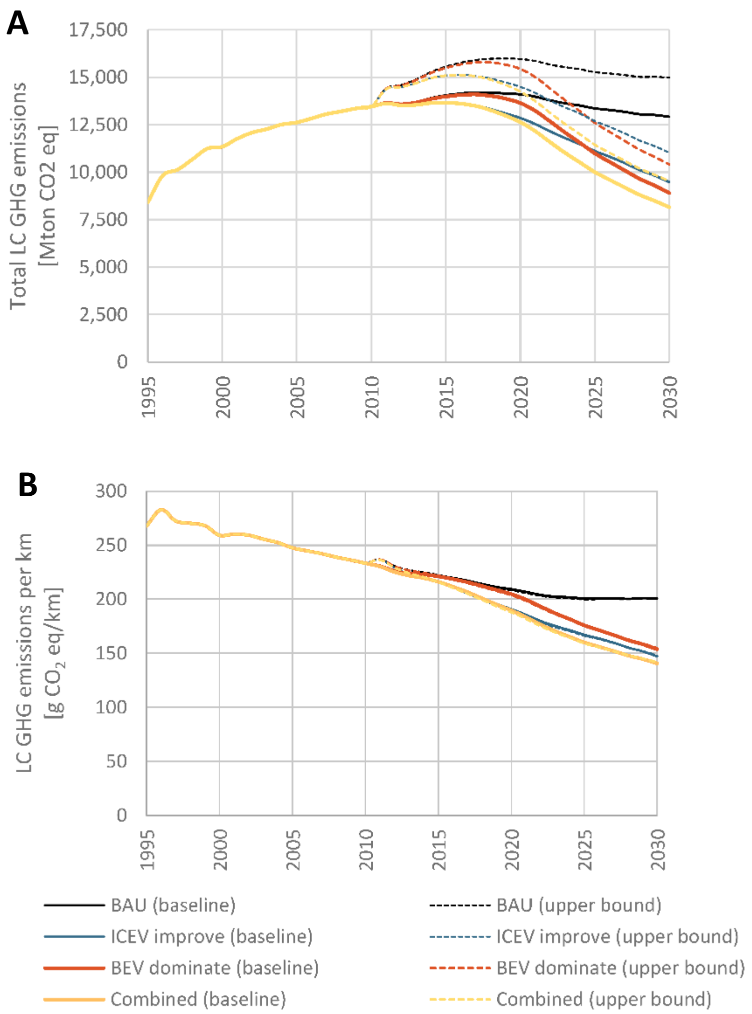

Fig. 3.10 Total life-cycle GHG emissions (A) and life-cycle GHG emissions per km (B) in each scenario for the model baseline and assuming a higher vehicle density (upper bound value in Table 3.4).

3.2.4.4 Parametric analysis

The sensitivity analysis showed that: (i) total fleet GHG emissions should be examined in order to account for the characteristics of the fleet (size and activity level) that affect the fleet emissions over time; and (ii) the diesel ICEV fuel consumption reduction rate and the electricity emission factor are the parameters that show higher variation in the 2030 GHG emissions (the latter being key for the ranking of scenarios). In this section, the analysis focus on how the fleet GHG emissions in 2030 change relative to the *BAU* scenario for different electricity emission factors, 2010-2030 new diesel ICEV fuel consumption reduction, and BEV market shares (Fig. 3.11).

At low BEV market shares, the BEV potential to reduce fleet GHG emissions compared to an all ICEV fleet is very low and the effect of the electricity emission factor is not very significant. As the BEV market share increases, the influence of the electricity emission factor in the GHG emission reduction also increases – at 100% market share, it increases by 4% per 100 g CO_2 eq kWh^{-1} decrease. All things equal, BEV introduction in the fleet has the potential to reduce total GHG emissions even if the electricity source is coal. However, as new ICEVs improve, the electricity emission factor becomes key for BEV introduction to be beneficial compared to ICEVs. Up to 780 g CO_2 eq kWh^{-1} (50/50 coal/natural gas mix), improving ICEVs (*ICEV improve* scenario) is better than introducing BEVs in the fleet, irrespective of the BEV market share. If ICEVs fuel consumption reduces by 80%, introducing BEVs is only better if the electricity emission factor is lower than 485 g CO_2 eq kWh^{-1} (current mix). A higher reduction (>15%) in fleet GHG emissions from the introduction of BEVs compared to improving ICEVs only occurs at high BEV market shares (>95% in 2030, corresponding to a fleet fraction of more than 25%) and electricity emission factors similar or lower to the current mix.

When a 50% BEV and 50% ICEV fleet is reached (100% BEV market share in 2030), results show that halving the GHG emissions from BEVs by reducing the electricity mix impacts is more effective than halving ICEVs emissions through reducing fuel consumption (13% and 9% reduction in fleet GHG emissions, respectively). This happens because fuel consumption reduction only affects new vehicles entering the fleet and it takes time for these new, higher-efficient vehicles to gain fleet share. On the contrary, the electricity emission factor affects all BEVs in the fleet, irrespective of their age. The lag between now and the time high BEV adoption is realized may allow for the decarbonization of the grid necessary for BEVs to reach their full potential. On the other hand, a lower rate of introduction of BEVs in the fleet may allow a quicker diffusion of high-efficient ICEVs, requiring a more aggressive decarbonization of the grid for BEV adoption to have a noticeable effect in overall fleet emissions.

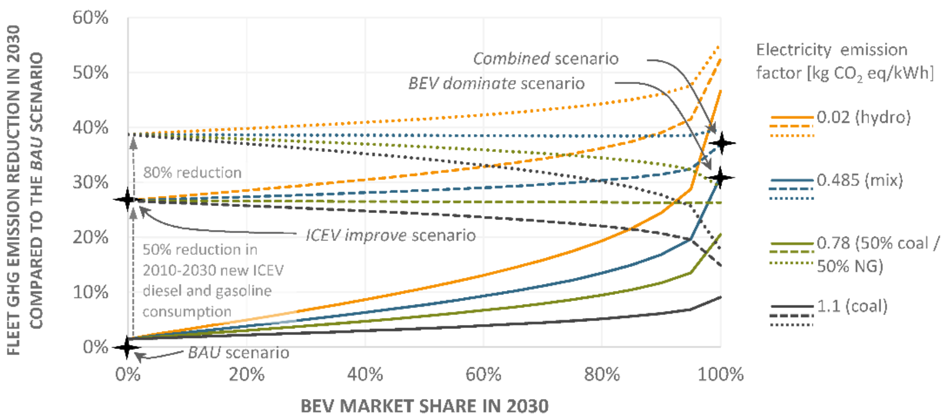

Fig. 3.11 Reduction in total fleet GHG emissions in 2030 as a function of the 2030 BEV market share and the electricity emission factor for 80%, 50%, and 0% 2010-2030 diesel ICEV fuel consumption reduction rates, compared to the *Business-as-usual (BAU)* scenario. NG: natural gas.

3.2.5 Concluding remarks

Fleet-wide environmental impacts of displacing ICEVs by EVs across different scenarios and metrics were assessed. The analysis took into account the dynamic behavior of the fleet, including fleet turnover, technology improvements (e.g., reduction in fuel consumption of new vehicles, weight reduction), and changes in background processes (e.g., electricity mix, material GHG intensity) and vehicle activity (e.g., annual distance traveled) within the same framework. The analysis spanned 15 years into the past (1995) and 20 years into the future (2030).

Results showed that it takes time for BEV share in the fleet to become significant and that only after 2025 does the effect of introducing BEVs in the fleet GHG emissions start to emerge. The reduction in the fleet GHG emissions from displacing ICEVs by BEVs is highly dependent on the BEV market share, new diesel ICEV fuel consumption reduction, and electricity emission factor. Emissions reductions at the end of the assessment period (2030) are between 1% and 47% when compared with a business-as-usual fleet (*BAU* scenario), and -16% and 38% if compared with an ICEV improved fleet that meets EU targets. For BEV introduction in the fleet to be beneficial compared to an increasingly more efficient ICEV fleet, a high BEV market share and electricity emission factor similar or lower to the current mix (485 g CO_2 eq kWh^{-1}) need to be realized; these conclusions hold for the different conditions analyzed. It was also found that halving the GHG emissions from BEVs by reducing electricity mix impacts has a larger effect on the overall fleet GHG emission reduction compared to halving GHG emissions from ICEVs by decreasing fuel consumption, but that effect may change depending on how the fleet evolves.

Besides the importance of the fuel consumption reduction rate of new ICEVs and the electricity mix emission factor, results were also sensitive to parameters that affect the fleet dynamic, such as those that change the vehicle stock, the scrappage rate, and the activity level of the fleet (11-19% variation in total GHG emissions in 2030). The influence of these parameters also varies over time, becoming more important as time passes. These effects can only be captured by assessing total fleet GHG emissions as opposed to the GHG emissions per km approach.

These results emphasize the importance of taking into account the dynamic evolution of the fleet, technology improvements over time, and changes in vehicle operation and background processes during the vehicle service life when assessing the potential benefits of displacing ICEVs by EVs in a fleet. These factors are usually not accounted for in the literature. Therefore, the fleet-based approach presented can provide a more comprehensive assessment of the adoption of an emerging technology, such as EVs, because it enables explicit assessment of improvements and developments over time, and also indirect effects related with the existing system, such as the effects of displacing ICEVs. It can also provide the scale and timing for assessing other indirect impacts, such as the effects of BEV load in the power grid, which is the main subject of Chapter 5. Moreover, this approach avoids fixed assumptions about vehicles service life, because impacts from vehicle production are accounted for in the year the vehicles are produced and are independent from the time the vehicle is scrapped. Assumptions about vehicle service life are often indicated as having a significant influence in the environmental impact results of vehicles (Hawkins et al. 2013; Nordelöf et al. 2014a). With this framework, it is also possible to assess the effect of other measures to decrease impacts from transportation, such as reducing the fleet size, decreasing the distance travelled by vehicles, and delaying or anticipating scrapping, and how they compare with EV adoption, which is left for future research.

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Chapter 4

COMPREHENSIVE LIFE-CYCLE MODELING OF THE PORTUGUESE ELECTRICITY SYSTEM

Abstract A comprehensive LCA of the electricity system in Portugal is performed, assessing annual environmental impacts from 2003-2014, and hourly GHG emissions from 2012-2014. The influence of the time of charging in BEV GHG emissions is assessed from an attributional perspective. The models developed provide the ground for the assessment of the effects of BEVs in the Portuguese electricity system presented in Chapter 5.

4.1 Attributional life-cycle assessment of electricity in Portugal[5]

4.1.1 Introduction

The Portuguese electricity mix is undergoing a significant shift away from the technologies that have dominated generation for the past decades. In the last 10 years, renewable energy installed capacity more than doubled, boosted by the large investments that took place especially on wind power. In 2012, Portugal was ranked on the top 10 of countries with higher wind power installed capacity and was the fifth in terms of installed capacity per capita (World Wind Energy Association 2012). Recent decisions on replacing and adding capacity to the Portuguese electricity system have been mostly driven by European Union (EU) policies, such as the EU Renewable Energy Sources (RES) Directive (European Commission 2009) targets, the EU Large Combustion Plant Directive (European Commision 2001) and the EU Cogeneration Directive (European Commission 2004). Regarding the EU RES Directive, Portugal has set the 5th more ambitious target in the EU for gross final energy consumption from RES: 31% in 2020 (the EU average is 20%). To achieve this, sector-specific targets have been defined, including 60% of electricity generated from RES in 2020. The Portuguese electricity technology portfolio has been

[5] Significant portions of this section appear in: Garcia R., Marques P., Freire F. (2014). Life-cycle assessment of electricity in Portugal. *Applied Energy* 134:563-572. http://dx.doi.org/10.1016/j.apenergy.2014.08.067

rapidly changing towards the diversification of energy sources and increasing security of energy supply. It is, therefore, important to understand how the recent changes in the electricity system influenced the environmental profile of electricity in Portugal in the last years.

Comprehensive life-cycle assessment studies of country or region electricity mixes, including a wide set of environmental impacts associated with the country/region specific technology portfolio, are not common. A LCA of electricity generation in Mexico was performed by Santoyo-Castelazo et al. (2011); Ou et al. (2011) and Mallia and Lewis (2012) assessed the LC greenhouse gas (GHG) emissions of electricity generation in China and Ontario, Canada, respectively. Emission factors for electricity generation mixes in different regions of the USA were determined by Weber et al. (2010). No study was found combining a comprehensive environmental assessment of the generation technologies of a country mix, electricity imports and the transmission and distribution grid impacts (i.e. which covered the cradle to the plug). This research contributes to the body of knowledge on the environmental assessment of a country/region electricity mix, including both generation and supply chains. Moreover, as far as the author is aware, no comprehensive LCA has been performed for the Portuguese electricity system.

This research aims at performing a LCA of electricity generation and supply in Portugal mainland (excluding the Azores and Madeira islands) from 2003 to 2014, including: (i) modeling the main electricity generation technologies available in Portugal, namely coal, natural gas, hydro and wind, which together represented 92% of the electricity generated in 2012; (ii) modeling the Portuguese transmission and distribution (T&D) grid infrastructure, including sulfur hexafluoride leakages and energy losses; (iii) characterizing the evolution of the electricity sector in Portugal in the last 10 years (2003-2014) regarding generation technology characteristics and its share in the electricity mix; and (iv) discussing how the recent changes in the technology portfolio affected the environmental performance of the electricity generated and supplied in Portugal. This retrospective analysis is relevant to obtain information on trends, for regulation at the country or product level (Soimakallio and Saikku 2012) and also in the context of LCA studies of several product systems using electricity. Furthermore, the application of LCA to electricity systems is important to understand how to meet future electricity demand with reduced environmental impacts.

4.1.2 Characterization of the Portuguese electricity system (2003-2014)

Total electricity generation in Portugal increased about 23% from 2003 to 2010, when it reached 50 TWh, decreasing to about 48 TWh in the subsequent years (except 2012, when it came down to about 41 TWh). On the other hand, total installed capacity increased about 43% since 2003, reaching 18.5 GW in 2012. Fig. 4.1 shows the power plant installed capacity per technology in Portugal from 2003 to 2014.

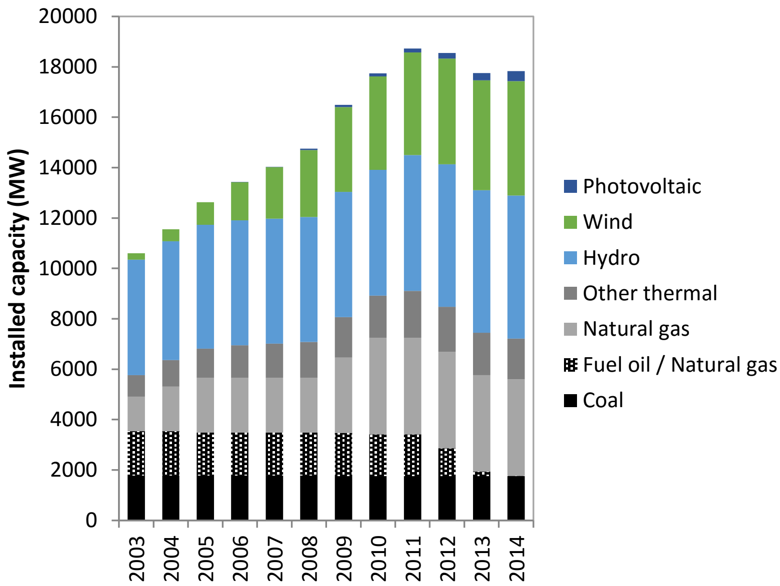

Fig. 4.1 Power plant installed capacity (MW) per technology (2003-2014) (REN 2015). Other thermal includes non-renewable CHP, biomass CHP, biomass, biogas and waste incineration. Large fuel oil PP started to be phased out in 2008 and stopped their operation in 2011, although some installed capacity still remained in 2012.

In the past, electricity generation in Portugal was mainly based on thermal power plants, mostly fueled by fossil resources, such as coal, natural gas and fuel oil, and large hydro power plants (PP). Due to hydropower generation dependence on meteorology, a high variation of its share in the electricity mix is observed, which is usually offset by thermal electricity generation (coal, natural gas and fuel oil). For instance, in 2005, a particularly dry year, only 20.5% of electricity was generated using renewable sources (REN 2015). Since 2003, however, there has been an increase in the use and diversity of renewable sources for electricity generation, namely wind, and, in a smaller scale, photovoltaic (PV), small-hydro, biomass and biogas, as a result of the implementation of policies which encouraged the generation of electricity from renewable endogenous resources (e.g., in the scope of the Energy Efficiency and Endogenous Energies (E4) Program).

Changes in the Portuguese electricity technology portfolio in the last 12 years can be observed in both renewable- and non-renewable-based technologies (detailed data for the Portuguese electricity generation and supply mix from 2003 to 2014 is shown in Table 4.1). Regarding renewable technologies, large investments took place, especially on wind power, and total renewable installed capacity more than doubled. In the last five years (except 2012), renewable electricity accounted for more than 50% of the total electricity generated in Portugal (see Table 4.1). In particular, from 2003 to 2014, the wind power installed capacity increased from 247 to 4541 MW (i.e. more than 18 times in 12 years), and wind power was the second highest contributor to electricity generation in the last three years (see Table 4.1). Biomass combined heat and power plant capacity also increased, reaching more than 5% of the electricity generated in Portugal in 2012. A growth of 7.5 times on PV installed capacity in seven years was observed (53 to 396 MW); however, PV share in the mix was still low in 2014 (less than 1%), as its high cost still prevented a large scale adoption. Although some large PV power plants were installed, namely the Amareleja PP (46 MW), located in the Alentejo region (South of Portugal), – the largest PV PP in the world in 2008, when its operation started – PV has mainly been promoted through a feed-in tariff scheme for small scale systems (under 3.68 kW of power).

Regarding non-renewable technologies, the decommissioning of large fuel oil PPs took place since 2008, following the implementation of the National Program for Climate Change (PNAC 2006) (Presidência do Concelho de Ministros 2006) and these do not contribute to the electricity mix since 2011. On the other hand, there was a significant investment in natural gas (NG) combined cycle (CC) capacity (more than doubled in 10 years). Nevertheless, electricity generation from NGCC was approximately constant from 2004 to 2011 and even decreased about 65% in the last three years due to higher costs compared to coal. Regarding coal PP, no new investments were made in the last 12 years, but the environmental performance of existing coal PP was improved, driven by the EU Large Combustion Plant Directive (European Commision 2001). In 2008, new flue gas treatment systems were installed, namely: (i) denitrification systems (DeNOx), based on selective catalytic reduction (SCR) systems, to reduce NO_x emissions; (ii) desulphurization systems (DeSOx), based on wet scrubbers, to remove SO_2; and (iii) electrostatic precipitators to remove particulate matter. The installation of combined heat and power (CHP) plants has also been encouraged through the EU Cogeneration Directive (European Commission 2004) and, as a result, natural gas CHP and, to a smaller extent, fuel oil CHP contribution to the electricity mix increased 274% in 2003-2012 and represented more than 10% of total electricity generation in Portugal in the last three years. To sum up, while in 2003 about 95% of all electricity generated in Portugal was mostly based on coal, natural gas and hydropower, 12 years later, a significant portion was generated by wind (23%), and the electricity mix is now more diversified.

4.1.3 Materials and methods

The environmental impacts associated with the electricity generated and supplied in Portugal mainland from 2003 to 2014 were evaluated using a process-based attributional LCA methodology (ISO 2006a; ISO 2006b).

4.1.3.1 Scope and system boundary

LC models and inventories for the main electricity generation systems and the transmission and distribution (T&D) grid were implemented for Portugal. The functional unit is 1 kWh of electricity. The following impact categories were assessed: cumulative non-renewable fossil energy demand (nREn), using the CED method (Althaus et al. 2007); global warming (GW), using the IPCC 2007 methodology (IPCC 2007); and abiotic depletion (AD), acidification (AC), eutrophication (ET), photochemical oxidation (PO) and ozone layer depletion (OD), using the CML 2 v2.05 LC impact assessment method (Guinée et al. 2002).

The system boundary included the Portuguese electricity generation systems from a cradle-to-grave perspective, namely coal, fuel oil, NGCC, NGCHP, hydro, wind, waste incineration, biogas and photovoltaic. The life-cycle stages included comprise extraction, processing and transport of fuels, operation of power plants, construction and decommissioning of power plants and waste management. The system boundary for the assessment of electricity supply included the previously described processes as well as the transmission and distribution (T&D) grid infrastructure; T&D grid losses, and electricity imports.

Table 4.1 Electricity generated by technology (E_j), total electricity generation (E_{gen}), and supply (E_{sup}) (GWh) (ERSE 2013; REN 2015).

Technologies	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
Non-renewables												
Coal	13641	13952	14291	14070	11663	10423	11942	6553	9128	12361	10856	10184
	33.9%	*35.8%*	*34.8%*	*32.2%*	*27.4%*	*25.3%*	*26.5%*	*13.2%*	*19.4%*	*30.2%*	*23.5%*	*21.8%*
Fuel oil	2648	1984	4840	1501	1268	800	303	47	0	0	0	0
	6.6%	*5.1%*	*11.8%*	*3.4%*	*3.0%*	*1.9%*	*0.7%*	*0.1%*	*0.0%*	*0.0%*	*0.0%*	*0.0%*
Natural gas	6105	9813	11490	9907	10494	12573	11463	10700	9600	5412	3486	3207
	15.2%	*25.1%*	*27.9%*	*22.7%*	*24.6%*	*30.6%*	*25.4%*	*21.6%*	*20.4%*	*13.2%*	*7.6%*	*6.9%*
Non-renewable CHP[a]	1550	2052	2540	2806	3252	3011	3590	4480	4767	4406	5152	4785
	3.8%	*5.3%*	*6.2%*	*6.4%*	*7.6%*	*7.3%*	*8.0%*	*9.0%*	*10.1%*	*10.8%*	*11.2%*	*10.3%*
Renewables												
Hydropower	15709	9911	4916	11196	10220	7095	8710	16243	11820	6423	13109	14354
	39.0%	*25.4%*	*12.0%*	*25.6%*	*24.0%*	*17.2%*	*19.3%*	*32.7%*	*25.1%*	*15.7%*	*28.4%*	*30.8%*
Wind	466	783	1728	2892	4018	5691	7480	9032	9105	9992	11494	11711
	1.2%	*2.0%*	*4.2%*	*6.6%*	*9.4%*	*13.8%*	*16.6%*	*18.2%*	*19.3%*	*24.4%*	*24.9%*	*25.1%*
Biomass CHP	129	463	1331	1508	1565	1519	1543	1734	1809	2316	1735	1724
	0.3%	*1.2%*	*3.2%*	*3.4%*	*3.7%*	*3.7%*	*3.4%*	*3.5%*	*3.8%*	*5.7%*	*3.8%*	*3.7%*
Biomass	43	54	60	71	149	146	305	612	688	676	679	741
	0.1%	*0.1%*	*0.1%*	*0.2%*	*0.3%*	*0.4%*	*0.7%*	*1.2%*	*1.5%*	*1.6%*	*1.5%*	*1.6%*
Waste incineration[b]	456	413	472	460	425	441	458	454	486	355	471	479
	1.1%	*1.1%*	*1.1%*	*1.1%*	*1.0%*	*1.1%*	*1.0%*	*0.9%*	*1.0%*	*0.9%*	*1.0%*	*1.0%*
Biogas	0	9	25	24	47	59	71	92	149	182	238	280
	0.0%	*0.0%*	*0.1%*	*0.1%*	*0.1%*	*0.1%*	*0.2%*	*0.2%*	*0.3%*	*0.4%*	*0.5%*	*0.6%*
Photovoltaic	0	0	0	0	20	33	140	167	187	227	395	456
	0.0%	*0.0%*	*0.0%*	*0.0%*	*0.0%*	*0.1%*	*0.3%*	*0.3%*	*0.4%*	*0.6%*	*0.9%*	*1.0%*
Pumping[c]	-485	-408	-568	-703	-541	-639	-929	-512	-587	-1379	-1459	-1276
	-1.2%	*-1.0%*	*-1.4%*	*-1.6%*	*-1.3%*	*-1.6%*	*-2.1%*	*-1.0%*	*-1.2%*	*-3.4%*	*-3.2%*	*-2.7%*
Total generation (E_{gen})	40261	39025	41125	43733	42581	41153	45075	49602	47152	40971	46156	46645
Imports (E_{imp})	4433	7460	7528	7649	9088	9478	5616	4350	4446	8297	5229	4084
Exports (E_{exp})	1633	976	702	2267	1591	40	827	1718	1635	403	2447	3184
T losses ($E_{loss\,T}$)	738	677	648	562	577	585	523	961	788	770	853	919
D losses ($E_{loss\,D}$)	3258	3451	3439	3168	2591	3633	3277	3778	3464	3904	4687	4302
Total supply (E_{sup})	39059	41377	43858	45444	46901	46366	46052	47486	45713	44190	43398	42324

In italics, contribution (%) of each technology to the generation mix.
T: transmission; D: distribution.
[a] Includes electricity generation from natural gas CHP and fuel oil CHP, to a smaller extent.
[b] The biodegradable fraction of municipal waste is 56% (Dias et al. 2006).
[c] The amount of electricity for pumping is explicitly accounted negatively, since this was not subtracted from the amount of energy generated by the remaining technologies.

4.1.3.2 *Life-cycle inventory and modeling*

Table 4.2 shows the main characteristics of the average PT technologies and data sources for the development of LC models and inventories (LCI). Background data was obtained from the ecoinvent v.2.2 database (ecoinvent 2007). Total electricity generated by coal, fuel-oil, natural gas CC and hydro power plants in each year was gathered from REN (*Redes Energéticas Nacionais*) technical reports (REN 2015). For the remaining technologies, data from ERSE (Energy Services Regulatory Authority) reports were used (ERSE 2013). LC environmental impacts per kWh of electricity generated was calculated based on Equation 4.1. LC environmental impacts per kWh of electricity supplied were calculated based on Equation 4.2. It should be noted that the environmental impacts of electricity generated within the country included electricity exports, but electricity supplied did not.

$$I_{gen,i} = \sum_j (E_j\, I_{ij}) / E_{gen} \tag{4.1}$$

in which,

$I_{gen,i}$ = Life-cycle impacts in environmental category *i* per kWh of electricity generation

E_j = Net electricity generation by technology *j* (kWh)

I_{ij} = Life-cycle impacts in environmental category *i* per kWh of electricity generated by technology *j*

E_{gen} = Electricity generation (kWh) (gross electricity generation – electricity consumption in power plants – pumped storage)

$$I_{sup,i} = [(E_{gen} - E_{exp})I_{gen,i} + E_{imp}\, I_{imp,i} + (E_{gen} - E_{exp} + E_{imp} - E_{loss\,T})I_{T,i} + (E_{gen} - E_{exp} + E_{imp} - E_{loss\,T} - E_{loss\,D})\, I_{D,i}] \,/\, E_{sup} \tag{4.2}$$

in which,

$I_{sup,i}$ = Life-cycle impacts in environmental category *i* per kWh of electricity supply

E_{exp} = Electricity exports (kWh)

E_{imp} = Electricity imports (kWh)

$I_{imp,i}$ = Life-cycle impacts in environmental category *i* per kWh of imported electricity

$I_{T,i}$ = Life-cycle impacts in environmental category *i* of the transmission grid infrastructure per kWh of electricity transported

$E_{loss\ T}$ = Electricity loss in the transmission network (kWh)

$E_{loss\ D}$ = Electricity loss in the distribution network (kWh)

$I_{D,i}$ = Life-cycle impacts in environmental category *i* of the distribution grid infrastructure per kWh of electricity distributed

E_{sup} = Electricity supply (kWh) (gross electricity generation – electricity consumption in power plants – pumped storage + electricity imports – electricity exports – T&D losses)

Electricity imports and exports were obtained from (REN 2015). Portugal trades electricity with Spain within the Iberian Electricity Market (MIBEL). The Iberian market has been almost cut off from the rest of Europe since cross-border capacity with neighboring power markets, such as France and other countries in the Central West European Market, through the Pyrenees, has been limited. Electricity imports were modelled assuming the Spanish (ES) electricity generation mix from 2003 to 2014 (see Table B-1 in the Appendix IV), based on Dones et al. (2007), Heck (2007), and Jungbluth et al. (2007; 2009). It should be noted that, if a country trades electricity with more than one country, impacts from electricity imports ($E_{imp}\, I_{imp,i}$ in Eq. 3.2) shall reflect the impacts from generating electricity within all those countries.

Table 4.2 Main characteristic of the Portuguese power plants (average technologies) and LCI data sources.

Energy source	Technology	Power	Efficiency	LCI data sources
Coal[a]	Boiler and steam turbine	300 MW	37.5% (36%)	Fuel and chemical consumption and direct emissions: Environmental Declarations registered in the Eco-Management and Audit Scheme (EMAS) of specific Portuguese plants (EDP 2012a; Tejo Energia 2012); infrastructure adapted from Dones et al. (2007).
Natural gas	Combined cycle	400 MW	57.8%	Fuel consumption and direct emissions: Environmental Declarations registered in EMAS of a specific power plant that reflects the technologies currently operating in Portugal (EDP 2012b); infrastructure adapted from Dones et al. (2007).
	CHP CC[b]	80 MW	40%[c]	Infrastructure and direct emissions adapted from Heck (2007).
	CHP gas engine[b]	1.5 MW	38%[d]	Infrastructure and direct emissions adapted from Dones et al. (2007).
Biomass	Boiler and steam turbine	10 MW	16.5%	Production of biomass (mainly eucalypt) based on Nunes and Freire (2007) and Nunes (2008); efficiency, power, service life (20 years) based on Nunes and Freire (2007) and Nunes (2008); infrastructure and direct emissions adapted from Dones et al. (2007).
	CHP[b]	12.8 MW	34%[e]	Production of biomass (mainly eucalypt) based on Nunes and Freire (2007) and Nunes (2008); efficiency, power, service life (20 years) based on Coelho (2010); infrastructure and direct emissions adapted from Dones et al. (2007).
Hydro	Run-of-river	8.6 MW	82%	Based on Flury and Frischknecht (2012).
	Reservoir	95 MW	78%	Based on Flury and Frischknecht (2012).
	Mini-hydro	0.18 MW	n/a	Based on Flury and Frischknecht (2012).
Wind	Onshore wind turbine	2 MW	93%	Capacity factor (24%) based on average PT conditions; infrastructure adapted from Dones et al. (2007).
Fuel oil	Boiler and steam turbine	500 MW	35.6%	Infrastructure and direct emissions for PT conditions based on Dones et al. (2007).
Waste incineration	Municipal waste incinerator	n/a	13%	Based on Jungbluth et al. (2007).
Biogas	CHP gas engine	160 MW	32%	Based on Jungbluth et al. (2007).
Photovoltaic	Mix of technologies	n/a	n/a	Based on Jungbluth et al. (2009) for PT conditions.

[a] Change in efficiency due to the new flue gas treatment systems (DeSOx & DeNOx) that were installed in coal PP in 2008 (37.5% >2008; 36% <2008).

[b] Allocation of burdens in CHP was performed using exergetic allocation.

[c] Electrical efficiency. The global efficiency is 80%.

[d] Electrical efficiency. The global efficiency is 82%.

[e] Electrical efficiency.

The Portuguese electricity transmission and distribution (T&D) grid infrastructure was modeled by assembling single component life-cycle inventories (LCI) into a grid system according to data on the infrastructure installed in the Portuguese grid collected from EDP and REN (EPD 2012; REN 2015). Data about the grid infrastructure installed in 2011 was used, being assumed that the grid was manufactured and assembled as a single unit with a life time of 40 years using current technology, despite the fact that the grid has been developed for many decades. The total kilometers of overhead lines and underground cables installed by voltage level in 2011 is shown in Table B-2 (Appendix IV). The number of transformers installed by load rating in 2011 is presented in Table B-3. The LCI data sources for T&D grid components are shown in Table B-4. The life-cycle stages included comprise: production and transportation of materials, manufacture and installation of components, operation/maintenance and end-of-life. Direct emissions resulting from sulfur hexafluoride (SF_6) leakages were included and were obtained from REN (2015) for the transmission grid (46.7 kg SF_6 year^{-1}) and EPD (2012) for the distribution grid (175.4 kg SF_6 year^{-1}). LC impacts of the T&D grid infrastructure were calculated per kWh of output electricity. Total power losses ranged between 6.3 and 9.6%, corresponding to a minimum of 3730 GWh in 2006 and a maximum of 4739 GWh in 2010 (EPD 2012; REN 2015) (see Table 4.1). More than 80% of T&D losses were in the distribution grid.

