Joana Martins Faria de Bastos

LINKING THE URBAN FORM TO ENVIRONMENTAL AND HEALTH IMPACTS WITH A LIFE-CYCLE APPROACH

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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UNIVERSIDADE DE COIMBRA

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Abstract

Today, over half of the world’s population lives in urban areas, which account for the heaviest consumption of natural resources and significant generation of emissions and waste. Urban planning and design play a central role in promoting sustainable urban development by evaluating and guiding changes in the urban form. Environmental assessments of urban areas are needed to anticipate how different urban form characteristics can affect the environmental performance of urban areas and help urban planners and decision-makers to identify and design strategies toward sustainable development. However, urban areas are complex systems and challenging to evaluate for many reasons: assessments are multidisciplinary in nature; many approaches and indicators can be used to evaluate environmental performance; data requirements can be large; and there are many limitations and uncertainties. There is a need for systematic, integrated and objective frameworks to model, evaluate and compare alternative strategies and scenarios that can promote sustainable urban development.

The overall goal of this dissertation is twofold: to provide insight on key linkages between urban form and environmental impacts of urban areas; and to address critical issues in the application of life-cycle assessment (LCA) to urban systems. The research investigates environmental and health impacts associated with residential buildings and transportation, examining the interaction with the urban form. The research evaluates modeling and parameter choices in the LCA of buildings, the potential significance of user commuting requirements, the comparative environmental and health assessment of commuting modes, and how to adequately address exposure and health effects associated with traffic-related air pollutants within a LCA framework. These issues are addressed using four case studies drawn from the Lisbon (Portugal) area.

The results show that larger dwelling floor area per occupant can significantly increase energy demand and greenhouse gas (GHG) emissions associated with residential buildings, due to higher construction, use phase and retrofit requirements, but different metrics can provide contrasting results. Energy and GHG emissions associated with commuting can exceed those of residential buildings if housing is located in a car-dependent location. Environmental impacts of different urban transportation modes vary significantly, and future shifts in transportation mode or technology can cause trade-offs between impact categories, e.g., a shift to electric vehicles may decrease GHG emissions and health impacts associated with traffic-related air pollutants, but increase freshwater eutrophication. In addition, it is important to consider local environmental and health impacts, particularly in areas near major roads that are densely populated. On a methodological level, results show the importance of applying a life-cycle perspective in the environmental assessment of urban areas, the varying results and implications of using different impact indicators and metrics, the significance of using system boundaries that encompass the greater urban area in order to avoid missed or shifted impacts, and the need for spatially-resolved analyses that more accurately estimate local environmental and health impacts.

Drawing on the approaches and results of this research, recommendations are provided for the evaluation of environmental and health impacts associated with urban areas and the urban form. The proposed approaches extend and can improve the application of LCA to urban systems, by addressing a wide variety of impacts and key methodological issues, identifying potential improvement opportunities, and revealing unintended impacts and trade-offs. The methods can be applied in other cities and different settings, since they use mostly statistical data from national or international sources. The findings obtained for Lisbon may be generalizable since Lisbon's urban form characteristics and development trends are common to those in many other cities; however, local and site-specific assessments are recommended. The dissertation shows the feasibility of integrating urban and transportation planning in a LCA context that considers environmental and health goals, and it suggests the benefits that can be attained by using this type of tools to develop more sustainable urban forms.

Keywords: air pollution; environmental assessment; health effects; life-cycle assessment (LCA); residential buildings; urban form; urban passenger transportation; urban planning.

Resumo

Actualmente, mais de metade da população mundial vive em zonas urbanas, e estas assumem o maior consumo de recursos naturais e produção de emissões e resíduos, a nível global. O urbanismo assume um papel central no desenvolvimento sustentável, pois determina e avalia as transformações e a evolução da forma urbana. A avaliação de impactes ambientais associados a áreas urbanas permite entender como diferentes características da forma urbana afectam a performance ambiental e, dessa forma, apoiar profissionais no processo de tomada de decisão e no desenvolvimento de estratégias para a sustentabilidade. No entanto, as áreas urbanas são sistemas complexos e difíceis de avaliar por diversos motivos: a avaliação de impactes ambientais associados a áreas urbanas tem um carácter multidisciplinar, há diversas metodologias e indicadores que podem ser aplicados neste contexto, que requerem muita informação e dados nem sempre disponíveis, e com limitações e incertezas significativas. Assim, são necessárias abordagens sistemáticas, integradas e objectivas para desenvolver modelos, avaliar e comparar estratégias e cenários alternativos, no sentido de promover um desenvolvimento urbano sustentável.

Os principais objectivos desta tese dividem-se em duas componentes: por um lado, a investigação procura contribuir para o entendimento das ligações entre a forma urbana e os impactes ambientais das áreas urbanas; e por outro, abordar aspectos críticos na aplicação da avaliação de ciclo de vida (ACV) a sistemas urbanos. A investigação incide sobre os impactes ambientais e na saúde pública associados aos edifícios de habitação e aos transportes, explorando a ligação com a forma urbana. Mais especificamente, a investigação aborda decisões na modelação e nos parâmetros utilizados na ACV de edifícios, a importância dos movimentos pendulares, a avaliação comparativa de impactes ambientais e na saúde de vários meios de transporte urbano, e a adequada avaliação da exposição e dos efeitos na saúde associados a emissões poluentes dos transportes no contexto da ACV. Estas questões são abordadas através de quatro estudos de caso aplicados a Lisboa, Portugal.

Os resultados indicam que uma maior área habitável por pessoa pode contribuir para aumentos significativos nos requisitos de energia e emissões de gases de efeito estufa (GEE), nas fases de construção, reabilitação e uso dos edifícios de habitação; no entanto, a utilização de diferentes métricas pode ter resultados contrastantes. A energia e os GEE associados aos movimentos pendulares dos residentes podem ser maiores do que os impactes associados aos edifícios de habitação, quando estes estão localizados em zonas dependentes da utilização do automóvel. Os impactes ambientais associados a diferentes meios de transporte podem variar significativamente, e alterações dos meios de transporte ou das tecnologias utilizadas podem contribuir para *trade-offs* entre cagorias de impacte ambiental, e.g., a utilização e veículos eléctricos pode contribuir para reduções nas emissões de GEE e para a melhoria da qualidade do ar e da saúde pública, mas provocar um aumento da eutrofização da água doce. É

importante considerar impactes ambientais locais e os efeitos na saúde humana, especialmente em áreas com alta densidade populacional junto a grandes eixos rodoviários. A nível metodológico, os resultados mostram a importância de utilizar uma perspectiva de ciclo de vida na avaliação ambiental de áreas urbanas, os diferentes resultados e as implicações de considerar diferentes indicadores e métricas, a importância de utilizar fronteiras de sistema alargadas à escala urbana para evitar o desvio de impactes ambientais, e a necessidade de desenvolver análises com resolução espacial para avaliar impactes ambientais locais e efeitos na saúde.

Com base nas abordagens desenvolvidas e nos resultados obtidos, são feitas recomendações para a avaliação de impactes ambientais e na saúde, associados a áreas e à forma urbana. Os métodos desenvolvidos contribuem para o desenvolvimento e melhoria da aplicação da ACV a sistemas urbanos, nomeadamente por considerar diversos impactes ambientais e questões metodológicas importantes, identificar oportunidades de melhoria, e revelar impactes não intencionados e *trade-offs.* Os métodos podem ser aplicados a outras cidades e contextos, pois utilizam sobretudo dados estatísticos nacionais e internacionais. Os resultados obtidos para Lisboa poderão ser generalizados uma vez que as características da forma e do desenvolvimento urbano nesta cidade são semelhantes em várias outras. No entanto, avaliações locais e específicas são recomendadas. A tese mostra que a integração do urbanismo, do desenho urbano e do planeamento de transportes com valores ambientais e de saúde pública num contexto de ACV é viável, e sugere os benefícios que podem resultar da aplicação deste tipo de metodologias no desenvolvimento de formas urbanas mais sustentáveis.

Palavras-chave: avaliação de ciclo de vida (ACV); avaliação de impactes ambientais; edifícios de habitação; efeitos na saúde; forma urbana; poluição do ar; transporte urbano de passageiros; urbanismo.

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Acronyms and abbreviations

AC	Acidification
BEV	Battery electric vehicle
CA	City apartment
CO_2	Carbon dioxide
CO_2 eq	Carbon dioxide equivalent
DALY	Disability adjusted life years
EF	Effect factor
EIO	Economic input-output
EU	European Union
EU-15	Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, Netherlands, Portugal, Spain, Sweden and United Kingdom.
EU-25	EU-15 and Cyprus, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Malta, Poland, Slovakia and Slovenia
EU-27	EU-25 and Bulgaria and Romania
EU-28	EU-27 and Croatia
EV	Electric vehicle
FE	Freshwater eutrophication
GBD	Global burden of disease
GHG	Greenhouse gas
GIS	Geographic information systems
HEV	Hybrid electric vehicle
HSR	High speed rail
ICE	Inventory of carbon and energy
ICEV	Internal combustion engine vehicle
IPCC	Intergovernmental Panel on Climate Change
iF	Intake fraction
IO	Input-output
ISO	International Organization for Standardization
JRC	Joint Research Centre
LC	Life-cycle
LCA	Life-cycle assessment
LCI	Life-cycle inventory
LCIA	Life-cycle impact assessment
LPG	Liquefied petroleum gas
ME	Marine eutrophication

NRE	Non-renewable primary energy
p	Person
PKT	Person-kilometer traveled
PM	Particulate matter
PM_{10}	Respirable particulate matter (aerodynamic diameter of 10 µm or less)
$PM_{2.5}$	Fine particulate matter (aerodynamic diameter of 2.5 µm or less)
ppm	Parts per million
ppt	Parts per trillion
SH	Suburban house
TE	Terrestrial eutrophication
TRAP	Traffic-related air pollutant
UCTE	Union for Co-ordination of Transmission of Electricity
UFP	Ultrafine particulate matter (aerodynamic diameter of 0.1 µm or less)
UK	United Kingdom
US	United States of America
USA	United States of America
US EPA	United States Environmental Protection Agency
VKT	Vehicle-kilometer traveled
VOC	Volatile organic compound
WHO	World Health Organization
y	Year

1 Introduction

1.1. Background and motivation

The demands of a growing population, together with the increasing migration of people to urban areas, technology developments and lifestyle trends, have driven cities to rapid development and growth (Kennedy et al. 2007; Stephan et al. 2013a; UN 2011). Today, more than half of the world's population lives in urban areas, including 73% of the population of Europe, and this share is expected to continue to increase (UN 2015). Urban areas embody the heaviest consumption of natural resources and production of pollution and waste: they are estimated to account for over 70% of the total greenhouse gas (GHG) emissions associated with anthropogenic activities worldwide, which are recognized as the main driver of global warming (UN 2007, 2015). A reduction of the resource requirements and emissions would be a major contribution at global and local scales, and cities offer the greatest mitigation opportunities (Breheny 1992a; Jenks et al. 1996; Schaffer & Vollmer 2010; UN 2007, 2015).

By concentrating economic, social and cultural activities, cities drive regional and national economic growth and competitiveness, in developed nations, thus playing a key role in sustainable development (WCED 1987; European Commission 2009). The concept of *sustainable development* lies on the core principles of inter- and intra-generational equity, and integrates economic growth, social and environmental values (WCED 1987; European Commission 2009; Zheng et al. 2014). Essentially, sustainable development focuses on ensuring that everyone can have their needs met (e.g., food, water and shelter) within earth's ecological carrying capacity (WCED 1987; Vojnovic 2014). Sustainable development challenges differ in the developing and developed regions of the world (Næss 2001; Vojnovic 2014). In developing nations, key priorities are mostly in meeting vital needs of the people (e.g., access to safe water, sanitation and health care). In developed countries, sustainable development is often centered on reducing environmental impacts associated with inhabitants' demands and lifestyles, which tend to account for a high share of the global resource use, pollution and waste, thus compromising global sustainability (WCED 1987; Næss 2001; Vojnovic 2014). In this dissertation, the focus is on developed world regions.

Sustainable urban development promotes efficient land use patterns with reliable infrastructure for supporting economic activity, social justice, and management of resources and waste associated with urban areas (European Commission 2009; Zheng et al. 2014). Along the environmental dimension, sustainable urban development specifically aims at mitigating effects from and adapting to climate change, promoting sustainable transportation, reducing dependency on fossil fuels, increasing eco-efficiency of consumption and production patterns, improving conservation and management of resources, protecting natural ecosystems and biological resources, and reducing health risks to the community (Næss 2001; European Commission 2009; Zheng et al. 2014; Michael et al. 2014).

Urban planning and design play a central role in sustainable urban development, as they drive the changes in urban form, especially in the developed world. *Urban form* is the term given to the description of the spatial configuration and shape of the city. For the purpose of this dissertation, urban form is the three-dimensional physical environment of an urban area, from the city as a whole at a regional scale, to the neighborhood, building or street scale (Rapoport 1977). It is the result of manmade elements (e.g., buildings, transport, water and energy infrastructure), natural elements (e.g., landscape, topography and rivers) and of the space and relationships between these elements (Rapoport 1977; Marquez & Smith 1999; Williams et al. 2000; Kropf 2009). Urban form is characterized by shape, size, density and land use, for example, and it evolves continually, reflecting the needs, culture and values of a society, which change with social, environmental, economic and technological developments (Lynch & Rodwin 1958; Rapoport 1977; Williams et al. 2000; Williams 2014). The concepts, limits and operating scales of urban planning and design are not clearly and consensually defined. Generally, *urban planning* is systems-driven and focuses on the functional organization of the built environment, laying the spatial framework for human activities and interactions in the urban context (Lynch & Rodwin 1958; Steinø 2004; UN-Habitat 2009). *Urban design* blends the scales of planning and architecture, articulating functional and aesthetical attributes to shape and enhance the qualities of urban space (Steinø 2004).

The urban form directly or indirectly influences environmental impacts associated with urban areas, including local water and air quality, non-renewable energy demand, GHG emissions and global climate change (Marquez & Smith 1999; Alberti & Marzluff 2004; Alberti 2005; Jabareen 2006; Chen et al. 2011; Heinonen & Junnila 2011). The physical features and relations between urban form elements directly affect the environment, e.g., buildings and infrastructure use construction materials, the spatial configuration and shape of the built environment influence air quality and local meteorological patterns (e.g., urban heat island), transportation infrastructure determine travel distances, and land use patterns affect water run-off and natural ecosystems (Marquez & Smith 1999; Jabareen 2006; Kennedy et al. 2007; Bohnet & Pert 2010). Indirectly, the urban form influences environmental impacts by affecting human behavior, e.g., the built environment influences people’s lifestyles, their daily activities, travel demand and transport mode choice (Newman & Kenworthy 1996; Saelens et al. 2003; Kennedy et al. 2007; Pan et al. 2009; Heinonen & Junnila 2011). Urban planning also plays a crucial role on the resilience and adaptation of settlements to climate change and other environmental hazards (UN-Habitat 2009).

The increasing awareness on sustainable urban development has led to an interest on identifying environmentally sustainable urban forms, which have been discussed by researchers, planners and others (Jenks et al. 1996; Jabareen 2006; Williams 2014). The effects of two archetypal concepts have been extensively debated: the *compact* and the *disperse* city (Næss 2001; Jabareen 2006; Williams 2014). While several planning and design principles are generally considered to improve environmental performance of urban areas, such as compact, dense, mixed use developments, sustainable transportation systems and green infrastructure (Jabareen 2006), the complexity of the multi-dimensional linkages between urban form and environmental impacts associated with urban activities makes it challenging to achieve consensual agreement, or universally applicable solutions (Sovacool & Brown 2010). *Environmental performance* refers to the measurable environmental impacts of a system, and it can be evaluated using indicators that compare environmental results to the systems' functional or environmental objectives (ISO 2015). For example, an indicator of the environmental performance for climate change effects in transportation can be the GHG emissions per vehicle-kilometer traveled (VKT). Environmental and sustainability assessments of urban areas can provide insight on how different urban form characteristics affect the environmental performance of urban areas and help urban planners and decision-makers identify pathways and strategies toward sustainable urban development (Huijuan Dong et al. 2016).

The importance of urban form on environmental sustainability is evident, but there has been limited progress in integrating environmental values and goals in urban planning. Urban areas are complex dynamic systems, which are challenging to model, with large data requirements from multiple sources and fields (Marquez & Smith 1999; Schwarz 2010). The slow pace at which changes in the built environment translate into observable effects is also an obstacle (Pan et al. 2009). There is a need for systematic, integrated and objective frameworks to model, evaluate and compare alternative approaches, strategies and scenarios, to inform planners, designers and decision-makers (Jenks et al. 1996; Jabareen 2006; Weisz & Steinberger 2010; Williams 2014). Such frameworks should rely on indicator-based approaches that can provide an understanding of the effects of one or several alternative strategies or scenarios, with a focus on potential risks and benefits, unintended impacts and trade-offs (Holden 2006). Unintended consequences and trade-offs are expected in this context, due to the complexity of urban systems and the wide range of environmental impacts (Holden 2006). There are many approaches and indicators to evaluate environmental performance of urban areas, and many limitations to their use and appropriateness (Weiland et al. 2011; Huijuan Dong et al. 2016).

This dissertation addresses several linkages between urban form and environmental and health impacts. Understanding these linkages can enable and promote the potential of urban planning and design for reducing these impacts, which is especially relevant due to the comprehensive and long lasting effects resulting from urban planning decisions (Rickwood et al. 2008; Weisz & Steinberger 2010; Heinonen & Junnila 2011). The research focuses on residential buildings and transportation systems, and a life-cycle approach is used to provide a comprehensive analysis (as noted later in section 1.1.2).