4.1.4 Results and discussion

4.1.4.1 Life-cycle impacts per electricity generation technology

Table 4.3 shows the environmental LC impacts calculated per kWh of electricity generated by technology (I_{ij} parameter of Equation 4.1). Hydro power presented the lower impacts per kWh generated for all impact categories. Nevertheless, other environmental aspects of hydro power should be kept in mind, such as significant changes to ecosystems, which were not analyzed in this thesis. Fuel oil power plants had the higher impacts per kWh in acidification (AC), ozone layer depletion (OD), photochemical oxidation (PO) and non-renewable fossil energy demand (nREn), whilst coal power plants showed a higher contribution in global warming (GW), abiotic depletion (AD) and eutrophication (EUT). Regarding coal power plants, a reduction of 74% in PO, 67% in AC, and 2% in EUT was achieved since 2008 due to the installation of desulphurization (DeSOx) and denitrification (DeNOx) systems. However, there was an increase in nREn, AD and GW as a result of the decrease in the power plant efficiency (1-2%) due to the operation of these new flue gas treatment systems, and an increase of 24% in OD due to the production of ammonia used in the DeNOx system.

Table 4.3 Environmental life-cycle impacts per kWh generated by technology.

Technologies		nREn ($MJ_{prim\ fossil}$)	AD (g Sb eq)	GW (g CO_2 eq)	AC (g SO_2 eq)	PO (mg C_2H_4 eq)	EUT (g PO_4^{3-} eq)	OD (µg CFC-11 eq)
Non-renewables								
Coal[a]	w/out DeSOx & DeNOx	11.04	7.55	988	8.72	291	**2.48**	6.52
	w/ DeSOx & DeNOx	11.48	**7.81**	**1021**	2.84	75	2.42	8.05
Fuel oil		**13.16**	5.86	912	**19.00**	**748**	0.57	**113.04**
NG CC		7.38	3.61	423	0.35	31	0.06	51.81
NG CHP	Gas engine	9.40	4.59	588	0.74	61	0.15	65.92
	CC	6.47	3.16	370	0.29	27	0.04	45.39
Renewables								
Hydro[b]	Reservoir	0.04	0.02	17	0.02	1	0.06	0.00
	Run-of-river	0.04	0.02	4	0.02	1	0.01	0.00
	Small-hydro	0.05	0.03	5	0.03	1	0.01	0.00
Wind		0.04	0.17	23	0.11	8	0.06	1.24
Biomass CHP		0.37	0.19	33	0.65	17	0.23	2.81
Biomass		0.60	0.29	56	1.40	31	0.44	4.55
Biogas		1.31	0.65	239	0.72	68	0.13	9.73
Photovoltaic		0.65	0.36	51	0.25	15	0.16	9.60
Waste incineration		1.71	0.83	147	1.28	44	1.19	14.85

nREn: non-renewable fossil energy; AD: abiotic depletion; AC: acidification; EUT: eutrophication; GW: global warming; OD: ozone layer depletion; PO: photochemical oxidation.
DeSOx & DeNOx: Desulphurization and denitrification systems.
In each column, values in bold indicate the highest value and underlined values the lowest value.
[a] New flue gas treatment systems (DeSOx & DeNOx) were installed in coal PP in 2008.
[b] Based on Flury and Frischknecht (2012).

4.1.4.2 *Life-cycle impacts of the transmission and distribution grid infrastructure*

Table 4.4 shows the environmental LC impacts of the transmissions and distribution grid infrastructure calculated per kWh of output electricity ($I_{T,i}$ and $I_{T,i}$ parameters of Equation 3.2), for the grid installed in 2011. The transmission grid presented 7-16% more impacts in nREn, AD, GW and OD than the distribution grid. On the other hand, the distribution grid had 70-90% more impacts in AC, PO and EUT due to the large amount of copper used in distribution lines (the production of copper contributed to more than 75% of the AC and PO impacts, while emissions of phosphate from disposal of sulfidic tailings from copper production contributed to more than 90% of EUT impacts). Impacts from the distribution grid may be overestimated because it was assumed that all lines and cables are 11 kV and the share of lower voltage cables is significant (about 50%).

Table 4.4 Environmental life-cycle impacts per kWh of the transmission (T) and distribution (D) grid infrastructure in 2011.

	nREn (MJ $_{prim\ fossil}$)	AD (mg Sb eq)	GW (g CO_2 eq)	AC (mg SO_2 eq)	PO (mg C_2H_4 eq)	EUT (mg PO_4^{3-} eq)	OD (µg CFC-11 eq)
T	0.01	3.72	0.64	3.04	0.21	1.63	0.04
D	0.01	3.39	0.53	17.49	0.67	19.38	0.04

nREn: non-renewable fossil energy; AD: abiotic depletion; AC: acidification; EUT: eutrophication; GW: global warming; OD: ozone layer depletion; PO: photochemical oxidation.

4.1.4.3 *Life-cycle impacts of the electricity generation mix*

Table 4.5 shows the LC impacts per kWh associated with the evolution of the Portuguese annual electricity generation mix ($I_{gen,i}$) calculated from Equation 3.1. Fig. 4.2 shows the environmental LC impacts per kWh by technology as well as the total electricity generated (2003-2014). The year 2010 had the higher electricity generation, but presented the lowest impacts per kWh in all categories (except OD), mainly due to a high share of renewables (56.5%) and a relatively low share of coal (13.1%). On the other hand, the 2005 mix presented the highest impacts per kWh in all categories, coinciding with the highest generation from non-renewable sources (79.5%). Non-renewable energy sources (coal, natural gas and fuel oil) were responsible for the majority of impacts in all categories (more than 84%). In particular, coal-based electricity dominated the impacts in all categories, except OD, as a result of a combination of high impacts per kWh (see Table 4.3) and a relatively high share in the mix (13.1 to 35.4%), while natural gas and fuel oil (until 2005) controlled the impacts in OD, mostly due to emissions of Halon 1211 and Halon 1301 from the utilization of fire extinguishers in the transport of natural gas by pipeline and in crude production in the fuel oil chain, respectively.

From 2003 to 2014, an overall reduction of the environmental impacts was achieved. In particular, since 2008, impacts in AC and PO dropped sharply (about 81% reduction in 2003-2014) as a result of the installation of desulphurization and denitrification systems in coal power plants, as well as the phase out of fuel oil power plants. AC impacts varied between 0.6 and 5.5 g SO_2 eq kWh^{-1} and PO impacts between 0.03 and 0.20 g C_2H_4 eq kWh^{-1} during the 2003-2014 period.

For the other impact categories, the reduction of impacts was less pronounced (33-39%), despite the increase in the renewable share, due to the still high share of coal-based electricity in the mix. Impacts on nREn, AD and GW had a similar progress and presented the same trend as non-renewable electricity generation until 2011. In 2012, although non-renewable generation decreased in relation to the previous year, there was an increase in nREn, AD and GW impacts as a result of an increase of 53% in the coal contribution to the electricity mix. Non-renewable energy demand (nREn) associated with the Portuguese electricity mix varied between 1.1 and 2.2 $MJ_{primfossil}$ MJ^{-1}, the impact on GW ranged between 287 and 609 g CO_2 eq kWh^{-1} and AD impacts fall between 2.3 and 4.6 g Sb eq kWh^{-1}. Regarding EUT, impacts were dominated by the share of coal-based electricity in the mix. The installation of new denitrification systems in these power plants had a very low effect in the reduction of those impacts. EUT impacts ranged between 0.38 and 0.98 g PO_4^{3-} eq kWh^{-1}. Concerning OD, the majority of impacts were associated with natural gas- and fuel oil-based electricity generation. The reduction of impacts first occurred as a result of a decrease in operation of fuel oil PP, and, recently, due to the reduction in natural gas-based generation. OD impacts ranged between 12 and 34 μg CFC-11 eq kWh^{-1}.

Table 4.5 Life-cycle impacts per kWh of the annual Portuguese electricity generation mix (2003-2012).

Impact categories	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
nREn ($MJ_{prim\ fossil}$)	6.12	7.00	**8.06**	6.31	5.98	6.15	5.79	4.00	4.70	5.46	4.31	3.99
AD (g Sb eq)	3.69	4.17	**4.63**	3.77	3.51	3.57	3.43	2.27	2.75	3.37	2.66	2.46
GW (kg CO_2 eq)	486	541	**609**	488	452	455	438	287	350	434	344	320
AC (g SO_2 eq)	4.31	4.25	**5.46**	3.64	3.16	1.32	1.10	0.62	0.78	1.08	086	0.81
PO (g C_2H_4 eq)	0.16	0.15	**0.20**	0.13	0.12	0.05	0.04	0.03	0.03	0.04	0.03	0.03
EUT (g PO_4^{3-} eq)	0.91	0.96	**0.98**	0.87	0.75	0.69	0.71	0.38	0.54	0.80	0.63	0.59
OD (μg CFC-11 eq)	20	25	**34**	22	23	25	22	19	19	17	13	12

nREn: non-renewable fossil energy; AD: abiotic depletion; AC: acidification; EUT: eutrophication; GW: global warming; OD: ozone layer depletion; PO: photochemical oxidation.
In each line, values in bold indicate the highest value and underlined values the lowest value.

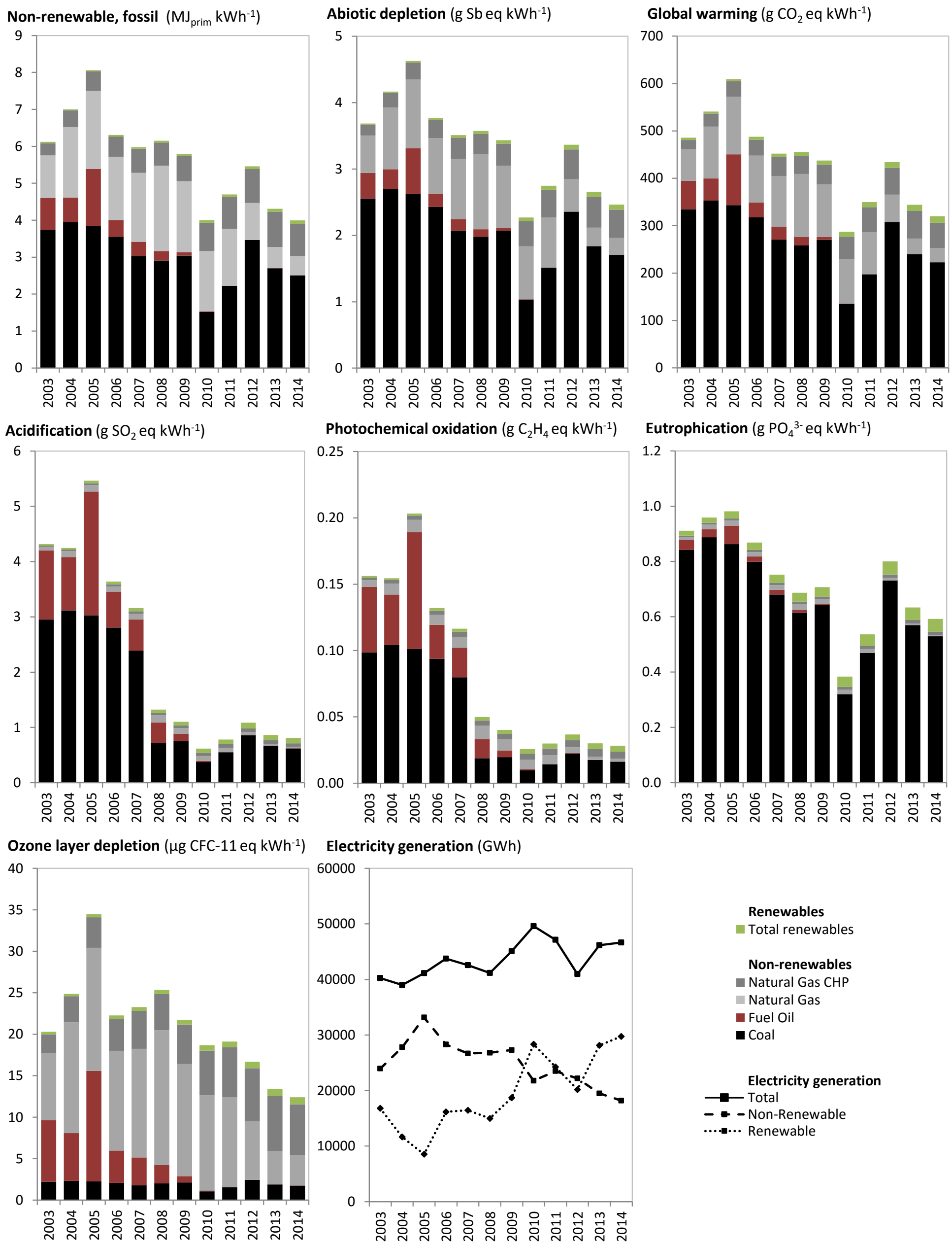

Fig. 4.2 Life-cycle impact assessment results of annual electricity generation mix in Portugal (2003-2012). Natural gas CHP includes 30% of electricity generated in combined cycle power plants and 70% in gas engines.

4.1.4.4 Life-cycle impacts of the electricity supply mix

Table 4.6 shows the LC impacts per kWh of electricity supply ($I_{sup,i}$) in Portugal from 2003 to 2014. Fig. 4.3 shows the contribution of different LC stages to the total LC impacts of electricity supply as well as the total electricity supply. Electricity imports contribution was highly variable between years and ranged from 5% in 2010 to 18% in 2007. Losses in T&D contributed between 5% in 2007 and 11% in 2013 to the environmental impacts. The transmission grid infrastructure had a negligible contribution to the environmental impacts (less than 0.8%). The distribution grid represented less than 4.5% of the impacts. Differences between generation (see Table 4.5) and supply (see Table 4.6) impacts varied between 3 and 13%. Similarly to what is verified for the electricity generation mix, the 2005 mix presented the highest impacts per kWh in all categories, and 2010 the lowest impacts (except OD). An overall reduction of impacts from electricity supply between 2003 and 2014 was also verified (80–81% in AC and PO; 32–36% in NREn, AD, GW, EUT and OD).

Table 4.6 Life-cycle impacts per kWh of the Portuguese annual electricity supply mix (2003-2014).

Impact categories	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
nREn ($MJ_{prim\ fossil}$)	6.67	7.46	**8.47**	6.72	6.30	6.44	6.09	<u>4.35</u>	5.08	5.72	4.70	4.42
AD (g Sb eq)	4.04	4.47	**4.91**	4.01	3.72	3.73	3.59	<u>2.46</u>	2.98	3.53	2.92	2.75
GW (g CO_2 eq)	533	585	**646**	523	480	477	459	<u>312</u>	380	456	376	355
AC (g SO_2 eq)	4.76	4.67	**5.74**	3.91	3.43	1.37	1.17	<u>0.69</u>	0.88	1.22	0.97	0.93
PO (g C_2H_4 eq)	0.17	0.17	**0.21**	0.14	0.13	0.05	0.04	<u>0.03</u>	0.03	0.04	0.03	0.03
EUT (g PO_4^{3-} eq)	1.01	1.04	**1.06**	0.92	0.80	0.72	0.74	<u>0.43</u>	0.60	0.86	0.70	0.67
OD (µg CFC-11 eq)	22	26	**36**	24	24	27	24	21	21	18	15	<u>14</u>

nREn: non-renewable fossil energy; AD: abiotic depletion; AC: acidification; EUT: eutrophication; GW: global warming; OD: ozone layer depletion; PO: photochemical oxidation.
In each line, values in bold indicate the highest value and underlined values the lowest value.

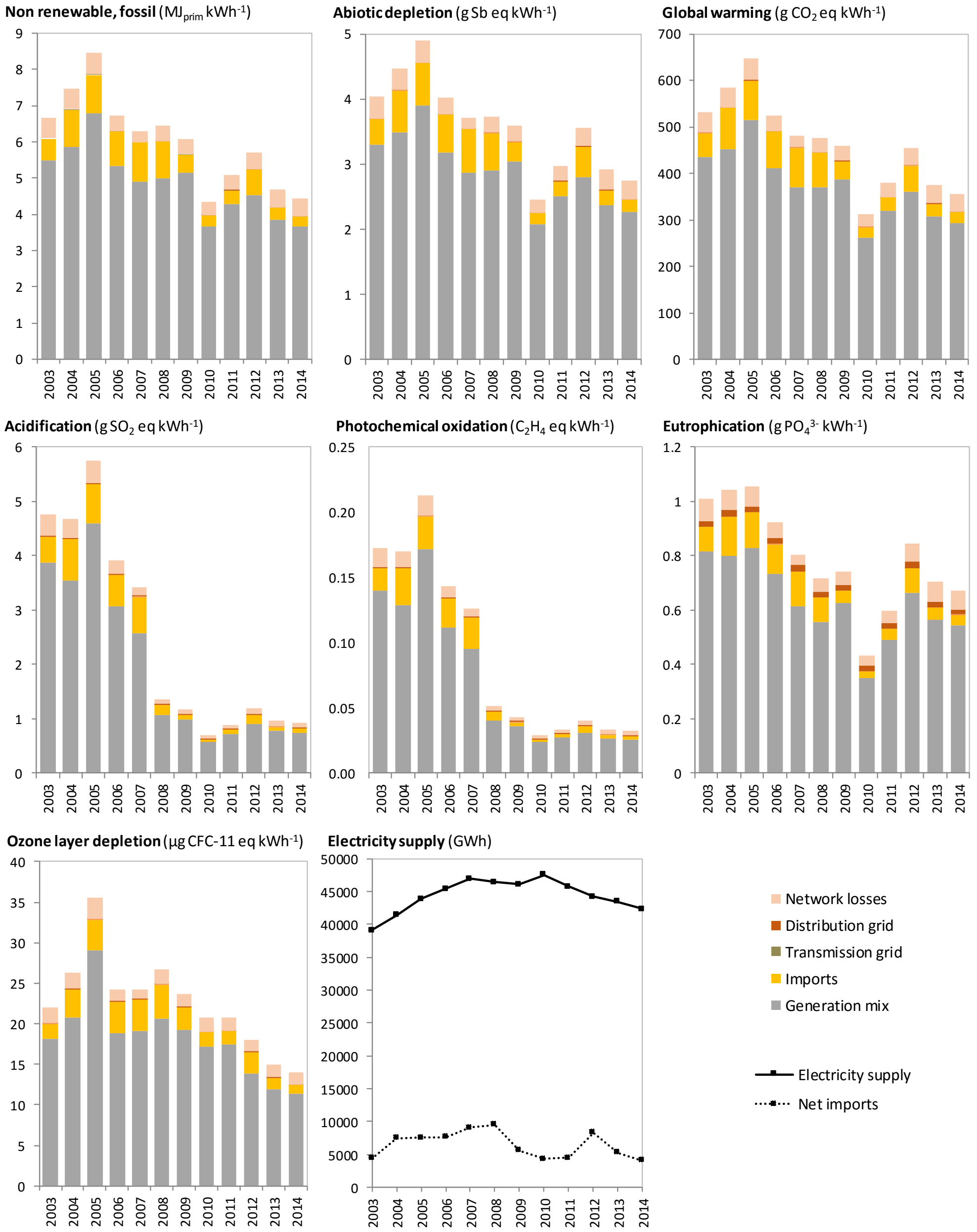

Fig. 4.3 Life-cycle impact assessment results of the annual electricity supply mix in Portugal (2003-2014).

4.1.5 Concluding remarks

A comprehensive life-cycle assessment of electricity generation and supply in Portugal from 2003-2014 was performed. An overall reduction in the environmental impacts was achieved between 2003 and 2014. The higher reductions were realized in AC and PO (-81%). The decommissioning of large fuel oil power plants and the installation of denitrification and desulfurization systems in coal power plants, which took place since 2008, had a significant effect in AC and PO impacts, by decreasing NO_x and SO_2 emissions (largely responsible for AC and PO impacts), despite decreasing the power plant efficiency from 37.5 to 36% and increasing upstream impacts due to ammonia production, particularly in OD (about 24%). For other impact categories, reductions varied between 33–39% for the generation mix and 32–36% for the supply mix. Reduction of impacts was mostly driven by the increase in renewable energy share. The T&D grid added 5-14% to the environmental impacts due to infrastructure (<5%) and T&D losses (5-11%).

Despite the growth in renewable capacity, the overall renewable energy generation was very variable in the last five years (decreasing 29% from 2010 to 2012 and increasing 30% in 2012-2014), due to the variability in hydro power generation, influenced by meteorological factors. Hydro variability between wet and dry years is still a limiting factor for the share of renewables in the mix and, therefore, for higher reductions in environmental impacts. In order to increase renewable penetration, further implementation of reversible or pumped hydropower plants combined with wind power is a key aspect. Moreover, the growth in NGCC installed capacity, which aimed at ensuring security of electricity supply in the medium and long term, did not translate into an increase in electricity generated by this source. Instead, in recent years, there has been a sharp decrease in the use of NGCC to the detriment of coal, with consequences regarding environmental impacts. In 2010, almost 70% of electricity from non-renewable sources was generated using natural gas; however, this value dropped to 44%, in 2014. If this trend continues, the contribution of the increasing renewable energy penetration to reduce environmental impacts is likely to be partly offset by the use of coal (e.g., in 2013 and 2014, the renewable share was higher than in 2010, but environmental impacts were higher, due to the higher share of coal). There is, however, potential to further reduce environmental impacts in key categories (NREn, AD and GW) through the exploitation of currently underutilized NGCC installed capacity.

4.2 Addressing temporal variability in the life-cycle assessment of electricity[6]

4.2.1 Introduction

The electricity generation mix typically varies yearly, seasonally, monthly, daily and hourly, depending on a number of factors (e.g., total demand, atmospheric conditions, fuel prices, technology portfolio). Correspondingly, environmental impacts associated with electricity generation and consumption also change over time making their assessment a difficult task. As the electricity mix changes, the environmental profile of electricity-using products, such as EVs, may also change over time. For EVs, this means that the timing of charging can determine their environmental impacts.

This section aims to assess the variability in environmental impacts (focusing on GHG emissions) associated with generating 1 kWh of electricity throughout the year in Portugal and how this variability affects the environmental impacts of a BEV as a function of the time of charging. Different electricity mixes were assessed (annual, monthly, and hourly) using data for 2012, 2013, and 2014. A comparison between a BEV and gasoline and diesel internal combustion engine vehicles (ICEV) was also performed (functional unit: 1 km, assuming a vehicle service life of 200 000 km), in order to assess the influence of the temporal variability in the results.

In this assessment, historic data of electricity generation and supply for recent years in Portugal retrieved from REN (2015) was used and BEVs were assumed to be part of the total load of the system. Therefore, a portion of the emissions of every power plant operating at a given hour were 'assigned' to the BEV based on the proportion of BEV demand (attributional approach). This is different from assessing the marginal change in electricity emissions due to EV charging (consequential approach), which is the aim of Section 5.1.

[6] Significant portions of this section appear in the article: Rangaraju S., Garcia R., De Vroey L., Marques P., Messagie M., Freire F., Van Mierlo J. (2015). Key parameters influencing the results of life cycle assessment of battery electric vehicles (in final preparation for submission to an ISI-indexed journal).

4.2.2 Temporal aspects of the Portuguese electricity mix

The hourly electricity generation in Portugal varies throughout the year: higher in the winter and lower in the summer; higher on weekdays and lower on weekends; higher during the day and lower during the night. In 2013, 5626 MWh of electricity was generated in each hour, on average. The maximum electricity generation per hour was 8296 MWh and the minimum 3550 MWh. The Portuguese electricity mix incorporates a high contribution from renewable energy sources (RES) (about 50-64%, in 2012-2014), in particular hydro and wind. The majority of fossil generation is based on coal (23-30%) and natural gas (11-24%). A high variability in the electricity mix between hours, months and seasons, and even years, as discussed in Section 4.1, is observed. One of the reasons for this variability is the intermittence of RES and the hydro dependence on meteorological conditions, which causes a variation in the availability of these energy sources.

Fig. 4.4 shows the contribution of different energy sources to the monthly electricity mix in 2012-2014. The years 2013 and 2014 presented higher hydro (1.17 and 1.27, respectively) and wind indexes (1.18 and 1.11, respectively) than 2012 (0.47 and 1.04, respectively), which explains the higher share of RES during the winter and spring. A higher variability in the share of energy sources between months and years in the first half of the year is observed, as a result of the variability in hydro availability. In the second half of the year, the share of energy sources does not change significantly from year to year.

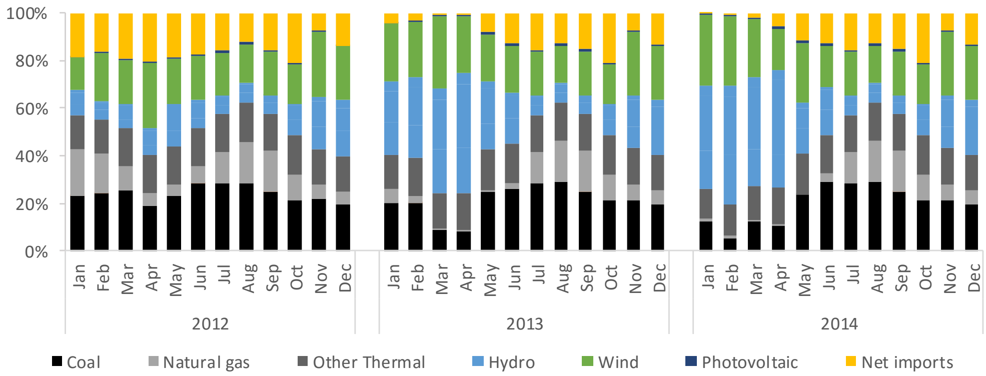

Fig. 4.4 Contribution of energy sources to the monthly electricity mix in Portugal in 2012-2014.

Fig. 4.5 and Fig. 4.6 show the contribution of energy sources to the hour electricity mix in a winter and summer weeks in 2013, respectively. In the winter week, a large share of the electricity is generated by RES during the night, to which coal and natural gas generation is added during the day to meet the additional demand. On the other hand, in the summer week, a high and almost constant generation of coal-based electricity and low hydro generation is observed, and significant imports of electricity from Spain are needed in order to meet demand. How this variability in the electricity generation from each energy source affects the environmental impacts of the electricity mix and, consequently, the battery electric vehicle impacts, will be analyzed in the next sections.

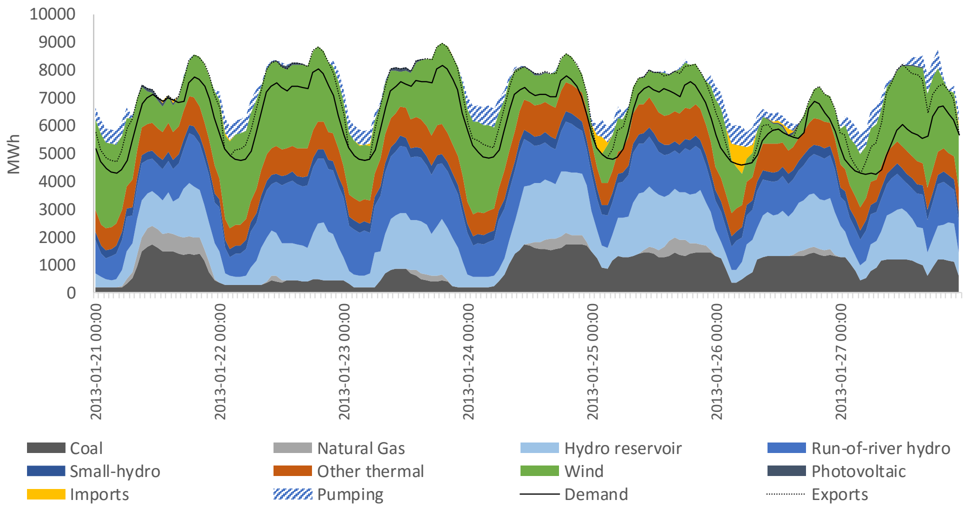

Fig. 4.5 Contribution of energy sources to the hourly electricity mix in Portugal in a winter week (21 to 27 January 2013). Consumption for pumping added on top of generation.

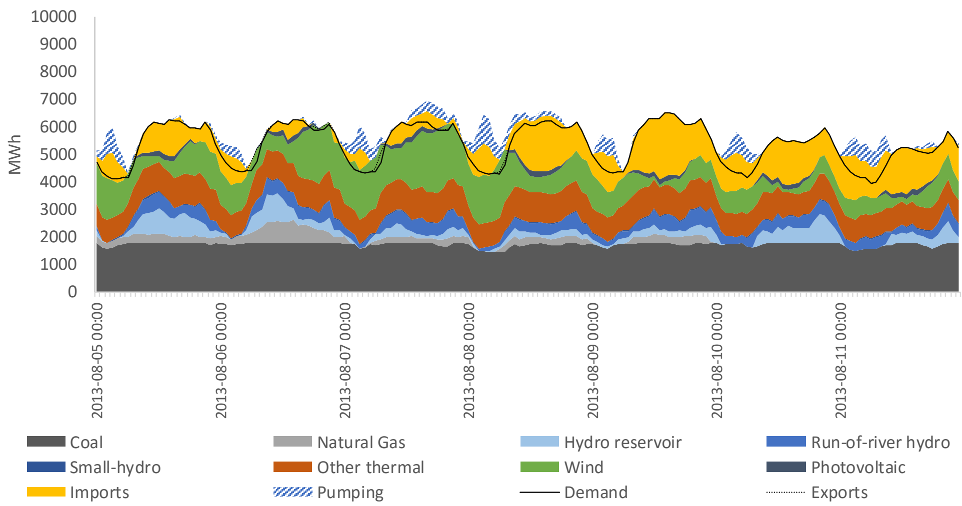

Fig. 4.6 Contribution of energy sources to the hourly electricity mix in Portugal in a summer week (5 to 11 August 2013). Consumption for pumping added on top of generation.

4.2.3 Materials and methods

Two life-cycle models were developed based on an attributional and process-based approach: (i) an hourly life-cycle model of electricity generation in Portugal, and (ii) a life-cycle model of a battery-electric vehicle as a function of the time of charging. The temporal scope is 2012-2014 and the geographical scope is Portugal mainland. Greenhouse gas (GHG) emissions were calculated using the IPCC 2007 methodology (IPCC 2007).

4.2.3.1 Hourly life-cycle model of electricity generation in Portugal

For the hourly life-cycle assessment of electricity generation, the LC models and inventories previously developed for the main electricity generation systems and the transmission and distribution (T&D) grid in Portugal were used (see Section 4.1). The functional unit is 1 kWh of electricity. A similar calculation procedure to that developed in Section 4.1 was implemented. However, instead of calculating the annual average mix impacts, hourly electricity generation data for the years 2012 to 2014 was used to calculate impacts per hour during that period. Hourly data was retrieved from REN (2015). Electricity imports were modeled using the Spanish annual mix, because hourly data was not available. This limitation may be particularly important for hours with a significant share of imports and should be taken into account when analyzing the results. Variations in T&D efficiency were not accounted for due to data unavailability.

4.2.3.2 Battery electric vehicle life-cycle model and inventory

Regarding the BEV life-cycle model, a compact passenger car (vehicle weight: 1525 kg; electricity consumption: 140 Wh km^{-1}; battery type: $LiMnO_4$; battery capacity: 24 kWh; energy density: 114 Wh kg^{-1}; battery weight: 300 kg) was used as reference, and impacts per km were assessed taking into account the complete life cycle of the vehicle (assuming a service life of 200 000 km). The LC modelling of the BEV comprised the stages of production of raw materials, manufacturing and distribution of components, assembly, operation, maintenance and repair of the vehicle, and its end-of-life. The operation stage included electricity generation, transport, and distribution, based on the hourly life-cycle model of electricity generation presented in Section 4.2.3.1. A simplified diagram of the system boundary is shown in Fig. 4.7. In order to account for variability in electricity consumption, a range of values (105–214 Wh km^{-1}) was considered to reflect different operation conditions (e.g., speed, climate control), based on Faria et al. (2014). The LC inventory (LCI) of the vehicle was compiled based on M. Spielmann et al. (2007) and Marques et al. (2013), and of the $LiMnO_2$ battery on Notter et al. (2009). LCI data for the materials and processes in the background system were taken from ecoinvent (2007). Impacts from BEV operation were modeled as a function of the time of charging, assuming that the vehicle is charged in a normal charger (single phase, 16–32 A, 95% efficiency) for about 8 hours (the sensitivity of results to a 2-hour charge was also analyzed). LCI data for the charger was retrieved from Lucas et al. (2012).

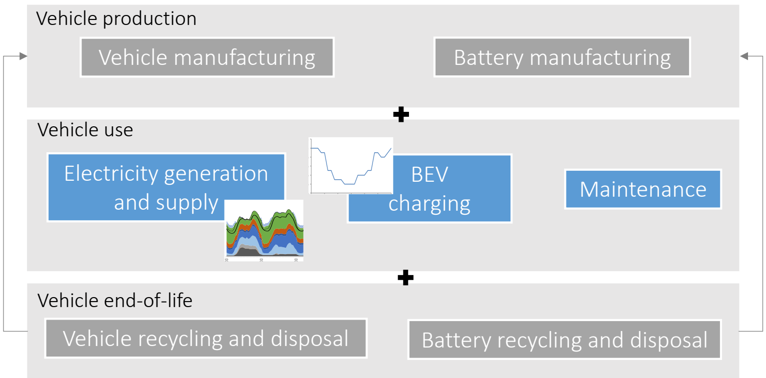

Fig. 4.7 System boundary for BEV life-cycle assessment.

4.2.3.3 *Internal combustion engine vehicle life-cycle model and inventory*

For the internal combustion engine vehicle (ICEV) life-cycle model, compact family cars with similar dimensions to the BEV were selected, but with different weights (gasoline ICEV – vehicle weight: 1058 kg; gasoline consumption: 5.2–6.1 L 100 km^{-1}; diesel ICEV – vehicle weight: 1058 kg; diesel consumption: 4.5–5.3 L 100 km^{-1}). In order to account for the variability in fuel consumption, a 17% increase in comparison to the values indicated by the car manufacturer was considered to reflect a range of operational condition, based on Nemry et al. (2008). The LC modelling of the ICEVs included the same stages as the BEV (except battery manufacturing and disposal). Regarding the fuel LC, the stages of crude oil extraction and refining, distribution, and the fuel combustion in the ICEV were taken into account and modeled in the same way as explained in Section 3.1.2.3. The functional unit was 1 km, assuming a vehicle service life similar to the BEV (200 000 km). GHG emissions from ICEVs typically do not depend on the time of energy use; thus, no temporal variation was considered.

4.2.4 Results and discussion

4.2.4.1 Hourly-average emissions from electricity

The hourly variability in electricity greenhouse gas (GHG) emissions per month in Portugal in 2012–2014 is shown in Fig. 4.8. The electricity mix had lower GHG emissions during the winter and spring when there was a high availability of wind and hydro, as in 2013 and 2014 (see Fig. 4.4), and higher emissions in the summer, in general. A higher variability in GHG emissions between months and years in the first semester was observed, as a result of the variability in hydro generation. In the second semester, smaller variations were perceived between years, as the share of energy sources remained nearly constant. Variations in GHG emissions between years in those months were mostly due to variations in the Spanish electricity mix (imports). The highest monthly average GHG emissions was 533 g CO_2 eq kWh^{-1} (August 2012), more than four times the lowest value: 126 g CO_2 eq kWh^{-1} (February 2014). A large spread in hourly GHG emissions in each month was also observed, with an interquartile range varying between 51 g CO_2 eq kWh^{-1} (February 2014; median: 208 g CO_2 eq kWh^{-1}) and 189 g CO_2 eq kWh^{-1} (December 2012; median: 383 g CO_2 eq kWh^{-1}). A minimum close to 75 g CO_2 eq kWh^{-1} was achieved in January-March 2014, suggesting that the lower bound for the system GHG emissions was reached.

GHG emissions associated with electricity consumption of an activity occurring mostly during the summer (e.g., air conditioning) would vary little between years compared to an activity occurring mostly on the first half of the year (e.g., heating) (see Table B-5, Appendix IV). Nevertheless, GHG emissions from electricity in the summer months would likely be above the annual average mix emissions, and using the latter to assess impacts, the most common approach, would underestimate the impacts from that activity. On the other hand, for an activity occurring in the winter, GHG emissions from electricity would more likely be below the annual average mix, if the hydro index is high, and impacts would be overestimated if the latter mix is used. For seasonal electricity-using activities, using a LCA approach that disaggregates impacts over time is key to obtain a more accurate assessment of electricity impacts and optimize the environmental performance of these systems.

Because the focus is on comparing BEV GHG emissions as a function of the time of charging, it is important to understand how GHG emissions in each hour of the day change along the year. Fig. 4.9 shows the monthly variability in electricity GHG emissions by hour in 2013. Emissions varied significantly between months (65-71% variation of the mean), with higher variations occurring at night (12 PM to 6 AM). On the other hand, the hours comprised between 9 AM and 11 PM had a similar pattern of variation along the year, as well as the hours between 12 PM and 8 AM. This means that consistently charging the BEV at the same hour will lead to similar emissions, on average, within those periods (for a year). Differences between those periods (less than 22%) is not as high as the difference between months, thus the variable month has a higher effect in the electricity GHG emissions variability than the variable hour.

Taking a closer look at how emissions vary over the day, Fig. 4.10 shows the variation in hourly GHG emissions for two representative winter and summer weeks in 2013 (the same weeks depicted in Fig. 4.5 and Fig. 4.6). GHG emissions ranged between 96–385 g CO_2 eq kWh^{-1}, in the winter week, and 439–691 g CO_2 eq kWh^{-1}, in the summer week. It should be noted that, because a significant share of electricity in the summer week came from imports, GHG emissions may be under- or over-estimated, depending on how the Spanish electricity mix at the time diverged from the annual average. Despite the higher range, GHG emissions in the winter week had lower variations along the day (expect on Monday and Thursday). In the summer week, emissions varied along the day in a similar pattern along the week: they had a peak during the night (at about 1 AM) and a valley in the late afternoon (at about 6 PM). Interestingly, emissions behavior was opposed to that of electricity demand, which was lower during the night and higher during the day (see Fig. 4.6). Because coal generation was almost constant along the summer week, the share of coal in the electricity mix in periods of low demand was higher, thus increasing electricity GHG emissions. Electricity GHG emissions at night were up to 50% higher than during the day. For the winter week, no distinguishable pattern can be observed.

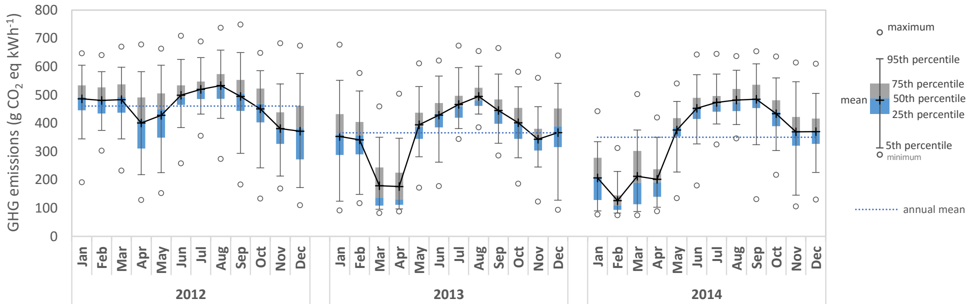

Fig. 4.8 Hourly variability in electricity greenhouse gas (GHG) emissions by month in Portugal in 2012-2014.

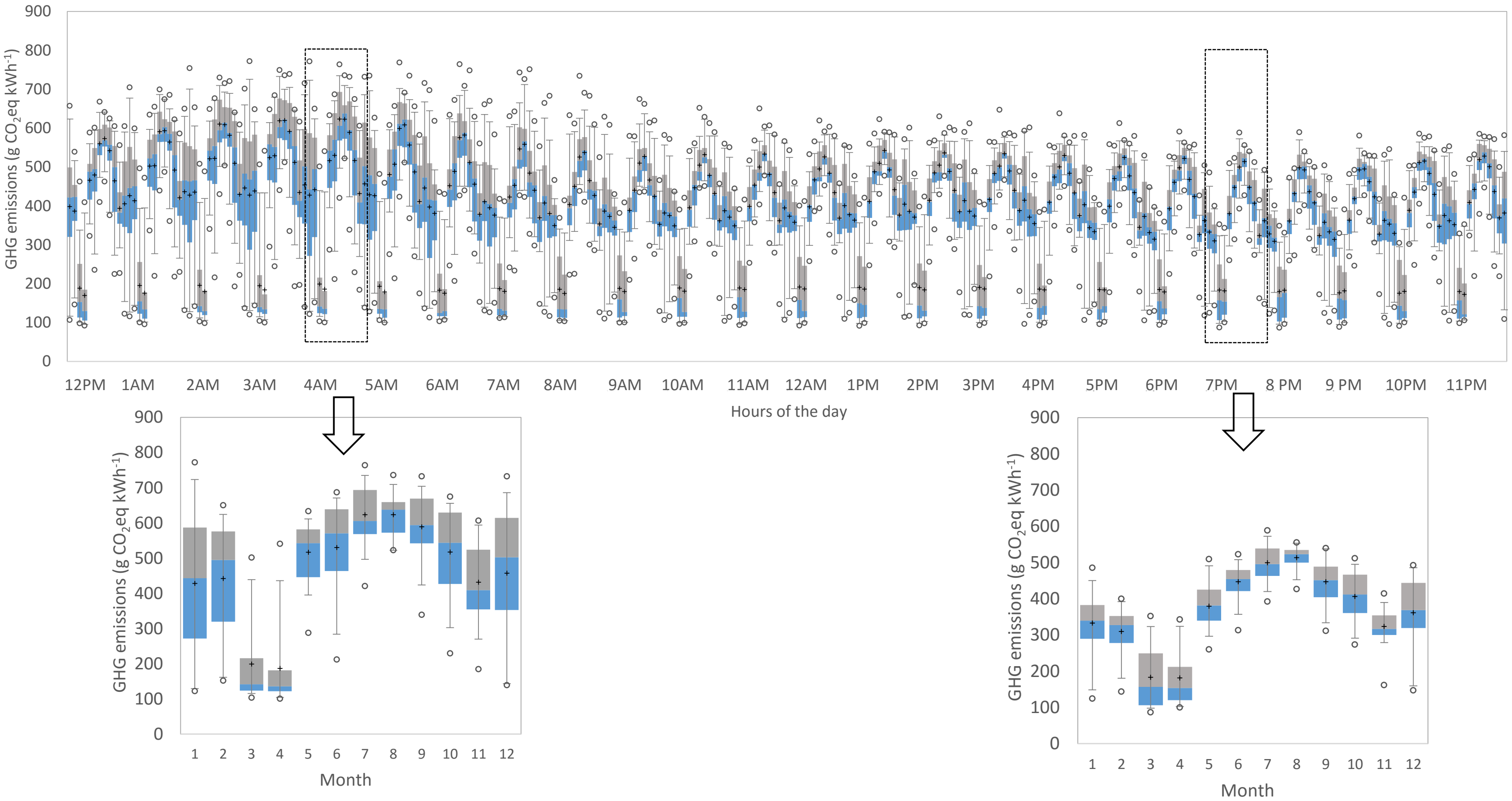

Fig. 4.9 Monthly variability in electricity greenhouse gas (GHG) emissions by hour (2013, Portugal). Each boxplot represents a month (January to December) in each hour. Error bars show the 5th and 95th percentiles of hourly emissions.

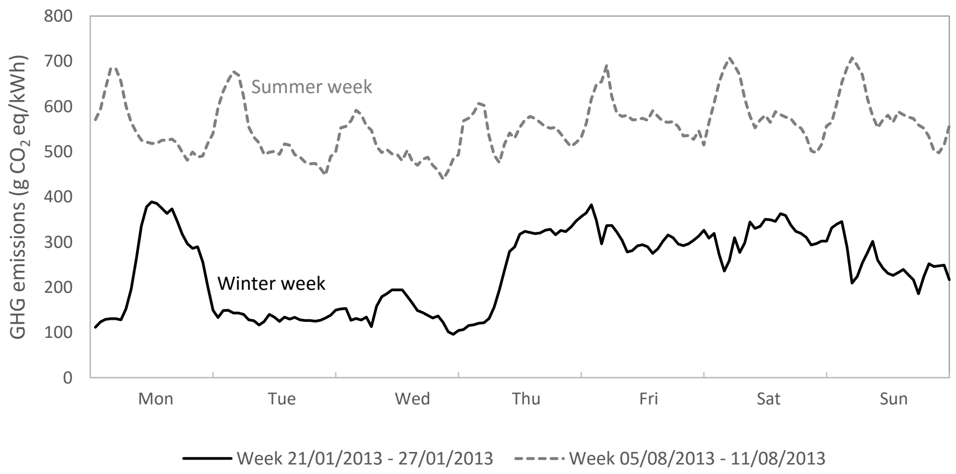

Fig. 4.10 Hourly greenhouse gas (GHG) emissions of a winter (21-27 January 2013) and summer (5-11 August 2013) weeks in Portugal.

4.2.4.2 Environmental impacts of battery electric vehicles as a function of the time of charging

Using the hourly life cycle model of electricity generation developed, the life-cycle GHG emissions of a BEV as a function of the time of charging were assessed. Fig. 4.11 shows the life-cycle GHG emissions per km of the BEV considering the hourly mix in each month (i.e. emissions in each hour of the day averaged over the month) and the annual mix in 2013 compared to the ICEVs (gasoline and diesel). Results for the hourly mix represent the average life-cycle GHG emissions of a BEV whose battery started charging in each hour of the day and charged for the next 8 hours (full charge) (e.g., if a vehicle starts to charge at 1 AM, it is assumed that it will continue charging for the next 8 h, until 9 AM, so GHG emissions from the operation of the BEV are the average GHG emissions for that 8-hour period). The range represents the variability in electricity or fuel consumption of the different vehicles.

GHG emissions of a BEV operating in Portugal may vary significantly throughout the year (lower bound: 63–107 g CO_2 eq km^{-1}; upper bound: 82–172 g CO_2 eq km^{-1}), with a maximum difference of 41%, considering the lower bound, and 52%, for the upper bound. When considering the annual mix to assess GHG emissions from BEVs (86–130 g CO_2 eq km^{-1}) this variability is not account for, and it becomes more relevant as consumption increases. The difference between the lower and the upper bound in each hourly scenario varied between 24-38%.

Regarding the comparison with ICEVs, the BEV presented lower emissions, on average, than the ICEVs, if the hourly mix was used, except in some scenarios in the summer months (which corresponds to a likely higher electricity consumption due to the use of air conditioning), in which similar impacts to the diesel ICEV were obtained. However, if the annual mix is used, the BEV would have lower impacts than the ICEVs (less 26-46% than the diesel ICEV and 35-51% than the gasoline ICEV).

Often in the literature, it is claimed that nighttime (referred here as off-peak) is the most favorable period to charge a BEV (Stephan and Sullivan 2008; Samaras and Meisterling 2008; Faria et al. 2013; Rangaraju et al. 2015), because it allows thermal generation units to have a more evenly distributed load and reduces wind power curtailment (Saunders et al. 2014). However, from a GHG standpoint, that may not be the case, as demonstrated by the results in Fig. 4.12 for 2013. In a scenario in which a BEV was fully-charged (8-hour charge) consistently at the same time, off-peak charging (i.e. charging between 12 PM and 8 AM) resulted most of the time in higher average emissions than peak charging (i.e. charging between 2 PM and 10 PM), although the cumulative difference was below 14%. This trend holds irrespective of the frequency of charging. If the charging period is decreased to 2 h per day (the "real" charging time would likely vary between 2 and 8 h, depending on the state-of-charge of the battery after each trip), the cumulative difference between off-peak (i.e. from 3 AM to 5 AM) and peak (i.e. 9 PM to 11 PM) charging emissions could reach 20%. The reason for the higher impacts of BEV charging during off-peak hours lay on the higher relative share of coal in the electricity mix during the night, when demand is the lowest.

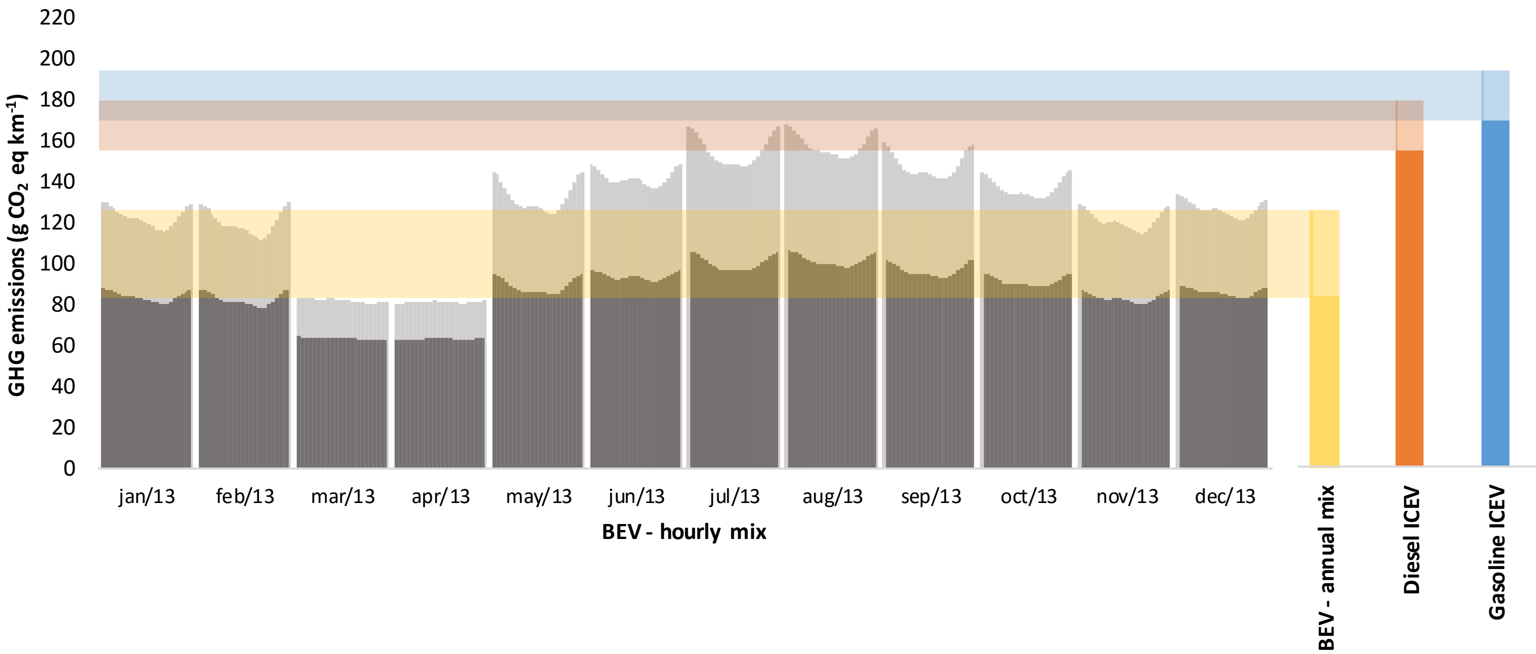

Fig. 4.11 Life-cycle greenhouse gas (GHG) emissions per kilometer of a battery electric vehicle (BEV) as a function of the hour of charging in each month (BEV – hourly mix) compared to the annual electricity mix (BEV – annual mix) and diesel and gasoline internal combustion engine vehicles (ICEVs). Results for Portugal, in 2013. Lighter areas correspond to the variability range in the energy consumption of each vehicle. Vehicle production, maintenance and end-of-life represent about 44 g CO_2 eq km^{-1} (BEV) and 15 g CO_2 eq km^{-1} (ICEVs) of the total LC GHG emissions.

Fig. 4.12 Comparison of the life-cycle greenhouse gas (GHG) emissions (in g CO_2 eq km^{-1}, left-hand axis) of a BEV (operation only) operating in Portugal throughout 2013 charged at off-peak and peak hours for 8 h (top) and 2 h (bottom). BEV electricity consumption is assumed to be 188 Wh km^{-1}. Percentage difference between off-peak and peak charging is also presented (right-hand axis).

4.2.5 Concluding remarks

The electricity mix changes over time and so do GHG emissions associated with electricity generation and consumption. Seasonal differences were found to be more significant than differences between hours along the year. This is mostly explained by the heavy reliance on hydro power, which is very dependent on meteorological conditions (more favorable in the winter), and, to a smaller extent, the variation in wind generation. Hydro availability also explains the difference in emissions between the last three years, since only small changes to the electricity technology portfolio occurred.

Using an hourly LCA of electricity provides a more accurate picture of the BEV impacts and the opportunities for improvement regarding the time of charging. A prescribed nighttime charging profile incentivized by a lower electricity pricing during off-peak hours (i.e. at night), for instance, will not necessarily lead to lower GHG emissions. A BEV operated in Portugal in the last three years would have lower life-cycle GHG emissions if charged during the day, in most cases, even though the cumulative difference was lower than 20%.

Accounting for the temporal variability in electricity GHG emissions may be more important for seasonal activities than for activities occurring during all year for a certain period of the day, such as BEV charging. Although the timing of electricity-use could be managed to correspond to periods of lower emissions, this could also lead to moving electricity consumption to periods of typically higher pricing (peak hours) and operationally critical from the system operator standpoint. Decreasing electricity GHG emissions during the summer by reducing coal generation could, on the one hand, reduce variability within the year and, on the other hand, reduce the influence of the time of charging in the GHG emissions of BEVs.

In this assessment, it was assumed that BEVs were part of the total load of the system and the goal was to assess the life-cycle GHG emissions of a kilometer driven by a BEV charged at different hours of the day in Portugal in the last three years, which is an attributional question. The next chapter will address how GHG emissions will change as a result of e introduction of BEVs in the Portuguese LDV fleet in the short- and long-term, which is a consequential question. Instead of average emissions from the electricity system, the focus is rather on marginal changes in emissions, i.e. on how the electricity system respond to an increase in demand for electricity due to a new fleet of BEVs.

Chapter 5

GHG CONSEQUENCES OF THE ADOPTION OF ELECTRIC VEHICLES

Abstract This chapter analyses the combined effects of EV adoption on the electricity system and on the displacement of ICEVs in the fleet, in order to understand the influence of the time of charging in the GHG emissions induced by EVs (Section 5.1) and the effects on GHG emissions of the interaction between EVs and electricity storage (Section 5.2). A dynamic fleet-based LCA framework is implemented, combining the dynamic fleet-based life-cycle model developed in Chapter 3 to assess displacement effects, with consequential LCA of electricity to assess effects on the operation of the electricity system, building on the life-cycle models of electricity generation in Chapter 4. In Section 5.1, a consequential life-cycle model of the Portuguese electricity system is implemented to assess hourly marginal emissions in the short-term and the effect of different BEV charging times and displacement options in GHG emissions. Section 5.2 explores the interactions between BEV and pumped hydro storage (PHS) and compares changes in GHG emissions considering different scenarios of technologies displaced by both BEVs and PHS in Portugal.

5.1 Short-term GHG consequences in the Portuguese electricity system

5.1.1 Introduction

The adoption of a new technology entails changes to the existing system. One of the effects of electric vehicle adoption is the displacement of conventional technologies, which was addressed in Chapter 3. Another effect that is important to analyze in light of the interaction between EVs and the existing system is the effect on the electricity demand patterns and, as a result, on the electricity system (Hedegaard et al. 2012).

Shifting from conventional vehicles to electric vehicles means shifting the energy source used for transportation, from mainly petroleum-based fuels to electricity (i.e. a mix of different energy sources). The adoption of EVs results in an increase in electricity demand

and a change in the grid load profile. How the electricity system will change in order to accommodate the new load and, consequently, how emissions from electricity will change is an indirect effect of EV adoption with implications in the environmental assessment of EVs. The nature and extent of the change is dependent on the timeframe of the analysis (short- versus long-term), on the scale of the intervention (EV penetration level) and on the charging management (controlled versus uncontrolled charging) (McCarthy and Yang 2010).

In the short-term (few years), the impact of EVs on the overall electricity demand is low, as EV penetration is low. For instance, in Germany, 1 million EVs, which is about 2% of the fleet, would require less than 1% of the total electricity demand (Schill and Gerbaulet 2015a); in Portugal, a BEV fleet share of 2% (about 100 thousand BEVs – 100 times more than the number of BEVs sold in Portugal until August 2015) would require less than 0.5%. These can be considered marginal changes in demand deemed to be accommodated by existing capacity and thus entail marginal changes in the operation of that capacity (Yang 2013). However, whilst electricity demand by EVs may be small compared to overall demand, hourly loads vary over time and can be rather high, depending on the charging strategy implemented (Schill and Gerbaulet 2015a).

Depending on the time of day and system load, the response of the electricity system to EV charging can be distinct, thus influencing GHG emissions. Therefore, it is important to understand how marginal emissions from electricity generation vary over time so that charging strategies can be designed to minimize environmental impacts from EVs. This section aims to assess the change in GHG emissions resulting from: (i) increasing electricity demand by 1 kWh in Portugal, and (ii) introducing BEVs in the Portuguese light-duty fleet in the short-term. A consequential life-cycle model of the Portuguese electricity system was developed and marginal emission factors were calculated. These provide a consistent metric to assess the short-term effect of the introduction of electric vehicles and other interventions. Hourly generation data and corresponding emissions from 2012 to 2014 was used to estimate marginal GHG emissions. Trends in marginal emissions regarding electricity demand, time of day, and month were explored, and a comparison between average and marginal emissions provided. The model was then applied, within the dynamic fleet-based LCA framework, to assess the effects of the introduction of BEVs in Portugal, for a range of displacement and charging scenarios.

5.1.2 Consequential life-cycle model of the Portuguese electricity system

A consequential life-cycle model of the Portuguese electricity system was implemented to assess the change in GHG emissions resulting from an increase in demand for electricity by 1 kWh in the short-tem. The next sections describe how system boundary was defined, as well as the method used to assess marginal supply and corresponding marginal emissions.

5.1.2.1 System boundary and identification of unconstrained technologies

The Portuguese electricity system was described in Chapter 4 and a life-cycle model was developed based on an attributional approach, whose system boundary included all electricity generators and energy sources contributing to the impacts. However, in a consequential approach (e.g., when the goal is to assess the change in emissions as a result of a change in demand), a change in demand does not affect all elements of the electricity system proportionally and only those generators that are able to respond to the change should be included in the system boundary.

In the short-term, only changes to the operation of the existing capacity are at stake, since the addition of new capacity to satisfy short-term demand is unlikely (building a new power plant implies long-term planning, high investments and may face policy constraints). Currently, there is excess capacity in the Portuguese electricity system (in 2014, total installed capacity was 18 GW – 11 GW dispatchable, 7 GW non-dispatchable – and peak load was 8 GW [REN 2014]), and a diversified set of generators is currently in operation, as described in Chapter 4. From those, constrained and unconstrained generators were identified, and the former excluded from the system boundary. Fig. 5.1 presents the system boundary for the assessment of the impacts of a marginal change in electricity demand.

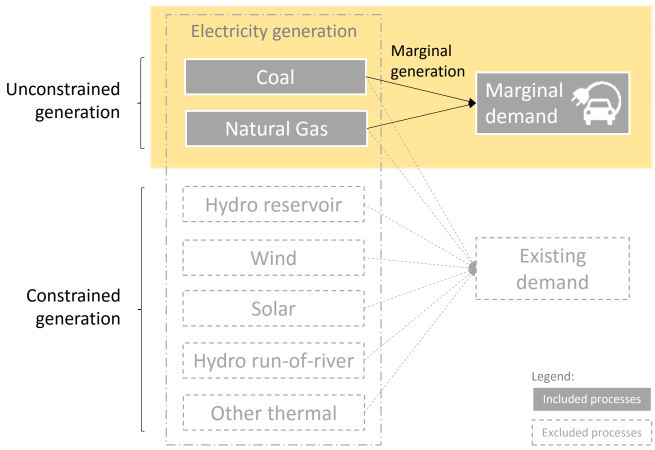

Fig. 5.1 System boundary for the short-term CLCA of the Portuguese electricity system.

The rationale for the identification of constrained and unconstrained generators is explained next. Combined heat and power generators (CHP) as well as biogas and waste incinerators are unlikely to affect marginal emissions because they are constrained as a by-product. Wind, solar, and run-of-river hydro generation occurs irrespective of demand; therefore, a change in demand does not generally have any effect in the output of these generators. There are particular cases when a change in demand can affect the utilization of wind or solar electricity: when surplus wind or solar generation that would be used for pumping, exported or curtailed is rather used to satisfy the additional demand. These effects are not captured in this particular analysis and are addressed in Section 5.2. Hydro reservoir, including pumping, can respond to changes in demand; however, only a fixed amount of water is available annually and total generation tends to be maximized over a long period, making it an energy-constrained resource (Ma et al. 2012). Consequently, only coal and natural gas power plants can respond to changes in electricity demand and take part of the marginal generation. The degree to which they respond to changes in demand and influence marginal emissions thus needs to be assessed. Electricity trading with Spain was not accounted for; therefore, only generators within Portugal were assumed to be on the margin.

5.1.2.2 Determining marginal supply and marginal GHG emissions

A data-driven approach accounting for time dynamics on marginal generators and emissions based on the actual behavior of the system was used to assess marginal electricity supply. The method is based on an analysis of historic generation data of the Portuguese electricity system and builds on Hawkes (2010), which calculated linear regression coefficients of change in the United Kingdom electricity system CO_2 emission rate versus the change in total system demand. The slope of such linear regression gives a good estimate of the average marginal emission factor.

Hawkes (2010) did not distinguish constrained and unconstrained generators and included both types in the assessment of marginal emission factors. The author obtained a good fit for the data considering the change in total system load versus the change in system CO_2 emissions (R^2=0.95), because the share of constrained generation (wind and hydro) was very low compared to unconstrained fossil generation. However, it is questionable whether constrained resources, such hydro, should be included in the assessment as their contribution would occur regardless of any change in demand. This method was then used by Siler-Evans et al. (2012) to estimate CO_2, SO_2 and NO_x emissions for the United States. Siler-Evans et al. (2012), on the other hand, only included large fossil-fueled generators in the assessment primarily due to data constraints. Both studies only assessed direct emissions.

The method applied in this research entails some modifications to Hawkes (2010) method, regarding mainly two aspects: (i) explicitly excludes constrained technologies from the assessment, focusing on how unconstrained generation respond to changes in demand; and (ii) takes a life-cycle perspective, by including fuel supply chain impacts (but excluding infrastructure, because a marginal change in electricity demand is not deemed to affect the existing infrastructure).

The Portuguese electricity system is comprised of a much larger share of renewable resources than any of the above systems (e.g., 50% against 14% in the UK [European Commission 2015] and 12% in U.S., in 2013 [IEA 2015]. Electricity from most renewables (such as wind, solar, and mini-hydro) has priority over electricity from other sources fed into the grid, serving as a kind of variable base load. For this reason, a high share of the change in demand may be randomly satisfied by this variable, non-dispatchable load. The unconstrained technology operation adapts to the variable renewable generation by either filling the need for additional generation, if the change in renewable generation is not enough to meet the additional demand, or reducing generation, if the change in renewable generation exceeds the additional demand. Both scenarios are of interest as to describe what the marginal electricity supply in a certain period of time is, because in both cases the response of the unconstrained technology to a change in the system is depicted. The analysis was thus focused on how unconstrained generation changes hourly and how does that affect marginal GHG emissions.

Hourly generation data for the Portuguese electricity system in 2012-2014 from REN was used (REN 2015) to calculate the change in unconstrained (coal and NGCC) generation (ΔG) and the corresponding change in GHG emissions (ΔE) between one hour and the previous (more than 26000 observations in 2012-2014). GHG emissions were assessed using the technology life-cycle emission factors estimated in Chapter 4 (Section 4.1.4.1), excluding infrastructure impacts (coal PP: 1006 g CO_2 eq kWh^{-1}; NGCC: 420 g CO_2 eq kWh^{-1}) (Garcia et al. 2014). The marginal emission factor corresponds to the slope of a linear regression of ΔE on ΔG, as plotted in Fig. 5.2. Increasing demand by 1 kWh (i.e. increasing fossil generation by 1 kWh) is expected to increase electricity system emissions by, on average, 723 g CO_2 eq, admitting that only fossil generation can change in response to a change in demand.

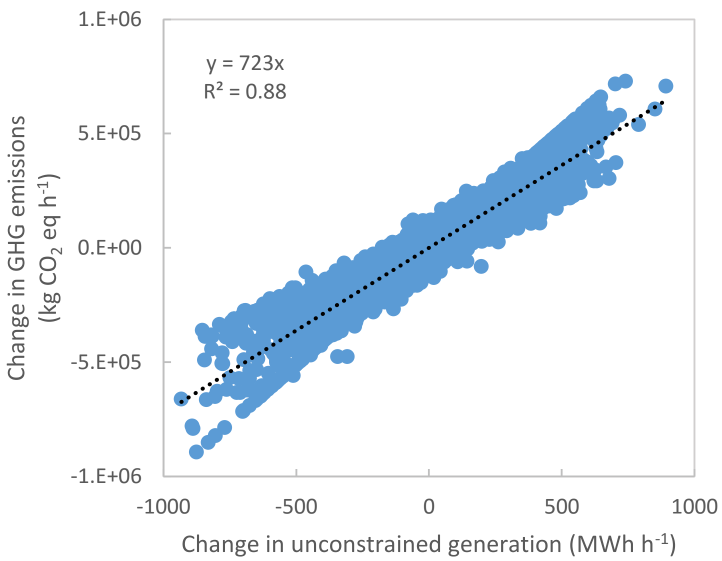

Fig. 5.2 Linear regression of ΔE on ΔG for Portugal from 2012 to 2014. The slope of the regression line gives the marginal GHG emission rate (723 kg CO_2 eq MWh^{-1}).