1.1.1. Urban form and environmental impacts

Most of the literature linking urban form and environmental impacts consists of empirical studies comparing cities, neighborhoods or households, for example, with different urban form characteristics (e.g., land use patterns or housing typologies), and focusing on energy demand and GHG emissions (e.g., Holden & Norland 2005; Brown et al. 2009; Weisz & Steinberger 2010; Heinonen & Junnila 2011, 2014; Du et al. 2016).

Buildings and building elements have been a focus of environmental and sustainability research. Buildings represent one of the most significant contributors to environmental impacts of urban areas – including their energy demand, GHG emissions, the associated resource depletion and waste (Cabeza et al. 2014; Lotteau et al. 2015; Chau et al. 2015). Several studies have explored the influence of urban form on the environmental impacts of buildings, mostly focusing on energy demand and GHG emissions (Holden & Norland 2005; Ewing & Rong 2008; Okeil 2010; Anderson et al. 2015; Huang et al. 2017). Urban form can affect dwelling energy demand through three pathways: (1) housing typologies and (2) urban heat island effects influence use phase energy requirements, and (3) dispersed urban forms can increase energy distribution losses (Ewing & Rong 2008). Regarding housing typologies, larger dwellings and single-family housing (e.g., detached houses) likely have larger heating and cooling requirements, due to the increase in floor and envelope surface areas (Ewing & Rong 2008; Takano et al. 2015). The dwelling area per person is highly variable, especially across different countries, and might strongly affect the environmental impacts of the built environment. This topic, however, has received little attention in the literature. In industrialized countries, in particular, the dwelling area per person has increased in the last decades, due to the steady growth of the building stock, combined with low population growth, and decrease of household sizes (Næss 2001; O'Broin 2007). The temperature increase associated with urban heat island effects, which is generally greater in larger and denser cities, can also contribute to larger cooling requirements in summer, but potentially smaller heating requirements in winter. Energy distribution losses are unlikely to be significant, since overall energy losses represent a small share of primary energy demand (Ewing & Rong 2008).

Another crucial link between urban form and the environmental impacts of urban areas is the connection between land use and transportation demand. Urban transportation strongly contributes to non-renewable energy demand and dependence on fossil fuels, GHG emissions, resource depletion, acidification, eutrophication, traffic-related injuries, ambient air pollution and noise (Castro et al. 2003; Litman & Burwell 2006; Khreis et al. 2016). Many studies have examined how the spatial distribution of key activities (e.g., residences, workplaces, education, leisure and shopping), infrastructure (e.g., energy and transportation) and other urban form characteristics affect users' transportation demand and environmental impacts (Handy 1996; Kenworthy & Laube 1996; Newman & Kenworthy 1996; Badoe & Miller 2000; Stead & Marshall

2001; Camagni et al. 2002; Holden & Norland 2005; Næss 2005; Woodcock et al. 2007; Grazi & van den Bergh 2008; Dulal et al. 2011; Mitchell et al. 2011).

While research on the linkages between urban form and transportation has mostly focused on travel demand, energy use and GHG emissions, transportation also affects urban sustainability since vehicle traffic is the largest emitter of air pollutants in most urban areas, and these pollutants are associated with numerous adverse health effects (Marquez & Smith 1999; Fenger 2009; Nieuwenhuijsen & Khreis 2016). Urban form affects the demand for transportation, the locations where transportation-related emissions occur (through infrastructure network layout), the dispersion and resulting pollutant concentrations (e.g., through the configuration of buildings and street canyons), and population exposure (e.g., through land use and population distribution). For example, denser mixed urban areas might contribute to less motorized transportation demand and promote active travel, but they can be associated with higher exposures to traffic-related noise and air pollution (Næss 2014). The considerable health burdens associated with exposure to traffic-related air pollutants has received intensive analysis and scrutiny in the research and regulatory areas (Künzli et al. 2000; Nazelle et al. 2011; Zhang & Batterman 2013; EEA 2015).

Several studies have explored potential linkages between urban form, environmental impacts and health (Dulal et al. 2011; Loon & Frank 2011; Gago et al. 2013; Wilson & Chakraborty 2013; Næss 2014). However, most research and current practice have focused on energy demand and GHG emissions of specific sectors and selected urban components, e.g., individual buildings or transportation modes, not addressing the urban context and its implications, such as the location of a building on users' transportation demand (Anderson et al. 2015). There are few examples of systematic approaches or analyses that provide an understanding of how urban form characteristics influence the global (e.g., non-renewable energy demand, GHG emissions, eutrophication and acidification) and local environmental impacts (e.g., air pollution) of urban areas that can support decision-making, in particular in identifying improvement opportunities and potential trade-offs. Such information would contribute to effective environmental management and increase the sustainability of cities through urban planning and design (Breheny 1992a; Heinonen & Junnila 2011).

More knowledge is needed on the linkages between urban form and environmental and health impacts. This thesis examines several of these linkages and how a life-cycle assessment (LCA) framework can improve understanding. In particular, the research focuses on the environmental assessment of residential buildings and transportation, with the intention of addressing several issues and research gaps, described next.

1.1.2. Environmental assessment of urban areas

Industrial ecology approaches and tools including urban metabolism, material flow analysis and life-cycle assessment (LCA), have been used to assess the environmental impacts of urban areas (Bai 2007). These tools have been applied to buildings and building elements (Ortiz-Rodríguez et al. 2009; Sharma et al. 2011), transportation systems (Castro et al. 2003; Hawkins et al. 2012; Chester, Nahlik et al. 2013), entire cities and metropolitan areas (Kennedy et al. 2007; Heinonen & Junnila 2011) and, to a lesser extent, to intermediate scale areas, such as neighborhoods (Codoban & Kennedy 2008; Lotteau et al. 2015).

LCA is a tool for systematically and comprehensively examining the potential environmental impacts of a product, system or service, over their entire life-cycle, i.e., from raw materials acquisition, through production and use, to end-of-life treatment, recycling and final disposal (Rebitzer et al. 2004; Pennington et al. 2004; Finnveden et al. 2009). Two international standards, ISO 14040:2006 and ISO 14044:2006, provide a framework and consolidate the procedures for LCA (ISO 2006a, 2006b). The framework is organized in four phases, which are implemented iteratively (Rebitzer et al. 2004; Pennington et al. 2004; Finnveden et al. 2009). After the study goals and scope definition, the product's life-cycle processes and the associated *input and output flows* (e.g., energy and material requirements, emissions and waste) are modeled, in relation to a defined *functional unit*, resulting in the *life-cycle inventory* (LCI). Then, the potential environmental impacts associated with these flows are estimated in the *life-cycle impact assessment* (LCIA) phase, which includes different impact categories (e.g., non-renewable energy demand, greenhouse gas intensity, resource depletion, acidification, eutrophication and human health impacts). Lastly, interpretation of results and their implications should be performed (Pennington et al. 2004; Rebitzer et al. 2004; Finnveden et al. 2009).

LCA has great potential for the assessment of urban areas because it quantifies the resources and environmental impacts associated with processes and life-cycle (LC) stages beyond geographic limits (e.g., raw materials extraction or energy production), with consistent metrics (Pincetl & Bunje 2009; Weisz & Steinberger 2010; Soares et al. 2017). The comprehensive and function-centered characteristics of LCA are particularly compelling for comparing alternative strategies or designs and for identifying improvement opportunities and unintended trade-offs (Heinonen & Junnila 2011).

In the *building sector*, LCA studies have focused on individual buildings, typically quantifying impacts associated with construction, maintenance and use (Sartori & Hestnes 2007; Ortiz-Rodríguez et al. 2009; Ramesh et al. 2010; Sharma et al. 2011; Anderson et al. 2015; Chau et al. 2015; Anand & Amor 2017). Some trends can be identified in the literature, such as the dominance of the *building use phase* in the overall results, especially due to heating and cooling energy requirements (Sartori & Hestnes 2007; Ortiz-Rodríguez et al. 2009; Nemry et al. 2010; Ramesh et al. 2010; Sharma et al. 2011; Anderson et al. 2015; Lotteau et al. 2015). Urban scale issues, such as urban density, building typologies, mobility demand, shared equipment and infrastructure also require analysis, as they affect the environmental impacts associated with the built environment (Rickwood et al. 2008; Anderson et al. 2015). These issues have motivated a recent trend towards the application of LCA at larger scales, e.g., neighborhoods (Lotteau et al. 2015).

LCA studies of buildings present many methodological issues and choices, several of which are associated with high uncertainty and variability, such as use phase energy requirements, occupancy, building lifespan, the final-to-primary energy conversion factor, and the functional unit (Cabeza et al. 2014; Lotteau et al. 2015; Anand & Amor 2017; Dixit 2017; Soares et al. 2017; Vilches et al. 2017). For example, building LCAs have mostly used three functional units: *absolute* (i.e., the overall results for the whole building lifespan), *area-based* (i.e., per square meter of floor area, most often on an annual basis), and *occupancy-based* (i.e., per person per year) (Sharma et al. 2011; Cabeza et al. 2014; Lotteau et al. 2015). While results per square meter or per person ease inter-study comparability, area/occupancy ratios can be highly variable and have strong implications on the results and their interpretation (e.g., low occupancy might contribute to lower impacts per square meter, without better environmental performance). The variability and many modeling choices for building LCAs lead to a large range of literature results, which may be difficult to interpret and compare (see section 2.1). Understanding the implications of the functional units used in LCA studies of buildings is important for examining effects of urban density, dwelling size and occupancy, and for improving the transparency, comparability and interpretation of results (Lotteau et al. 2015; Anand & Amor 2017).

Research has generally focused on in-dwelling energy and environmental impacts (e.g., building construction, maintenance and use); however, a building's location can significantly affect the transportation demand of its inhabitants and associated environmental impacts (Maat & Timmermans 2009; Chester, Nahlik et al. 2013). Focusing on only the direct impacts associated with buildings can *shift impacts* and overlook improvement opportunities for more sustainable urban development (Rickwood et al. 2008; Anderson et al. 2015). As an example, an energy-efficient building in a suburban low-density development might have better environmental performance in terms of (lower) energy requirements per unit floor area or per capita than a conventional building in the city center, yet cause significantly higher overall impacts due to additional transportation-related energy requirements and environmental impacts (Stephan et al. 2013b). In order to support decision-making at the urban scale and avoid these shifts, impacts associated with buildings and transportation should be addressed together, with a comprehensive approach centered on users, not just on buildings, to consider energy and environmental impacts associated with buildings and their location (Lotteau et al. 2015). A review of LCA of buildings integrating user transportation requirements is provided in section 3.1.

In the *transportation sector*, LCAs have focused on energy demand and GHG emissions associated with alternative technologies and fuels within the same mode or a limited number of modes, e.g., comparing hybrid and electric vehicles (Hawkins et al. 2012). Very few studies have addressed pollution-related health effects (e.g., Chester, Nahlik et al. 2013), which are particularly relevant for urban areas. A broader set of environmental indicators is needed to identify and avoid unintended trade-offs in mitigation strategies (Bauer et al. 2015). The need to address exposure to traffic-related air pollutants and potential health effects is evident by the recent efforts to improve characterization in life-cycle impact assessment (LCIA) (Humbert et al. 2011; Apte et al. 2012).

The integration of health effects associated with traffic-related air pollutants in LCA requires the application of *spatially differentiated characterization factors* that account for the much higher exposure to emissions occurring within urban areas, and near major roads in particular (JRC 2011; UNEP/SETAC 2016). The exposure and consequent health effects per mass unit of fine particulate matter ($PM_{2.5}$) emissions, for example, from electricity generation, which generally occur from tall stacks located outside urban areas, can differ significantly from those associated with exhaust emissions in urban areas. A brief review on transportation LCAs, including details on estimating traffic-related air pollutant emissions and associated health effects, is presented in section 4.1.

The *intake fraction* (iF), defined as the fraction of a pollutant emitted from one or more sources that is inhaled by a population (Bennett et al. 2002; Marshall & Nazaroff 2006), has been used to quantify population exposure to traffic-related air pollutants (Greco, Wilson, Hanna, et al. 2007; Luo et al. 2010), and it has been recommended for application in LCIA (JRC 2011; UNEP/SETAC 2016). City-wide estimates of iFs for transportation emissions in urban areas in the literature (e.g., Stevens et al. 2007; Apte et al. 2012) appear to have significant uncertainty since (intra-urban) exposure variation, important for estimating health impacts, is not captured (Tainio et al. 2010; Tainio et al. 2014; Lamancusa et al. 2017), as discussed in section 5.2. For traffic-related air pollutants in particular, urban form is a key determinant of the spatial distribution of emissions and population, which govern exposure. Understanding the spatial and potentially the temporal variation of exposures is needed to provide reliable estimates of health impacts (Vienneau et al. 2009).

1.2. Research objectives

The overall goal of this thesis is twofold: to provide insight on the linkages between urban form and the environmental impacts of urban areas; and to address critical issues in the application of LCA to urban systems. The research investigates environmental and health impacts associated with residential buildings and transportation, addressing several issues on their interaction with the urban form within a LCA framework. Drawing on the overall goal and the identified literature gaps, the objectives are formulated as four research questions, presented next, together with the specific aims and research tasks.

1) How do modeling and parameter choices, such as the selection of functional units, affect the LC energy and GHG emissions of residential buildings?

The LC energy requirements and GHG emissions of three residential building designs are compared, focusing on building construction, retrofit and use phases. The assessment considers apartment buildings of different sizes and compares results using two functional units: per square meter-year and per person-year. Other modeling and parameter choices, such as final-to-primary energy conversion factors and building lifespan, are analyzed. The approach uses an econometric model based on statistical and consumer expenditure data to estimate in-dwelling energy consumption based on the number of occupants, building age, dwelling area, dwelling type, urbanization level and region, using statistical data for many of the required parameters. The model is broadly applicable to circumstances when historical and representative energy data is not available, and it circumvents the need for many of the assumptions and parameters required by engineering or demand-type models of household energy consumption. By comparing three apartment building designs in the same settings and built with similar construction materials, the assessment also provides insight on the effect of design, dwelling size and occupancy on the environmental impacts of residential buildings.

2) What is the potential significance of user commuting requirements on the LC energy and GHG emissions of residential buildings?

A comparative LC energy and GHG analysis of two representative buildings is performed at two locations: an apartment building in a central urban area and a semi-detached house in a suburban area. By comparing two building typologies and the different locations, the assessment provides insight on the potential significance of planning and design decisions. The analysis builds on the previous LC model implemented in question 1, but expands the assessment boundaries to the urban scale in order to examine the influence of a building’s location on user commuting (i.e., trips between home and work) energy demand and GHG emissions. The model includes building construction (construction and retrofit materials), building use and user commuting (with public and private transportation). Sensitivity analyses are performed to evaluate the effect of alternative residence and workplace locations. The need to consider user transportation requirements to better inform decision-making and avoid problem shifting is discussed.

3) How should the potential environmental and health effects associated with alternative commuting modes be assessed and compared?

A comprehensive LCA of six commuting modes (car, bus, overground train, subway, motorcycle and bicycle) in an urban setting is presented. A broad set of environmental indicators is used, which notably includes fine particulate matter ($PM_{2.5}$) emissions and the associated health effects. To incorporate potential health effects within a LCA framework, and specifically to appropriately account for exposure to emissions occurring in urban and non-urban environments, the assessment uses two intake fractions (iF), and an "effect factor" that combines dose response and severity factors. The approach is compared with the use of a single iF for all emissions. The implications of differentiating emission characterization to adequately address exposure is discussed. A scenario analysis illustrates an application of the approach and the potential trade-offs of five strategies that can reduce the environmental and health impacts associated with transportation.

4) How can exposure and health effects associated with traffic-related air pollution be addressed within a LCA framework?

Building on the application using iFs to represent exposure to urban and non-urban pollutant emissions (question 3), an approach is developed to calculate spatially-resolved iF estimates to improve the accuracy of health impact estimates for traffic-related air pollutants. As noted, city-wide estimates typically obtained using one-compartment models do not represent the spatial variation in exposures, which might be important for health impact estimates. The approach combines geographic information systems (GIS) and dispersion modeling to account for small-scale or intra-urban variation due to the distribution of emissions and population, and meteorological patterns. An application is performed for estimating spatially-resolved iFs for traffic-related primary $PM_{2.5}$ and results are compared with a city-scale one-compartment model iF estimate.

1.3. Importance and novelty

The four questions examined in this dissertation address aspects of how urban form affects environmental and health impacts associated with residential buildings and transportation, which assume a key role on the environmental performance of urban areas and, thus, on global sustainability. As described in the background and motivation (section 1.1), the understanding of how urban form affects environmental and health impacts, e.g., *what characteristics can be associated with lower impacts, what is the potential of different planning and design strategies to reduce environmental impacts, what are the potential unintended burdens,* is useful to inform and support urban planning. The approaches, which extend and improve the application of LCA to urban systems, are oriented to support and inform decision-making and urban planning by addressing a wide variety of impacts and key methodological choices, by identifying potential improvement opportunities, and by revealing unintended impacts and trade-offs.

Each of the questions is addressed using case studies drawn from the Lisbon area, in Portugal, based on statistical and publicly available data. Within the EU, Portugal is generally below average in terms of socio-economic conditions, but significant increases in quality of life have been observed in the last decades despite recent effects of the international economic crisis (Guedes et al. 2009; UN-Habitat 2016). Following the global trends, there has been a strong migration of population from inland rural areas to large urban regions near the coast. Currently, about 27% of the country's population lives in the Lisbon metropolitan area (2.8 million people; INE 2012). However, like many other cities in developed countries, the city has lost population to suburban areas in the last decades (INE 2012). The dispersion of population across suburban areas and other clusters in metropolitan areas has contributed to increased environmental impacts; controlling or even reversing this dispersion presents one of the main challenges of urban planning (Catalán et al. 2008; Guedes et al. 2009). These applications provide insight on real-world issues, and the approaches used can be replicated and applied to other urban areas.