The marginal emission factor estimated in Fig. 5.2 gives an average result for 2012-2014, but does not provide any insight on the underlying trends of marginal emissions. For instance, how do marginal emissions factors vary with electricity demand, time of day or between months? Trends in marginal emissions factors were explored by applying the method explained above to different subsets of the data, as performed by Hawkes (2010), and Siler-Evans et al. (2012). Regarding electricity demand, marginal emission factors were calculated by disaggregating the data (ΔE and ΔG) by every fifth percentile of the corresponding electricity demand, and performing separate regressions for each set of data. The first set includes the 5% of data occurring at the lowest-demand hours, and the last set the 5% of data occurring during the highest-demand hours. Hourly and monthly marginal emission factors were estimated by performing 24 and 12 separate regressions of ΔE on ΔG for all observations occurring at a given hour and month, respectively.

The degree to which different generators respond to changes in demand (i.e. the share of marginal generation from coal and NG generators) was also assessed using a variation of the above method. The change in fossil generation between one hour and the previous (ΔG) was calculated as well as the corresponding change in coal-based (ΔF_{coal}) and natural gas-based (ΔF_{NG}) generation and then separate regressions of ΔG on ΔF were performed to estimate the share of marginal generation for each fuel.

5.1.3 Results and discussion

5.1.3.1 Marginal electricity supply and GHG emissions

Marginal emissions as a function of total demand

The trend in marginal emissions as a function of total demand is depicted in Fig. 5.3. In low demand hours, coal was the dominant marginal technology, with about 84% share, whilst natural gas dominated at high demand hours with 66% share. The share of coal in marginal generation tended to decrease as load increased. Conversely, marginal GHG emissions also decreased as load increased, varying between 626 and 925 g CO_2 eq kWh^{-1}. There are situations, however, in which marginal emissions may increase during high demand hours, such as in 2014, as a result of a high hydro availability in the winter, when higher loads tend to occur, that satisfies most of the generation, making coal capacity available to feed the margins.

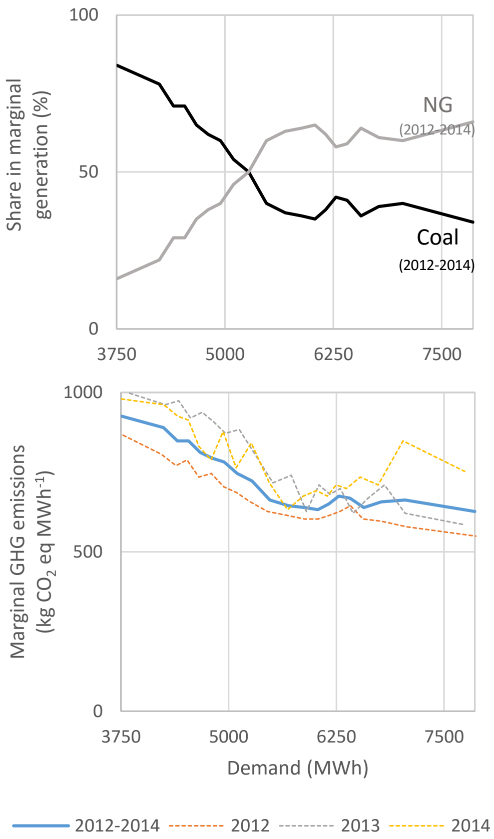

Fig. 5.3 Share in marginal generation and marginal GHG emissions in 2012-2014 as a function of total demand.

Temporal trends

As regards temporal trends (Fig. 5.4), marginal GHG emissions were higher during late-night (1-5 AM), corresponding to the off-peak period, and lower in the early-morning (6-7 AM), corresponding to the beginning of the morning peak, and evening (9-11 PM), corresponding to the declining of the evening peak, with an overall maximum difference of 35%. During day-time, fluctuations in marginal emissions were lower: differences were below 18%. Increasing demand for electricity by 1 kWh at night could result in an additional emission of 943 g CO_2 eq; during the day an additional kWh demanded from the grid could result in an emission of at least 644 g CO_2 eq. Marginal emission rates were higher during spring and fall, and lower in the summer. Between 2012 and 2013, marginal GHG emissions increased 15%, but stabilized in 2014. 2012 was considered a dry year, with less hydro availability; consequently, natural gas was more often on the margin than in 2013 and 2014, both wet years.

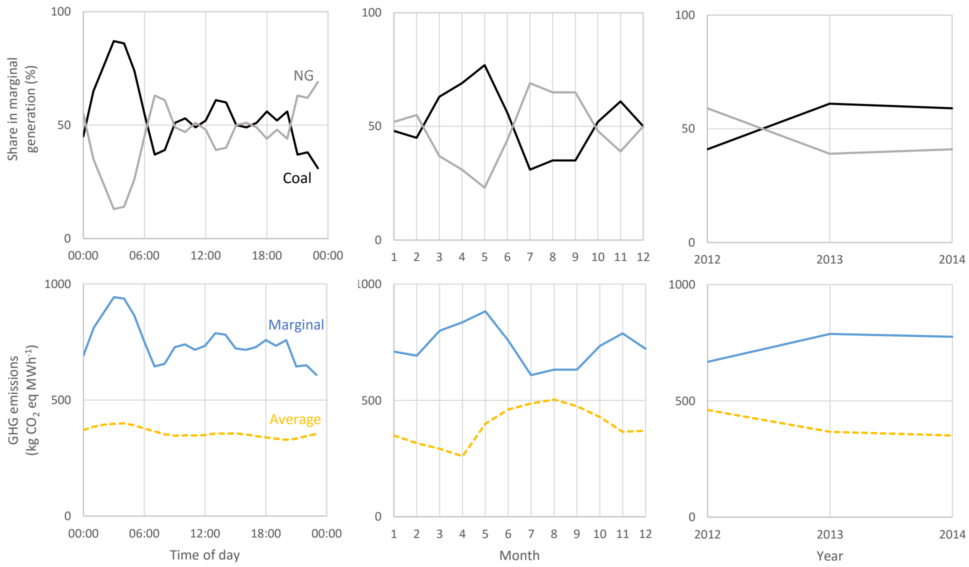

Fig. 5.4 Temporal variations in the contribution of technologies to the marginal generation, and marginal and average GHG emissions, based on data for 2012 through 2014.

Marginal versus average emissions

Marginal emissions were consistently higher than average emissions (42-58% higher considering the time of day), as the latter included high shares of low-carbon renewable sources (Table 5.1). Emissions followed a similar trend along the day (higher during the night and lower during the day), but marginal emissions showed much higher variation (Fig. 5.4). Conversely, marginal and average emissions were negatively correlated on a monthly and annual basis. Further analyzing the data, it is apparent that as the availability of hydro power increases (dry versus wet years; summer versus winter), the utilization of natural gas power plants decreases, due to its high operation costs. As a result, coal is more often on the margin, increasing marginal emissions, but, at the same time, there is more hydro providing power, decreasing average emissions.

Whilst average emissions describe the life-cycle impacts of generating 1 kWh of electricity, marginal emissions depict the life-cycle impacts of increasing electricity generation by 1 kWh. For the Portuguese electricity system, with a high share of non-dispatchable renewable power and excess capacity for the short-term, marginal emissions are considerably higher than average emissions. Increasing electricity generation by 1 kWh means increasing fossil-based generation (either coal or natural gas), resulting in higher emissions than the renewable-based average. Therefore, using average emissions to assess the impacts of implementing a new technology which uses or displaces electricity can underestimate the burdens and the savings achieved, respectively.

Table 5.1 Marginal fuel sources, marginal emission factors and comparison with average emission factors for each hour of the day for electricity generation in Portugal in 2012-2014.

Time of day	**Marginal fuel source**		**Marginal emission factor** (g CO_2 eq kWh^{-1})	**Average emission factor** (g CO_2 eq kWh^{-1})	**% difference**
	Coal (%)	**NG** (%)			
1 AM	65	35	812	385	53
2 AM	76	24	877	394	55
3 AM	87	13	943	397	58
4 AM	86	14	937	400	57
5 AM	74	26	866	391	55
6 AM	55	45	752	377	50
7 AM	37	63	644	366	43
8 AM	39	61	656	352	46
9 AM	51	49	728	347	52
10 AM	53	47	740	348	53
11 AM	49	51	716	347	52
12 AM	52	48	734	349	52
1 PM	61	39	788	356	55
2 PM	60	40	782	356	54
3 PM	50	50	722	356	51
4 PM	49	51	716	351	51
5 PM	51	49	728	345	53
6 PM	56	44	758	339	55
7 PM	52	48	734	334	55
8 PM	56	44	758	329	57
9 PM	37	63	644	333	48
10 PM	38	62	650	345	47
11 PM	31	69	608	354	42
12 PM	45	55	692	371	46

5.1.3.2 Limitations

The marginal emission factors used in this analysis were calculated based on regression models that use historical data on power plant generation and emissions. Whilst these models describe the electricity system historically, they do not capture potential changes in the system over time, and thus can only be used to assess changes in electricity demand in the short-term. Nevertheless, only small changes to the electricity system portfolio are expected to occur in the next few years, and, as new data becomes available, it is possible to regularly update the analysis to reflect the changes in the electricity sector.

The analysis presented in this Section assumed that hydro reservoir is constrained. While the total output of hydro reservoir may be fixed, making it a constrained technology, its distribution along time may vary. An increase in hydro reservoir generation in one hour may result in less hydro power available at some time in the future, and thus in an increase in the use of the marginal generation at that future time (e.g., coal or natural gas). Therefore, hydro reservoir may not be part of the marginal supply, but may influence the operation of marginal generators. By excluding hydro reservoir from the analysis, this shifting effect is not accounted for. While a shift in hydro reservoir generation during day-time (9 AM to 20 PM) will likely have a minor effect in marginal emissions, as marginal emission factors vary little (maximum difference is below 8%), the impact of a shift from night- to day-time (or vice-versa) may be non-negligible. If the shift occurs from night- to day-time, marginal emissions will likely decrease; if the shift is from day- to night-time, marginal emissions will likely increase. As a result, differences between day and night marginal emissions would decrease.

5.1.4 Application to battery electric vehicles

5.1.4.1 Change in electricity GHG emissions due to BEV charging

The effect on GHG emissions of adding BEVs charging to the Portuguese electricity system was assessed for 2012-2017. The consequential model of the Portuguese electricity system developed in Section 5.1.2 was used to assess the effects of BEV charging in the grid. The additional electricity demand from BEVs was calculated using the dynamic fleet-based life-cycle model developed in Section 3.1 (Garcia et al. 2015), considering a projection of BEV penetration until 2017.

The consequential model developed is only valid to describe marginal changes in electricity demand in the short-term; therefore, it is necessary to verify: (i) if the change in demand induced by EVs can be considered marginal as regards total demand from the system, and (ii) if the additional hourly load to the system from BEV charging did not entail changes to the marginal operation of the system depicted in the model (i.e. if the probability of load exceeding remaining capacity of marginal generators is low).

During the period under analysis (2012-2017), the maximum annual amount of electricity requested to the grid was calculated as 11 GWh (average electricity consumption of 175 kWh km^{-1}), corresponding to an addition of less than 0.03% to baseline electricity demand, which was assumed to be a small-enough change to be considered marginal on a yearly basis. If all BEVs were charging at the same time in a 3.3 kW charger (normal charger), the maximum power requested to the grid would be below 36 MW. The remaining capacity of natural gas CC generators was always above 1100 MW in all hours of 2012-2014, while coal PP remaining capacity was below 36 MW in about 25% of the hours (Fig. 5.5). Because the probability of simultaneous charging of all BEVs should be very low, the effect in GHG emissions due to the additional electricity requested to the grid by the BEV fleet may thus be described by the model developed in Section 5.1.2.

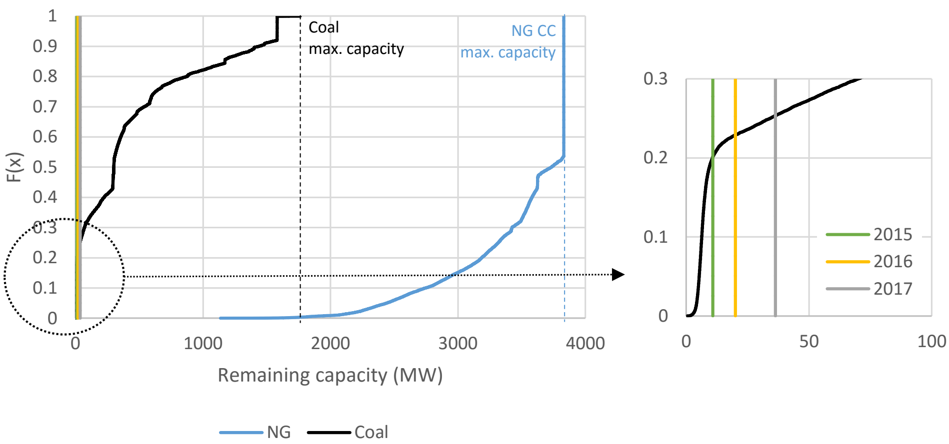

Fig. 5.5 Cumulative probability distribution of remaining coal and NG CC capacity in each hour of 2012-2014 and comparison with maximum BEV load for 2015, 2016 and 2017.

Four generic BEV charging scenarios were analyzed, considering different charging times and durations, as depicted in Table 5.2. The additional emissions resulting from BEV charging were calculated by determining the average additional electricity demand in each hour (assuming an average electricity T&D and charging efficiencies of 92% and 95%, respectively), using the dynamic fleet-based LC fleet model presented in Chapter 3, and applying the marginal emission factor calculated for that hour (Table 5.1). Cumulative emissions from 2012 to 2017 were calculated and are presented in Table 5.3. The addition of BEVs to the Portuguese electricity system in the short-term (2012-2017) would entail a higher increase in GHG emissions if vehicles were charged during off-peak hours. The temporal difference in BEV charging was about 11% for an 8-hour charge, but could reach 32% for a 2-hour charge.

Table 5.2 BEV charging scenarios.

Duration of charging	Charging time	
	Peak	Off-peak
8-h charge	2 PM – 10 PM	12 PM – 8 AM
2-h charge	9 PM – 11 PM	3 AM – 5 AM

Table 5.3 Cumulative increase in electricity GHG emissions (Mton CO_2 eq) as a result of the introduction of BEVs in Portugal in the short-term considering different scenarios for vehicle charging.

Duration of charging	Cumulative change in electricity GHG emissions (Mton CO_2 eq)		Δ
	Peak	Off-peak	
8-hour charge	31.3	35.0	11%
2-hour charge	27.7	40.6	32%

5.1.4.2 Overall change in GHG emissions resulting from the introduction of BEVs in the short-term

In addition to the change in the operation of the electricity system and corresponding change in GHG emissions as a result of the additional electricity demand by BEV charging, the short-term effects of the introduction of BEVs in Portugal also entail the displacement of conventional technologies in the fleet and corresponding change in GHG emissions. Fig. 5.6 presents the system boundary for the assessment of both effects in the short-term. Whilst the effect of the new BEV fleet over the electricity system translates into an increase in emissions (as shown in Table 5.3), the displacement of the ICEV fleet may result in GHG savings depending on how BEVs compare with the displaced technologies.

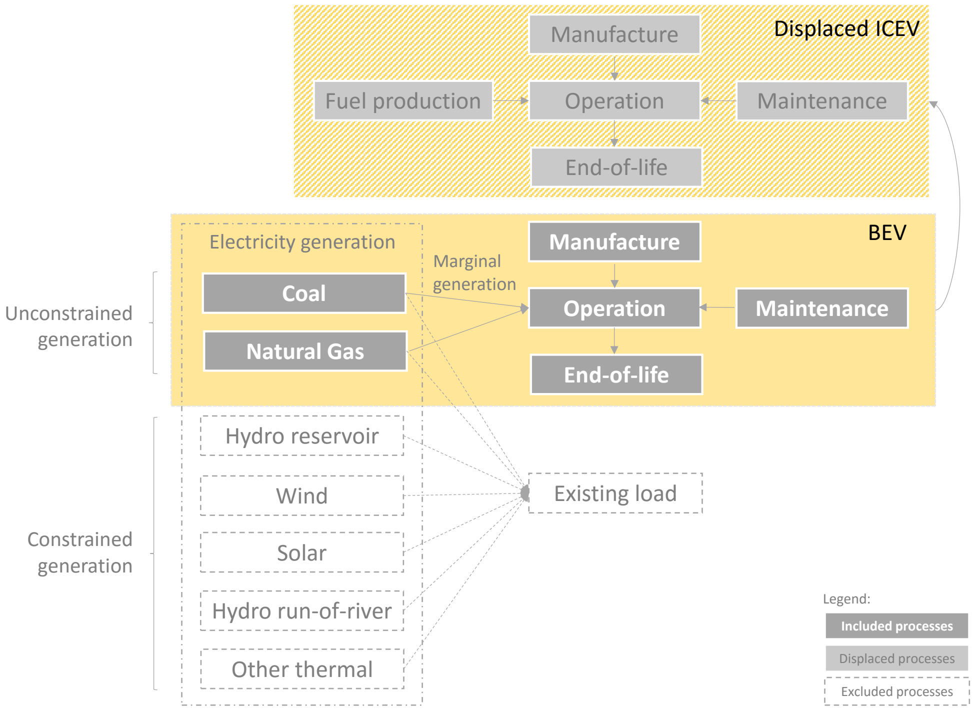

Fig. 5.6 System boundary for the assessment of the effects of the introduction of BEVs in Portugal in the short-term.

Apart from the two options for BEV charging (peak and off-peak), two options for the displaced technology were considered and are depicted in Table 5.4: (1) BEVs displace an average new ICEV according to the *ICEV improve* scenario from Table 3.3, which assumes a 70/30 share in sales of diesel/gasoline ICEVs; and (2) BEV displaces a new gasoline ICEV according to the same scenario. The cumulative GHG emissions due to the introduction of BEVs for the scenarios in Table 5.4 are presented in Fig. 5.7. Displacing an average new ICEV (scn1) leads to GHG savings if BEVs are charged during peak hours and also during off-peak hours for an 8-hour charge. This is mostly because the vehicles displaced are mainly diesel ICEV with higher VKT than BEVs. Conversely, displacing gasoline ICEVs (scn2) results in an increase in GHG emissions, as BEVs were assumed to be driven more than gasoline ICEVs. Off-peak charging can more than double cumulative emissions compared to peak charging.

Table 5.4 Displacement and charging scenarios.

Scenario	Displaced technology	Charging
scn1_p	Average new vehicle according to *ICEV improve* scenario (Table 3.3) (30% gasoline ICEV; 70% diesel ICEV)	Peak (Table 5.2)
scn1_off-p	Average new vehicle according to *ICEV improve* scenario (Table 3.3) (30% gasoline ICEV; 70% diesel ICEV)	Off-peak (Table 5.2)
scn2_p	New gasoline ICEV according to *ICEV improve* scenario (Table 3.3)	Peak (Table 5.2)
scn2_off-p	New gasoline ICEV according to *ICEV improve* scenario (Table 3.3)	Off-peak (Table 5.2)

A sensitivity analysis was performed to assess the effect of assumptions regarding BEV VKT on the results (Fig. 5.8). BEV VKT was changed to match that of the displaced technology in each scenario, i.e. for scn1, BEV VKT took the upper bound value in Table 3.4 (sc1_p*; scn1_off-p*); for scn2, the lower bound value (sc2_p*; scn2_off-p*). Only if BEVs displace an average new ICEV in the scenario of 2-hour charge during off-peak hours (scn1-peak*) BEV introduction results in GHG savings. This is because the marginal GHG emissions of the grid are low enough to make the GHG emissions per km driven by a BEV lower than an average new ICEV. However, when BEVs displace gasoline ICEVs, an increase in emissions is verified in all scenarios, though lower compared to Fig. 5.7. This results from the increase in vehicle manufacturing GHG emissions resulting from BEV introduction which represents a higher share in the overall change in GHG emissions as VKT decreases, making the displacement of gasoline ICEVs in scn2_p* and sc2_off-p* worse than the displacement of an average new ICEV in scn1_p* and scn1_off-p*, respectively.

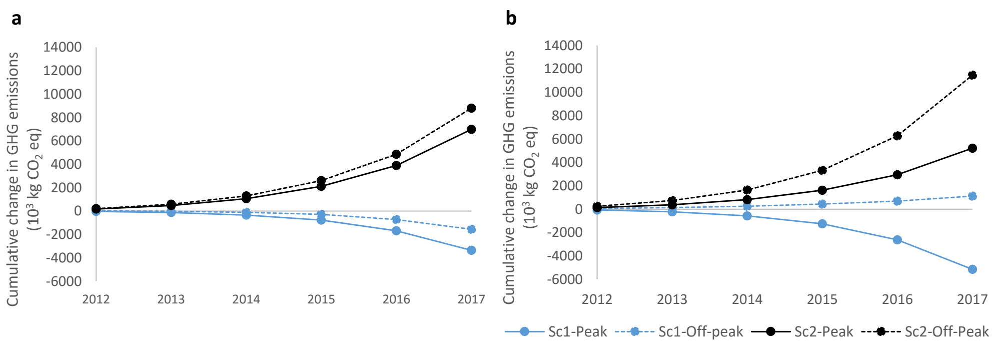

Fig. 5.7 Cumulative change in greenhouse gas (GHG) emissions due to the introduction of BEVs in the Portuguese light-duty fleet in 2012-2017 for the scenarios in Table 5.4 considering (a) 8-hour charge; (b) 2-hour charge.

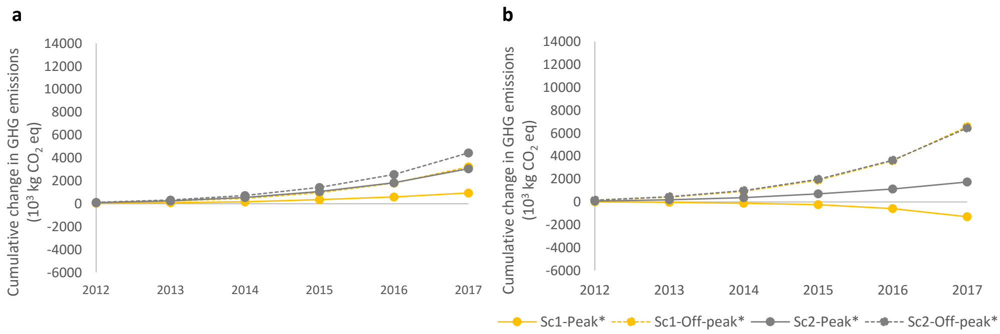

Fig. 5.8 Sensitivity analysis of the cumulative change in greenhouse gas (GHG) emissions due to the introduction of BEVs in the Portuguese light-duty fleet in 2012-2017 to BEV VKT for (a) 8-hour charge; (b) 2-hour charge.

5.1.5 Concluding remarks

A consequential model of the Portuguese electricity system was developed to assess the GHG emissions caused by a change in electricity demand in the short-term. The model was applied, within the dynamic fleet-based LCA framework, to assess the effects of the introduction of BEVs in Portugal, for a range of displacement and charging scenarios.

Coal and natural gas were the marginal energy sources identified, but their contribution to the margin depended on the hour of the day, time of year and system load, causing marginal emission factors to vary significantly. Increasing electricity consumption during off-peak hours was found to induce a higher increase in GHG emissions than in peak hours, due to a higher contribution of coal to the margin. In periods of low demand or high hydro availability, coal is often the marginal technology, as a result of the lower operation costs combined with the low price of CO_2.

For the Portuguese electricity system, with a high share of non-dispatchable renewable power and excess capacity in the short-term, marginal emissions are considerably higher than average emissions. Increasing electricity generation by 1 kWh means increasing fossil-based generation (either coal or natural gas), resulting in higher emissions than the renewable-based average. When the goal is to assess the effect on GHG emissions of implementing a technology which entails a change in electricity consumption (may it be increasing or decreasing consumption), marginal emission factors should be used, as marginal effects have a distinct and larger magnitude than the average behavior of the electricity system.

The application of the model to assess the GHG effects of BEVs in Portugal showed that BEVs induce, in the short-term, a much higher burden than an attributional approach can depict. Even considering the displacement of ICEVs, BEVs cause and increase in overall GHG emissions in the majority of scenarios. However, BEV effects on GHG emissions are very dependent on the time of charging – off-peak charging can more than double cumulative emissions compared to peak charging – and on the assumptions about the displaced technology, including the activity level of both BEVs and displaced ICEVs.

In this analysis, the potential use of surplus renewable energy by BEVs, particularly in off-peak hours, is not accounted for; therefore, it represents a worst-case scenario. Maximizing the use of renewable power would reduce GHG emissions from the introduction of BEVs. However, interactions with other strategies to enable renewables, such as electricity storage, need to be assessed, which are addressed in Section 5.2.

5.2 Effects on GHG emissions of introducing electric vehicles into an electricity system with large storage capacity[7]

5.2.1 Introduction

Meeting European targets for 20% of renewable energy in 2020 and 80-95% reduction in greenhouse gas (GHG) emissions by 2050 compared to 1990 will require a large penetration of renewable energy sources (RES) in the electricity mix, with intermittent RES, such as wind and solar, playing an important role. However, increasing intermittent RES generation increases the likelihood of temporal mismatch between electricity generation and demand, thus requiring more flexibility in the electricity system with sufficient dispatchable back-up generation and storage to balance demand. Electric vehicles (EVs), as a potentially controllable load, are one of the possible solutions for load balancing (Verzijlbergh et al. 2014; Mwasilu et al. 2014).

EVs have been investigated for their two-fold potential to reduce environmental impacts: (i) by displacing internal combustion engine vehicles (ICEVs) and (ii) by enabling more intermittent RES by charging with surplus power in periods of low demand (Richardson

[7] Significant portions of this section appear in: Garcia, R., Freire, F., Clift, R. (2015). Effects on greenhouse gas emissions of introducing electric vehicles into an electricity system with large storage capacity. (submitted)

2013). Shifting from ICEVs to EVs means shifting the energy source used for transportation from petroleum-based fuels to the mix of energy sources used for electricity generation. GHG emissions associated with use of EVs are highly dependent on the GHG intensity of the electricity used for charging (Stephan and Sullivan 2008; MacPherson et al. 2012; Hawkins et al. 2013). If charged with RES-based electricity, EV GHG emissions are lower than ICEVs; if charged with coal-based electricity, their impacts exceed those of comparable conventional fossil-fueled vehicles (Samaras and Meisterling 2008; Hawkins et al. 2012; Hawkins et al. 2013; Nordelöf et al. 2014a). These general conclusions, however, do not take into account how EV charging may influence grid emissions.

The charging strategy is key to maximizing renewable energy use by EVs. In principle, adoption of EVs can enable increased use of wind power (Soares et al. 2012; Hedegaard et al. 2012; Liu et al. 2013; Ekman 2011; Lund and Kempton 2008; Dallinger et al. 2013). Although the interaction with solar power has been less studied, EVs can also enable more solar power to be accommodated through day-time charging (Zhang et al. 2012; Nunes et al. 2015). However, the effects of EVs depend on the detailed structure and dynamics of the electrical system. EVs represent an additional demand for electrical energy so their adoption will, in some cases, lead to an increase in fossil-based generation and associated GHG emissions (Hedegaard et al. 2012; Banez-Chicharro et al. 2014; Verzijlbergh et al. 2014). Despite the increase in RES penetration due to EVs, the associated increase in fossil-based generation can offset the benefits of additional RES (Hedegaard et al. 2012; Schill and Gerbaulet 2015b). However, the interaction between EVs and different technologies and strategies to enable RES, such as large storage capacity, have generally not been accounted for.

Pumped hydro storage (PHS) is considered the most flexible and widespread technology for large-scale storage of electricity, representing about 99% of the installed storage capacity worldwide, with a considerable expansion potential in Europe (Gimeno-Gutiérrez and Lacal-Arántegui 2013). PHS can foster the integration of RES by storing surplus energy to be used when demand is higher, displacing generation from fossil-based power plants, albeit with an efficiency penalty. Unlike thermal plants, PHS combines flexible operation with low running costs and is therefore a good complement to intermittent renewables in general (Göransson and Lundberg 2014). Other strategies to increase the flexibility of the electricity system and enable RES penetration include cross-border transmission, back-up plants (usually fossil-fueled) and vehicle-to-grid energy transfer, but these are not addressed in this article.

The interaction between different technologies and strategies to enable RES needs to be better understood, including the GHG consequences of introducing EVs allowing both for their effect on the electricity system and for displacement of ICEVs. Despite numerous studies on the effects of EVs on the electricity grid, in particular on integration of RES, the combination of these effects has received little attention. The effects of EVs depend on the existing electricity technology portfolio, the availability of renewable energy and the constraints on its use, the charging strategy of EVs and other options, including storage, to increase the flexibility of the electricity system. These interactions are explored in this paper, using life cycle assessment to compare possible scenarios in Portugal, selected as a case study for its high share of both wind and PHS capacity and favorable conditions for EV deployment with a charging network and policy incentives for buying EVs already in place.

5.2.2 Electric vehicles and pumped hydro storage

Different scenarios of introduction of EVs and PHS into a system for generating and distributing electricity are considered, summarized in Table 5.5. The changes in GHG emissions for those scenarios are assessed, starting with the effects of introducing PHS and EVs separately into the system (Scenarios A - Grid without storage or EVs) and then analyzing the interactions between PHS and EVs (Scenarios B - Grid with PHS). The electric vehicles considered are purely battery driven without an internal combustion engine (Ellingsen et al. 2014). Energy consumption of the vehicle technologies (mid-sized passenger vehicles) and efficiency of the power plants considered are displayed in Table 5.6; current and future (reference year: 2030) technologies are considered.

Both PHS and EVs can use the surplus power resulting from periods of low demand and high RES generation. PHS is usually operated to maximize the storage of surplus RES during low demand and to generate electricity when demand is high. Thus, the interaction between PHS and EVs is more significant for large scale wind penetration, associated with higher generation in off-peak hours, than solar, whose generation roughly coincides with peak hours. PHS can be managed as baseload or peak-load power plant; therefore, it can displace either baseload power (e.g. coal) or peak-load power (e.g. natural gas). The GHG emission savings of using surplus renewable energy in PHS thus depend on the technology displaced (Scenarios PHS-1a; PHS-1b).

As a flexible load, EVs can be managed to charge mainly during off-peak hours in order to maximize the use of renewable energy that would otherwise have been curtailed (Scenarios EV-1c and EV-1d). However, if surplus renewable energy is not enough to meet EV demand or all renewable power is utilized by other loads, other plants need to be brought online to satisfy the additional EV demand. In Portugal, as in many other countries, these plants would be fueled by coal or natural gas (NG) (Scenarios EV-2c, EV-2d, EV-3c, EV-3d). This means that, whilst the additional demand from EVs may in some instances be satisfied by RES, it is also possible for EVs to induce an increase in coal and/or NG generation. The overall GHG benefits of EVs also depend on the vehicle technology displaced; gasoline and diesel mid-sized passenger ICEVs are considered here.

5.2.2.1 Separate introduction of storage and electric vehicles

The changes in GHG emissions resulting from introducing PHS and EVs separately into a system for generating and distributing electricity (Scenarios A- Grid without storage or EVs in table 1) are displayed in Fig. 5.9. Storing 1 MWh of surplus energy in PHS enables 0.7 MWh to be recovered to displace fossil generation, avoiding about 266 kg CO_2 eq from a state-of-the-art natural gas (NG) combined cycle (CC) power plant (PP) (Scenario PHS-1a) or 576 kg CO_2 eq from a state-of-the-art hard coal PP (Scenario PHS-1b). With current technologies, controlled charging of EVs can ensure that the use of surplus RES is maximized (Scenarios EV-1c and EV-1d), leading to higher GHG savings because the fossil fuel displaced is gasoline or diesel burned in an ICEV for which the efficiency is lower than for electricity generation. Future GHG savings related to PHS are projected to increase as the efficiency of PHS improves; where the PHS displaces coal, the projected increased savings are even larger than those from displacing gasoline or diesel ICEVs (Scenarios PHS-1b versus EV-1c or EV-1d).

Table 5.5 Scenario description (generic case).