1.4. Dissertation organization

This dissertation consists of six chapters, including this introduction, which summarizes the background, objectives, knowledge gaps, research goals and significance. Chapters 2 through 5, written as stand-alone sections for publication (see below), are structured according to the research questions presented in section 1.2. Chapter 2 addresses question 1 and presents a comparative LC analysis of three apartment building designs, quantifying and comparing the primary energy requirements and GHG emissions per square meter-year and per person-year. Chapter 3 (question 2) extends the assessment to address the implications of a building's location on users' commuting requirements, and compares the LC energy and GHG emissions of two residential typologies: an apartment building in the city center and a semidetached house in a suburban area. Chapter 4 (question 3) provides a comprehensive LCA of six commuting modes using a wide set of indicators, including an assessment of health effects associated with $PM_{2.5}$ emissions, which are calculated with two iFs, for urban and non-urban emissions, together with an effect factor. Chapter 5 (question 4) addresses some of the limitations of using a city-wide iF to estimate health impacts in urban areas. It describes and applies an approach for estimating spatially-resolved iFs for traffic-related air pollutants. An application for $PM_{2.5}$ is presented, and results are compared with those obtained using a city-wide one-compartment model iF estimate. Chapter 6 synthesizes conclusions. It includes a summary of key findings and contributions of the dissertation, and presents recommendations and suggestions for further research.

Chapters 2 to 5 are based on four manuscripts or articles that are published or under review, in ISI-indexed journals. Abstracts of these four articles are presented in Appendix I, and a full list of publications related with the research is provided in Appendix II.

2 LCA of three residential building designs addressing population- and area-based functional units

This chapter presents a life-cycle energy and greenhouse gas analysis of three representative residential building designs in a well-known area in Lisbon, *Bairro de Alvalade*. The life-cycle model focuses on building construction, retrofit and use phases, applies an econometric model to estimate energy use in Portuguese households, and considers two functional units: per square meter-year and per person-year. [1]

Section 2.1 presents a short review on life-cycle studies of buildings, identifying key methodological issues and choices. Section 2.2 provides an overview of the materials and methods, the life-cycle model and the buildings. Section 2.3 presents the results and discusses the importance of the size/occupancy relationship in life-cycle studies of buildings, as well as the implications of using different metrics. Section 2.4 summarizes the conclusions.

2.1. Introduction

In 2010, residential buildings accounted for around 27% of the final energy consumption in the EU-27 and about 16% in Portugal (EU 2012). Residential buildings represent a major opportunity for reducing energy requirements and greenhouse gas (GHG) emissions (Stephan et al. 2012). As noted in chapter 1, the potential of urban planning and design to reduce energy and GHG emissions has been discussed for some decades (Burchell & Listokin 1982; Breheny, 1992b; Jabareen 2006), and research is needed to assess its specific influence on energy requirements and GHG emissions (Breheny, 1992b; Norman et al. 2006). In the LCA context, studies on buildings present many methodological issues and choices, some of which are associated with high uncertainty and variability regarding use phase energy requirements, building lifespan, energy production mix, and other factors that lead to a large range of results and that can impede inter-study comparisons (Anand & Amor 2017; Dixit 2017).

[1] Significant portions of this chapter are from Bastos, J., Batterman, S. & Freire, F., 2014. Life-cycle energy and greenhouse gas analysis of three building types in a residential area in Lisbon, *Energy and Buildings* 69: 344-353.

This chapter presents a life-cycle (LC) energy and GHG analysis of three residential building designs, including construction, retrofit and use phases. The main objectives are to quantify the primary energy requirements and GHG intensity of the buildings and the relative contributions of each LC phase, with two functional units: per square meter-year and per person-year. The subsequent sections of the chapter review LC studies of residential buildings in urban areas, characterize the building types, describe the LC model, present and discuss the results, and summarize the chapter conclusions.

2.1.1. Life-cycle studies of residential buildings

Over the last several decades, many authors have highlighted the importance of a LC perspective to understand the environmental impacts associated with buildings (e.g., Blanchard & Reppe 1998; Fay et al. 2000; Adalberth et al. 2001; Nemry et al. 2010; Soares et al. 2017). Table 2.1 summarizes selected LC studies of residential buildings, focusing on conventional buildings, i.e., built according to practice prevailing at the time and location (Sartori & Hestnes 2007), as opposed to passive or low energy designs. For simplification, the LC stages are listed in the table as construction, use and end-of-life, but differences might exist in the approaches. For example, in the construction phase, some studies consider only materials manufacture, while others estimate impacts for materials transport and on site activities. These and most other studies examining residential buildings have several common findings, such as the operation phase of buildings being responsible for a major share of the energy consumption and GHG emissions (e.g., Adalberth 1997a; Thormark 2002; Norman et al. 2006). However, the studies have many methodological differences, such as the building lifespan, the LC phases considered, whether final or primary energy is considered, the final-to-primary energy conversion factor, and the functional unit considered (Anand & Amor 2017; Dixit 2017), as discussed next.

Building lifespan can be highly variable and difficult to predict. Most LC studies have considered new buildings and a 50-year use phase (e.g., Adalberth 1997b; Adalberth et al. 2001); however, lifespans from 40 (Blengini 2009) to 100 years (Fay et al. 2000; Fuller & Crawford 2011) have been considered. Several authors have considered existing buildings (and end-of-life), e.g., evaluating the actual lifespan of their case studies (Fay et al. 2000; Blengini 2009), or considering different lifespan estimates, for existing and new buildings (Nemry et al. 2010).

Table 2.1 Life-cycle (LC) studies of residential buildings

Reference	Indicators	Case-study	Dwelling area (m^2)	Occupancy (persons)	Location	LC phases	Lifespan (years)	Functional units	Selected results[b] (total and share of LC phases)
Adalberth 1997a, 1997b	Energy	3 single-unit dwellings	129 - 138	5	Sweden	1) construction 2) use 3) end-of-life	50	1 m^2 x y 1m^2 x 50 y	Total: 152 to 176 kWh m^2/y (1) 11-12% (2) 88-89% (3) <1%
Fay et al. 2000	Energy	Detached house (1 variant)	128	n/a	Melbourne, Australia	1) construction[c] 2) use	100	1 m^2 x 100 y	Total: 1.3 to 1.4 GJ m^2/y (1) 25–27% (2) 72-75%
Adalberth 2001	Energy, GHG and others	4 apartment buildings	74 - 145	n/a	Sweden	1) construction 2) use 3) end-of-life	50	1 m^2 x 50 y	Total: 122 to 184 kWh m^2/y; 26 to 32 kg CO_2 eq m^2/y (2) 70 – 90%
Keoleian et al. 2000	Energy, GHG and costs	Standard house and efficient house	228	n/a	Michigan, USA	1) construction 2) use 3) end-of-life	50	1 house 1 m^2 x y	Total: 0.6 to 1.4 GJ m^2/y; 33 to 89 kg CO_2e q m^2/y (1) 9-26% (2) 74-91% (3) <1%
Norman et al. 2006	Energy and GHG	Apartment building and detached dwellings	Apart.: 77 House: 242	Apart.: 1.8 House: 3	Toronto, Canada	1) construction 2) use 3) user transport	50	1 m^2 x year 1 person x y	Total: 0.9 to 2.2 GJ m^2/y; 78 to 107 kg CO_2 eq m^2/y (1) 7-12% (2) 32-70% (3) 18-61%
Citherlet & Defaux 2007	Energy and GHG	Detached house (3 variants)	266	2	Lausanne, Switzerland	1) construction 2) use 3) end-of-life	n/a	1 m^2 x y	Total: 200 to 580 MJ m^2/y; 10 to 28 kg CO_2 eq m^2/y (1) 10-45% (2) 45-85% (3) <10%
Blengini 2009	Energy, GHG and others	Apartment building	76	n/a	Turin, Italy	1) construction 2) use 3) end-of-life	40	1 m^2 x y	Total: 1.0 GJ m^2/y; 67 kg CO_2 eq m^2/y (1) 9-12% (2) 90-93% (3) -2%
Ortiz-Rodríguez et al. 2010	Energy, GHG and others	2 houses	Colombia: 125 Spain: 145	4	Spain and Colombia	1) construction 2) use 3) end-of-life	50	1 m^2 x 50 y	Total: 17 to 49 kg CO_2 eq m^2/y (1) 8-28% (2) 69-91% (3) 1-3%
Nemry et al. 2010	Energy, GHG and others	72 building types based on EU-25	n/a	n/a	EU-25	1) construction 2) use 3) end-of-life	20 - 40	1m^2 x y	Total: 0.2 to 2.5 GJ m^2/y (1) 12-20% (2) 80-89% (3) -3-5%
Monteiro & Freire 2012	Energy, GHG and others	Detached house (with variants)	132	4	Coimbra, Portugal	1) construction 2) use	50	1 house x 50 y	Total: 136 to 265 MJ m^2/y (1) 25-57% (2) 43-75%

[a] Livable area per dwelling; in cases, average livable areas were calculated with total living area and number of dwellings

[b] results focused on energy and GHG; where needed, results were adjusted to 1 m^2/year to ease comparison and approximate values were based on graphic figures

[c] construction and renovation materials included

Different energy metrics have been used in LC studies of buildings. Although most studies have used primary energy (e.g., Blanchard & Reppe 1998), some present results in final energy (e.g., Adalberth 1997b), and others do not specify whether the analysis used final or primary energy (Sartori & Hestnes 2007). In a study for a detached house in Switzerland (Citherlet & Defaux 2007), primary non-renewable energy requirements were calculated considering a set of alternative design and equipment choices (three variants in total), both for the Swiss and for the Union for Co-ordination of Transmission of Electricity (UCTE) electricity production mix. Non-renewable primary energy varied by 20 to 37% between the two electricity scenarios.

Building LC studies have mostly used three functional units. Some have provided the total energy demand over the whole building lifespan (e.g., Keoleian et al. 2000). Others have provided results with reference to the living area, on an annual basis (per m^2 per year, e.g., Adalberth et al. 2001; Blengini 2009; Citherlet & Defaux 2007), or for the whole lifespan (e.g., per m^2 for 50 years, in Adalberth 1997a), and Norman et al*.* (2006) used both area and number of occupants to compare a high density residential development near the Toronto city core and a low density development in the periphery. The analysis considered the urban context, including building materials, surrounding infrastructure, operational building requirements and transportation of users. Overall, the low density development had higher energy requirements and GHG emissions both per inhabitant and per square meter, but the two functional units showed significant differences, especially for building construction and use phases. For example, the low density development had 1.5 times higher building construction impacts per inhabitant; however, the high-density settlement was 1.25 more intensive in terms of energy and GHG emissions per square meter. Transportation was the only LC component that was higher in the low-density development both on the basis of inhabitants (3.7 times higher) and living area (2 times higher).

2.1.2. Key issues in building life-cycle studies

Differences in methodology, climate, building type, behavior and functional unit, as well as uncertainty and variability, can lead to a large range of LC results and impede comparisons between studies. In their review, Ramesh et al. (2010) found that LC energy demand in conventional residential buildings ranged from 150 to 400 kWh m^{-2} $year^{-1}$; Sartori & Hestnes (2007) estimated a range from 290 to 1180 kWh m^{-2} $year^{-1}$.

Most published LC studies of buildings have been completed in developed countries and in cold regions, such as Norway and Sweden (Sartori & Hestnes 2007). Citherlet & Defaux (2007) examined a family house in Switzerland, comparing three alternatives with different insulation, energy production systems and use of renewable energies. For the standard house, and considering the Swiss electricity production mix, the LC energy was 580 MJ m^{-2} $year^{-1}$ and the GHG intensity was 27 kg CO_2 eq m^{-2} $year^{-1}$.

Only a few LC studies of buildings have been completed in Southern Europe, none was found comparing existing buildings. Blengini (2009) performed a detailed LCA of a residential building in Italy, focusing on end-of-life (demolition and recycling potential) and alternative waste disposal scenarios, and estimated a recycling potential of 29% and 18% in energy and GHG emissions, respectively. Ortiz-Rodríguez et al. (2010) compared the LC energy and environmental impacts of dwellings in Spain and Colombia. The Spanish house emitted approximately 2470 kg CO_2 eq/m^2 during the 50-year lifespan (2248 in the use phase, including maintenance, 198 in the construction and 25 kg CO_2 eq/m^2 in the end-of-life), while the Colombian dwelling emitted 862 kg CO_2 eq/m^2 (595, 241 and 26 in the same three phases). Different results in the use phase were attributed to differences in climate and consumption behavior in the two countries. Few LC studies of buildings have been developed for Portugal (Pinto 2008; Monteiro & Freire 2012). Monteiro & Freire (2012) considered a single-family house in Portugal with seven alternative exterior wall types and two operational patterns, and differing in occupancy and comfort levels. LC primary energy ranged from 800 to 1600 GJ eq and GHG emissions from 58 and 115 ton CO_2 eq. Assuming the average operational pattern, a 50-year lifespan, and a living area of 132 m^2, the primary energy requirement was 182 MJ m^{-2} year^{-1} and the GHG emissions were 13 kg CO_2 eq m^{-2} year^{-1}.

The linkage between building design, energy use and GHG emissions is dependent on and sensitive to climate and socio-demographic characteristics that are geographically and culturally variable. Thus, it is highly relevant to provide comparative studies of existing buildings in different regions. The case study we describe in this chapter compares three long-lived apartment buildings with the same construction principles, materials and location, which allows an analysis of the effect of building design, a topic that has received little attention in the literature. In addition, a comprehensive econometric model recently developed for Portugal is applied, which considers building design and socio-demographic characteristics. This model estimates household energy consumption based on the number of occupants, building age, dwelling area, dwelling type, urbanization level and region using statistical data. The approach is efficient and broadly applicable to circumstances when historical and representative energy data is not available, and it circumvents the need for many assumptions and parameters used in engineering or demand-type models of household energy consumption. Lastly, the analysis explores and discusses the influence of key methodological choices on the results, including the final-to-primary energy conversion factor, building lifespan and functional unit selected.

2.2. Materials and methods

2.2.1. Residential case study

The building designs considered are in *Bairro de Alvalade,* Lisbon. The master plan for *Bairro de Alvalade* was the most significant public development for the expansion of Lisbon in the 1940s, and was planned by the architect João Faria da Costa (Alegre & Heitor 2003). The development consists of a low rent housing area, presented in Figure 2.1, designed by Jacobetty Rosa. The area is characterized by a regular urban morphology with *standardized* elements: dwellings, buildings and techniques were repeatedly used. The analysis compares three building types (of the nine existing in the area), described next.

Figure 2.2 presents schematic drawings of the three selected building types. The buildings have three or four stories, two dwellings per story, and a common staircase. Type 2 is the smallest: it has a gross area of 122 m^2 per story, three stories (total gross area of 367 m^2), and each dwelling unit has two bedrooms. Type 3 has a gross area of 157 m^2 per story, three stories (total gross area of 472 m^2), and three bedrooms per dwelling unit. Type 8, the largest, has a gross area of 260 m^2 per story, four stories (total gross area of 1041 m^2), and the dwelling units have five bedrooms.

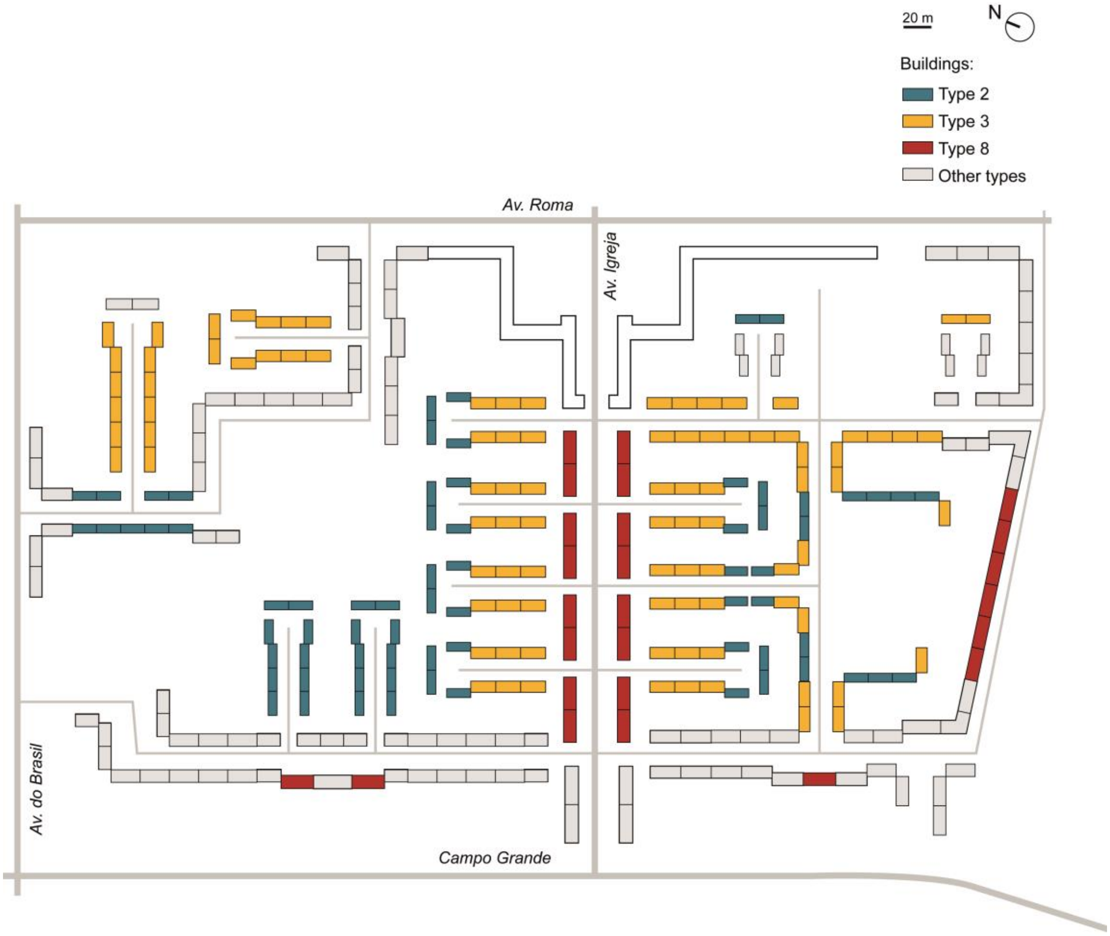

Figure 2.1 Schematic plan of the urban area including the residential buildings considered (colored)