Scenarios	Technology introduced	PHS		EV	
		Energy source used	Energy source displaced	Energy source used	Technology displaced
A (Grid without storage or EVs)					
PHS-1a	PHS	1. RES	a. NG CC	-	-
PHS-1b			b. Hard coal	-	-
EV-1c	EVs	-	-	1. RES	c. Gasoline ICEVs
EV-1d		-	-		d. Diesel ICEVs
EV-2c	EVs	-	-	2. NG CC	c. Gasoline ICEVs
EV-2d		-	-		d. Diesel ICEVs
EV-3c	EVs	-	-	3. Hard coal	c. Gasoline ICEVs
EV-3d		-	-		d. Diesel ICEVs
B (Grid with PHS)					
PHS-1a&EV-4c	PHS&EVs	1. RES	a. NG CC	4. RES and/or NG CC	c. Gasoline ICEVs
PHS-1a&EV-4d					d. Diesel ICEVs
PHS-1b&EV-4c[a]			b. Hard coal		c. Gasoline ICEVs
PHS-1b&EV-4d[a]					d. Diesel ICEVs
PHS-1a&EV-5c	PHS&EVs	1. RES	a. NG CC	5. RES and/or Hard coal	c. Gasoline ICEVs
PHS-1a&EV-5d					d. Diesel ICEVs
PHS-1b&EV-5c[a]			b. Hard coal		c. Gasoline ICEVs
PHS-1b&EV-5d[a]					d. Diesel ICEVs

[a] Results for the scenarios in which PHS displaces coal (B) are presented in Annex V (Fig. C-1).

Table 5.6 Energy consumption of vehicles and efficiencies of power plants for the generic case (Section 5.2.2) and the Portuguese case (Section 5.2.3).

	Generic case[a] (current | *future*)		**Portuguese case**[b]		
Vehicles	**Energy consumption** (MJ km^{-1})		**Energy consumption** (MJ km^{-1})		
			2015	2020	2025
EV[c]	0.91 | *0.75*		0.64	0.60	0.56
Gasoline ICEV	2.80 | *2.17*		1.96	1.73	1.44
Diesel ICEV	2.43 | *1.93*		1.85	1.58	1.32

Power plants	**Efficiency**	**Emission factor** (kg CO_2 eq MWh^{-1})	**Efficiency**	**Emission factor** (kg CO_2 eq MWh^{-1})
PHS	0.70 | *0.85*	13 | *13*	0.70	13
NG CC	0.58 | *0.62*	398 | *366*	0.58	423[d]
Coal	0.45 | *0.49*	841 | *800*	0.38	1021

Electricity transmission and distribution efficiency of 92% not accounted in the emission factors. Additional details about the technologies are presented in Annex V (Table C-1).

[a] For the generic case, all vehicles are comparable mid-sized European passenger vehicles (based on Bauer et al.[2015]). Efficiencies and emissions factors for generic current and future power plants are based on Bauer et al. (2008) and Volkart et al. (2013).

[b] For the Portuguese case, characteristics of future average new vehicles are based on Section 3.1.2.3 (Garcia et al. 2015). Efficiencies and emission factors of Portuguese power plants are those in Table 4.2 (Garcia et al. 2014).

[c] The vehicle type considered is battery electric with no internal combustion engine.

[d] Despite the same efficiency, the emission factor for the Portuguese NG CC power plants is higher than for the generic case, mostly due to the lower efficiency of the natural gas supply chain to Portugal (75%), which incorporates a high percentage of liquefied natural gas from Nigeria (Safaei et al. 2015).

5.2.2.2 Electric vehicles combined with storage

The interactions between PHS and EVs are analyzed in this Section. When EVs with controlled charging are added to a grid with PHS capacity (Scenarios B in Table 5.5), PHS and EVs compete for the surplus renewable energy generated during periods of low demand. The total energy available for storage is reduced by the EV demand so that RES is only available for storage if supply exceeds EV demand. The decrease in energy stored requires other plants to be brought online or to increase their generation during periods of high demand.

The net effects on GHG emissions, allowing both for increased electricity generation from NG CC or coal and for reduced use of conventional vehicles, are shown in Fig. 5.10 for the Scenarios B (see Table 5.5). The abscissa in Fig. 5.10 shows the proportion of RES in the energy supplied to EVs. The lines refer to the fuel used in the grid when sufficient surplus RES is not available (full lines for NG CC; broken lines for coal) and also the type of ICEV displaced (blue for gasoline; black for diesel). Results are presented for both current and future technologies (see Table 5.6).

Compared with introducing EVs into a grid without storage (previous section), overall savings are lower and in some cases negative, resulting from the additional natural gas or coal generation required. Introducing EVs charged with surplus RES or NG is seen to result in overall savings (PHS-1a&EV-4c and PHS-1a&EV-4d) but, comparing Fig. 5.10(a) and (b), the savings are projected to reduce over time because the efficiency of the displaced vehicle technologies increases more than the efficiencies of the power plants. The projected gains in efficiency of PHS are calculated to make charging EVs with NG better than using surplus energy. However, if EVs are charged mainly with coal-based electricity, an increase in GHG emissions results in all scenarios (PHS-1a&EV-5c and PHS-1a&EV-5d).

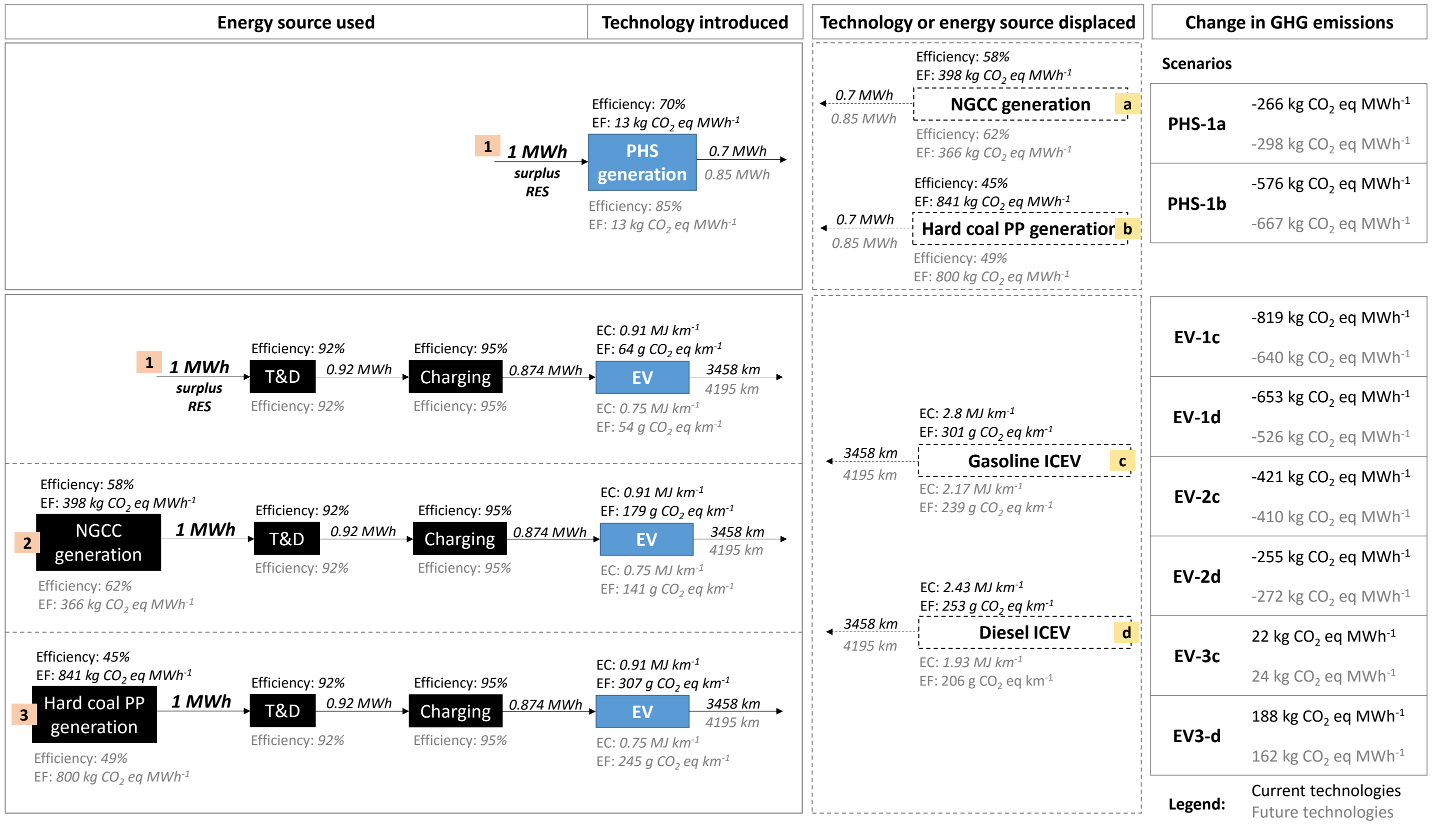

Fig. 5.9 Change in GHG emissions per MWh generated due to the introduction of EVs or PHS in a grid for the scenarios defined in Table 5.5 (A- Grid without storage or EVs). All vehicles are comparable mid-sized European passenger vehicles (based on Bauer et al.[2015]). An efficiency of 92% for electricity transmission and distribution (T&D) (Garcia et al. 2014) and 95% for charging (based on a normal charger – single phase, 16–32 A [Mwasilu et al. 2014]) was assumed. Data sources and additional details about the technologies are presented in Table 5.6. EC = energy consumption; EF = emission factor.

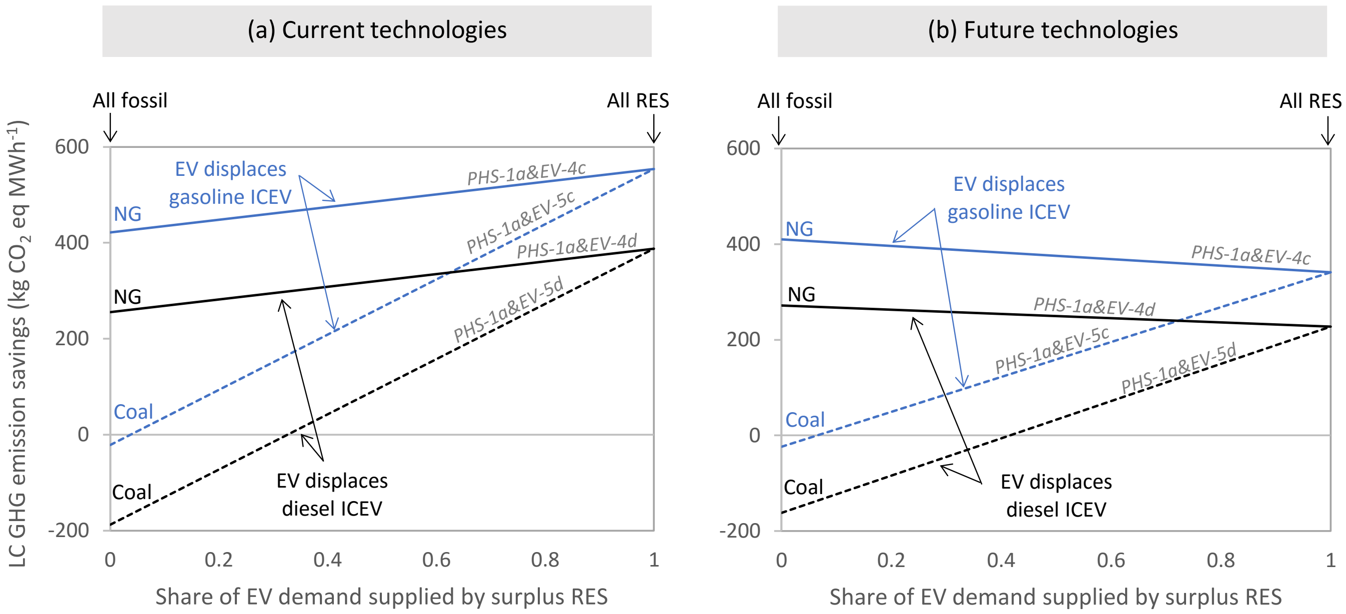

Fig. 5.10 LC GHG emission savings per MWh of electricity consumed by EVs as a function of the proportion of renewable energy sources (RES) in the energy supplied to EVs, for Scenarios B in which PHS displaces natural gas combined cycle (PHS-1a&EV; Table 5.5). Results for (a) current technologies and (b) future technologies.

5.2.3 Application to the introduction of EVs in Portugal

5.2.3.1 The Portuguese electricity system

The effects of introducing EVs in a grid system with large storage capacity are assessed using Portugal as a case-study. In the last decade, Portugal has more than doubled RES installed capacity, especially due to a large increase in wind power plants, and renewable energy supplied more than 60% of total electricity consumption in 2014. According to the National Renewable Energy Action Plan (Presidência do Concelho de Ministros 2013), RES installed capacity is planned to increase at an average rate of 5% per year up to 2020. The continuous growth in wind capacity will be supported by the construction of new pumped hydro storage (PHS) capacity, which will reach 5050 MW in 2022. Smaller expansion is projected in other renewables: photovoltaic, solar thermal, biomass and wave.

Compared to the European mix, Portugal has a larger share of both installed capacity and hydro-generation, with no nuclear sector (Fig. 5.11). Natural gas is expected to replace coal generation at a higher rate than in the average European mix. Despite the favorable insolation, investments in solar power are modest compared to the European average (Fig. 5.11). Solar capacity is expected to increase due to the foreseen reduction in the price of the technology, but the largest investments will continue to be in wind. For that reason, we focus our analysis on the interactions between wind, PHS and EVs.

The role of pumping in the Portuguese electricity system

The current generating capacity of PHS in the Portuguese electricity system is 1515 MW, about 8% of total capacity and 27% of installed hydro. In Portugal, pumping occurs mostly during off-peak hours (see Fig. 4.5 and Fig. 4.6). Especially in the summer, Portugal imports electricity (from Spain) to store through PHS, mainly low cost nuclear electricity; on the other hand, surplus wind energy is frequently exported at low price hours, especially in the winter. Increasing wind capacity would reduce the need to import electricity for storage, while increasing PHS capacity would reduce electricity exports at low price hours or curtailment of wind generation when interconnection capacity is insufficient or the Iberian electricity market is saturated; according to DGEG (2012), the planned expansion of both wind and PHS capacity will reduce wind curtailment in off-peak hours to 0.03% in 2025.

Plants displaced by PHS are fired by either natural gas or coal (see Fig. 4.5 and Fig. 4.6), which are the marginal sources in Portugal (see Section 5.1). In 2013, during the top 5% highest-demand hours, natural gas was the dominant marginal energy source, with a share of 73%, against 27% for coal generation (see Section 5.1). This indicates that PHS is replacing natural gas more than coal generation. In the future this trend should continue. We therefore consider NG CC as the marginal generation in all cases.

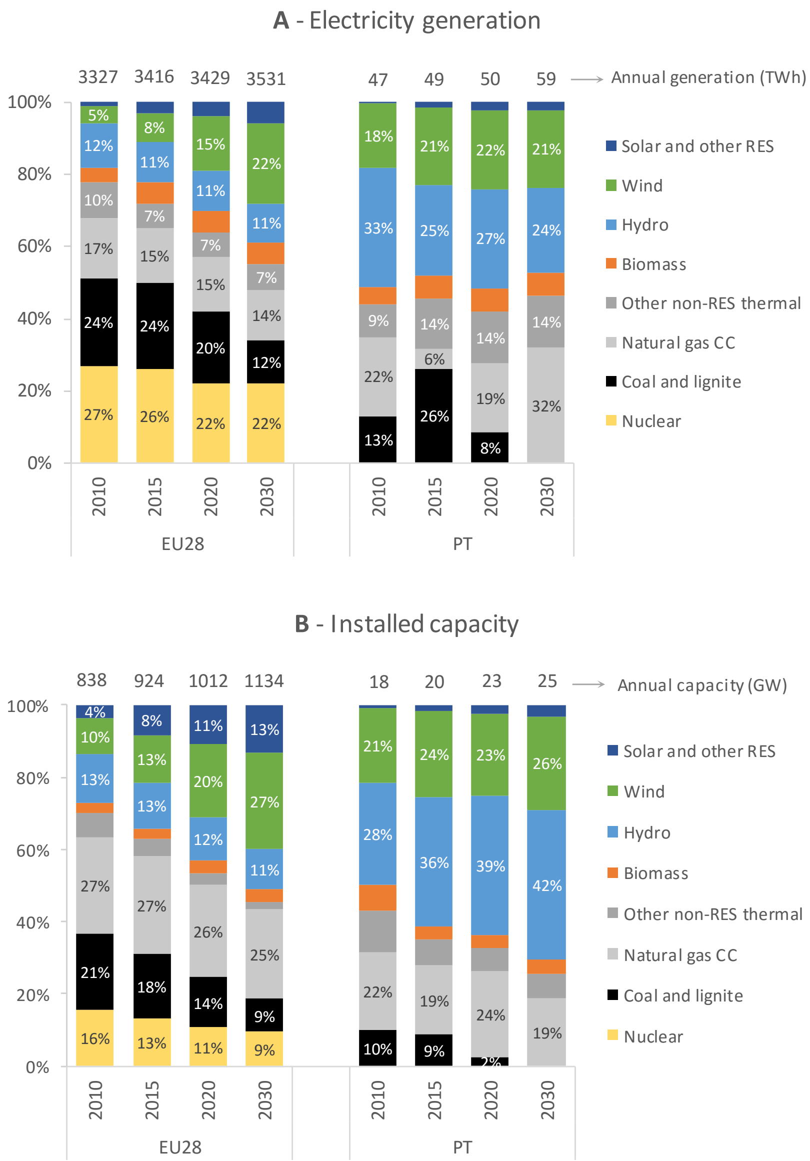

Fig. 5.11 Electricity generation (A) and installed capacity (B) by energy source in Europe (Capros et al. 2014) and Portugal (DGEG 2012) in 2010-2030.

5.2.3.2 Scenarios

Two scenarios, summarized in Table 5.7, have been developed to assess how the addition of EVs to a grid with large storage capacity would influence, firstly, the GHG emissions of the Portuguese electricity system, and, secondly, the overall GHG emissions. The main characteristics of vehicle technologies and Portuguese (PT) power plants are displayed in Table 5.6 (Portuguese case). These differ from the technologies considered in the generic case, as vehicles in Portugal are on average smaller (sub-compact and compact), while coal power plants are older and less efficient than state-of-the-art plants. Scenarios have been assessed for three years (2015, 2020, and 2025) to explore different levels of EV deployment and different configurations of the electricity system.

The effect of adding EVs to a grid with (PT-PHS+EV) and without storage (PT-EV) has been assessed by calculating the change in electricity generation by technology and the corresponding change in GHG emissions compared to the same scenario without EVs. It has been assumed that EVs are charged during off-peak hours and, in the cases in which EV demand exceeds the available surplus wind energy, the excess is generated from coal (in 2015 and 2020) or NGCC (in 2025).

Table 5.7 Scenario description (Portuguese case).

Scenario	Description	Grid capacity, including storage	Energy source used by EVs	Energy source displaced by PHS
PT-PHS+EV	EVs are added to the PT grid with PHS capacity	According to DGEG (2012)	Surplus wind + coal (2015; 2020)/NG CC (2025)	NG CC
PT-EV	PHS capacity in the PT grid is limited and EVs enable wind energy	According to DGEG (2012), without storage	Surplus wind + coal (2015; 2020)/NG CC (2025)	-

Evolution of the electricity mix

A study by the Directorate-General for Geology and Energy (DGEG) on the development of the Portuguese electricity system to 2030 (DGEG 2012) provided the basis for scenario PT-PHS+EV (Table 5.8). According to DGEG (2012), four thermal plants will be decommissioned, including the two existing coal plants in 2018 and 2021, respectively, when their service contracts are worked out. Two new NG CC plants are planned. However, in recent years, coal power plants have been used heavily in place of NG, despite the recent investment in the latter technology, due to the low prices of coal compared to natural gas, combined with the low price of CO_2. Decommissioning coal PP will lead to a loss of competitiveness of the electricity system, which will become dependent on a single fossil energy source. For these reasons, it is possible that the coal plants will not be shut down and that no new NG power plants will be built in the planned period. The implications of these considerations on the results from the present work are addressed in Section 5.2.3.4.

Hydro power capacity is planned to increase significantly, including PHS capacity to reduce surplus energy from wind generation in off-peak hours to less than 5 GWh in 2025 (i.e. 0.03% of wind energy generation) (Table 5.8). Thus, the planned PHS system should be able to absorb about 99% of off-peak surplus energy. For scenario PT-EV, it has been assumed that the expanded hydro-electric capacity in the DGEG scenario is regular generating capacity without storage. Because there is overcapacity in the Portuguese electricity system, it is unlikely that EV deployment will induce capacity expansion beyond that already planned in the timeframe of this analysis; therefore, this effect has not been considered here.

Table 5.8 Planned installed capacity in the Portuguese electricity system in 2015-2025, and wind curtailment and surplus energy in off-peak hours with and without PHS capacity (based on DGEG 2012). Installed capacity up to 2020 based on PNAER 2020 (Presidência do Concelho de Ministros 2013).

	2015	**2020**	**2025**
Installed capacity (MW)			
Coal[a]	1756	576	0
Natural gas CC[b]	3829	5595	4605
Non-renewable CHP	1420	1460	1570
Hydro[c]	6713	8550	9650
PHS	2655	3950	5050
Wind	4800	5300	5820
Other RES	1549	1911	2108
Wind curtailment			
with PHS (%)	0.16%	0.05%	0.03%
without PHS (%)	5.11%	5.99%	6.01%
Surplus energy			
with PHS (GWh)	16.8	6.3	4.2
without PHS (GWh)	537	667	674

[a] Decommissioning of Sines power plant is planned to 2017 and Pego power plant to 2022.

[b] New power plant in Lavos and Sines are planned to be installed; Decommissioning of Tapada do Outeiro power plant is planned to 2024.

[c] Includes hydro reservoir (including PHS) and run-of-river.

EV penetration

We consider here an extreme development with regard to electric vehicle deployment by assuming that the proportion of EVs in all new vehicles entering the fleet increases sigmoidally from 2015 to reach 100% in 2030, corresponding to a stock of 192 thousand EVs by 2020 (4% of fleet share), 1.2 million by 2025 (24%), and 2.4 million by 2030 (50%). This scenario represents an upper bound for EV demand. The electricity consumption associated with the EV out to 2030 has been assessed using a dynamic fleet-based model of the Portuguese light-duty fleet (Chapter 3; Garcia et al. 2015) which considers all relevant variables into account, most importantly how the vehicle stock and technology evolve over time. Baseline electricity demand and the additional EV demand in 2015-2030 are shown in Table 5.9.

Table 5.9 Baseline demand and battery electric vehicle (EV) demand scenarios.

	2015	2020	2025
Baseline demand[a] (GWh)	48812	50087	53604
EV demand[b] (GWh)	11	559	3262

[a] Baseline demand based on the central scenario by DGEG (2012)

[b] Market share of EVs sigmoidally increases up to 100% in 2030; total electricity consumption takes into account the fleet evolution and EV technology evolution, according to the *Combined* scenario in Chapter 3 (Garcia et al. 2015).

5.2.3.3 Results

Changes in electricity GHG emissions

Fig. 5.12 shows the changes in the GHG emissions from the electricity sector due to the progressive introduction of EVs for the scenarios with and without PHS. At very low penetration, as in 2015, EVs have a negligible effect, irrespective of the PHS capacity. As EV penetration increases but the associated demand is still below the amount of surplus RES available, as in 2020, EVs contribute to increase wind generation in both scenarios. Without PHS (scenario PT-EV), all the increase in demand is met by wind and other power plants keep their generation fixed. However, in the scenario with PHS (PT-PHS+EV), the surplus wind energy used by EVs is no longer available for storage in PHS; natural gas generation now increases to compensate, resulting in higher changes in GHG emissions. By 2025, EV demand exceeds surplus wind power. The effect of the EVs is now to increase GHG emissions in both scenarios but the increase is larger for the scenario with PHS.

Table 5.10 compares marginal and average GHG emissions from the PT grid with and without PHS. Average grid emissions decrease over time, mainly as a result of the decommissioning of coal plants, and are lower when there is already PHS capacity in the system. However, marginal emissions due to increased use of EVs are higher with PHS capacity, given that the marginal demand is met by natural gas, and higher than the average GHG emissions when EV penetration is high (as in 2025).

Overall change in GHG emissions

In order to assess the overall change in GHG emissions due to the introduction of EVs in Portugal, both the effects in the electricity grid and the displacement of conventional technologies in the fleet need to be accounted for. Displacement effects have been estimated using the ICEV improve scenario in Chapter 3 (Garcia et al. 2015) as baseline and assuming that EVs displace average new ICEVs. This scenario assumes that the market share of diesel and gasoline ICEVs stabilizes at 70/30% and the fuel consumption of new vehicles decreases over time according to the European Union targets. Adding the displacement effects to the effects on the grid shows that introduction of EVs leads to overall reduction in GHG emissions (see Table 5.10). The savings would initially be about 45% higher (2015, 2020) in the absence of PHS capacity but the difference decreases (to 22% in 2025) as more EVs enter the fleet.

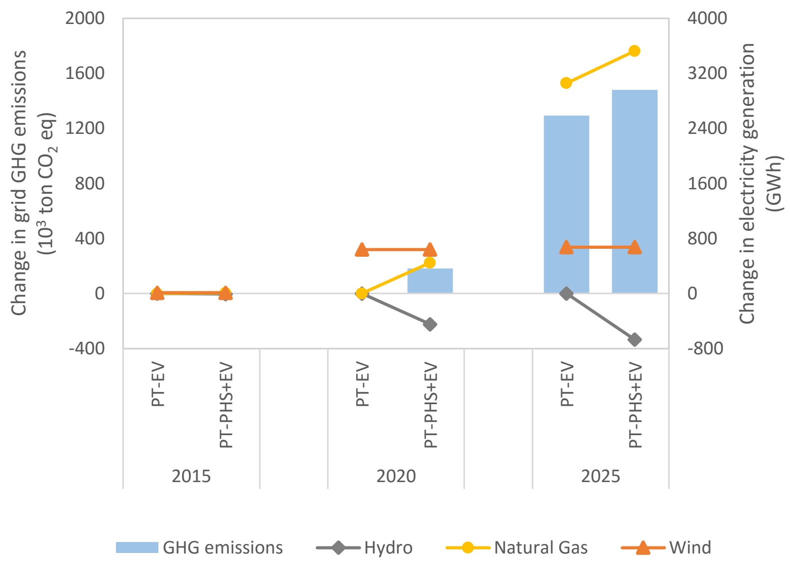

Fig. 5.12 Changes in hydro, natural gas and wind electricity generation (markers, right axis), and overall grid GHG emissions (columns, left axis) due to the introduction of EVs in the Portuguese grid with and without pumped hydro storage.

Table 5.10 Average grid emissions, marginal grid emissions and overall change in GHG emissions due to the introduction of EVs in Portugal in 2020 and 2025 for the scenarios in Table 5.7.

Scenarios		Grid GHG emissions (kg CO_2 eq kWh^{-1})		Change in GHG emissions (10^6 kg CO_2 eq)			
		Marginal	Average	Change in electricity	Change in vehicle manufacturing, maintenance and EoL	Change in ICEV operation	Total
PT-PHS+EVs	2015	0.284	0.375	3	2	-10	-4
	2020	0.284	0.257	182	31	-462	-249
	2025	0.397	0.232	1480	161	-2418	-776
PT-EVs	2015	0.000	0.380	0	2	-10	-8
	2020	0.000	0.259	0	31	-462	-431
	2025	0.347	0.233	1294	161	-2418	-963

EoL: end-of-life.

5.2.3.4 Discussion

The Portuguese scenarios explored above show that EV charging using surplus wind power does not necessarily lead to decreasing GHG emissions from the electricity system. With PHS, using surplus wind energy to charge EVs reduces the amount of energy available for storage, leading to an increase in natural gas generation which results in additional emissions per kWh of wind power used in EVs; however, without PHS, surplus wind energy use by EVs has no marginal effect on grid GHG emissions. The difference between the emissions per kWh demanded by EVs in a grid with and without storage is more significant for low EV penetrations (below 6%). As EV penetration increases, that difference will decline, given that the amount of surplus energy available annually is fixed.

The uncertainties over the future development of the electricity system in Portugal have a strong influence on the effects of introducing EVs. If it turns out that the existing coal plants are not replaced by new NG capacity, introducing EVs in Portugal will increase overall GHG emissions and emissions will again be significantly larger with PHS than in the absence of storage (Fig. 5.13).

The analysis is based on the assumption that EVs absorb surplus RES as long as it is available; this represents the best case scenario for EVs but the worst for the net effect on storage. However, the timing of wind generation and charging demand will generally not coincide, which increases the demand for power from fossil-fueled plants. The effects also depend on exchanges between Spain and Portugal and on the marginal generating capacity in Spain, which have not been considered here.

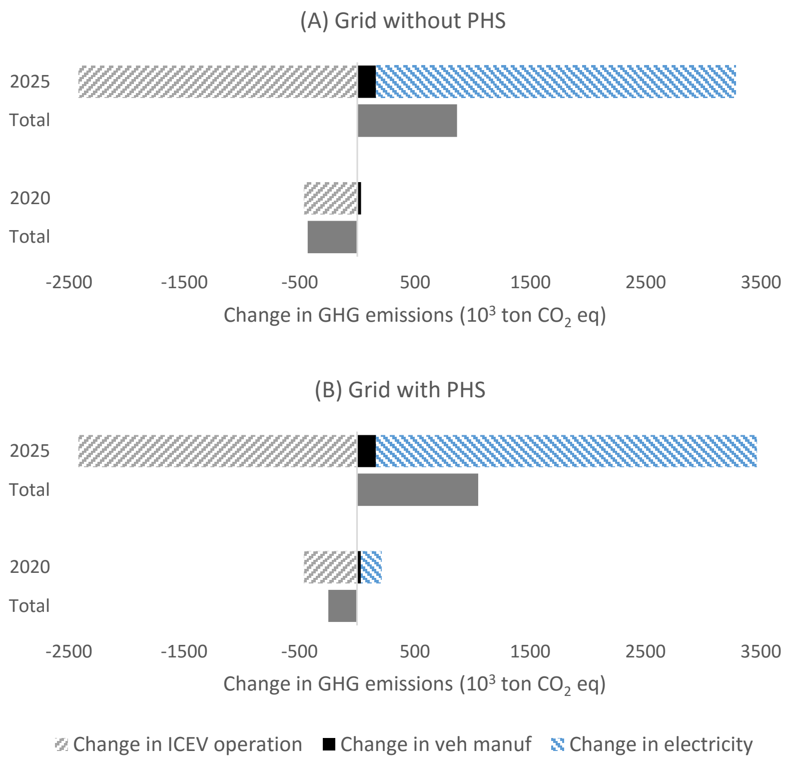

Fig. 5.13 Overall change in GHG emissions due to the introduction of EVs in Portugal in 2020 and 2025 considering that coal PPs provide the additional power requested by EVs.

5.2.4 Concluding remarks

The net effects on GHG emissions of adding electric vehicles into a national or regional electricity system are complex when the system includes renewable energy sources and large scale storage capacity. The Portuguese electricity system has been explored in detail as a specific example, because it is characterized by relatively high capacities of wind generation and pumped hydroelectric storage. When the system includes significant storage of energy from intermittent sources, the effects of introducing EVs go beyond the straightforward displacement of ICEVs and increase in electricity demand, to include significant indirect effects from the dynamics of storage. In the absence of storage, introducing EVs charged at times of low demand generally increases the penetration of RES and therefore leads to major reductions in GHG emissions. However, with storage and high EV penetration, the net GHG savings are lower: diversion of surplus electricity from storage to EVs means that there is an overall increase in the load of the electricity system. For the specific case of Portugal, the savings are still positive if the EVs are charged from RES or natural gas combined cycle plants but, if the marginal production is from coal, increasing penetration of EVs leads to increased GHG emissions.

The net effects on GHG emissions are very dependent on the technologies displaced both by PHS and by EVs, so that detailed analysis is needed for any specific energy system, allowing for future technological improvements. While new vehicles enter the fleet at a relatively high rate, power plants have a long service life so that the electricity system is subject to technological lock-in; therefore, the net GHG benefits of introducing EVs decline over time because the efficiencies of the displaced ICEVs improves more rapidly than the average efficiency of generating plants. Displacing gasoline ICEVs leads to higher GHG savings than displacing diesel, due to the lower efficiency of gasoline engines.

Chapter 6

CONCLUSIONS

6.1 Key findings and contributions

In this thesis, a dynamic fleet-based LCA framework was developed and implemented to assess the effects on greenhouse gas emissions of the introduction of electric vehicles in a fleet. The framework combines fleet modeling and dynamic life-cycle inventories of vehicle technologies to assess the displacement of conventional vehicles over time, and consequential life-cycle assessment of electricity to assess the changes induced in the operation of the electricity system due to EV charging. The analysis focused on the introduction of battery electric vehicles (BEVs) in the Portuguese light-duty fleet and included the following steps:

- A dynamic fleet-based LC model of the Portuguese light-duty fleet was developed to assess fleet-wide environmental impacts of displacing ICEVs by EVs across different scenarios (Chapter 3). The analysis took into account the dynamic behavior of the fleet, including fleet turnover, technology improvements, and changes in background processes, and vehicle activity within the same framework.
- A comprehensive (attributional) life-cycle assessment of electricity generation and supply in Portugal was performed (Chapter 4), assessing annual environmental impacts in the last decade, and hourly GHG emissions from 2012-2014, to identify the main drivers of impacts, how impacts change over time, and how charging time influences BEV GHG emissions.
- A consequential model of the Portuguese electricity system was developed to assess GHG emissions caused by a change in electricity demand in the short-term and how they vary over time. The model was applied, within the dynamic fleet-based LCA framework, to assess the effects of the introduction of BEVs in Portugal, for a range of charging and displacement scenarios (Section 5.1).
- The interactions between BEVs and pumped hydro storage were explored using the dynamic fleet-based life-cycle framework developed to compare changes in GHG emissions for different scenarios of technologies displaced by both BEVs and PHS in Portugal. (Section 5.2).

- Insights on different methodological issues were also provided, namely on the identification of the marginal electricity supply in consequential studies (Chapter 5), on the choice between attributional and consequential modeling (Chapter 4 and 5), and on the relevance of including dynamic aspects in LCA (Chapter 3).

A key contribution from this research is the dynamic fleet-based life-cycle framework developed to assess the environmental impacts of the adoption of a new technology. Contrary to most of the LCA literature which perform static analyses of single products, the framework proposed in this thesis enables the explicit assessment of changes in technologies and background systems over time and of the transient effects occurring during a technology shift by considering the dynamic behavior of the product fleet from a life-cycle perspective. Therefore, the dynamic effects of displacing an older technology can be assessed through this framework.

Another aspect that distinguishes the approach developed in this research is the integration of both fleet displacement effects and effects in electricity impacts within the same framework. It is thus appropriate to assess the effects on environmental impacts of other electricity-using products in a fleet perspective, particularly other transportation or industrial systems, and can also be used to assess the environmental effects of measures that improve the energy efficiency of end-use applications, such as lighting systems, or that shift the use of electricity, such as storage of electricity in batteries or load scheduling in households. The change-oriented approach pursued can contribute to understand the environmental effects of policies and strategies that enable and promote the use of electricity over other fuels. In particular, this thesis sheds light on the effects of the interaction between large storage capacity (i.e. pumped hydro storage) and EV charging, an issue that has been seldom investigated, and contributes to the discussion on the merits of peak versus off-peak charging for EVs.

Comprehensive LCA studies of countries or regions electricity mixes including a wide set of environmental impacts associated with the country/region specific portfolio are not common. This research contributes to the body of knowledge on the environmental assessment of a country/region electricity mix including both generation and supply chains considering different temporal resolutions. It draws attention to the importance of considering a higher temporal resolution in the assessment of products or activities which seasonally use electricity in their life-cycle, in particular for electricity mixes which include a high share of RES (especially hydro). It also provides updated environmental impacts for the Portuguese electricity system, which can be used in LCAs of products that require electricity along its life cycle and are produced or used in Portugal, in environmental product declarations and carbon footprint analyses, and in the establishment of emission factors in national regulations (e.g., regarding the energy performance certification of buildings [Marques et al. 2015a]). The emission factors calculated for the main technologies operating in Portugal as well as the transmission and distribution infrastructure can be used to assess future electricity mix impacts. The marginal emission factors provided are suitable for assessing short-term changes in the operation of the electricity system beyond the applications presented in this thesis.

The key findings from this thesis are presented in the following paragraphs with respect to the five research questions formulated in Chapter 1 (Table 1.1).

1. *What are the conditions under which the displacement of conventional vehicles by electric vehicles in the Portuguese light-duty fleet reduces transportation GHG emissions?*

   Reducing fleet-wide GHG emissions by displacing ICEVs by BEVs in Portugal depends heavily on the GHG intensity of the Portuguese electricity system, on the degree of reduction in fuel consumption of new ICEVs and on the level of penetration of BEVs.

   For achieving significant reductions in the Portuguese light-duty fleet through the introduction of BEVs compared to an increasingly more efficient ICEV fleet, a high BEV market share and electricity GHG intensity similar or lower to the current mix (485 g CO_2 eq kWh^{-1}) need to be realized. Furthermore, it was found that halving the GHG emissions from BEVs by reducing the electricity mix impacts is more effective than halving ICEVs emissions through reducing fuel consumption, because the first effect affects the entire BEV fleet, whilst the latter is only applicable to new ICEVs.

   Overall fleet GHG emission reductions are also dependent on parameters that affect the vehicle stock, the scrappage rate and the activity level of the fleet.

   The influence of these parameters also varies over time, becoming more important as time passes. Therefore, acting upon decreasing vehicle ownership, optimizing vehicle scrappage and reducing overall distance travel should complement policies that promote EV adoption, so that system-wide impacts are actually reduced.

2. *What are the life-cycle environmental impacts attributed to electricity generation and supply in Portugal in the last decade and how do they vary between years and throughout the year? How does this variability affect the environmental impacts of an EV as a function of the time of charging?*

Environmental impacts of the Portuguese electricity system have been reducing since 2003, as a consequence of investments in RES, decommissioning of fuel oil PP and installment of gas treatment systems in coal PP, but annual variability is significant.

The higher reductions were realized in acidification and photochemical oxidation (-81%), and resulted from the decommissioning of large fuel oil power plants and the installation of denitrification and desulfurization systems in coal power plants. For other impact categories, reductions were as high as 39% (36% for GHG emissions), and were mostly driven by the increase in renewable energy share. However, the overall renewable energy generation varied significantly in the last five years (decreasing 29% from 2010 to 2012 and increasing 30% in 2012-2014), due to the variability in hydro power generation, leading to significant variations in environmental impacts between years (e.g., GHG emissions varied between 287 and 434 g CO_2 eq kWh^{-1}) and limiting steady reductions in impacts.

The contribution of the increasing renewable energy penetration to reduce environmental impacts is likely to be partly offset by the increasing use of coal in place of natural gas.

Although NGCC installed capacity has increased significantly in the last decade, there has been a sharp decrease in the use of NGCC to the detriment of coal in the last years, with important consequences regarding environmental impacts. If this trend continues, the contribution of the increasing renewable energy penetration to reduce environmental impacts is likely to be partly offset by the use of coal. Increasing the utilization of the currently underutilized NGCC installed capacity has the potential to further reduce GHG emissions and non-renewable energy consumption associated with electricity in Portugal.

Temporal variability in GHG emissions of electricity in Portugal is significant, particularly between seasons.

Hourly GHG emissions associated with electricity generation and consumption in Portugal change considerably along the year, with hours with more than 80% of electricity generated by renewable sources contrasting with others in which coal supplied more than half. However, seasonal differences were found to be more significant than differences between hours along the year, mostly due to the heavy reliance on hydro power generation, which changes seasonally. This suggests that accounting for the temporal variability in electricity GHG emissions may have more impact for seasonal activities than for activities occurring during all year for a certain period of the day, such as BEV charging. Moreover, variability between years was also significant, thus using a single GHG emission factor to assess the environmental profile of products with a relatively long service life may not be representative.

Using an hourly LCA of electricity can provide a more accurate picture of the BEV impacts and the opportunities for improvement regarding both the vehicle (e.g., time of charging) and the electricity system.

Due to the high contribution of electricity generation to the GHG emissions of BEVs, GHG emissions of a BEV operating in Portugal may vary significantly throughout the year (up to 52%). Considering the annual mix to assess GHG emissions from BEVs does not account for this variability, which becomes more relevant as BEV energy consumption increases and the duration of charging decreases. Assessing GHG emissions of BEVs as a function the time of charging can shed light on the most favorable periods to charge BEVs.

Decreasing electricity GHG emissions during the summer by reducing coal generation (or increasing renewable generation) could, on the one hand, reduce variability within the year and, on the other hand, reduce the influence of the time of charging in the GHG emissions of BEVs. Although reduction in the use of coal is not foreseen in the near future if CO_2 pricing continues to follow recent trends, the time lag until large BEV penetration could provide an opportunity to act upon the electricity system towards decarbonization, for instance, by investing in increasingly more affordable solar power.

Off-peak charging incentivized by a lower electricity pricing will not necessarily lead to lower GHG emissions because of the higher relative share of coal in the electricity mix during the night, when demand is the lowest.

A BEV operated in Portugal in the last three years would have lower life-cycle GHG emissions if charged during the day (i.e. peak hours), in most cases, even though the cumulative difference was lower than 20%. However, this could also lead to increasing electricity consumption in periods of typically higher pricing (peak hours) and operationally critical from the system operator standpoint.

3. *What are the potential effects of EV deployment in the GHG emissions of the Portuguese electricity system in the short-term?*

Coal and natural gas combined cycle were identified as the marginal technologies in the short-term, but their contribution to the margin depend on the hour of the day, time of year and system load, causing marginal emission factors to vary significantly.

Increasing electricity consumption during off-peak hours was found to induce a higher increase in GHG emissions than in peak hours, due to a higher contribution of coal to the margin. In periods of low demand or high hydro availability, coal is often the marginal technology, as a result of the lower operation costs combined with the low price of CO_2.

Off-peak charging can more than double cumulative emissions compared to peak charging.

The addition of BEVs to the Portuguese electricity system would entail a higher increase in GHG emissions if vehicles were charged during off-peak hours. Therefore, understanding how marginal emissions from electricity generation vary over time can aid in the design of charging strategies that minimize environmental impacts from EVs.

The increase in electricity GHG emissions due to BEV charging in the short-term is not compensated by the displacement of ICEVs, so that the introduction of BEVs results in a net increase in GHG emissions in most cases.

The marginal supply in the short-term is composed by a high percentage of coal generation, so that the additional electricity demand by BEVs induces high GHG emissions, which, together with the increase in vehicle manufacturing impacts, does not compensate the emissions savings by displacing ICEVs in most scenarios. However, BEV effects on GHG emissions are very dependent on the time of charging and on the assumptions about the displaced technology, including the activity level of both BEVs and displaced ICEVs.

4. *How does the addition of EVs to a grid with a large storage capacity influence the GHG emissions of the electricity system?*

When the electricity system includes significant storage of energy from intermittent sources, the effects of introducing BEVs go beyond the straightforward displacement of ICEVs and increase in electricity demand, to include significant indirect effects from the dynamics of storage.

In the absence of storage, introducing BEVs charged at times of low demand may increase the penetration of RES and therefore lead to major reductions in GHG emissions. However, with storage and high BEV penetration, the net GHG savings are lower: diversion of surplus electricity from storage to BEVs means that there is an overall increase in the load on the electricity system. For the specific case of Portugal, the savings are still positive if the BEVs are charged from RES or natural gas combined cycle plants but, if the marginal production is from coal, increasing penetration of BEVs leads to increased GHG emissions.

The net effects on GHG emissions are very dependent on the technologies displaced both by PHS and by BEVs, so that detailed analysis is needed for any specific energy system, allowing for future technological improvements.

While new vehicles enter the fleet at a relatively high rate, power plants have a long service life so that the electricity system is subject to technological lock-in; therefore, the net GHG benefits of introducing BEVs decline over time because the efficiencies of the displaced ICEVs improves more rapidly than the average efficiency of generating plants. Displacing gasoline ICEVs leads to higher GHG savings than displacing diesel, due to the lower efficiency of gasoline engines.

5. *What insights does one gain by applying an attributional and a consequential LCA approach to the same electricity system?*

For an electricity system with a high share of non-dispatchable renewable power, such as the Portuguese system, marginal emissions in the short-term are generally higher than average emissions.

Increasing electricity demand generally means increasing fossil-based generation (e.g., coal or natural gas), resulting in higher emissions than the renewable-based average. For the Portuguese system, marginal GHG emissions can be up to 58% higher than average emissions considering the time of day. Whilst average emissions describe the life-cycle impacts of generating 1 kWh of electricity, marginal emissions depict the life-cycle impacts of increasing electricity generation by 1 kWh. Therefore, using average emissions to assess the impacts of implementing a new technology which uses or displaces electricity can underestimate the burdens or the savings achieved, respectively. It was shown that the introduction of BEVs in Portugal induce, in the short-term, a much higher burden than an attributional approach can depict. This extends to the other environmental impact categories assessed for electricity generation (e.g., acidification, eutrophication, ozone layer depletion), as emission factors calculated for both coal- and natural gas-based generation were also higher than for the remaining power plants.

The existence of storage in the electricity system to balance intermittent renewables tends to increase marginal emissions resulting from the introduction of BEVs, but decreases average emissions compared to a system without storage.

Marginal emissions due to increased use of BEVs are higher with PHS capacity, given that the marginal demand is met by fossil plants (such as natural gas, in Portugal), and higher than the average GHG emissions, particularly when BEV penetration is high. On the other hand, average emissions are generally lower for a system with storage, because hydro generation is higher. In the long-term, average emissions tend to decrease over time as low-carbon policies are enforced, but marginal emissions may increase as increased EV demand exceeds surplus renewable power.

The findings from this research emphasize the importance of taking into account the dynamic evolution of the fleet, technology improvements over time, and changes in vehicle operation and background processes during the vehicle service life when assessing the potential benefits of displacing ICEVs by EVs in a fleet. Furthermore, they highlight the importance of a detailed analysis of the specific electricity system addressing temporal variability in electricity GHG emissions for the identification of the overall effects of EV adoption and the charging strategies that minimize environmental impacts.

6.2 Limitations and topics for future research

The indirect effects assessed in this thesis are not the sole effects of EV adoption. Effects over electricity transmission and distribution network as well as charging infrastructure are also worth exploring. Consequences outside the system considered, such as the effect of decreasing the use of gasoline or diesel in ICEVs or increasing the use of coal or natural gas for electricity generation on the use of these fuels elsewhere, were not accounted for. Moreover, future research is required to improve our understanding of the potential environmental rebound effects arising from the different cost of electric and conventional vehicles and the different operation conditions of these technologies (e.g., range, re-fueling/re-charging convenience, which may divert some of the VKT to alternative transportation modes).

The analysis focused on GHG emissions, for the sake of simplicity (with the exception of the annual electricity generation impact assessment which comprised other impact categories); nevertheless, other environmental impacts should be accounted for in order to make the analysis more comprehensive and potential trade-offs explicit. Emissions of pollutants such as nitrogen oxides, hydrocarbon, non-methane hydrocarbons, carbon monoxide and particulate matter are regulated for ICEVs (e.g., by the European emissions standards) and are important sources of air pollution in cities. Conversely, depletion of abiotic resources, such as metals, and toxicity impacts, particularly related to leakage of toxic substances from mining, may be a source of significant impacts from EVs (Nordelöf et al. 2014a). Electricity generation can have important impacts in water demand and quality (Masanet et al. 2013). Impacts on biodiversity should also be analyzed, as they may be important, for example, for hydropower.

The future changes considered in this research are uncertain and depend on a number of factors, including technical developments, policy measures, consumer acceptance, and other external factors. It is not the aim of this thesis to project vehicle sales, fleet growth or vehicle technology development and associated environmental impacts. Instead, it aims to provide a framework for the consideration of this factors in an integrated manner, allowing for the assessment of a range of scenarios intended to illustrate the extent of the effects of the introduction of EVs in a fleet. Nevertheless, the analysis would benefit from a more detailed approach to parameter uncertainty (e.g., by using Monte Carlo simulation techniques), in order to increase the robustness of the results. Probabilistic scenario analysis techniques, such as the one presented in Noshadravan et al. (2015), could be useful in this context to understand the overall variation in GHG emissions from different technologies and to compare the different scenarios.

The actual penetration of EVs will depend on a number of factors, including the relative economics of EVs and other alternative technologies. Consumer preferences towards EVs were not accounted for in this analysis. Future research could combine consumer-choice models capturing the preferences of consumers towards electric vehicles to inform on the future market share of EVs (e.g., Oliveira et al. 2015) and on the most probable displacement options. Disaggregation of technologies by vehicle segment in the current model could also improve the assessment of trade-offs between displacement options.

This thesis exposed potential interactions and effects of EVs on the Portuguese electricity system considering the current official plan of development of the electricity system for the next 15 years. A different pathway, but outside the scope of this thesis, would have been to use energy system modeling tools to optimize the investment in electricity generation capacity in face of EV demand and then assess the corresponding changes in operation of that capacity (an example of such analysis is the work by Pina et al. [2013; 2014]). Such tools could be useful, for instance, to shed light on the optimal level of PHS and RES capacity required to reduce GHG emissions in face of increasing EV demand.

The model could be expanded to consider other technologies than ICEVs and BEVs (e.g., plug-in hybrid electric vehicles) and other fuel pathways, such as biofuels. The effect of other measures to decrease impacts from transportation and how they compare with EV adoption could also be assessed using the framework presented in this thesis. These include, for instance, reducing the fleet size, decreasing the distance travelled by vehicles, delaying or anticipating scrapping, and incorporating biofuels. The framework developed could also be applied to other geographical contexts, with different fleet compositions and dynamics, and distinct electricity systems to identify the regions for which EVs could yield significant environmental benefits.

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APPENDIX I: CORE PUBLICATIONS (ABSTRACTS)

Dynamic fleet-based life-cycle greenhouse gas assessment of the introduction of electric vehicles in the Portuguese light-duty fleet[8]

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Abstract

Purpose: Reducing greenhouse gas (GHG) emissions from the transportation sector is the goal of several current policies and battery electric vehicles (BEVs) are seen as one option to achieve this goal. However, the introduction of BEVs in the fleet is gradual and their benefits will depend on how they compare with increasingly more energy-efficient internal combustion engine vehicles (ICEVs). The aim of this article is to assess whether displacing ICEVs by BEVs in the Portuguese light-duty fleet is environmentally beneficial (focusing on GHG emissions), taking into account the dynamic behavior of the fleet.

Methods: A dynamic fleet-based life-cycle assessment (LCA) of the Portuguese light-duty fleet was performed, addressing life-cycle (LC) GHG emissions through 2030 across different scenarios. A model was developed, integrating: (i) a vehicle stock sub-model of the Portuguese light-duty fleet; and (ii) dynamic LC sub-models of three vehicle technologies (gasoline ICEV, diesel ICEV and BEV). Two metrics were analyzed: (i) *Total fleet LC GHG emissions* (in Mton CO_2 eq); and (ii) *Fleet LC GHG emissions per km* (in g CO_2

[8] Garcia, R., Gregory, J., Freire, F. (2015). Dynamic fleet-based life-cycle greenhouse gas assessment of the introduction of electric vehicles in the Portuguese light-duty fleet. *The International Journal of Life Cycle Assessment* 20(9):1287-1299. http://dx.doi.org/10.1007/s11367-015-0921-8

eq/km). A sensitivity analysis was performed to assess the influence of different parameters in the results and ranking of scenarios.

Results and discussion: The model baseline projected a reduction of 30-39% in the 2010-2030 fleet LC GHG emissions depending on the BEV fleet penetration rate and ICEV fuel consumption improvements. However, for BEV introduction in the fleet to be beneficial compared to an increasingly more efficient ICEV fleet, a high BEV market share and electricity emission factor similar or lower to the current mix (485 g CO_2 eq kWh-1) need to be realized; these conclusions hold for the different conditions analyzed. Results were also sensitive to parameters that affect the fleet composition, such as those that change the vehicle stock, the scrappage rate, and the activity level of the fleet (11-19% variation in GHG emissions in 2030), which are seldom assessed in the LCA of vehicles. The influence of these parameters also varies over time, becoming more important as time passes. These effects can only be captured by assessing *Total fleet GHG emissions* over time as opposed to the *GHG emissions per km* metric.

Conclusions: These results emphasize the importance of taking into account the dynamic behavior of the fleet, technology improvements over time, and changes in vehicle operation and background processes during the vehicle service life when assessing the potential benefits of displacing ICEVs by BEVs.

Keywords: battery electric vehicles; internal combustion engine vehicles; greenhouse gas emissions; fleet model; life-cycle assessment.

Life-cycle assessment of electricity in Portugal[9]

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Abstract

The Portuguese electricity mix is undergoing a significant shift away from the technologies that have dominated generation for the past decades. This article aims at assessing the environmental life-cycle impacts of electricity generation and supply in Portugal (PT), including: i) modeling the main electricity generation systems; ii) modeling the transmission and distribution (T&D) grid infrastructure; iii) characterizing the evolution of the electricity sector in PT from 2003-2012; and iv) discussing how the recent changes in the technology portfolio affected the environmental performance of the electricity generated and supplied. The life-cycle assessment methodology was used to quantify impacts in: non-renewable fossil energy demand (nREn), global warming (GW), abiotic depletion (AD), acidification (AC), eutrophication (ET), photochemical oxidation (PO) and ozone layer depletion (OD). From 2003 to 2012, an overall reduction of the environmental impacts was achieved. In particular, since 2008, electricity generation impacts in AC and PO dropped sharply as a result of the installation of desulphurization (62% reduction in AC; 74% reduction in PO) and denitrification (5% reduction in AC) systems in coal power plants (PP), as well as the phase out of large fuel oil PP. For NREn, AD and GW, the reduction of impacts was less pronounced (9-22% in the generation mix; 14-22% in the supply mix). T&D grid added 5-14% to the environmental impacts due to infrastructure (<5%) and T&D losses (5-9%). Despite the large increase in renewable capacity (especially wind) and the investments in lower-carbon fossil fuel technologies (natural gas), electricity generation still relies heavily on coal. There is, however, potential to further reduce environmental impacts in key

[9] Garcia, R., Marques, P., Freire, F. (2014). Life-cycle assessment of electricity in Portugal. *Applied Energy* 134:563-572. http://dx.doi.org/10.1016/j.apenergy.2014.08.067

categories (NREn, AD and GW) since there is significant available capacity of natural gas combined cycle currently underutilized.

Keywords: electricity mix; environmental impacts; LCA; power; primary energy.

Fleet-based life-cycle approaches: a review focusing on energy and environmental impacts of vehicles[10]

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Abstract

Alternative vehicle technologies are being promoted to reduce energy consumption and environmental impacts in the transportation sector. Life-cycle assessment (LCA) is often used to assess and compare the environmental impacts of vehicle technologies, but, in its traditional form, it lacks the ability to capture the transient effects as new vehicles displace older vehicles in the fleet. Fleet-based life-cycle (LC) approaches, which combines the LCA methodology with fleet models that describe the stocks and flows associated with a class of products over time, has been proposed as a modeling approach to circumvent these issues. This article presents a critical review of the literature addressing fleet-based LC approaches by providing: (i) an overview of the modelling approach; (ii) its main applications; and (iii) an analysis of the key aspects underlying energy and environmental impacts of vehicle fleets (focusing on electrification pathways).

Fleet-based LC approaches have been applied with different purposes (e.g., to model recycling processes, to assess trade-offs between manufacturing and use impacts; to optimize products service life). The issue of evaluating the impacts of introducing alternative vehicle technologies is appropriately addressed by a fleet-based LC approach, because it allows for the capture of displacement effects, technological improvements over time, and changes in background processes. Several studies have used such an approach to assess scenarios of evolution of the light-duty transportation sector. The main key aspects identified were: fleet penetration rate, electricity source, fuel economy improvements, and vehicle weight reduction. Emission reductions were found to be very dependent on the

[10] Garcia, R., Freire, F. (2015). Fleet-based life-cycle approaches: a review focusing on energy and environmental impacts of vehicles. (submitted)

underlying assumptions of the study. Reducing fuel consumption is one of the key ways to reduce fleet GHG emissions, but it needs to be combined with other measures, such as high penetration of alternative technologies, to bring about significant reductions. The electricity generation source was also found to have a large impact on the fleet GHG emissions and increasing renewable energy penetration is key to reduce overall emissions.

Keywords: electric vehicles; life-cycle assessment (LCA); vehicle fleets; greenhouse gas emissions (GHG).

Effects on greenhouse gas emissions of introducing electric vehicles into an electricity system with large storage capacity[11]

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Abstract

Electric Vehicles (EVs) can potentially reduce environmental impacts by displacing Internal Combustion Engine Vehicles (ICEVs) and by enabling more intermittent Renewable Energy Sources (RES) by charging with surplus power in periods of low demand. However, the net effects on Greenhouse Gas (GHG) emissions of adding EVs into a national or regional electricity system are complex, particularly when the system includes RES and large scale storage capacity such as Pumped Hydro Storage (PHS). This paper explores the interactions between EVs and PHS, using life cycle assessment to compare changes in GHG emissions for different scenarios. The Portuguese electricity system is taken as a specific example, characterized by relatively high capacities of wind generation and PHS.

When there is significant storage of energy from intermittent sources, the effects of introducing EVs go beyond straightforward displacement of ICEVs and increase in electricity demand, to include significant indirect effects from the dynamics of storage. In the absence of storage, introducing EVs charged at times of low demand increases the penetration of RES, leading to major reductions in GHG emissions. However, with storage and high EV penetration, the net savings are lower: diversion of surplus electricity from storage to EVs means that there is an overall increase in the load on the electricity system. The net effects on GHG emissions depend on the technologies displaced by both PHS

[11] Garcia, R., Freire, F., Clift, R. (2015). Effects on greenhouse gas emissions of introducing electric vehicles into an electricity system with large storage capacity. (submitted)

and EVs, so that detailed analysis is needed for any specific energy system, allowing for future technological improvements.

Keywords: Industrial Ecology (IE), renewable energy sources (RES), pumped hydro storage (PHS), life cycle assessment (LCA), greenhouse gas (GHG), wind power.

APPENDIX II: FULL LIST OF PUBLICATIONS

Articles in international journals with scientific refereeing

Published

1. **Garcia, R.**, Gregory, J., Freire, F. (2015). Dynamic fleet-based life-cycle greenhouse gas assessment of the introduction of electric vehicles in the Portuguese light-duty fleet. *The International Journal of Life Cycle Assessment* 20(9):1287-1299.
   http://dx.doi.org/10.1007/s11367-015-0921-8
   JCR® impact factor (2014): 3.988

2. **Garcia, R.**, Marques, P., Freire, F. (2014). Life-cycle assessment of electricity in Portugal. *Applied Energy* 134:563-572.
   http://dx.doi.org/10.1016/j.apenergy.2014.08.067
   Citations (Scopus): 6 JCR® impact factor (2014): 5.613

3. Domingues, A.R., Marques, P., **Garcia, R.**, Freire, F., Dias, L. (2015). Applying Multi-Criteria Decision Analysis to the Life-Cycle Assessment of Vehicles. *Journal of Cleaner Production* 107:749-759.
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4. Faria, R., Marques, P., **Garcia, R.**, Moura, P., Freire, F., Delgado, J., Almeida, A. (2014). Primary and Secondary Use of Electric Mobility Batteries from a Life Cycle Perspective. *Journal of Power Sources* 262:169-177.
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5. **Garcia, R.**, Freire, F. (2014). Carbon footprint of particleboard: a comparison between ISO/TS 14067, GHG Protocol, PAS 2050 and Climate Declaration. *Journal of Cleaner Production* 66:199-209.
   http://dx.doi.org/10.1016/j.jclepro.2013.11.073
   Citations (Scopus): 11 JCR® impact factor (2014): 3.844

Submitted or in final preparation for submission to ISI-indexed journals

6. **Garcia, R.**, Freire, F., Clift R. (2015). Effects on greenhouse gas emissions of introducing electric vehicles into an electricity system with large storage capacity. (submitted)

7. **Garcia, R.**, Freire, F. (2015). Fleet-based life-cycle approaches: a review focusing on energy and environmental impacts of vehicles. (submitted)

8. Rangaraju, S., **Garcia, R.**, De Vroey, L., Marques, P., Messagie, M., Freire, F., Van Mierlo, J. (2015). Key parameters influencing the results of life cycle assessment of battery electric vehicle (in final preparation)

9. Marques, P., **Garcia, R.**, Kulay, L., Freire, F. (2015). Comparative life-cycle assessment of $LiMn_2O_4$ and $LiFePO_4$ batteries for electric vehicles. (in final preparation)

LCA research articles not directly related to the thesis (written during the PhD program)

10. Silva, D., Lahr, F., **Garcia, R.**, Freire, F., Ometto, A. (2013). Life cycle assessment of medium density particleboard (MDP) produced in Brazil. *The International Journal of Life Cycle Assessment* 18(7):1404-1411. http://dx.doi.org/10.1007/s11367-013-0583-3

    Citations (Scopus): 21 JCR® impact factor (2014): 3.988

11. **Garcia, R.**, Freire, F. (2012). Environmental assessment of wood-based panels: a comparison of life-cycle-based tools. *International Journal of Sustainable Construction* 1(1):63-71. http://ijsc.greenlines-institute.org/volumes/2012/IJSC_2012_V01_01_08_Garcia_63_71.pdf

Papers in proceedings of national and international conferences with scientific refereeing

12. **Garcia, R.**, Freire, F., Gregory, J. (2015). Life-cycle greenhouse gas emissions of displacing internal combustion engine vehicles by electric vehicles in the Portuguese passenger car fleet. In: Proceedings of Energy for Sustainability 2015 – Designing for People and the Planet, May 14-15, Coimbra, Portugal.

13. Marques, P., **Garcia, R.**, Freire, F. (2015). Primary energy and greenhouse gas emission factors for electricity in Portugal. In: Proceedings of Energy for Sustainability 2015 – Designing for People and the Planet, May 14-15, Coimbra, Portugal.

14. Marques, P., **Garcia, R.**, Kulay, L., Freire, F. (2015). Life-cycle assessment of electric vehicle batteries – $LiMn_2O_4$ versus $LiFePO_4$. In: Proceedings of Energy for Sustainability 2015 – Designing for People and the Planet, May 14-15, Coimbra, Portugal.

15. **Garcia, R.**, Marques, P., Freire, F. (2014). Avaliação de ciclo de vida da eletricidade em Portugal. In: VII Encontro Nacional do Colégio de Engenharia Mecânica: A Indústria e a Engenharia Portuguesa – Desafios e Perspetivas, March 14-15, Lisbon, Portugal.

16. Marques, P., **Garcia, R.**, Freire, F. (2013). Life cycle assessment of electric and conventional cars in Portugal. In: Proceedings of Energy for Sustainability 2013 - Sustainable Cities: Designing for People and the Planet, September 8-10, Coimbra, Portugal.

17. **Garcia, R.**, Marques, P., Freire, F. (2013). Emissões de ciclo de vida de gases com efeito de estufa de um veículo híbrido plug-in: modo elétrico versus modo convencional. In: 10ª Conferência Nacional do Ambiente / Congresso Nacional de Engenharia do Ambiente, November 6-8, Aveiro, Portugal.

18. **Garcia, R.**; Freire, F. (2013) A fleet-based life cycle assessment of electric vehicles in Portugal. In: Proceedings of Energy for Sustainability 2013 - Sustainable Cities: Designing for People and the Planet, September 8-10. Coimbra, Portugal.

19. Domingues, A.R., Marques, P., **Garcia, R.**, Freire, F., Dias, L.C. (2013). Communicating life-cycle assessment results: A comparison of vehicle alternatives

in Portugal. In: Proceedings of Energy for Sustainability 2013 - Sustainable Cities: Designing for People and the Planet, September 8-10. Coimbra, Portugal.

Oral communications and posters

20. **Garcia, R.**; Gregory, J.; Freire, F. (2015). The introduction of electric vehicles in the Portuguese car fleet: A dynamic fleet-based life-cycle approach. ISIE Conference 2015 - Taking stock of Industrial Ecology, July 7-10, Guilford, UK. (Poster)

21. **Garcia, R.**; Marques, P.; Freire, F. (2015). Life-cycle assessment of electricity in Portugal – attributional and consequential approaches. In: Proceedings of WAM 2015: Workshop on Assessment Methodologies – Energy, mobility and other real world applications, June 19, Coimbra, Portugal. (Oral communication)

22. **Garcia, R.**; Gregory, J.; Freire, F. (2015). A dynamic fleet-based life-cycle approach to assess greenhouse gas emissions from electric vehicle adoption. In: Book of Abstracts of the 2nd Discussion Forum on Industrial Ecology and Life-Cycle Management, March 5-6, Coimbra, Portugal. (Oral communication)

23. **Garcia, R.**; Gregory, J.; Freire, F. (2014). Dynamic fleet-based LCA of the Portuguese light-duty fleet: the introduction of electric vehicles. LCA XIV Conference, October 6-8, San Francisco, USA. (Oral communication)

24. **Garcia, R.**; Freire, F. (2014). Dynamic fleet-based LCA of electric vehicle systems. In: Book of Abstracts of the Discussion Forum on Industrial Ecology and Life-Cycle Management, April 7-8, Coimbra, Portugal. (Oral communication)

APPENDIX III: DYNAMIC FLEET-BASED LCA – SUPPLEMENTARY INFORMATION[12]

1. Vehicle stock sub-model

1.1 Fleet composition

Table A-1 Portuguese light-duty vehicle fleet composition in 1995 (Ceuster et al. 2007), used to initiate the vehicle stock simulation.