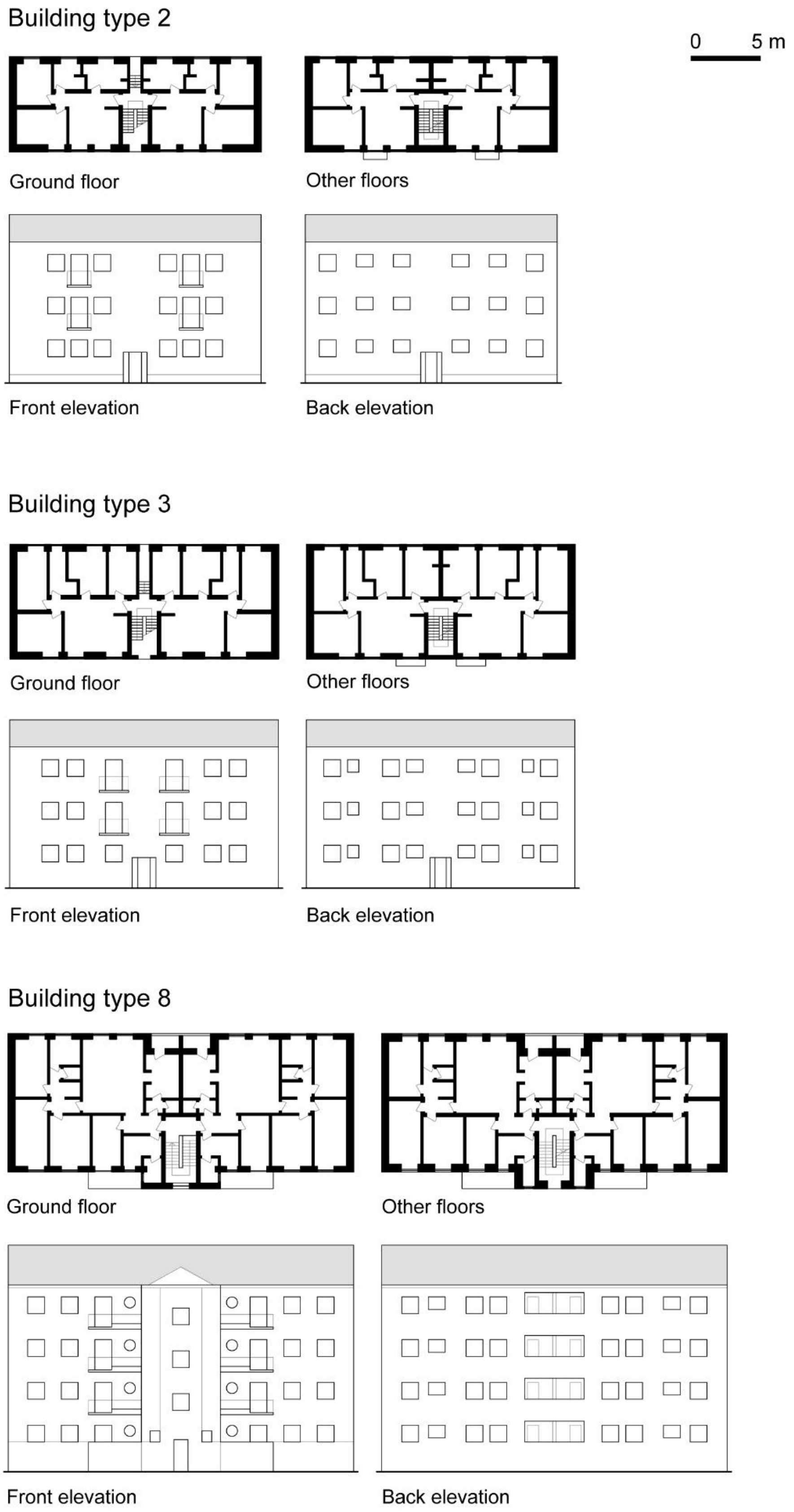

Figure 2.2 Floor plans and elevations for the three building types

2.2.2. Functional units and building lifetime

The building life was assumed to be 75 years, since the buildings date from the 1940s. Two functional units were selected: per floor area over a year (m^2-year) and per inhabitant over a year (person-year). The model assumes an average occupancy of 1.5 persons per dwelling unit, based on statistical urban area data from 2011 (INE 2012). The average occupancy was calculated from block-scale statistical units in the case-study area, which contained 88 to 276 people and 10 to 31 buildings each.

2.2.3. Construction phase

For the construction phase, primary energy requirements and GHG emissions were calculated using the Inventory of Carbon and Energy (ICE) Version 2.0 (Hammond & Jones 2011). The ICE lists the embodied energy, carbon and GHG (measured in grams of CO_2 equivalent, g CO_2 eq) for a large number of building materials. The "embodied energy", defined as the total primary energy (MJ_{prim}) required by the building materials, is the energy consumed in the extraction of raw materials, production of building materials and transportation to the building site ("cradle-to-site") (Hammond & Jones 2008). Similarly, the "embodied GHG" emissions comprise the GHG emissions from the extraction of raw materials to the building site. In the ICE, the term "embodied carbon" is used for both carbon and GHG emissions. The present analysis addresses GHG intensity on a 100-year time horizon.

Seven building elements were considered:

- external walls using hydraulic stone masonry and hollow brick masonry,
- interior walls using solid and hollow brick masonry,
- floors, both wooden beams/planks and reinforced concrete slabs,
- staircases in concrete with reinforced concrete landings,
- roofs, with wood structure and roof tiles,
- fenestrations in glass, and
- interior doors in wood.

For external walls, quantity was provided in volume (m^3) because thicknesses vary. In building types 2 and 3, external walls are 0.50 m thick in the ground floor and 5 cm less every upper floor. In building type 8, the external ground floor walls are 0.55 m in thickness, and 5 cm less on upper floors. Interior walls vary depending on structural and functional characteristics (0.15 to 0.25 m).

Details regarding building materials were obtained from the original drawings and other project documents maintained at the Municipal Archive of Lisbon (also in Alegre 1999). The type of stone used in exterior wall masonry was assumed to be limestone, based on contemporary construction materials (Aguiar et al. 2002; Appleton 2005). For each building element or material, volume was based on project documents, and density was on construction material providers and a technical reference (Messier 1974).

2.2.4. Retrofit phase

Energy requirements for the building retrofit phase used an intervention scenario with the measures considered listed in Table 2.2. Based on the survey by Alegre (1999), roughly half of the buildings in the case-study area have replaced the wooden floors and windows. Energy conservation measures considered included the addition of insulation in external walls and roof, replacement of the roof tiles, and a partial replacement of wall masonry. The embodied energy and GHG emissions associated with these retrofit measures were based on the ICE (Hammond & Jones 2011) (see construction phase).

Table 2.2 Retrofit phase: intervention measures

Exterior walls	Replacement of 20% of stone and brick masonry
	Additional 40 mm mineral wool insulation
Floors	Replacement of all wood floors with reinforced concrete slabs and terrazzo tiles
Roof	Replacement of 100% of clay roofing tiles
	Additional 40 mm mineral wool insulation
Fenestration	Replacement of 100% fenestrations with double glass

2.2.5. Use phase

The use phase represents household energy demand. Buildings use electricity and natural gas or liquefied petroleum gas (LPG). The total energy use per year was calculated based on the ratio between residential electricity use and natural gas or LPG from the *Lisbon Energy Matrix* (Lisboa E-Nova 2005), which provides estimates of energy use in Lisbon building stock using 2002 data. Electricity accounts for 60% of the final energy consumption in residential buildings, while natural gas or LPG account for 40%.

Annual electricity consumption was calculated using regression model 2a proposed by Wiesmann et al. (2011). These authors developed an econometric analysis of residential electricity expenditures in mainland Portugal using 2005 data. The total electricity consumption was based on a price of 0.141€/kWh. At the household level, electricity consumption in model 2a depends on ten variables: (1) persons per household (1.5); (2) building age (65 years, based on the 2005 reference year); (3) dwelling area (46, 62 and 100 m^2 for buildings type 2, 3 and 8, respectively); (4) dwelling type (apartment in building with less than 10 apartments); (5) urbanization level (mainly urban); and (6) region (Lisbon). Regarding (7) income and (8) number of appliances, the average for mainland Portugal was considered (Wiesmann et al. 2011). Finally, (9) children were considered to be present in half of the dwellings for each building type (mainland average was 58%), and (10) all dwellings were considered to be owned by the household.

The primary conversion factor for electricity, used to calculate the primary energy requirement, depends largely on the mix of generation technologies. Two conversion factors were considered: 2.5 MJ_{prim}/MJ_{fin} (suggested in the European Directive 2006/32/EC (European Commission 2006), which allows comparisons with other studies), and 2.0 MJ_{prim}/MJ_{fin} (based on the average of the Portuguese electricity system between 2003 and 2012; Garcia et al. 2014). The GHG intensity for electricity generation was 450 g CO_2 eq/kWh, based on the average for Portuguese generation between 2003 and 2012 (Garcia et al. 2014). For natural gas, the primary energy conversion factor was 1.13 MJ_{prim}/MJ_{fin} (Faist-Emmenegger et al. 2007), and the GHG emission factor was 72 g CO_2 eq/MJ (Faist-Emmenegger et al. 2007).

2.3. Results and discussion

2.3.1. Construction phase

Table 2.3 presents the LC inventories for the three building types, including the quantity of each construction element and the ratio between the quantity and the building's gross built area. Table 2.4 characterizes the main construction elements in terms of volume, mass, density, embodied energy and GHG.

Table 2.5 presents energy and GHG emissions associated with the construction phase, by building type. On a square meter basis, larger buildings have lower embodied energy and GHG, primarily due to the smaller contributions of walls. Building type 2 has the highest embodied energy (3433 MJ/m^2) and GHG (212 kg CO_2 eq/m^2). In comparison, the type 3 building attains a reduction of 2% in embodied energy and 5% in GHG, and the type 8 has a 10% decrease in both energy and GHG emissions.

Table 2.3 Life-cycle inventory: construction elements by building type

Element	Description	Building type 2		Building type 3		Building type 8	
		total	per m^2	total	per m^2	total	per m^2
Walls							
Exterior	Hydraulic stone masonry (m^3)	116	0.32	139	0.29	268	0.26
	Hollow brick masonry (m^3)	6	0.02	7	0.01	14	0.01
Interior	Solid brick masonry (m^3)	23	0.06	23	0.05	33	0.03
	Hollow brick masonry (m^3)	54	0.15	72	0.15	166	0.16
Floors							
Wood	Wooden beams and planks (m^2)	208	0.57	289	0.61	653	0.63
Concrete	Reinforced concrete slabs (m^2)	90	0.25	104	0.22	265	0.25
Staircases							
Landings	Reinforced concrete landings (m^2)	16	0.04	18	0.04	27	0.03
Stairs	Concrete stairs (m^3)	1	0.004	1	0.004	2	0.002
Roofs	Wood structure and roof tiles (m^2)	141	0.38	174	0.37	282	0.27
Fenestration	Glass doors and windows (m^2)	59	0.16	66	0.14	115	0.11
Interior doors	Wooden doors (m^2)	48	0.13	57	0.12	158	0.15

Table 2.4 Main characteristics of the construction elements, energy and greenhouse gas (GHG) emissions

Elements	Description	Material	Volume (m^3)	Density (kg/m^3)	Mass (kg)	Energy (MJ)	GHG (kg CO_2 eq)
Exterior walls	Hydraulic stone masonry (m^3)	Limestone	0.860	2400	2064	3096	186
		Mortar	0.140	2100	294	285	46
	Hollow brick masonry (m^3)	Brick	0.840	1300	1092	3276	262
		Mortar	0.160	2100	336	326	52
Interior walls	Solid brick masonry (m^3)	Brick	0.840	1800	1512	4536	363
		Mortar	0.160	2100	336	326	52
	Hollow brick masonry (m^3)	Brick	0.840	1300	1092	3276	262
		Mortar	0.160	2100	336	326	52
Floors	Wooden beams and planks (m^2)	Wood	0.287	600	172	1722	53
	Concrete slabs (m^2)	Reinforced concrete	0.100	2500	250	305	35
Staircases	Concrete slabs (m^2)	Reinforced concrete	0.100	2500	250	305	35
	Concrete stairs (m^3)	Concrete	1.000	2400	2400	1680	240
Roofs	Wooden structure (m^3)	Wood	0.064	600	38	384	12
	Roof tiles (m^2)	Ceramic tiles	-	-	44	528	34
Fenestration	Doors and windows (m^2)	Glass	0.004	2500	10	150	9
Interior doors	Wooden doors (m^2)	Wood	0.030	600	18	180	6

Table 2.5 Construction phase: primary energy requirement and greenhouse gas (GHG) emissions per building type

Elements	Building type 2				Building type 3				Building type 8			
	Energy (MJ)		GHG (kg CO_2 eq)		Energy (MJ)		GHG (kg CO_2 eq)		Energy (MJ)		GHG (kg CO_2 eq)	
	total (10^3)	per m^2	total	per m^2	total (10^3)	per m^2	total	per m^2	total (10^3)	per m^2	total	per m^2
Walls												
Exterior	414.1	1128	2876	78	494.8	1048	34371	73	955.0	917	66349	64
Interior	307.6	838	26645	73	371.4	787	32211	68	756.4	726	65740	63
Floors												
Wood	358.1	975	11102	30	498.1	1055	15440	33	1123.9	1079	34841	34
Concrete	27.5	75	3124	9	31.6	67	3584	8	80.8	78	9174	9
Staircase												
Landings	4.9	13	554	2	5.4	12	15440	1	8.3	8	945	1
Stairs	2.2	6	317	1	2.2	5	3584	1	3.5	3	506	1
Roof	128.7	351	351	18	159.0	337	8059	17	257.2	247	13038	13
Fenestration	8.9	24	24	2	9.9	21	599	1	17.2	16	1045	1
Interior doors	8.6	23	23	1	10.2	22	317	1	28.5	27	884	1
Total	1260.6	3433	77847	212	1582.5	3351	95515	202	3231.0	3102	195523	185

2.3.2. Retrofit phase

The retrofit energy requirement for the 75-year period is presented in Table 2.6. The total energy and GHG emissions are higher in larger buildings. However, on a per square meter basis, energy requirement and GHG emissions are slightly lower in larger buildings. This is probably due to the higher ratio of building envelope/floor area in smaller buildings. The only retrofit measure that has higher impacts per square meter in building type 8 is the replacement of floors, which is the only measure that does not affect the building envelope.

Table 2.6 Retrofit phase: primary energy requirement and greenhouse gas (GHG) emissions

Elements	Building type 2				Building type 3				Building type 8			
	Energy (MJ)		GHG (kg CO_2 eq)		Energy (MJ)		GHG (kg CO_2 eq)		Energy (MJ)		GHG (kg CO_2 eq)	
	total (10^3)	per m^2	total	per m^2	total (10^3)	per m^2	total	per m^2	total (10^3)	per m^2	total	per m^2
Exterior walls	99.3	271	7028	19	118.7	252	8399	20	229.1	220	16210	16
Floors	83.1	226	8886	24	115.6	245	12358	26	260.7	250	27886	27
Roof	81.5	222	5385	15	100.7	213	6653	14	163.0	157	10764	10
Fenestration	8.9	24	540	2	9.9	21	599	1	17.2	17	1045	1
Total	272.9	743	21839	60	344.9	731	28009	59	670.1	644	55904	55

2.3.3. Use phase

Table 2.7 presents the annual primary energy requirement and GHG emissions of the use phase for the different building types. In absolute terms, the smallest building (type 2) is associated with the lowest energy demand and GHG emissions; the largest building (type 8) has 44% higher energy requirements and emissions. However, the trend is reversed on a per square meter basis for building types 3 and 8 where energy and GHG emissions are 20 and 49% lower than building type 2, respectively. The lower energy requirement per square meter in larger buildings is due to the area/volume and area/occupancy ratios. The area/volume ratio is generally higher in larger buildings (i.e. the same living space requires less building envelope surface), which can result in lower energy consumption. The area per person is the highest in building type 8 (87 m^2/person), followed by type 3 (52 m^2/person) and by type 2 (41 m^2/person). A larger area per inhabitant can contribute to lower energy requirement per square meter.

Table 2.7 Use phase: household primary energy requirement and greenhouse gas (GHG) emissions (per year)

	Energy (MJ)				GHG emissions (kg CO_2 eq)	
	Factor 2.0		Factor 2.5			
	total	per m^2	total	per m^2	total	per m^2
Building type 2	125 308	341	148 027	403	7 872	21
Building type 3	128 083	271	151 305	320	8 046	17
Building type 8	179 897	173	212 513	204	11 301	11

2.3.4. Life-cycle analysis

Figures 2.3 and 2.4 present the LC primary energy requirements and GHG emissions per building type, respectively. The error bars result from the two primary energy conversion factors used for household electricity consumption. The use phase has the greatest primary energy demand and GHG emissions for the three building types, representing 78-88% of both. The construction phase accounts for 10-19% of both energy and GHG emissions, while the retrofit phase accounts for less than 4% in all cases.

Figure 2.5 shows the LC primary energy requirements using the two functional units. On a per square meter basis, building type 2 has the highest annual requirement (397 to 468 MJ m^{-2} $year^{-1}$), followed by type 3 and type 8, which are approximately 19% and 45% lower, respectively. This pattern is reversed when energy requirements are expressed on a per person basis: building type 2 has the lowest annual requirements (16 195 to 19 083 MJ $person^{-1}$ $year^{-1}$), while building types 3 and 8 are approximately 4% and 18% higher, respectively.