	Vehicle stock F(*i,k,1995*)	
Vehicle age (*k*)	Vehicle type (*i*)	
	Gasoline ICEV (*g*)	Diesel ICEV (*d*)
0	188355	30208
1	209387	38217
2	219271	28274
3	273539	27491
4	139329	20316
5	181312	17575
6	176471	8406
7	160020	8161
8	143560	5123
9	127237	5730
10	111214	5415
11	95732	8621
12	81050	7218
13	67415	5943
14	55034	4334
15	44056	3525
16	34557	2433
17	26543	956
18	19955	522
19	14678	331
20	10562	118

[12] Significant portions of this section appear in the Supplementary Materials of: Garcia R., Gregory J., Fausto F. (2015). Dynamic fleet-based life-cycle greenhouse gas emissions of the introduction of electric vehicles in the Portuguese light-duty fleet. *Int J Life Cycle Assess* 20(9): 1287-1299. http://dx.doi.org/10.1007/s11367-015-0921-8

1.2 Vehicle density

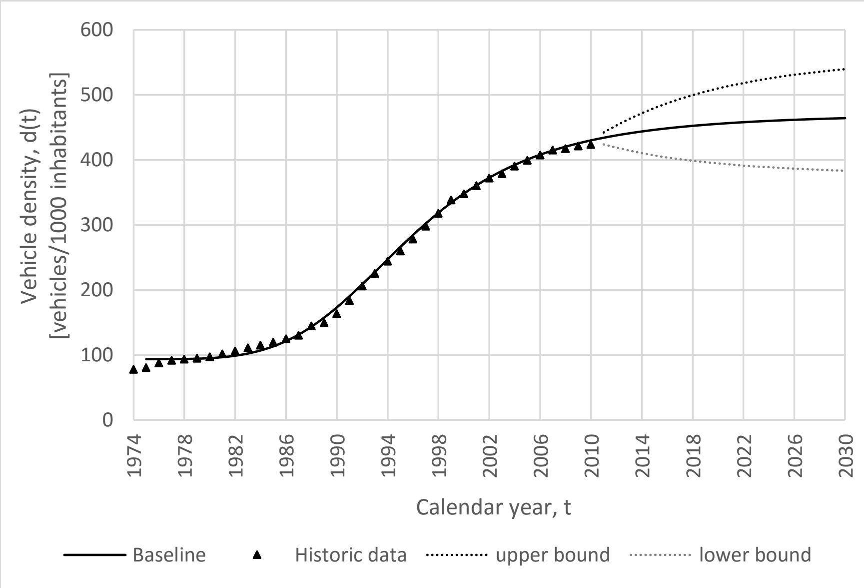


Fig. A-1 Light-duty vehicle density in Portugal: historic data, logistic curve calibrated for Portugal, and upper and lower bounds for sensitivity analysis.

1.3 Probability of surviving

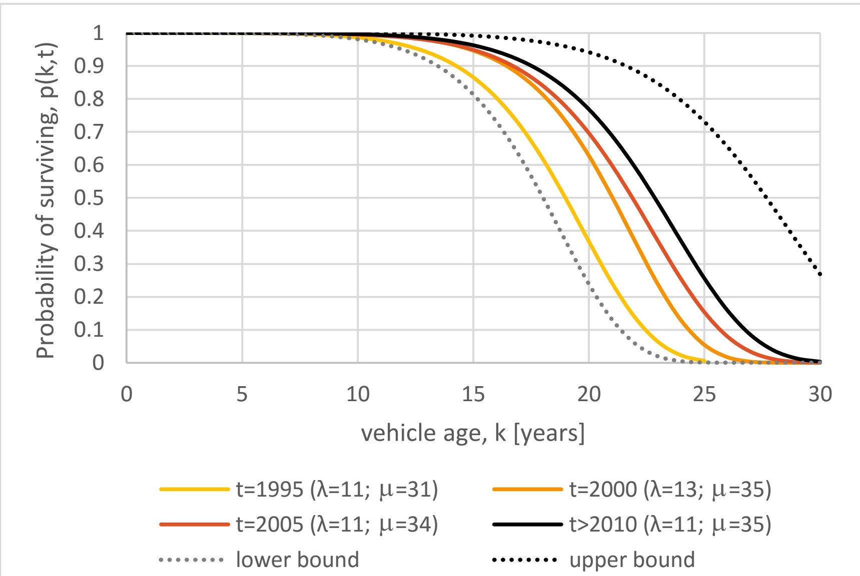


Fig. A-2 Probability of a light-duty vehicle surviving in the Portuguese fleet for different calendar years (based on Moura 2009), and lower and upper bounds for sensitivity analysis. As the vehicle age increases, its probability of surviving in the fleet decreases. The curves also indicate a later retirement of vehicles as time passes.

2. Dynamic life-cycle sub-models

2.1 Vehicle and battery weight and material composition

Fig. A-3 shows the evolution of the new light-duty vehicle curb weight in the Portuguese fleet for the three vehicle types (gasoline and diesel ICEV, and BEV). Future curb weight was assumed to decrease for ICEVs, since it is expected that weight reduction measures will take place as a means of reducing fuel consumption of new vehicles. A decrease of 16% in vehicle curb weight in 2010-2030 (0.8% per year) for ICEV vehicle technologies was assumed, to reflect the lightweighting of vehicles based on Bandivadekar et al. (2008). The upper bound for sensitivity analysis was set based on Cheah (2010), which showed that combining the use of lightweight materials and vehicle redesign for minimal weight can reduce vehicle weight up to 35%.

Electric vehicle batteries were also assumed to become lighter, as it is expected that the energy density of the battery packs will increase with model year. It was assumed that Li-ion battery pack energy density increases from 80 Wh/kg today (24 kWh capacity) to 235 Wh/kg in 2020 (45 kWh capacity), according to the USA Battery Consortium (USABC 2014), and constant thereafter.

Material composition of ICEVs was assumed to change over time according to Cheah (2010). The main changes are related to the substitution of cast iron and conventional steel by lightweight materials such as high-strength steel (HSS), aluminum, and plastics. Material composition of BEVs and batteries was assumed constant. Iron, steel, aluminum, and magnesium material production (i.e. extraction and processing) was assumed to become more energy-efficient and less GHG intensive over time (evolution according to Cheah 2010). Regarding other materials, energy use and GHG emissions were assumed constant over time. Energy intensity and GHG emissions from 1995-1999 were assumed equal to 2000.

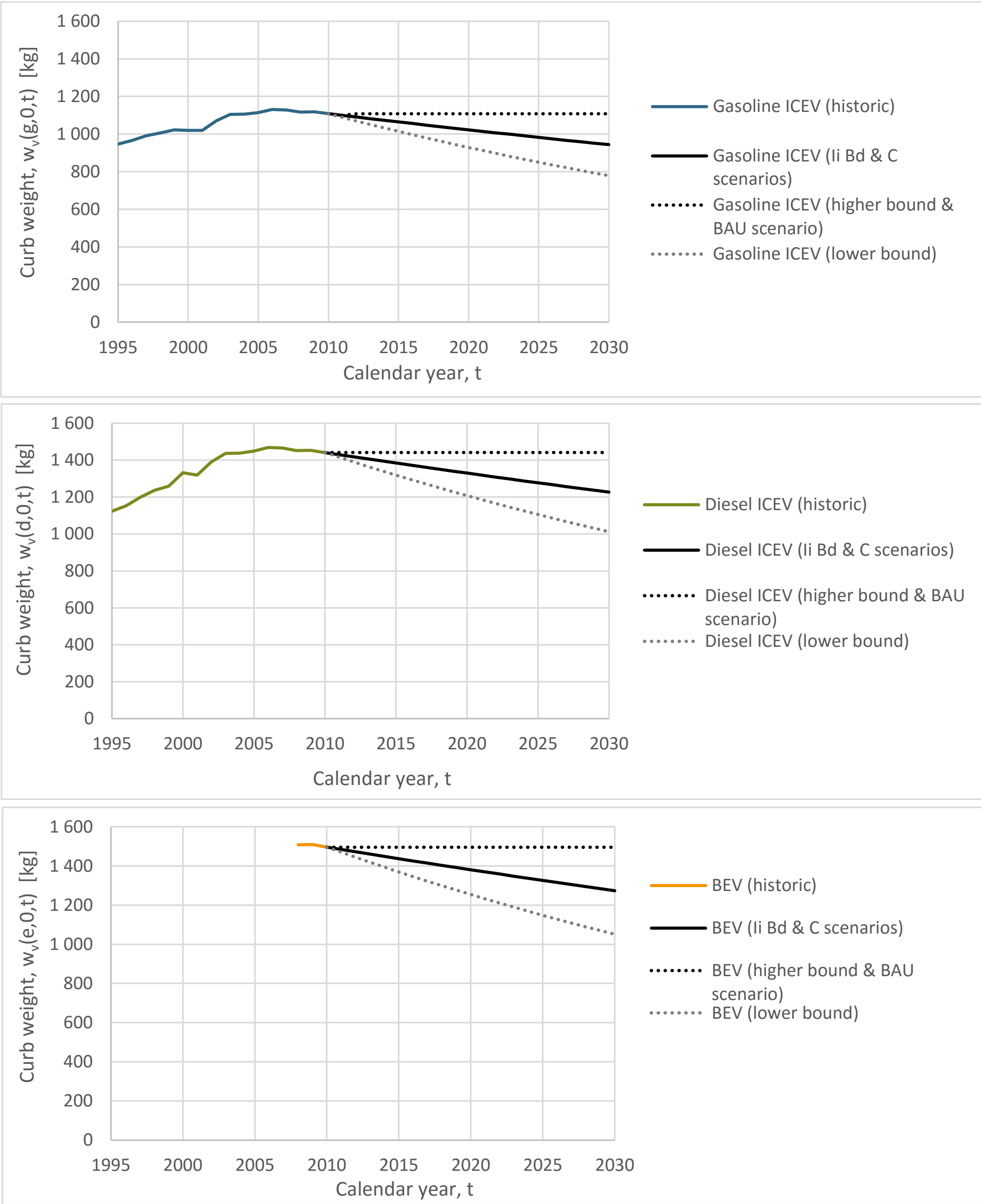


Fig. A-3 New light-duty vehicle curb weight evolution in the Portuguese fleet: historic data, scenarios and, lower and upper bound for sensitivity analysis.

Table A-2 BEV battery weight for different calendar years. It was assumed that Li-ion battery pack energy density increases from 80 Wh kg^{-1} today (24 kWh capacity) to 235 Wh kg^{-1} in 2020 (45 kWh capacity), according to the USA Battery Consortium (USABC 2014), and constant thereafter.

Calendar year, t	**Battery weight,** $w_b(e,0,t)$ [kg]
2010-2015	300
2016	254
2017	228
2018	212
2019	200
2020-2030	191

2.2 Maintenance

Table A-3 Maintenance operation schedule (g: gasoline ICEV; d: diesel ICEV; e: BEV; y: model year).

Maintenance operation, *m*	**Vehicle type,** *i*	**Cumulative vehicle distance traveled,** $t(i,m)$ [km]	**Emission factor,** $e_m(i,m)$ [kg CO_2 eq per operation]	**Source**
Battery replacement	*e*	100000	0.85[a]	Faria et al. (2014); Keoleian et al. (2012)
Tire replacement	*g, d, e*	80000	227.05[b]	Keoleian et al. (2012)
Engine oil substitution	*g, d*	10000 (y<2000); 30000 (y≥2000)	12.58[c]	Keoleian et al. (2012)

[a] Emission factor per kg of battery. Includes impacts from battery disposal and production of new battery.

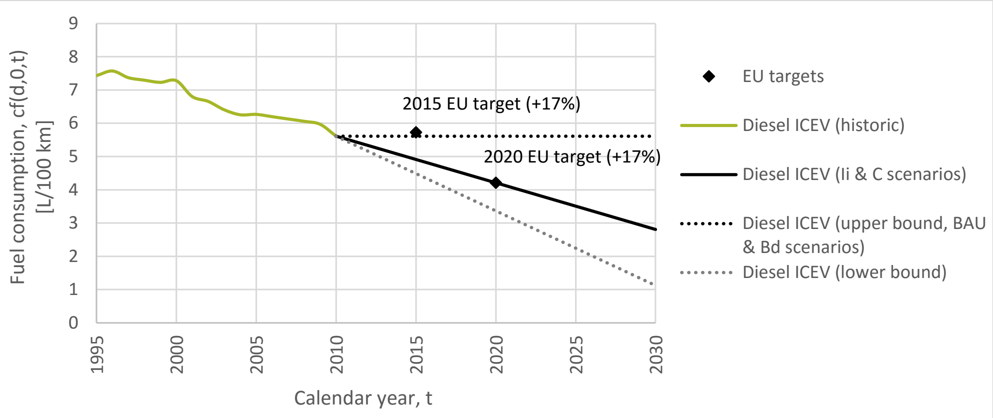
[b] Includes impacts from production and disposal of four tires.

[c] Includes impacts from engine oil production and disposal.

2.3 Fuel and electricity consumption

Vehicle fuel consumption varies with model year. Fig. A-4 shows the average fuel consumption of new internal combustion engine vehicles (ICEVs) from 1995 to 2030. Historic fuel consumption of new light-duty gasoline ICEVs and diesel ICEVs was obtained from European Commission (2012b), based on the New European Driving Cycle (NEDC) figures. Since fuel consumption in real-world conditions is considerably higher than measured in test-cycles, mainly due to the use of energy consuming devices such as air conditioners, a 17% increase in real-world consumption factors compared with test-cycle figures was assumed, according to Nemry et al. (2008).

For those scenarios in which improvements on new ICEVs (*ICEV improves* and *Combined,* see Table 3.3) were considered, future fuel consumption figures were set so that the EU targets for 2020 (4.1 L/100 km for gasoline ICEVs and 3.6 L/100 km for diesel ICEVs) would be met (25% decrease in 2010-2020), following a linear trend up to 2030 (resulting in a 50% decrease in 2010-2030). For the sensitivity analysis, the lower bound was defined based on the fuel consumption of diesel concept cars (1 L/100 km), assuming that that figure is reached in 2030 (resulting in an 80% decrease in 2010-2030; the same relative reduction for gasoline ICEVs was assumed).



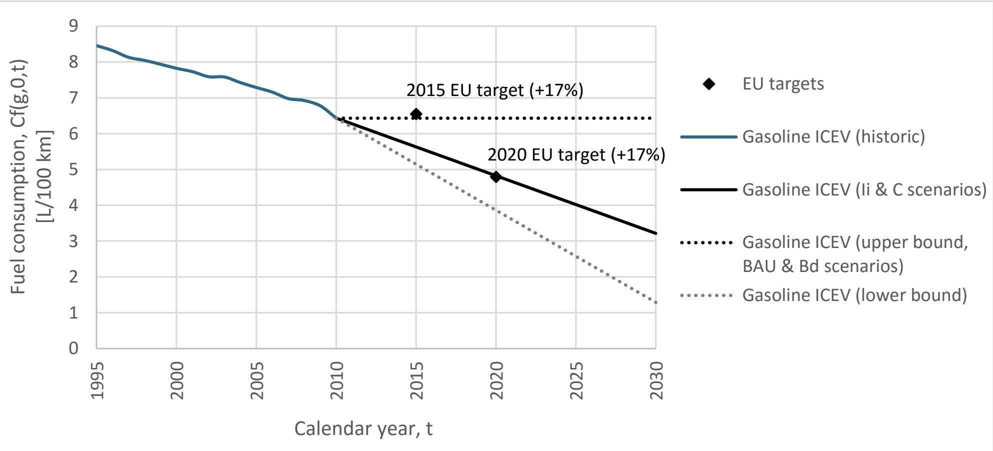


Fig. A-4 Fuel consumption of new light-duty ICEVs in Portugal: historic data, scenarios and, lower and upper bound for sensitivity analysis.

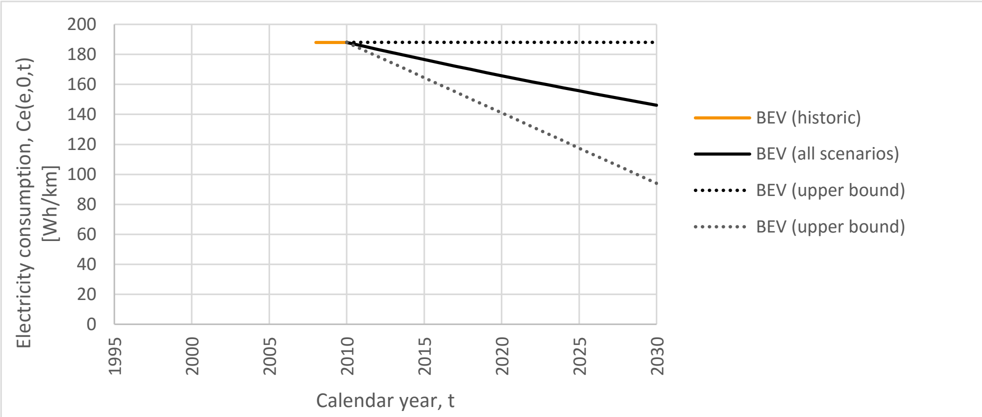


Fig. A-5 Electricity consumption of new BEVs in Portugal: historic data, scenarios and, lower and upper bound for sensitivity analysis.

2.4 Vehicle distance traveled

The distance travelled by a vehicle (vehicle kilometers traveled, VKT) varies depending on a number of factors, such as vehicle age, technology, and utilization purpose. VKT may decrease with vehicle age due to deterioration, reduced reliability, and shifting of primary to secondary car usage (Kim 2003; Moura 2009). Annual VKT estimations were based on vehicle inspection data for Portugal for 2005 from Azevedo (2007). A mileage reduction index curve was obtained for both gasoline and diesel ICEVs through adjustment of a logarithmic function to the vehicle inspection data set (r^2=0.9797 for gasoline ICEV; r^2=0.9893 for diesel ICEV) (Fig A-4). For BEVs the same VKT profile as gasoline ICEVs was assumed. However, since BEVs are about 70% more energy efficient than gasoline ICEVs, a higher VKT was assumed in order to account for the expected rebound effect, in line with Silva (2011). The same mileage reduction curve was used for all model years, but different first-year VKT values were assumed, as explained in the next paragraph. A sensitivity analysis was also performed to determine the effect of the VKT profile in the results considering higher or lower yearly reduction rates according to the curves presented in Fig. A-6. Fig. A-7 shows the VKT profile for model year 2010 for different technologies. Yearly VKT of diesel ICEVs is higher than gasoline ICEVs. A model year 2010 gasoline ICEV is driven around 140 000 km over a 15-year lifetime, a BEV around 170 000 km, and a diesel ICEV around 250 000 km.

First-year VKT for model year 2005 was derived from data from Azevedo (2007) (see Table A-4). For gasoline ICEV, first-year VKT figures for model years 1995-2010 were adjusted so that total fleet gasoline consumption approximates gasoline sales in Portugal reported by DGEG (2014) (see Fig. A-8; it was assumed that gasoline sales in Portugal are allocated to light-duty vehicles only). A reduction of 1%/year was assumed in 1995-2010, with reference to the 2005 data. It was assumed that first-year VKT of gasoline ICEVs continues to decrease until 2020, but at a slower rate (0.5%/year), and constant thereafter. According to ADEME (2012), the average light-duty vehicle VKT in Portugal has stabilized since 2000. Our baseline model assumes that this trend will continue in the future. This means that, since diesel ICEVs are driven more than gasoline ICEVs and their share in the fleet is expected to increase, their VKT should decrease. First-year VKT figures for diesel ICEVs were assumed to decrease 0.5%/year from 2000 to 2030 so that estimated average LDV VKT remains approximately unaltered. First-year VKT figures for BEVs were assumed constant in 2010-2020 and to increase 0.5%/year in 2020-2030, as battery technology improves. Since BEV VKT is assumed to be lower than diesel ICEV VKT, the fleet VKT decreases as BEV penetration rate increases. In order to understand the effect of BEV VKT in the results, a sensitivity analysis to the first-year VKT of BEVs (in 2010) was performed. The upper bound was set so that total fleet VKT is similar to that of the all ICEVs scenarios. The lower bound mirrors that value (relative to the baseline value).

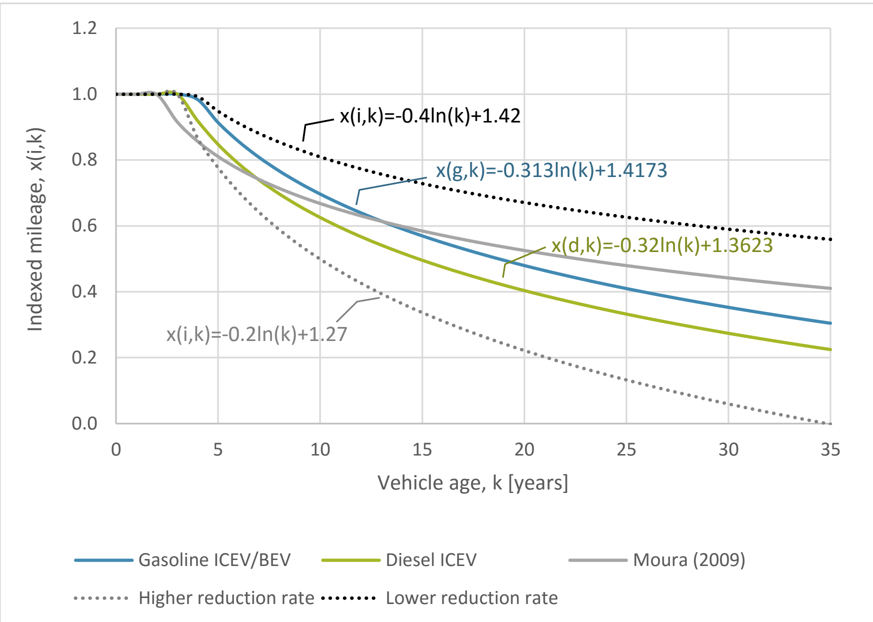


Fig. A-6 Indexed mileage for gasoline ICEVs, BEVs and diesel ICEVs estimated based on Azevedo (2007); higher and lower bounds for the sensitivity analysis; and comparison with the indexed curve used in Moura (2009).

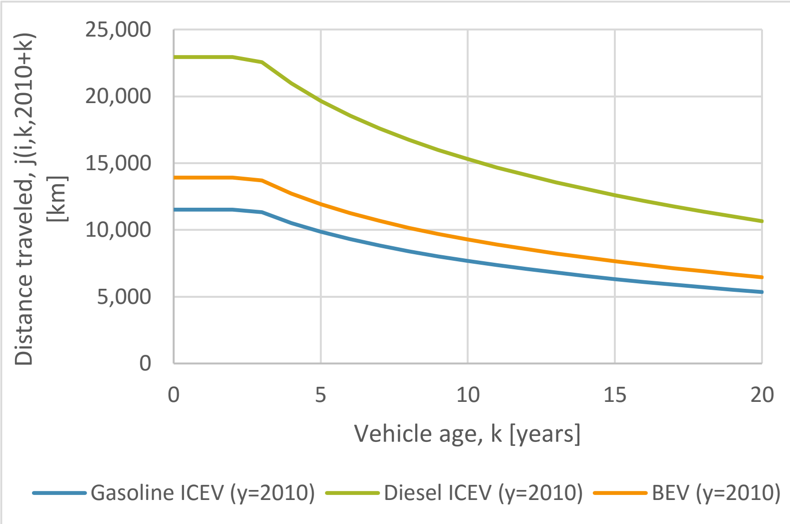


Fig. A-7 Annual vehicle distance traveled by powertrain for model year 2010.

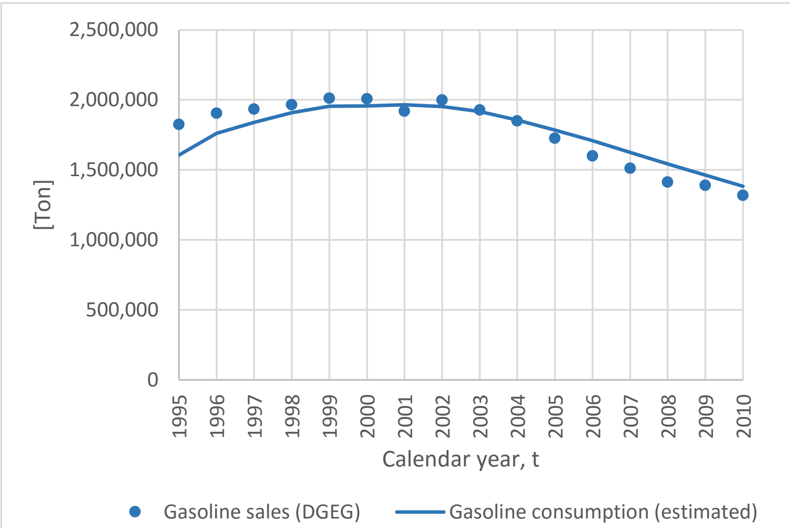


Fig. A-8 Gasoline sales in Portugal reported by DGEG (2014) and gasoline consumption calculated by the model.

Table A-4 First-year VKT for light-duty vehicles in Portugal. Data for model years 2005 was based on Azevedo (2007); for the remaining years, figures were estimated based on the yearly growth rates shown in Table A-5. Values in brackets are the lower and upper bound for sensitivity analysis.

Calendar year (*t*)	**y(*i,0,t*)**		
	Vehicle type (*i*)		
	Gasoline ICEV (*g*)	**Diesel ICEV** (*d*)	**BEV** (*e*)
2005	12 105	24 133	--
2010	11 512	22 950	13 929 (*10 500-17 500*)
2015	11 227	21 825	13 929 (*10 500-17 500*)
2020	10 949	20 756	13 929 (*10 500-17 500*)
2025	10 949	20 242	14 281 (*10 765-17 942*)
2030	10 949	19 741	14 642 (*11 037-18 395*)

Table A-5 First-year VKT yearly growth rate for light-duty vehicles in Portugal.

Calendar year (*t*)	**ϱ(*i,t*)**		
	Vehicle type (*i*)		
	Gasoline ICEV (*g*)	**Diesel ICEV** (*d*)	**BEV** (*e*)
1995-2010	-1.0%	-1.0%	--
2010-2020	-0.5%	-1.0%	0.0%
2020-2030	0.0%	-0.5%	0.5%

2.5 Fuel production and electricity generation

GHG emissions from gasoline and diesel production were obtained from Jungbluth (2007). According to ICCT (2010), GHG emissions from crude oil (extraction-to-refinery output) in Europe (and derivatives) are expected to increase by 7% up to 2020, due to increasing need to exploit deeper reservoirs, deeper waters, and emission-intensive sources, such as tar sands. This value was used as upper bound for the fuel production emission factor rate of change in the sensitivity analysis (2010-2020 reduction, constant thereafter). For the lower bound, a 6% reduction was assumed based on the European Commission reduction target for fuel upstream GHG emissions by 2020 (European Parliament and Council of the European Union 2009).

Historic GHG emissions from electricity generation and supply (e_e) in Portugal were obtained from Garcia et al. (2014). The average of the emission factors for the last 10 years (2003-2012) was assumed up to 2030 (constant value), in order to account for the variability between years (baseline). For sensitivity analysis, the upper bound (1.1 kg CO_2 eq/kWh) was set to simulate a coal-based mix and the lower bound a hydro-based mix (0.02 kg CO_2 eq/kWh), based on the technologies available in Portugal (Garcia et al. 2014).

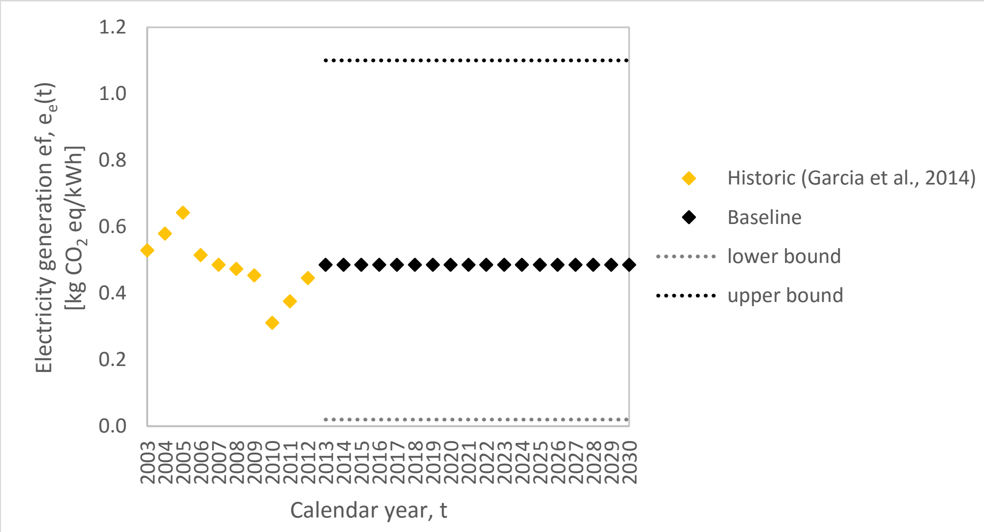


Fig. A-9 Electricity generation emission factors for Portugal: historic data (2003-2012), baseline value (average of 2003-2012), and lower (hydro) and upper (coal) bounds for sensitivity analysis.

3. Sensitivity analysis

3.1 Total fleet life-cycle GHG emissions

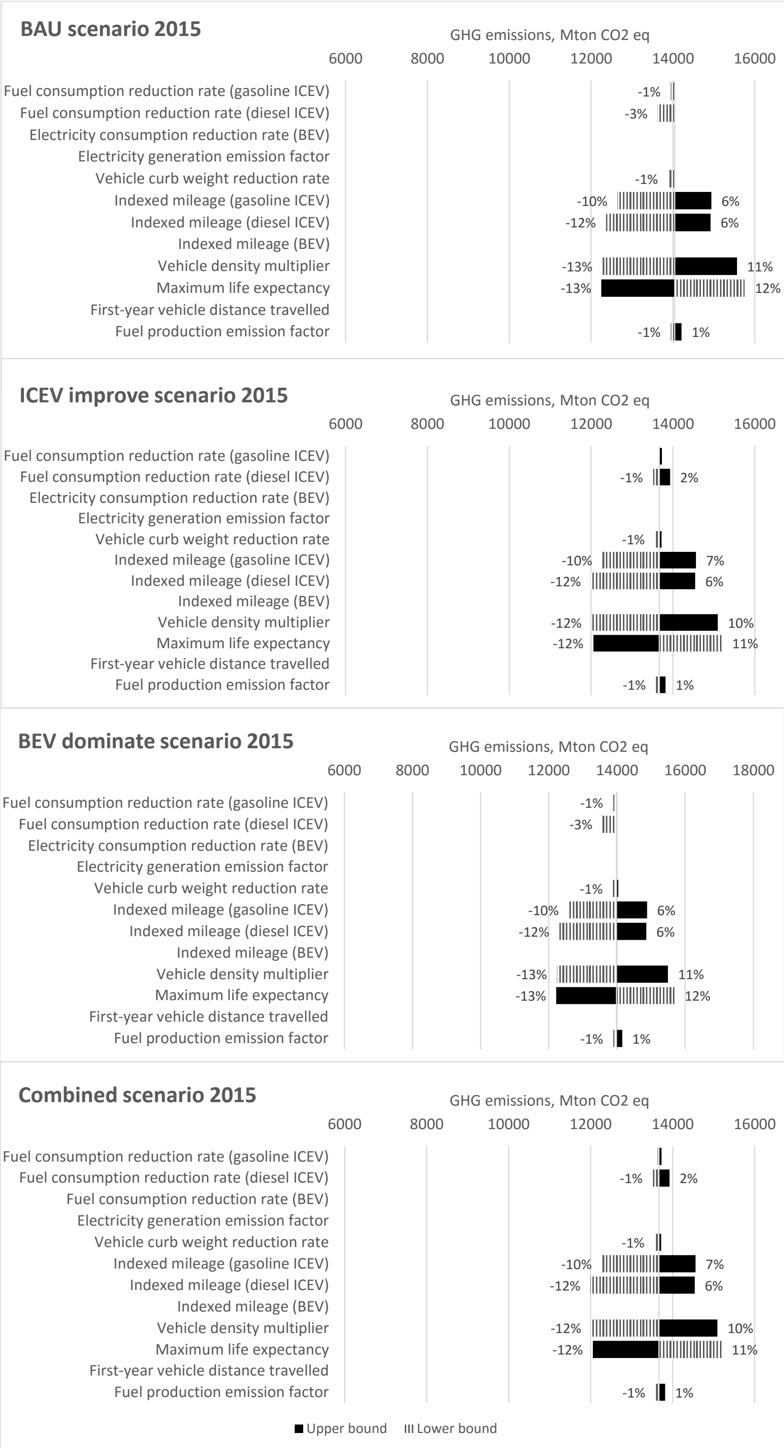


Fig. A-10 Sensitivity analysis results for total fleet LC GHG emissions in 2015.

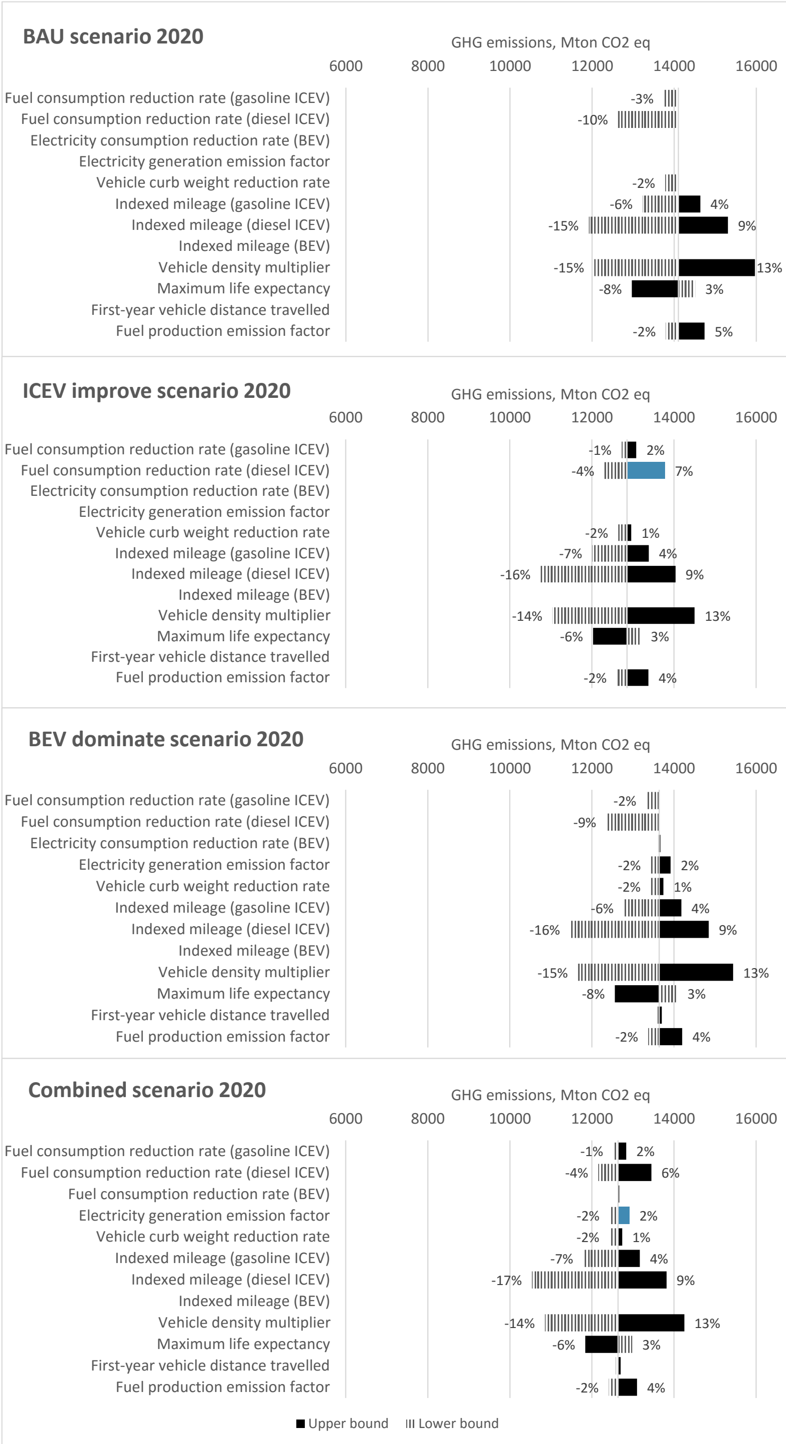


Fig. A-11 Sensitivity analysis results for total fleet LC GHG emissions in 2020. Blue bars indicate that the ranking of scenarios changes when the respective parameter is at its lower or upper bound.

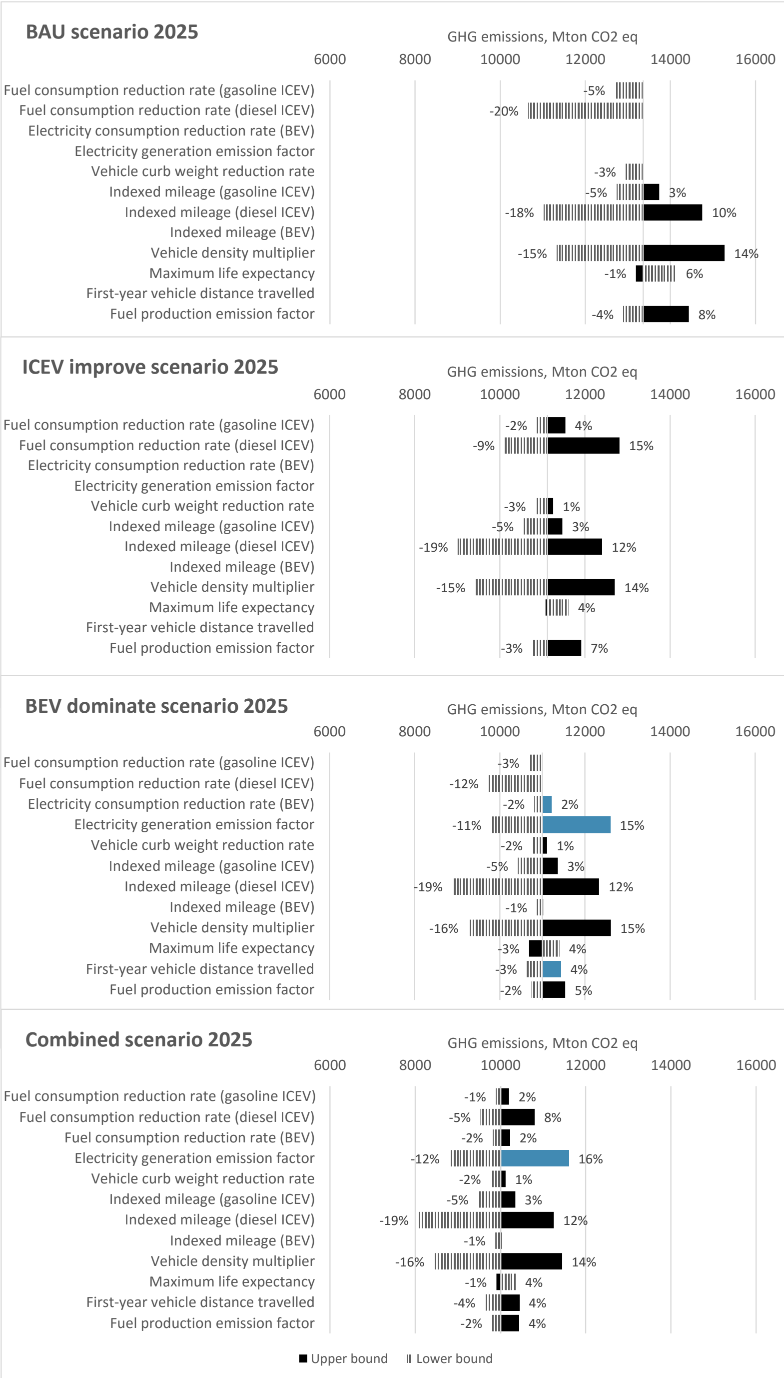


Fig. A-12 Sensitivity analysis results for total fleet LC GHG emissions in 2025. Blue bars indicate that the ranking of scenarios changes when the respective parameter is at its lower or upper bound.

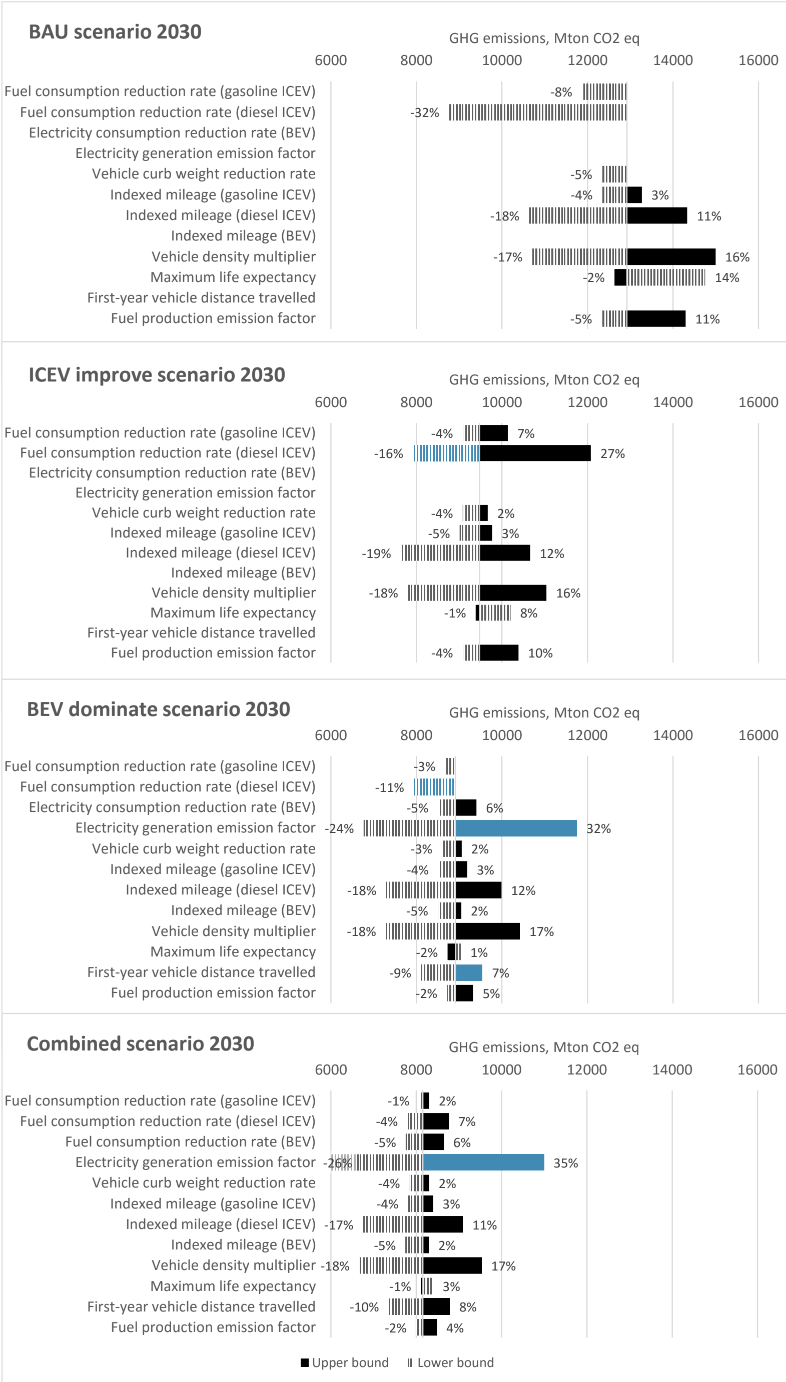


Fig. A-13 Sensitivity analysis results for total fleet LC GHG emissions in 2030. Blue bars indicate that the ranking of scenarios changes when the respective parameter is at its lower or upper bound.

3.2 Life-cycle GHG emissions per kilometer

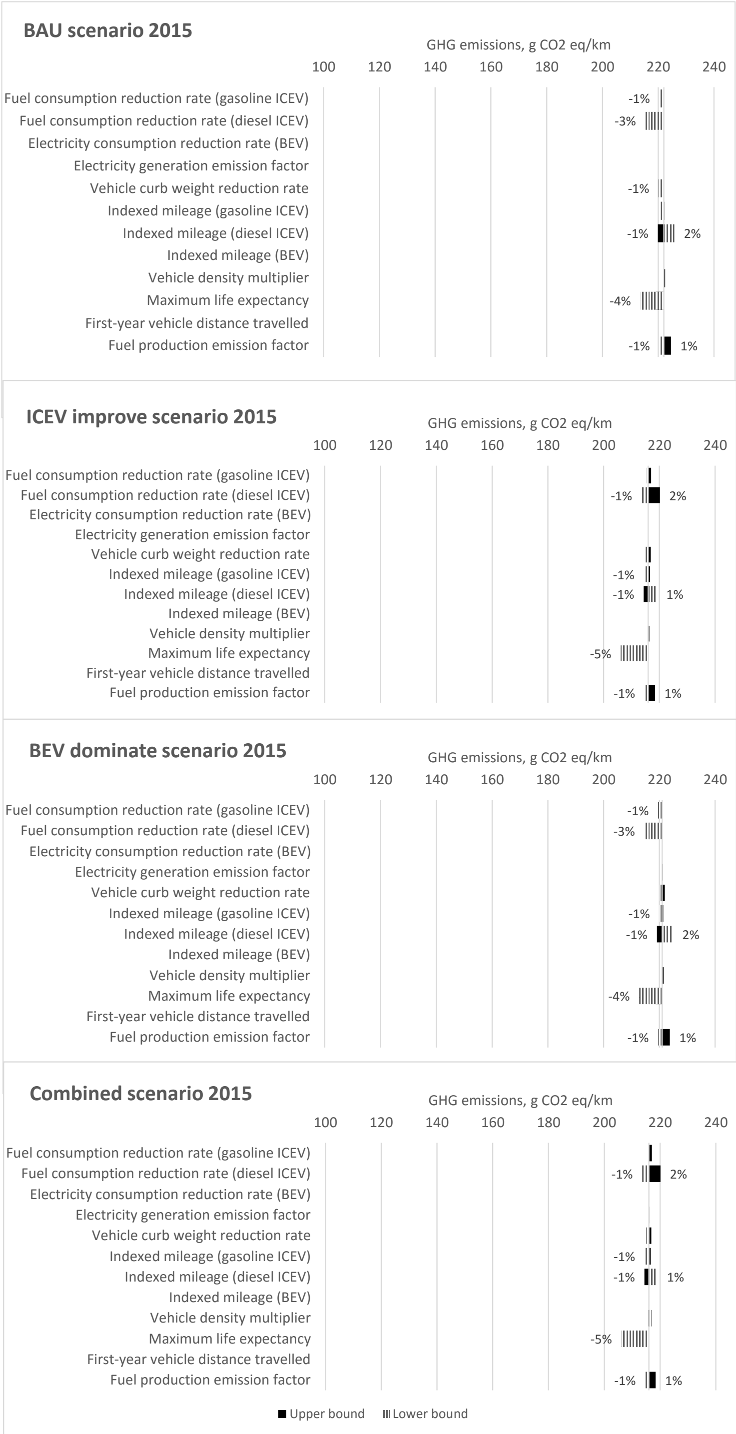


Fig. A-14 Sensitivity analysis results for LC GHG emissions per km in 2015.

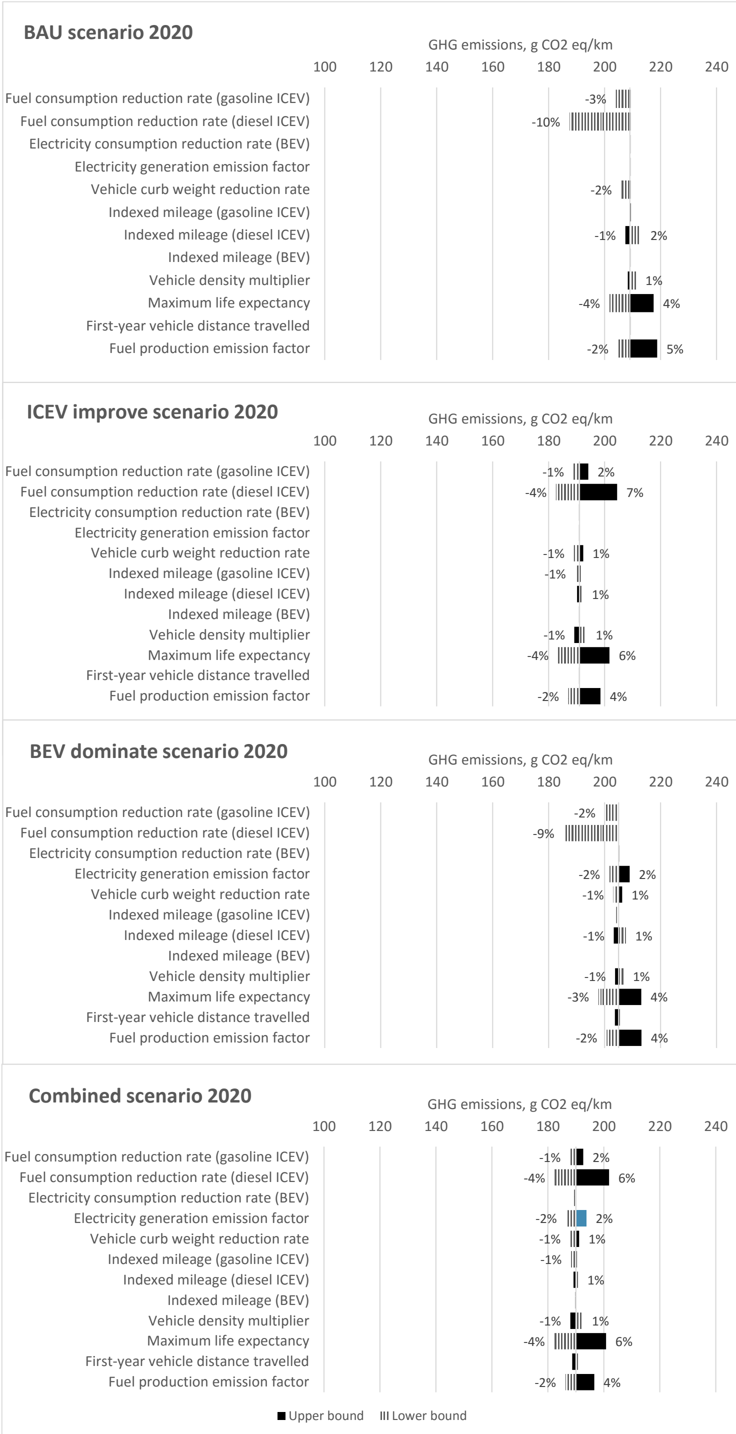


Fig. A-15 Sensitivity analysis results for LC GHG emissions per km in 2020. Blue bars indicate that the ranking of scenarios changes when the respective parameter is at its lower or upper bound.

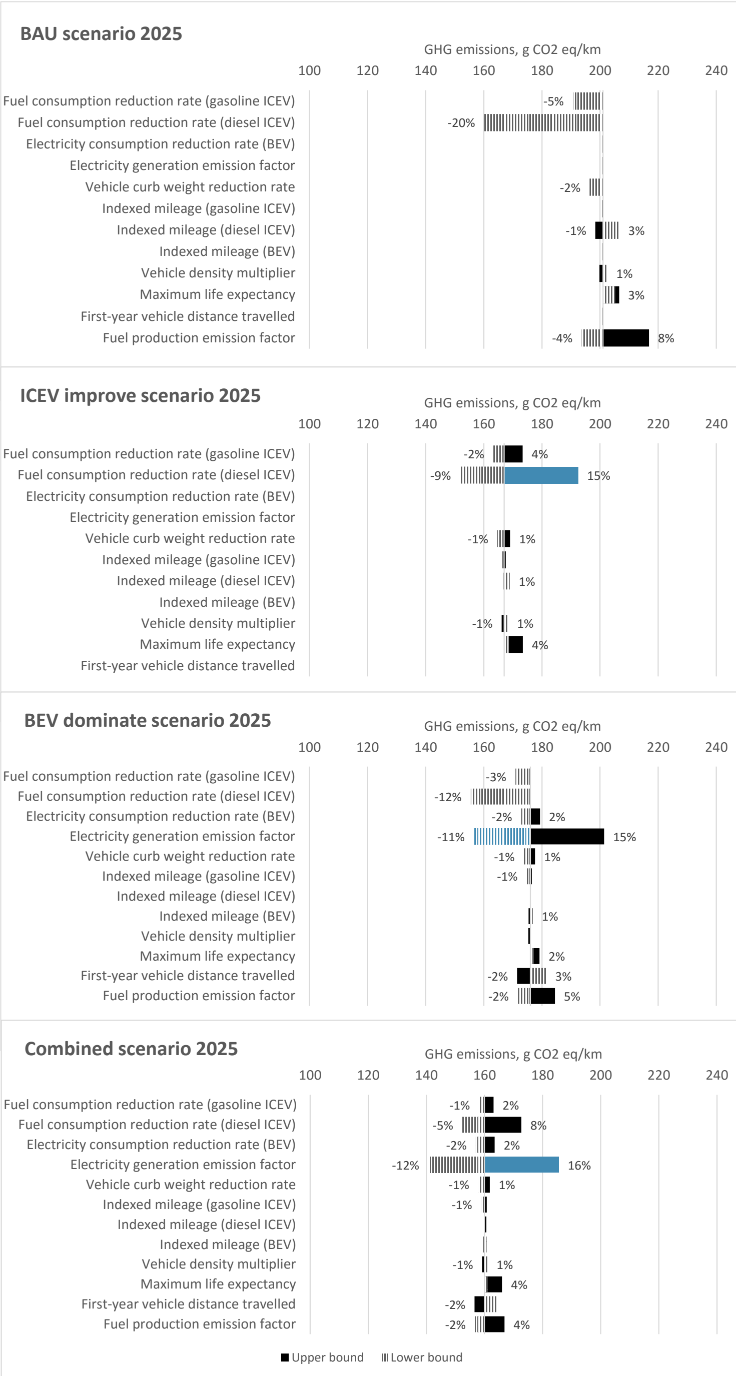


Fig. A-16 Sensitivity analysis results for LC GHG emissions per km in 2025. Blue bars indicate that the ranking of scenarios changes when the respective parameter is at its lower or upper bound.

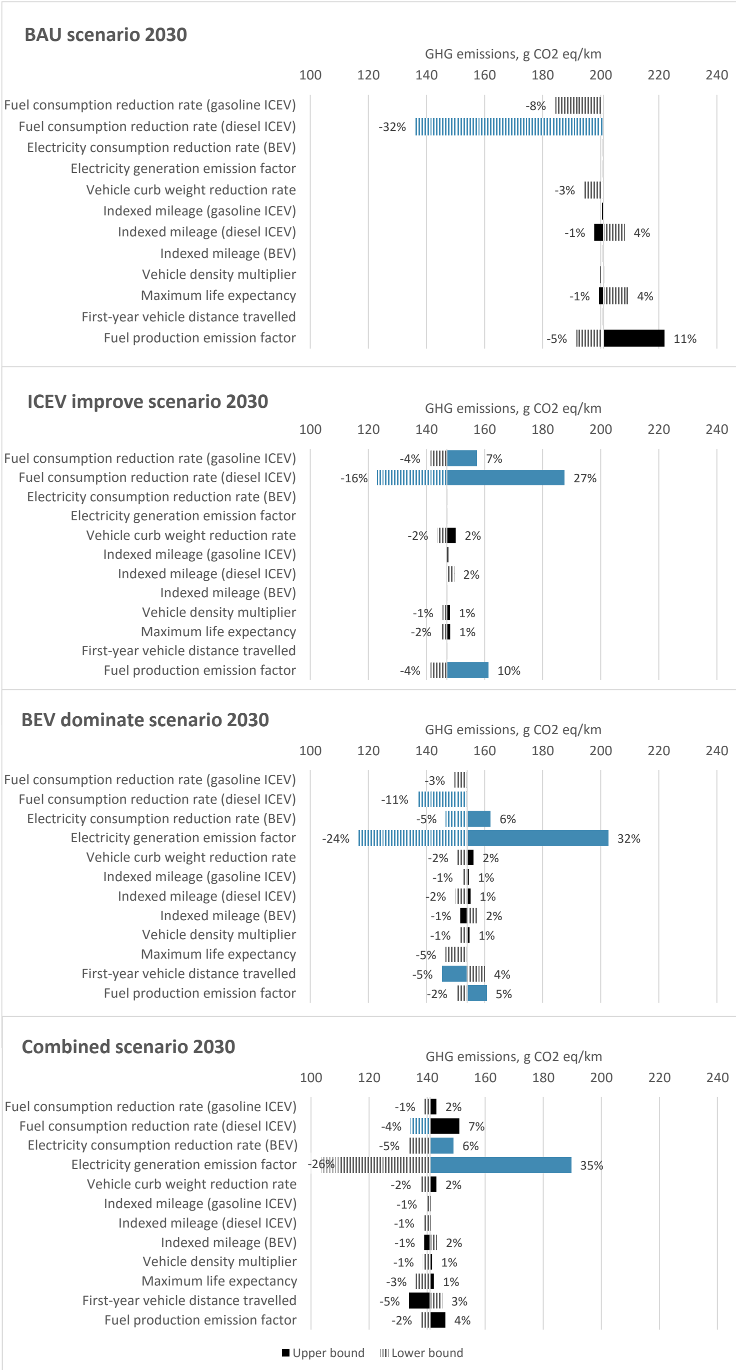


Fig. A-17 Sensitivity analysis results for LC GHG emissions per km in 2030. Blue bars indicate that the ranking of scenarios changes when the respective parameter is at its lower or upper bound.

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APPENDIX IV: LCA OF ELECTRICITY IN PORTUGAL – SUPPLEMENTARY INFORMATION

1. Attributional LCA of electricity in Portugal[13]

Table B-1 Contribution (%) of different technologies to the annual electricity mix in Spain (2003-2014) (REE 2014).

	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
Coal	30.3	30.1	29.2	24.2	25.5	16.0	12.4	7.9	16.0	19.8	14.9	16.9
Fuel Oil	3.4	3.0	3.8	2.2	0.8	0.8	0.8	0.7	0.0	0.0	0.0	0.0
Natural Gas CC	6.3	11.4	18.5	23.3	24.2	31.6	28.7	23.0	18.7	13.9	9.4	8.4
Natural Gas CHP	8.0	7.6	7.1	6.1	6.3	7.3	8.1	8.5	9.2	9.7	12.0	10.0
Nuclear	26.0	25.1	21.7	22.0	19.5	20.4	19.4	22.1	21.2	22.2	21.3	22.0
Hydropower	18.5	13.6	8.7	10.8	10.8	9.0	10.7	16.2	12.1	8.7	15.4	16.4
Wind	5.1	6.4	8.0	8.5	9.8	11.1	14.0	15.4	15.4	17.4	20.4	19.6
Biomass	0.7	0.7	0.8	0.8	0.8	0.9	1.1	1.1	1.4	1.7	1.9	1.8
Waste incineration	1.0	0.9	1.0	0.9	1.0	0.9	1.1	1.1	1.1	0.9	0.0	0.0
Biogas	0.9	1.1	1.2	1.2	1.2	1.1	1.4	1.5	1.6	1.5	0.0	0.0
Solar/Photovoltaic	0.0	0.0	0.0	0.0	0.2	0.9	2.3	2.5	3.4	4.1	4.6	4.9

Table B-2 Total length (km) of lines and cables installed in the Portuguese T&D grid by voltage level in 2011.

	Voltage level (kV)				**Total** (kV)
	Distribution	Transmission			
	<60	**150**	**220**	**400**	
Overhead lines	177306	2643	3465	2236	185650
Underground cables	48216		27		48243

Table B-3 Number of transformers installed in the Portuguese T&D grid by load rating in 2011.

	Load rating (MVA)					**Total** (MVA)
	Distribution			Transmission		
	5-12	**12-20**	**20-50**	**50-100**	**>100**	
Transformers	43	280	349	25	165	862

[13] Significant portions of this section appear in the Appendix of: Garcia R., Marques P., Freire F. (2014). Life-cycle assessment of electricity in Portugal. Applied Energy 134:563-572. http://dx.doi.org/10.1016/j.apenergy.2014.08.067

Table B-4 LCI data sources for T&D grid components.

	Transmission	Distribution
Overhead lines & underground cables	Jorge et al. (2011b)[a], Jorge and Hertwich (2013)[b]	Jones and McManus (2010)[c]
Transformers	Jorge et al. (2011a)	
Substations	Harrison et al. (2010)	

[a] Data source for LCI data on foundations, masts and insulations. LCI data for 220 kV overhead lines was adjusted from the LCI of 150 kV lines.
[b] Data source for LCI data on conductors.
[c] LCI data for 11 kV power lines and cables was used to model the distribution grid.

2. Temporal variability in the LCA of the Portuguese electricity system

Table B-5 Life-cycle GHG emissions of the Portuguese electricity system by season from 2012 to 2014 compared to the annual average emissions.

	2012		2013		2014	
	GHG emissions (g CO_2 eq kWh^{-1})	Δ from annual	GHG emissions (g CO_2 eq kWh^{-1})	Δ from annual	GHG emissions (g CO_2 eq kWh^{-1})	Δ from annual
Winter	484	*5%*	291	*-26%*	182	*-93%*
Spring	443	*-4%*	333	*-10%*	344	*-2%*
Summer	516	*11%*	469	*22%*	480	*27%*
Fall	401	*-15%*	371	*1%*	392	*10%*
Annual	461	-	367	-	351	-

APPENDIX V: EFFECTS ON GHG EMISSIONS OF INTRODUCING ELECTRIC VEHICLES INTO AN ELECTRICITY SYSTEM WITH LARGE STORAGE CAPACITY – SUPPLEMENTARY INFORMATION[14]

Table C-1 Energy consumption and LC GHG emissions of current and future mid-sized passenger vehicle technologies (generic case).

Technology	Energy source	Reference year	Energy consumption (MJ/km)[a]	**LC GHG emissions** (g CO_2 eq km^{-1})		
				Vehicle production and maintenance [a]	**Vehicle use**	**Total**
EV[c]	Surplus RES	2012	0.91	64	0	64
		2030	0.75	54	0	54
	NG CC	2012	0.91	64	109 [b]	173
		2030	0.75	54	83 [b]	137
	Coal	2012	0.91	64	231 [b]	295
		2030	0.75	54	181 [b]	235
ICEV	Gasoline	2012	2.8	41	260 [a]	301
		2030	2.17	37	198 [a]	239
	Diesel	2012	2.43	41	215 [a]	253
		2030	1.93	38	206 [a]	206

[a] Bauer et al. (2015)
[b] Calculated based on the electricity emission factors from Table 5.6, assuming an electricity transmission and distribution efficiency of 92%.
[c] The vehicle type considered is battery electric with no internal combustion engine.

[14] Significant portions of this section appear in the Supporting Information of: Garcia, R., Freire, F., Clift, R. (2015). "Effects on greenhouse gas emissions of introducing electric vehicles into an electricity system with large storage capacity". (submitted)

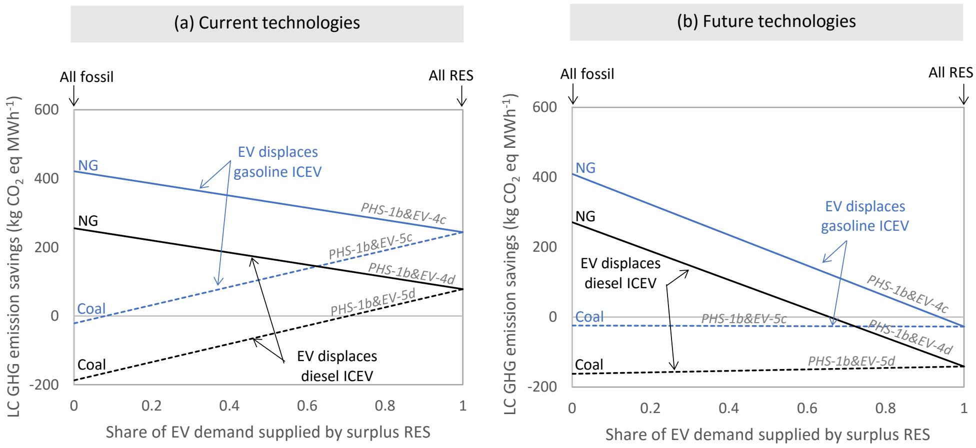

Fig. C-1 LC GHG emission savings per MWh of electricity consumed by EVs as a function of the proportion of renewable energy sources (RES) in the energy supplied to EVs, for Scenarios B in which pumped hydro storage (PHS) displaces coal (PHS-1b&BEV;Table 5.5). Results for (a) current technologies and (b) future technologies.