Figure 2.6 shows GHG emissions for the two functional units. Building type 2 has the highest GHG emissions per square meter (25 kg CO_2 eq m^{-2} $year^{-1}$), followed by types 3 and 8 (lower by 18% and 44%, respectively). On a per person basis, type 2 has the lowest emissions (1022 kg CO_2 eq $person^{-1}$ $year^{-1}$), while types 3 and 8 are 5% and 19% higher, respectively.

The estimated LC energy use for the three building types is comparable to that in recent literature. The single-family house in Barcelona, Spain (Ortiz-Rodríguez et al. 2010), had LC GHG emissions of 49 kg CO_2 eq/m^2, higher than found here (15 to 27 kg CO_2 eq/m^2 if a 50-year lifespan is considered). The difference is partially associated with differences in the building typologies. A single-family house is generally associated with higher use phase energy consumption on a square meter basis due to its relatively larger envelope area (Nemry et al. 2008; Takano et al. 2015). The Portuguese single-family house (Monteiro & Freire 2012) had a primary energy requirement of about 136 MJ m^{-2} $year^{-1}$ in a base case scenario with moderate occupancy. The study considered use phase energy consumption for heating, cooling and maintenance, but excluded other uses, such as lighting, water heating, cooking and washing appliances. In our study, primary energy requirement ranged from 247 to 496 MJ m^{-2} $year^{-1}$, for a 50-year lifespan. The present analysis uses an econometric model recently developed for Portugal that accounts for all household energy use, and thus represents a LC estimate that is improved over earlier studies.

Comparison with other LC studies, as noted in the chapter introduction, is affected by methodological choices in the LC analysis methods, climate, the uniqueness of each building, consumption habits of occupants, and other factors. For example, Sartori & Hestnes (2007) found somewhat higher use in their review of 60 case studies, 1040 to 4250 MJ m^{-2} $year^{-1}$. The studies of residential buildings (33), considered only six countries, mostly in cooler climates: Sweden (14), Australia (3), Germany (6), USA (2), New Zealand (3) and Norway (5); only two studies considered multi-family buildings, and all studies were completed between 1978 and 2004. Ramesh et al. (2010) estimated 540 to 1440 MJ m^{-2} $year^{-1}$, which is higher than our results. Our analysis calculated energy consumption in the use phase based on case-specific characteristics. This is likely to be lower than average comfort standards and other studies, because this is currently a residential area with low occupancy. Despite the inherent variability and uncertainty associated with LC analyses of buildings, the estimated LC energy and GHG emissions are comparable to the range of results provided by the studies in south European context.

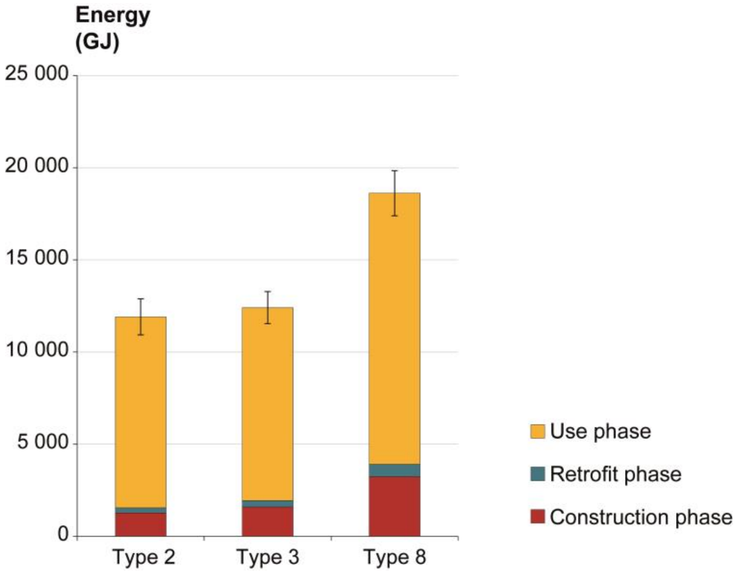

Figure 2.3 Life-cycle primary energy requirements by building type
(error bars present the use phase primary energy calculated with 2.0 and 2.5 factors)

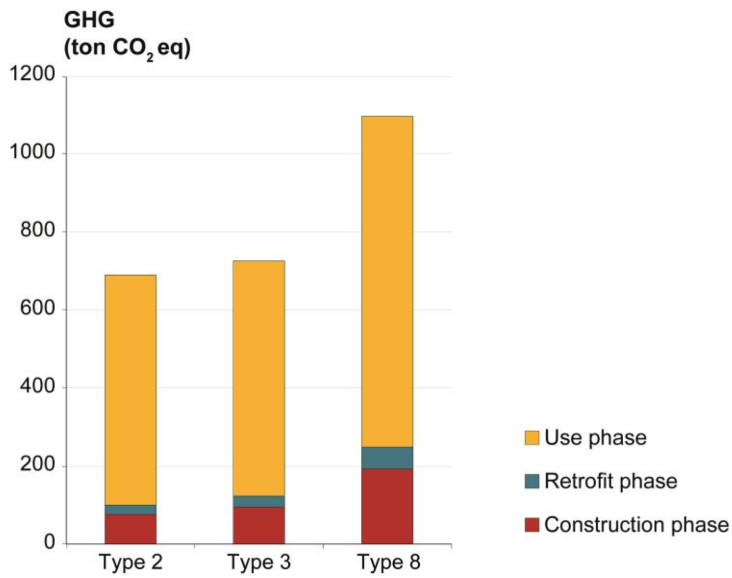

Figure 2.4 Life-cycle greenhouse gas (GHG) emissions by building type

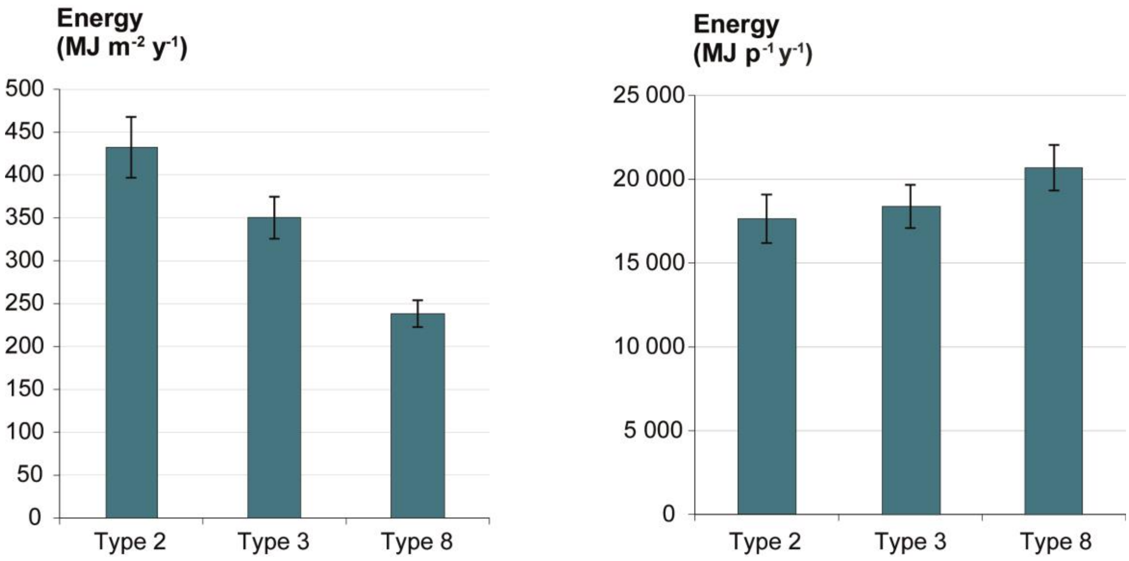

Figure 2.5 Annual energy requirement per square meter and per person
(error bars present the use phase primary energy calculated with 2.0 and 2.5 factors)

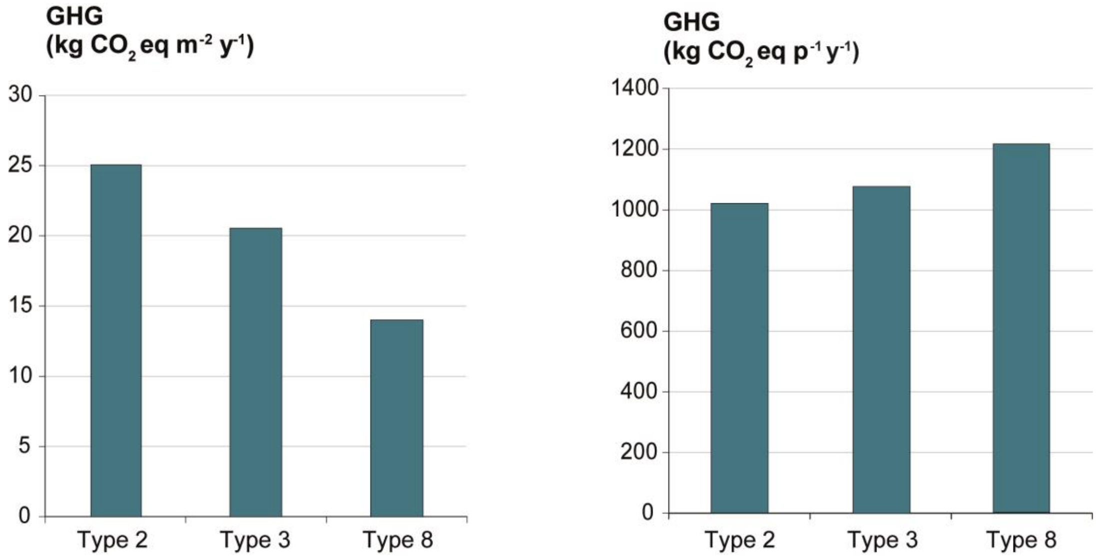

Figure 2.6 Annual greenhouse gas emissions (GHG) per square meter and per person

2.3.5. Model assumptions and uncertainties

LC analyses of buildings involve many assumptions and simplifications associated with the energy production mix, building use phase energy requirements, building lifespan, LC phases considered, functional units and building data (Anand & Amor 2017).

Primary energy incorporates not only final energy consumption, but also the (upstream) energy used to produce and deliver it. Energy use should be quantified in terms of primary energy since this incorporates the life-cycle efficiency of the different energy types and electricity generation mix and reflects the true environmental implications of energy use (Malça & Freire 2006; Sartori & Hestnes 2007; Dixit 2017). However, the technology and generation mix can evolve and change significantly during a building's long lifespan. In this analysis, two primary energy conversion factors for electricity were evaluated (2.0 and 2.5 MJ_{prim}/MJ_{fin}) which changed the building's total LC energy use by 11-13%. The impact of electricity production mix is important for identifying potential improvements that can reduce energy requirements and associated environmental impacts (Adalberth et al. 2001; Citherlet & Defaux 2007); however, it can make comparisons between LC studies of buildings more difficult, especially when methodological choices and disaggregated results are unclear.

Energy consumption during the use phase also changes, and predictions over the building LC (e.g., 75 years) are highly uncertain. We assumed a constant consumption rate based on data from 2002 and 2005. Energy consumption depends mainly on the energy use per capita, number of persons per household, and floor area per capita. Historically, energy use per capita in Portuguese households increased from 0.23 in 1989 to 0.30 toe/capita in 2009 (INE & DGEG 2011) (1 tonne of oil equivalent (toe) corresponds to approximately 42 GJ). While the dwellings in the study area were designed for an average occupancy of 4 inhabitants in 1940, occupancy rates had decreased to 1.5 inhabitants per dwelling by 2011 (INE 2012), which greatly increased floor area per capita. These trends are similar to many urban areas in the EU-25: between 1990 and 2004, energy use per capita in residential buildings increased from around 25 to 28 GJ, persons per household decreased from 2.8 to 2.5, and floor area per capita increased from 30 to 35 m^2 (O'Broin 2007). Such trends, also difficult to anticipate, highlight the importance of considering functional units other than building area, such as occupancy.

Building lifespan is also variable and difficult to predict (Nemry et al. 2010). While many buildings in Europe were built in the last few decades, over 40% of residential buildings were built before the 1960s and some are hundreds of years old (BPIE 2011). We considered a 75-year lifespan (buildings were constructed in the 1940s), which has the effect of *lowering annual energy and environmental burdens* compared to the 50-year lifespan used in most previous studies (while increasing the overall LC impacts). For the three building types considered, a 50-year life would give primary energy requirements from 247 to 496 MJ m^{-2} $year^{-1}$ and GHG emissions between 15.6 and 26.9 kg CO_2 eq m^{-2} $year^{-1}$. The construction, use and retrofit phases would account for 14-24%, 70-83% and 3-7% of the overall results, respectively. Considering a 50-year lifespan would reduce the overall LC energy and GHG emissions by 26 to 29%.

Building end-of-life phase is considered negligible in the overall energy requirement and GHG emissions (Sartori & Hestnes 2007; Ramesh et al. 2010; Nemry et al. 2010 ; Chau et al. 2015), and thus was not considered in the present analysis. In addition, dismantlement and waste treatment scenarios can be difficult to foresee. The exclusion of this phase is not expected to substantially alter results.

The selection of functional units depends on the goal and scope of the LC study. Most LC studies of buildings have adopted area-based functional units, which allow the comparison of design alternatives for a house, for example. Using an area-based functional unit, larger dwellings have lower energy requirements and lower GHG emissions for the same occupancy, but these indicators do not necessarily translate to better environmental performance. In contrast, the use of an occupancy-based functional unit (often used in studies at the urban scale) can overlook the building's performance, e.g., high occupancy could compensate for poor environmental performance. Thus, to provide comprehensive and useful insight on the environmental impacts associated with buildings, we recommend the use of both functional units.

The building design and materials were obtained mainly from original project documents. Few project data were unavailable, i.e., the type of stone in exterior walls masonry and material densities. Embodied energy and GHG emissions of building materials were based on data provided by the ICE (Hammond & Jones 2011), which is derived from UK production processes. Although these uncertainties are not expected to significantly change results, more appropriate and site-specific data would improve the accuracy of the analysis .

2.4. Conclusions

This chapter presented a comparative LC analysis of energy and GHG emissions for three residential building types, accounting for building construction, retrofit and use phases, across a 75-year lifespan. The use phase was dominant, accounting for 76-88% of the total energy requirement and GHG emissions. In the construction phase, walls represented the largest embedded energy requirement and GHG emissions, e.g., across the three building types, exterior walls represented 30-33% and 34-37% of energy and GHG burdens, respectively; interior walls accounted for 23-24% and 34%, and floors contributed 30-37% and 18-23%. In the largest building, these burdens were lower by 9-11% for energy and GHG emissions expressed on a per square meter basis. However, these differences are relatively small since the construction phase accounted for less than 20% of the overall life-cycle burden.

LC studies of buildings are associated with considerable variability and uncertainty, and present many methodological differences that impede comparisons. The results highlight the importance of functional units when comparing among different building types. Results expressed on the basis of built area or occupancy showed opposite trends, e.g., larger buildings had higher energy and GHG emissions per person, but lower energy and GHG emissions per square meter. To provide LC analyses that are consistent and that account for site-specific differences, the use of both occupancy- and area-based functional units is recommended. To more accurately express the potential energy requirements of buildings, the use of primary energy is also recommended.

The research highlights the importance of addressing dwelling size and occupancy in urban development strategies. In-dwelling energy consumption strongly on the number of persons per household and floor area per capita. Urban planning and policies can control the growth of the building stock, in particular, focusing on the costruction and renovation of smaller building typologies and limiting construction of large houses.

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3 Integrating location and user commuting requirements in the LCA of residential buildings

While most LCAs of buildings have focused on construction and use phases, the location of a building can significantly affect the transportation demand of its inhabitants. This chapter presents a comparative life-cycle energy and greenhouse gas analysis of two representative buildings: an apartment building in the city center and a semidetached house in a suburban area. The life-cycle model includes building construction, building use and user transportation, and sensitivity analyses are used to evaluate impacts for multiple locations. [2]

Section 3.1 lays the background and motivation for this analysis and summarizes the previous LCA studies of buildings that addressed user transportation. Section 3.2 describes the model, materials and methods used. Section 3.4 provides the results, including two sensitive analyses, for a set of residence and work locations. Section 3.4 summarizes the key findings and wraps the conclusions up.

3.1. Introduction

While a large body of literature has addressed environmental impacts associated with buildings and transportation separately, few studies have considered these environmental impacts together (Rickwood et al. 2008; Anderson et al. 2015). Transportation needs and the associated environmental impacts, which in part are determined by a building's location, can represent a potentially large share of the life-cycle (LC) impacts. Both the design and the location of a building affect environmental performance, and focusing on one or the other can shift impacts and overlook improvement opportunities for more sustainable urban development (Rickwood et al. 2008; Stephan et al. 2011; Stephan et al. 2013b; Anderson et al. 2015; Huang et al. 2017; Soares et al. 2017).

This chapter explores linkages between urban planning and environmental impacts associated with residential buildings. It presents a comprehensive energy and greenhouse gas

[2] Significant portions of this chapter are from Bastos, J., Batterman, S. & Freire, F., 2015. Significance of mobility in the life-cycle assessment of buildings. *Building Research and Information* 44 (4): 376-393.

(GHG) analysis of residential buildings in Lisbon, Portugal, integrating user transportation to assess its contribution to the overall non-renewable primary energy (NRE) and GHG emissions. It first reviews the literature that jointly considers building and user transportation, and then discusses methodological issues. Next, a case study compares two typical residential typologies in Lisbon: an apartment building in a central area and a semidetached house in a suburban area. A novel and integrated bottom-up LC approach is used to quantify energy and GHG emissions associated with (1) building construction (including construction and retrofit materials), (2) building use and (3) user transportation (both public and private). Sensitivity analyses evaluate impacts for other residence and workplace locations. The chapter concludes with a discussion of opportunities to improve the energy and GHG performance of housing and transportation policies.

3.1.1. LCA of buildings integrating transportation

LC approaches have been increasingly used in environmental assessments of buildings (Sartori & Hestnes 2007; Ramesh et al. 2010; Cabeza et al. 2014; Rodrigues & Freire 2014; Sharma et al. 2011; Monteiro & Freire 2012; Chau et al. 2015; Anand & Amor 2017). As noted in chapter 1, most assessments have focused on and been typically limited to specific buildings, quantifying impacts associated with building construction, maintenance and use (Anderson et al. 2015). However, a building's location can significantly affect the transportation demand of its inhabitants (Norman et al. 2006; Codoban & Kennedy 2008; Rickwood et al. 2008; Lotteau et al. 2015), and the omission of transportation requirements can shift impacts to user transportation. As an example, an energy-efficient building in a suburban low-density development might have better environmental performance in terms of energy requirements per unit floor area or per capita than a conventional building in the city center, yet result in significantly higher overall impacts due to the additional transportation-related energy requirements and environmental impacts (Stephan et al. 2013b). Thus, a comprehensive LC analysis should center on users, not just buildings, and should consider energy and environmental impacts associated with the building, its location, and transportation and other requirements of the users that are affected by urban planning and building design. In Switzerland, the Society of Engineers and Architects has developed a process-analysis based methodology for calculating the energy consumption of buildings that includes induced mobility, which takes into account the building's location (SIA 2011). Applied in two studies that included residential, office and school buildings (over 30 case studies in Switzerland) (Perez & Rey 2013; Frischknecht et al. 2014; Wyss et al. 2014) and considering a 60-year period, mobility accounted for a significant share of the LC energy.

Nine LC studies were identified in the literature that integrated transportation and buildings. Table 3.1 summarizes these analyses and lists study locations, dwelling areas, occupancy rates, LC phases, functional units, the life-cycle inventory (LCI) approach for calculating embodied impacts, environmental indicators, and the main results expressed as per person-year to provide comparability. Treloar et al. (2000) argued that in the analysis of buildings, a diversity of activities undertaken by the occupants should be considered, and estimated the embodied and operational energy requirement for a semidetached house (including appliances), together with belongings, consumables, financial services, motor vehicles (excluding public transport) and vacations of the occupants. Subsequent studies have focused mainly on comparisons between city and suburban residential typologies. Norman et al. (2006) compared a central high-density and suburban low-density development in Toronto, considering construction materials for building, utility and road construction, building use, and user transportation. Results depended on the functional unit, e.g., energy and GHG emissions from material production were 1.5 times higher for the low-density settlement per inhabitant; however, per square meter, they were 1.25 times higher in the high-density settlement.

In Australia, Perkins et al. (2009) performed a comparative LC energy and GHG analysis of three residential areas mainly characterized by: apartment buildings in the city center, two-story attached houses in inner suburbs and one-story detached houses. They concluded that, on a per capita basis, apartment households had similar energy requirement to the alternatives, and higher GHG emissions, mainly due to lower occupancy rates and higher dwelling operational and embodied energy usage, despite the higher share of trips by walking. Fuller & Crawford (2011) compared different residential development patterns to explore the influence of housing size, style and location. Energy requirements per capita varied from 29 GJ person^{-1} year^{-1} for the inner suburban apartments, to 80 GJ person^{-1} year^{-1} in the energy-efficient (rated seven-star) outer suburban house. Location and house size were the dominant factors determining energy use and GHG emissions.

Stephan et al. (2011) provided the first European study to compare a city apartment and a suburban house in three scenarios with different energy efficiency levels in Brussels, Belgium. Only car travel was considered. In another study, Stephan et al. (2012) compared energy-efficient houses in Belgium (330 m^2 of gross floor area, four people) and Australia (297 m^2, five people), adding the energy requirement associated with the maintenance of infrastructure near the building. In a subsequent study, Stephan et al. (2013a) performed an energy and GHG analysis of a low-density neighborhood in Melbourne, Australia. Drawing on a base case study (214 houses, 749 people) and a set of scenarios in which house size, transport and housing technology varied, the study concluded that energy and GHG emissions associated with all LC phases could be reduced. Stephan et al. (2013b) performed an LC energy analysis of a suburban passive house (330 m^2, four people) in Brussels, for a 100-year lifespan, which was compared with a retrofitted apartment in the city (80 m^2, two people) for which a public transportation scenario was considered. They found that on a per capita basis the retrofitted apartment had lower energy requirements, partly due to the different floor area per person ratio, which was significantly lower in the city apartment. Stephan and Stephan (2014) provided a LC energy analysis of a residential building in Lebanon, showing that the relative contributions of embodied, operational and transportation energy differed significantly from studies in Belgium and Australia, probably due to the mild Mediterranean climate (associated with relatively low use phase energy requirements), energy supply and highly car-dependent transportation systems.

The nine studies vary significantly in terms of the methodologies and the cases analyzed. The Belgium, Australia and Canada studies have very different climates, and all studies have specific socio-economic contexts. For the construction phase, the studies used different LCI approaches, namely input–output (IO) analysis and hybrid analysis, which integrates process and IO data. IO data are extensive in some countries, such as Australia, but unreliable and highly aggregated in other countries (see next section). In the Belgium and Lebanon studies, the Australian IO-based hybrid database was used due to the unavailability of quality IO data. For the use phase, most studies relied on operation or simulation models based on comfort standards, which may not be representative of the actual energy demand in buildings. For the transportation phase, the studies focused on car use and excluded public transport, with the exception of Stephan et al. (2013a, 2013b). While several other studies have examined building and transportation factors at the city or metropolitan scale (e.g., Ramaswami et al. 2008; Heinonen et al. 2013), these studies included energy demands associated with food, clothes, vacations and other factors that are not expected to be directly linked with building or urban design. To date, the overall LC energy and GHG emissions of residential buildings in southern Europe have not been evaluated; however, the climatic, geographic, socio-economic and cultural contexts are expected to yield results that differ from the existing literature. Thus, such assessments are needed to identify robust strategies than can reduce the energy and GHG emissions of residential buildings.

Table 3.1 Life-cycle studies of buildings integrating user transportation *(continues on next page)*

Reference	Analysis	Location	Lifespan (years)	Dwelling area (m^2)	Occupancy (persons)	LC phases	Metrics	LCI[a]	Indicators	Results (per person-year)		
										Constr.	Operation	Transport
Treloar et al. 2000	2-bedroom house	Australia	30	123	2	1) constr. 2) retrofit 3) operation 4) transport[b]	1 house x 30 y 1 person x 1 y	Hybrid	Energy (GJ)	31.7	50.7	50.3
Norman et al. 2006	2 residential areas: HD) high density in city LD) low density in suburb	Canada	50	-	1.8 – 3.0	1) constr. 2) operation 3) transport	1m² x 1year 1 person x 1 y	EIO	Energy (GJ)	LD) 7.4 HD) 4.7	LD) 49.8 HD) 27.5	LD) 28.8 HD) 7.9
									GHG (t CO_2 eq)	LD) 0.6 HD)0.4	LD) 2.7 HD)1.5	LD) 5.3 HD) 1.4
Perkins et al. 2009	CA) city apartment IS) inner suburb house OS) outer suburb house	Australia	60	113 – 117	1.6 – 2.75	1) constr. 2) operation 3) transport[b] 4) transport embodied	1 house x 1 y 1 person x 1 y 1m² x 1 y	EIO	Energy (GJ)	CA) 16 IS) 8 OS) 6	CA) 20 IS) 14 OS) 13	CA) 19 IS) 31 OS) 56
									GHG (t CO_2 eq)	CA) 1.4 IS) 0.9 OS) 0.7	CA) 5.8 IS) 3.0 OS) 2.0	CA) 2.9 IS) 3.2 OS) 4.7
Fuller and Crawford 2011	CA) city apartment IS) inner suburb apartment OE) outer suburb efficient house OC) outer suburb conventional house	Australia	100	64 – 238	1.5 – 2.5	1) constr. 2) operation 3) work travel	1 person x 1 y	Hybrid	Energy (GJ)	CA) 15 IS) 12 OC) 18 OE) 20	CA) 15 IS) 12 OC) 18 OE) 20	CA) 2 IS) 6 OC) 47 OE) 40
									GHG (t CO_2 eq)	CA) 1.0 IS) 0.8 OC) 1.3 OE) 1.3	CA) 2.8 IS) 1.1 OC) 3.2 OE) 2.8	CA) 0.1 IS) 0.5 OC) 3.8 OE) 3.3

Table 3.1 *(continuation)* Life-cycle studies of buildings integrating user transportation

Reference	Analysis	Location	Lifespan	Dwelling area	Occupancy	LC phases	Metrics	LCI[a]	Indicators	Results (per person-year)		
			(years)	(m^2)	(people)					Constr.	Operation	Transport
Stephan et al. 2011	CA) city apartment SH) suburb house	Belgium	50	90 – 130	2	1) constr. 2) operation 3) transport[b]	1 house x 50 y	Hybrid	Energy (GJ)	CA) 12-15 SH) 17-22	CA) 21-29 SH) 28-39	CA) 20.7 SH) 51.2
Stephan et al. 2012	2 single-family energy efficient houses in: AU) Australia BE) Belgium	Australia and Belgium	5	297 – 300	4 – 5	1) constr. 2) operation 3) transport[b]	1 house x 50 y	Hybrid	Energy (GJ)	AU) 27.8 BE) 42.6	AU) 34.3 BE) 29.5	AU) 31.7 BE) 24.5
Stephan et al. 2013a	Low density neighborhood and scenarios[c]	Australia	50	180 – 230	3 – 4	1) constr. 2) operation 3) transport	1 neighborhood x 50 y 1 km^2 x 50 y 1 p x 50 y	Hybrid	Energy (GJ)	26.2	38.4	32.8
									GHG (t CO_2 eq)	1.6	3.1	2.6
Stephan et al. 2013b	PH) suburb passive house CA) city apartment	Belgium	100	90 – 297	2 – 4	1) constr. 2) operation 3) transport	1 house x 100 y 1 m^2 x 100 y 1 p x 100 y	Hybrid	Energy (GJ)	PH) 36.1 CA) 13.3	PH) 29.5 CA) 25.0	PH) 24.5 CA) 14.4
Stephan and Stephan 2014	Apartment building	Lebanon	50	113 – 154	4	1) constr. 2) operation 3) transport	1 building x 50 y	Hybrid	Energy (GJ)	15.4	27.3	41.3

[a] Life-cycle inventory (LCI) approach type applied to calculated embodied impacts: process-based, economic input-output (EIO) or hybrid
[b] only car transportation
[c] results for the base case study

3.1.2. Life-cycle inventory approaches

Several LCI approaches have been used to calculate embodied impacts of buildings (Table 3.1). *Process-based* approaches, which are the most common in LC studies, use a bottom-up approach to assess the main production processes in detail for a specific product or system (Suh et al. 2004). Process-based LCA may have limitations associated with the definition of a system boundary and the consequent truncation errors (Treloar 1997; Lenzen 2000; Bilec 2007; Chau et al. 2015; Dixit 2017). The accumulation of neglected processes in upstream stages can significantly underestimate production or manufacturing impacts, mainly due to the exclusion of capital goods and services (Fay et al. 2000; Crawford 2004; Suh et al. 2004; Frischknecht et al. 2007; Dixit 2017). Truncation errors have been quantified and they depend on the type of product and depth of study (Lave et al. 1995; Lenzen 2000, 2002; Crawford 2004; Nässén et al. 2007).

The second major approach uses *IO analysis* (IO-LCA), which bases the environmental impacts of a product or system on national statistical data describing economic flows between sectors and their linkage with environmental data (Fay et al. 2000; Lenzen 2000; Bilec 2007). Although systemically complete (considering the whole economy), this approach gives results for a national 'average product', and it is not as reliable for assessing individual specific products or for identifying potential process improvements (Treloar 1997; Fay et al. 2000; Crawford 2004; Suh et al. 2004; Bilec 2007; Chau et al. 2015). IO-LCA limitations mainly arise from the level of aggregation of data and assumptions regarding tariffs, homogeneity and proportionality, in economic flows and sectors. Another limitation is that IO-LCA is used mainly for the production phase, and is combined with process data for the use and end-of-life phases (Bilec 2007).

Hybrid approaches have been developed to combine the strengths of process- and IO-based analyses, namely reliable specific data and a complete framework (e.g., Treloar 1997; Shipworth 2002; Junnila & Horvath 2003; Bilec 2007; Dixit 2017). Although hybrid approaches should reduce truncation errors due to the inclusion of the whole economy, defining the location of the boundary between process and IO data remains subjective (Suh et al. 2004). While IO data are critical for buildings' LCA (Bilec 2007), further research and improvements are needed to integrate process and IO data (Wan Omar et al. 2014). In addition, the reliability of IO data in some countries might be questionable because the availability of quality economic and environmental data suitable and disaggregated for IO- and hybrid-LCA are very limited in countries other than the US or Australia (Crawford 2004; Bilec 2007). In an evaluation of the IO-based hybrid LCI method used in the Australian case studies noted previously (Table 3.1), Crawford (2008) concluded this was the preferred approach for Australian buildings due to its level of completeness.

Several studies have explored potential biases associated with the LCI approach on the embodied energy associated with buildings. Crawford (2011) highlighted that truncation errors (estimated to be 50–87%) depended on the product complexity and the contribution of the main processes. Nässén et al. (2007) compared an IO based analysis of the Swedish building sector with results from 18 studies with process-LCA in Scandinavia. Considering material production, transport and on-site construction, primary energy was 5.1 and 3.1 GJ/m^2 for IO- and process-based LCAs, respectively, representing a 64% difference. Recently, two studies (Crawford & Stephan 2013; Stephan & Stephan 2014) performed LC energy analyses of residential buildings in Belgium and Lebanon and explored different LCI approaches. Embodied impacts were about four times higher for the IO based hybrid analysis compared with the process-based approach. These analyses considered an IO-hybrid database developed for Australia which contributed to a high level of uncertainty (+40%), and the authors highlighted the need for a hybrid embodied energy-coefficient database for Europe (Crawford & Stephan 2013). As noted above, however, the incompleteness of process-LCA in comparative analyses is not expected to have a significant impact since a common boundary selection should similarly affect the different products or systems (Crawford 2004).

In summary, process-, IO- and hybrid-based LCAs have different advantages and disadvantages (Treloar 1997; Lenzen 2000; Suh et al. 2004; Nässén et al. 2007; Crawford 2011; Crawford & Stephan 2013; Wan Omar et al. 2014), and the selection of an approach should be based on the study goals and the availability of data (Bilec 2007). The LCI implemented in our analysis followed a process-based approach, as explained in the next section.

3.2. Materials and methods

A LC model for two residential typologies was developed addressing three phases: building construction, building use and user transportation. The LC model builds on the research presented in chapter 2 and the apartment building in this analysis is building type 8 in chapter 2. In building construction, the amount of materials was quantified based on project documents. Building end-of-life was not considered because it is not expected to be significant in the overall LC of conventional buildings (Nemry et al. 2010), and it is hardly predictable. Building lifespan was assumed to be 50 years, the most common timeframe used in the literature (Grant & Ries 2013). Three functional units were selected: (1) one dwelling × 50 years; (2) 1 m^2 of net usable floor area × 1 year; and (3) one person × 1 year. Multiple functional units were used due to the complexity and heterogeneity of the built environment and to provide comparability with other studies (chapter 2). The combination of different types of functional units (absolute, spatial and per capita) has been recommended in several recent studies (Stephan et al. 2013a; Bastos et al. 2014; Lotteau et al. 2015). Materials and transportation LCIs were based on the ecoinvent database v2.2. The impact assessment, performed using Simapro 7.3.3 software, focused on NRE, within Cumulative Energy Demand v1.08 (Frischknecht & Jungbluth 2007), and GHG intensity, based on the Intergovernmental Panel on Climate Change (IPCC) method v1.02, with a 100-year time horizon (IPCC 2007).

Two dwellings similar in terms of rooms and area were selected for comparison. Figure 3.1 presents schematic drawings of the two housing typologies: a city apartment (CA) building and a suburban semidetached house (SH). Table 3.2 summarizes their main characteristics. The CA building had four stories and two dwellings per story, and each of the eight apartments had 102 m^2 of net usable floor area (total gross area of 848 m^2). The SH had one dwelling, two stories and a net usable floor area of 104 m^2. Both typologies were based on buildings designed for the master plan of *Bairro de Alvalade* in Lisbon. The buildings were selected because they were representative of residential buildings in Portugal and because there were high-quality public data describing their design and construction. All design data, such as plans and construction materials, were based on project documents maintained in the Municipal Archive of Lisbon (details described in chapter 2). Both buildings were designed by the same architect, with similar construction solutions and materials. Inner-city and suburban locations in Lisbon were selected to calculate transportation requirements. The CA was located in *Campo Grande* (location 1) and the SH in *Santo Antão do Tojal* (location 5).

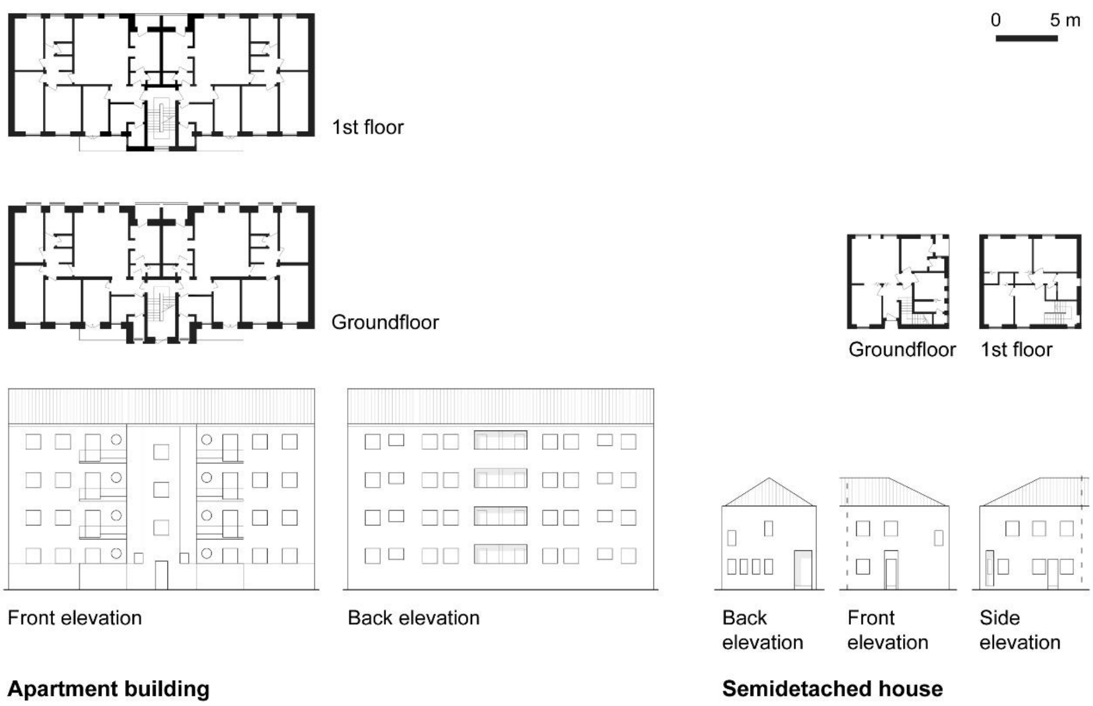

Figure 3.1 Schematic plans and elevations of the two housing typologies: an apartment building and a semidetached house

Table 3.2 Main characteristics of the case study dwellings

Characteristics	Unit	City apartment (CA)	Suburban house (SH)
Building lifespan	years	50	50
Usable floor area	m^2	102	104
Location (base case)[a]	-	1	5
Number of occupants	p/household	3	3
Living area per person	m^2/p	34.0	34.7
Income	€/household	1500[b]	1500[b]
Appliances	number/household	14.4[b]	14.4[b]
Commuters	p/household	2	2
Road distance to workplace (base case)	km	5	19
Construction type	Load-bearing stone masonry walls; wood framed floors and reinforced concrete slabs; wood framed roof with ceramic tiles		

[a] see Figure 3.2
[b] based on Wiesmann et al. 2011

The LCI implemented in this analysis followed a process-based approach. The main goal was to explore the relationship between urban design and environmental impacts along the LC by comparing two residential typologies. The data on both buildings were specific and provided in physical units; and possible truncation errors would be expected to have a similar effect on both buildings, with no significant consequences in the comparison. Lastly, process-based data in Europe are extensive, but IO data in Portugal are highly aggregated and much less extensive than in the US or Australia. The potential truncation errors that might arise from this methodological choice are discussed in section 3.4.

3.2.1. Building construction

NRE and GHG emissions of materials production (for construction and retrofit) were calculated based on the LCIs of building products (Kellenberger et al. 2007; Werner et al. 2007). Table 3.3 lists the construction materials by building element and respective quantities. Eight building elements were considered: (1) external walls using hydraulic stone masonry and hollow brick masonry, (2) foundations using hydraulic stone masonry, (3) interior walls using solid and hollow brick masonry (4) floors, both wooden beams/planks and reinforced concrete slabs (5) staircases in concrete with reinforced concrete landings (6) roofs, with wood structure and roof tiles (7) fenestrations in glass and wood (8) interior doors in wood.

Table 3.3 Construction materials by building element (including retrofit)

Building element[a]	Description	Apartment building (CA) Area (m²)	Apartment building (CA) Volume (m³)	Suburban house (SH) Area (m²)	Suburban house (SH) Volume (m³)	Materials	Unit	Retrofit Factor	CA Input	SH Input
Exterior walls	Hydraulic stone masonry	761.2	342.6	156.6	54.8	Cement mortar	kg	1.2	107 903	17 266
						Limestone	kg	1.2	873 500	139 774
	Hollow brick masonry	83.2	16.64	13.58	2.0	Cement mortar	kg	1.2	5 242	642
						Hollow brick	kg	1.2	22 065	2 701
	Rock wool insulation	844.4	33.8	170.2	6.8	Rock wool	kg	1.0	3 378	681
Foundation	Hydraulic stone masonry	-	150.9	-	41.6	Cement mortar	kg	-	39 616	10 908
						Limestone	kg	-	320 701	88 304
Interior walls	Solid brick masonry	249.0	62.3	30.4	4.6	Cement mortar	kg	-	16 340	1 195
						Solid brick	kg	-	116 403	8 512
	Hollow brick masonry	919.1	137.9	96.9	14.5	Cement mortar	kg	-	36 190	3 816
						Hollow brick	kg	-	152 341	16 063
Floors	Wooden floors	-	34.0	-	6.0	Sawn timber	m³	-	34	6
	Reinforced concrete slabs[b]	184.5	33.2	21.4	3.9	Concrete	m³	-	33	4
						Steel	kg	-	1 594	185
	Ceramic tiles	147.1	4.4	14.0	0.4	Ceramic tiles	kg	2.0	17 647	1 684
Roofs	Wood structure	-	20.5		3.0	Sawn timber	m³	-	20	3
	Ceramic roof tiles	271.7	-	70.6	-	Ceramic tiles	kg	2.0	24 779	6 435
	Rock wool insulation	271.7	10.9	70.6	2.8	Rock wool	kg	1.0	1 087	282
Fenestration	Windows	31.8	-	7.6	-	Wood frame	m²	2.0	64	15
	Windows	46.4	-	9.3	-	Double glazing	m²	2.0	93	19
	Exterior doors	36.6	-	5.0	-	Wood and glass	m²	2.0	73	9.9
Interior doors	Interior doors	154.8	-	30.9	-	Wood	m²	-	155	31
Staircase	Reinforced concrete landings	36.6	6.6	2.5	0.4	Concrete	m³	-	7	0
						Steel	kg	-	316	21
	Stairs	-	3.7	-	0.8	Concrete stairs	m³	-	4	1

[a] Only major materials were included in the inventory (e.g., paint was excluded)
[b] Steel reinforcement per m² is slightly lower than typical figures

To account for maintenance and retrofits along the 50-year lifespan, a set of intervention measures was considered (based on the survey by Alegre 1999), namely the partial (20%) replacement of stone and brick masonry in walls, the replacement of all ceramic floor tiles, the replacement of all roofing tiles and fenestrations, and the addition of insulation in external walls and roof (40 mm of rockwool). Initially, the buildings did not utilize thermal insulation. Details regarding building materials were obtained from the original drawings and other project documents maintained at the Municipal Archive of Lisbon (also in Alegre, 1999). The type of stone used in exterior wall masonry was assumed to be limestone, based on contemporary construction material references (Aguiar et al. 2002; Appleton 2005). Basements were excluded from the model. An additional 10% was added to the energy and GHG emissions to account for transportation of materials and onsite construction activities in this phase, based on the average building construction share of process-based LCA studies (Nässén et al. 2007).

3.2.2. Building use

The use phase includes all household (building) energy demands, including electricity, natural gas and liquefied petroleum gas (LPG). The total energy use per year was calculated based on the ratio between residential electricity use and natural gas or LPG from the Lisbon Energy Matrix (Lisboa E-Nova 2005), which estimates energy use in Lisbon building stock using 2002 data. Electricity accounted for 60% of the final energy use in residential buildings, while natural gas or LPG accounted for 40%. In the residential building stock of Lisbon, gas is mainly used for cooking and water heating, and also accounts for a small share of heating; electricity is mainly used for cooling, lighting, heating and other uses (Lisboa E-Nova 2005).

Annual electricity use was calculated using an econometric model for Portugal proposed by Wiesmann et al. (2011), based on the analysis of residential electricity expenditures using 2005 data. We used model 2a (Wiesmann et al. 2011), which incorporated 10 variables: (1) persons per household (three); (2) building age (50 years); (3) dwelling area (102 m^2 for the CA and 104 m^2 for the SH); (4) dwelling type (apartment in building with fewer than 10 apartments and semidetached house); (5) urbanization level (mainly urban); and (6) region (Lisbon). Regarding (7) income and (8) number of appliances, the average for mainland Portugal was considered (Wiesmann et al. 2011). Lastly, (9) children were considered to be present in half the dwellings for each building type (mainland average was 58%); and (10) all dwellings were considered to be owned by the household. The model allowed us to account for the influence of dwelling type on energy use; however, socio-economic and demographic characteristics were considered to be the same in both households. Controlling for behavior and self-selection effects can be very complex (Rickwood et al. 2008) and was not within the scope of the analysis.

NRE and GHG emissions associated with electricity were considered to be 6.33 $MJ_{prim\ fossil}$ and 485 g CO_2 eq/kWh, respectively, the average of the Portuguese annual electricity supply mix from 2003 to 2012, based on Garcia et al. (2014). We have not used a total primary energy factor because there is no unified approach to account for renewable energy (Molenbroek et al. 2011). In this 10-year period, the share of renewables in the generation mix in Portugal ranged from 20% to 56% (2005 and 2010, respectively), and the factor found by Garcia et al. (2014) (1.76 $MJ_{prim\ fossil}/MJ_{fin}$) is comparable with the NRE found for European countries with a high share of renewables (1.6 $MJ_{prim\ fossil}/MJ_{fin}$ for Sweden; 1.78 MJ_{prim} fossil/MJ_{fin} for Spain, with 50% and 22% of renewable energy, respectively). Natural gas and LPG used factors of 1.13 MJ_{prim} and 72.2 g CO_2 eq per MJ_{fin} (Faist-Emmenegger et al. 2007).

3.2.3. User transportation

Energy requirements and GHG emissions from commuting were calculated for the CA (in the city) and for the SH (in a suburb about 20 km from the city center), considering commutes between the residence location and workplace or school. Figure 3.2 presents the parishes (administrative subdivisions of a municipality) considered for the CA (1) and SH (5). Four additional locations (2, 3, 4 and 6) were used in a sensitivity analysis for the SH (see below).

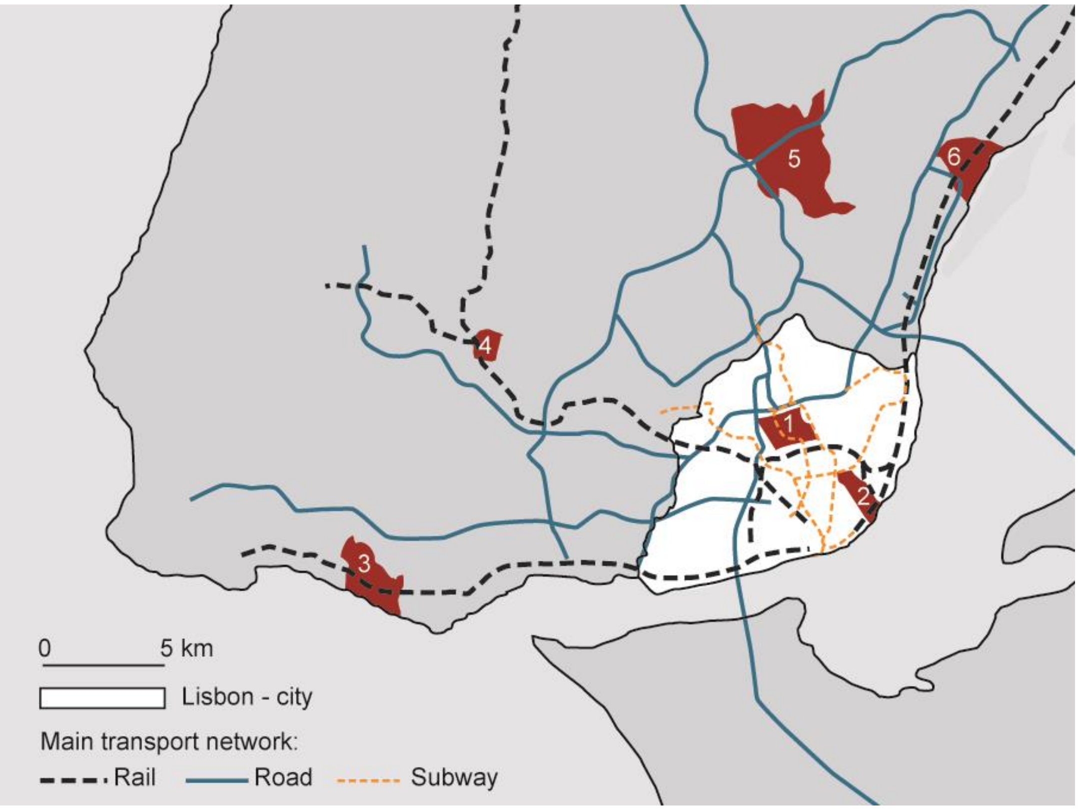

Figure 3.2 Considered parishes and main transportation network. (1) Campo Grande, (2) São João, (3) Parede, (4) Mira-Sintra, (5) Santo Antão doTojal and (6) Póvoa de Santa Iria

Two of the three people in each household were assumed to commute. Census data (INE 2012) show that an average of 65% of the population in these parishes were commuters in 2011 (range from 46% to 72% for locations 4 and 6, respectively). The workplace/school was assumed to be in central Lisbon in the base case. Two other locations were considered in a sensitivity analysis for the SH: a workplace in the residence parish and a workplace in another parish in the same municipality. Existing data (INE 2012) show that about 59% of the commuters from the six residential parishes work and live in the same municipality. However, in Lisbon parishes (1 and 2) this share exceeds 85%, while in suburban parishes the share ranges from 43% (6) to 66% (5). In suburban parishes, most commuters who leave the municipality go to Lisbon (21% and 35% of the overall commuters for locations 5 and 6, respectively). Among the suburban locations, 20–31% of commuters work in the parish where they live, while 16–39% work in another parish in the same municipality.

Transportation mode fractions were based on statistical data (INE 2012). Energy use and GHG emissions for each mode were based on a LC inventory (Spielmann et al. 2007), which included vehicle manufacturing. The LC data represent European or Swiss averages. This is not expected to affect results significantly since vehicles are mostly imported, and most transport infrastructures should be comparable. Table 3.4 presents the share of commuters using six transportation modes (by parish in 2011), which include car, bus, collective company/school transport, subway, train and motorcycle (INE 2012). These categories were selected from the statistical data and incorporate all motorized land transport modes. The other categories were boat (excluded because it is mainly used by commuters who cross the river, and not relevant for the locations considered), bicycle and walking. These last two modes, which have very low NRE and GHG emissions, were excluded because they are likely limited to short distances and could be partially attributed to the third (non-commuting) person in the household. The dominance of the car mode is highlighted: it accounted for over half of all commuters (except in parishes 2 and 4, where it was about 40%), and ranged to 65% at location 3, probably due to a higher socio-economic context and good road infrastructure. Bus accounted for 3–30% of commuters; subway (only in parishes 1 and 2) accounted for 12%, and train accounted for under 2% within the city and 16–18% in suburban locations (except for 5, where it was not available). Company or school collective transport accounted for less than 2%, and motorcycle for less than 1%. Other transportation modes, which were mostly walking, accounted for 11–20% of commutes.

Table 3.4 Share of commuters by main transportation mode in the six residential locations (2011)

Place of residence [a]	Number of commuters	Car (driver)	Car (passenger)	Bus	Company /school transport	Subway	Train	Motorcycle	Others
		(%)	(%)	(%)	(%)	(%)	(%)	(%)	(%)
1	6 114	37.7	13.7	13.4	1.0	12.2	1.1	1.0	19.9
2	7 818	29.0	10.6	26.1	0.8	12.2	1.7	0.9	18.9
3	12 368	44.3	20.3	2.8	0.7	0.6[b]	18.1	1.0	12.1
4	2 406	27.9	11.5	22.0	0.6	0.5[b]	18.1	0.5	19.0
5	2 412	41.3	15.5	27.2	3.5	1.0[b]	0.1[b]	0.5	11.0
6	21 239	41.6	16.7	9.8	1.5	0.9[b]	16.2	0.5	12.8

[a] See Figure 3.2

[b] Cases where the nearest station was more than10 km from the parish were not considered

Table 3.5 Commuting distance from the six residential locations to Lisbon city center

Place of residence [a]	Travel distance (km)					
	Car	Bus	Company/ school collective transport	Subway	Train	Motorcycle
	1.4 p/vehicle	20 p/vehicle	20 p/vehicle	130 p/vehicle	205 p/vehicle	1 p/vehicle
1	5.0	5.8	5.3	5.0	5.8	5.0
2	3.5	4.0	3.7	4.7	6.2	3.5
3	22.0	25.3	23.1	-	19.5	22.0
4	19.5	22.4	20.5	-	20.7	19.5
5	19.0	21.9	20.0	-	-	19.0
6	22.5	25.9	23.6	-	14.9	22.5

[a] See Figure 3.2

Table 3.6 Commuting distance from location 5 to the three workplace locations

Workplace	Travel distance (km)			
	Car	Bus	Company/ school collective transport	Motorcycle
	1.4 p/vehicle	20 p/vehicle	20 p/vehicle	1 p/vehicle
Lisbon center	19.0	21.9	20.0	19.0
Residence parish (5)	1.5	1.7	1.6	1.5
Another parish in the municipality	10.0	11.5	10.5	10.0

Commuting distances were calculated for each residential area and transportation mode using maps and other transport-provider documents. Table 3.5 shows travel distances from the residence location to the city center by transportation mode; Table 3.6 shows travel distances from location 5 to three alternative workplace locations. For cars and motorcycles, distances were measured on maps considering the full transportation network. Where the shortest distance included tolls, an average between that distance and an alternative without tolls was used. For buses and company/school collective transport, distances were increased by 15% and 5%, respectively. For subways and trains, the network routes were considered. The occupancy rate of cars was 1.4 persons/vehicle (Table 3.5), based on statistical data (INE 2012) at the parish level (range: 1.36 - 1.46). Bus, subway and train occupancy rates of 20, 130 and 205 persons/vehicle, respectively, were based on reports from the main providers (Carris 2012; Comboios de Portugal 2012; Metropolitano de Lisboa 2012). Collective company/school transport occupancy was assumed to be the same as the bus (20 persons/vehicle). Motorcycle occupancy was assumed to be one person/vehicle.

One round trip per day for weekdays was assumed (255 days/year). The statistical commuting data referred to all people living in a parish at least 15 years of age and professionally or academically active in 2011 (INE 2012). The main transportation mode was assumed to be the only transportation mode used. Cases where the nearest train or subway station was more than 10 km from the parish and designated as the main transportation mode were not considered (other modes were more likely), but such commutes represented less than 1% of the population across all parishes (Table 3.4).

3.3. Results

3.3.1. Life-cycle analysis

The results highlight the large contribution of transportation to the overall LC demands of residential buildings. Figure 3.3 shows the overall LC NRE and GHG intensity, and the relative contributions of each LC phase over the 50-year lifespan. In the overall LC, the SH had 75–100% higher energy and GHG emissions than the CA. However, considering only construction (including retrofit) and use phases, the difference was only 16%. For the CA, building use accounted for the largest share of NRE and GHG (63–64%), while for the SH, most (51–57%) of the energy requirements and GHG emissions were associated with transportation. The second highest contributions were transportation (20–24%) in the CA and building use (35–38%) in the SH. Construction accounted for about 15% of the impacts for the CA and even less, 10%, for the SH. Table 3.7 presents results for the three functional units considered. The relative contributions of the three LC phases were similar across the units because the area and occupancy of dwellings considered in both typologies were similar.

Table 3.7 Life-cycle non-renewable primary energy (NRE) and greenhouse gas (GHG) emissions for the city apartment (CA) and the suburban house (SH)

	City apartment (CA)			Suburban house (SH)		
NRE	GJ per dwelling 50 y	MJ per m^2 y	MJ per p y	GJ per dwelling 50 y	MJ per m^2 y	MJ per p y
Construction	287	54.1	1 910	332	63.9	2 215
Use	1 351	254.8	9 004	1 408	270.7	9 386
Transportation	521	98.3	3 475	2 298	442.0	15 323
Total	1 867	352.3	12 447	3 735	718.3	24 900
GHG	t CO_2 eq per dwelling 50 y	t CO_2eq per m^2 y	t CO_2 eq per p y	t CO_2 eq per dwelling 50 y	t CO_2 eq per m^2 y	t CO_2 eq per p y
Construction	24.9	4.7	165.8	28.9	5.6	193.0
Use	98.5	18.6	656.3	102.6	19.7	684.2
Transportation	30.7	5.8	204.7	139.0	26.7	926.7
Total	154.0	29.1	1 026.8	270.6	52.0	1 803.9

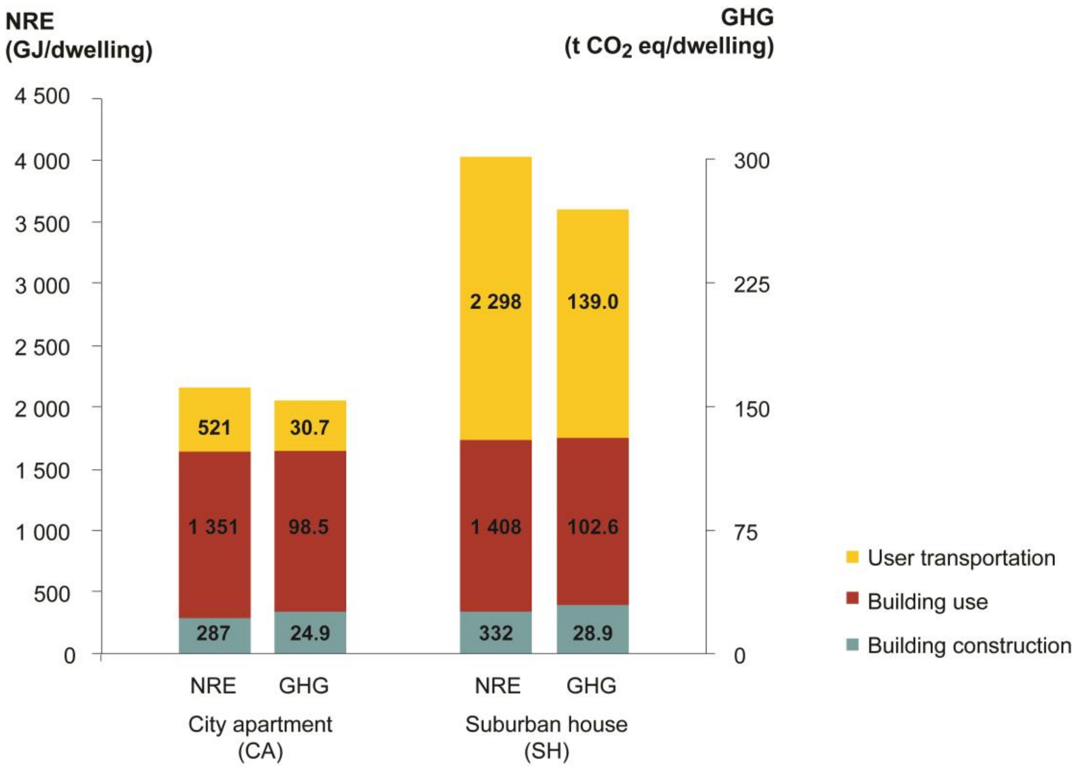

Figure 3.3 Life-cycle non-renewable primary energy (NRE) and greenhouse gas (GHG) emissions for the city apartment (CA) and the suburban house (SH) for a 50-year lifespan

3.3.2. Building construction

Tables 3.8 and 3.9 detail the NRE and GHG emissions associated with material production per unit of building element (in volume or area units) for both typologies. The overall NRE and GHG emissions were similar for both typologies. Interior walls had significantly higher NRE and GHG emissions per unit of construction than exterior walls (by 4-fold), mainly because the interior walls are brick masonry, which is associated with much higher energy and GHG emissions than limestone masonry per cubic meter. Concrete and wooden floors also showed significant differences (about four and six times higher NRE and GHG emissions in concrete floors, respectively).

Table 3.8 Construction phase: non-renewable primary energy (NRE) and greenhouse gas (GHG) emissions by building element (per dwelling and per m^2 of floor area)

Building element	City apartment (CA)				Suburban house (SH)			
	NRE (MJ)		GHG (kg CO_2 eq)		NRE (MJ)		GHG (kg CO_2 eq)	
	total	per m^2	total	per m^2	total	per m^2	total	per m^2
Exterior walls	37 403	353	4 037	38.1	48 321	465	5 091	49.0
Interior walls	95 168	898	9 241	87.2	69 896	672	6 800	65.4
Wooden floors	8 139	77	442	4.2	11 597	112	630	6.1
Concrete floors	41 176	388	3 074	29.0	33 189	319	2 588	24.9
Staircase	2 625	25	388	3.7	2 214	21	362	3.5
Roof	18 534	175	1 478	13.9	34 907	336	2 870	27.6
Windows	23 640	223	1 401	13.2	49 705	478	2 893	27.8
Foundations	7 677	72	1 020	9.6	16 910	163	2 247	21.6
Exterior doors	12 276	116	818	7.7	13 300	128	886	8.5
Interior doors	13 847	131	710	6.7	22 076	212	1 132	10.9
Material production	260 485	2 457	22 609	213.3	302 113	2 905	25 499	245.2
Transportation and on-site activity (10%)	26 049	246	2 261	21.3	30 211	291	2 550	24.5
Total	286 534	2 703	24 870	234.6	332 324	3 195	28 048	269.7

Table 3.9 Material production: non-renewable primary energy (NRE) and greenhouse gas (GHG) emissions by building element (per m^2 or m^3 of building element)

Building element	City apartment (CA)			Suburban house (SH)		
	(m^3)	(MJ/m^3)	(kg CO_2 eq/m^3)	(m^3)	(MJ/m^3)	(kg CO_2 eq/m^3)
Exterior walls	44.9	833	89.9	56.8	851	89.6
Interior walls	25.0	3 803	369.3	19.1	3 659	356.0
Wooden floors	4.3	1 915	104.0	6	1 933	105.0
Concrete floors	4.7	8 761	654.0	4.3	7 718	601.9
Staircase	1.3	2 039	301.4	1.2	1 845	301.7
Foundations	18.9	407	54.1	41.6	406	54.0
	(m^2)	(MJ/m^2)	(kg CO_2 eq/m^2)	(m^2)	(MJ/m^2)	(kg CO_2 eq/m^2)
Roof	34.0	546	43.5	70.6	494	40.7
Windows	9.8	2 418	143.3	16.9	2 941	171.2
Exterior doors	4.6	2 683	178.8	5	2 660	177.2
Interior doors	19.4	716	36.7	30.9	714	36.6

3.3.3. Building use

Table 3.10 presents the final energy demand, NRE and GHG emissions for the use phase. In absolute terms, the CA had 4% lower energy demand and GHG emissions. The trend was similar for the other functional units, e.g., the SH had 6% higher NRE and GHG per square meter and 4% per inhabitant.

Table 3.10 Building use: Final energy demand, non-renewable primary energy (NRE) and greenhouse gas (GHG) emissions

	Final energy demand		NRE	GHG
	(kWh/year)	(MJ/year)	(MJ_{prim}/year)	(kg CO_2 eq/year)
City apartment (CA)				
Electricity	2 989	10 761	18 921	1 450
Natural gas/LPG	2 004	7 216	8 090	519
Total	4 994	17 977	27 011	1 969
Suburban house (SH)				
Electricity	3 116	11 217	19 723	1 512
Natural gas/LPG	2 090	7 522	8 433	541
Total	5 205	18 739	28 157	2 053

3.3.4. User transportation

Transportation impacts, including NRE and GHG per commuter per km and NRE and GHG per commuter per year, are presented in Tables 3.11 and 3.12, respectively. Passenger car use accounted for 74% of impacts associated with transportation in all parishes. Cars were not only the dominant transportation mode, but also had the highest NRE demand and GHG emissions due to low environmental performance and occupancy rate.

Table 3.11 User transportation: non-renewable primary energy (NRE) and greenhouse gas (GHG) emissions per commuter per km (person-kilometer traveled, PKT)

	NRE ($MJ\ p^{-1}\ km^{-1}$)	GHG ($kg\ CO_2\ eq\ p^{-1}\ km^{-1}$)
Car	3.0	0.18
Bus	1.7	0.10
Company/school collective transport	1.7	0.10
Subway	0.5	0.01
Train	0.5	0.01
Motorcycle	1.5	0.10

Table 3.12 User transportation: non-renewable primary energy (NRE) and greenhouse gas (GHG) emissions per commuter over a year, for each residence location

Location	Total		Car		Bus		Company/ school transport		Subway		Train		Motorcycle	
	NRE (MJ)	GHG (kg CO_2 eq)	NRE (%)	GHG (%)	NRE (%)	GHG (%)	NRE (%)	GHG (%)	NRE (%)	GHG (%)	NRE (%)	GHG (%)	NRE (%)	GHG (%)
1	5 212	307	86.7	88.1	8.7	9.3	0.6	0.6	2.9	0.9	0.3	0.1	0.8	1.0
2	3 264	191	74.6	76.5	18.9	20.4	0.5	0.5	4.3	1.4	0.8	0.2	0.8	1.0
3	26 633	1 559	93.9	96.1	1.6	1.7	0.4	0.4	-	-	3.4	0.9	0.8	0.9
4	17 548	1 019	77.0	79.6	16.6	18.0	0.4	0.4	-	-	5.5	1.5	0.4	0.5
5	22 984	1 390	82.6	81.9	15.2	15.8	1.8	1.9	-	-	-	-	0.4	0.4
6	25 496	1 507	90.4	91.8	5.9	6.3	0.8	0.9	-	-	2.5	0.6	0.4	0.5

3.3.5. Sensitivity analyses

Residence location

Figure 3.4 shows the NRE demand and GHG emissions associated with the semidetached house in the six alternative parishes. Residential location significantly affected results. Locations in the central city, near the workplace/school, had better performance, e.g., location 2 had the lowest NRE and GHG emissions (construction, use and transport accounted for 16%, 68% and

16% of the total, respectively). Overall NRE and GHG emissions were 50–115% higher at suburban locations. Since the same building type (semidetached house) was considered, these differences are due to the differences in transportation demand associated with the six locations.

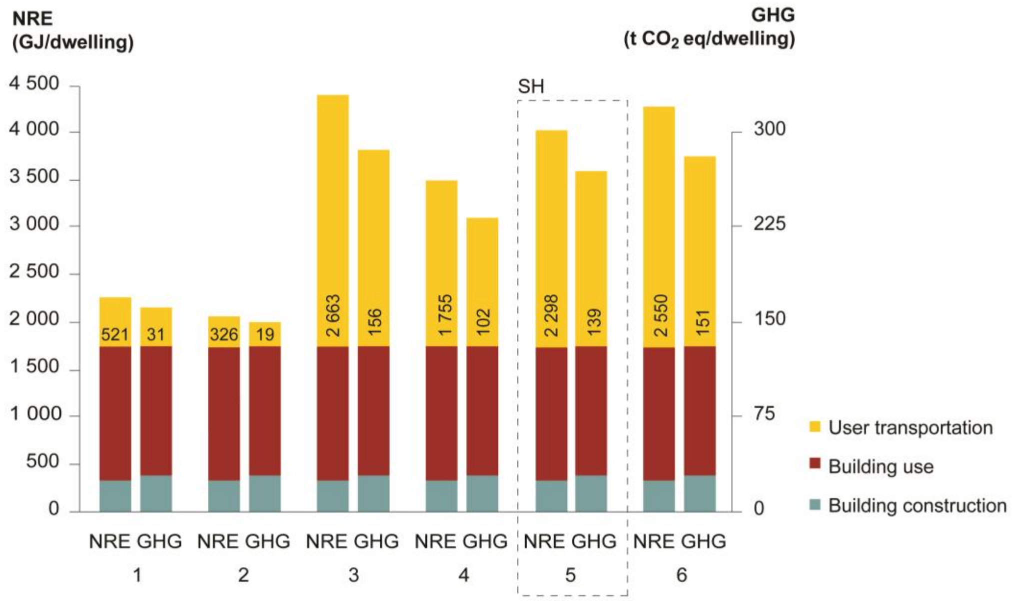

Figure 3.4 Sensitivity analysis: non-renewable primary energy (NRE) and greenhouse gas (GHG) emissions of the semidetached house in six alternative locations (50-year lifespan)

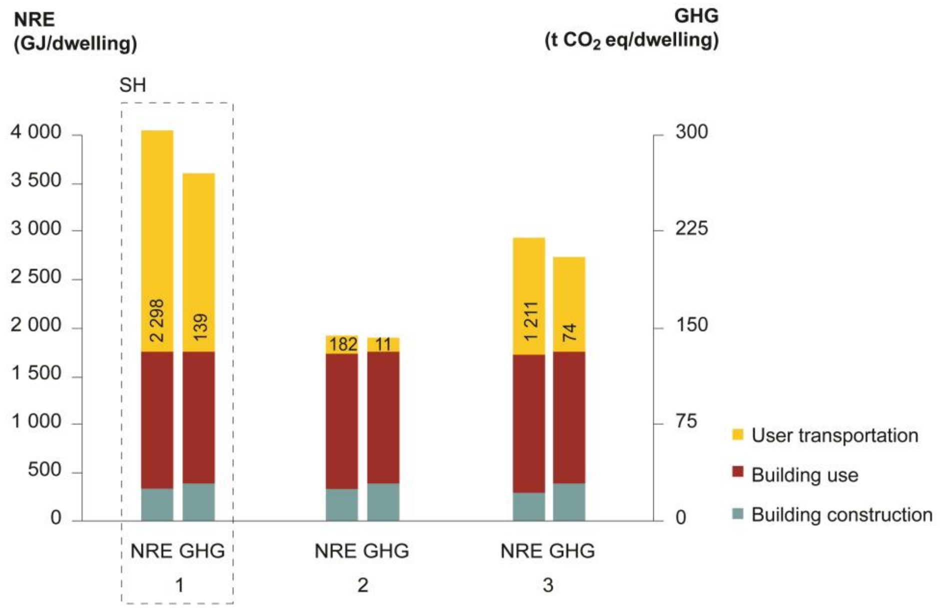

Figure 3.5 Sensitivity analysis: non-renewable primary energy (NRE) and greenhouse gas (GHG) emissions of the semidetached house in location 5, for three workplace locations: (1) in Lisbon city center, (2) in the residence parish and (3) in another parish in the same municipality (50-year lifespan)

Workplace location

Figure 3.5 shows the energy and GHG emissions associated with the semidetached house at location 5 for three alternative workplace locations: Lisbon city center (base case), the parish of the residence and another parish in the residence municipality. Compared with the base case, in which the workplace is located in the city center, a workplace located in the residence parish reduces the overall LC NRE and GHG emissions by about 50% (case 2), and by about 75% for a workplace in another parish of the municipality. Even in the case 3, transportation needs remain highly significant, accounting for 36–41% of the overall NRE and GHG emissions. In 2011, only 20–31% of suburban commuters travelled within their residence parish (INE 2012), which highlights the importance of transportation impacts for cases 1 and 3.

3.4. Discussion

The comparative analysis shows that the suburban house (SH) had almost twice the energy and GHG emissions of the city apartment (CA) due to transportation demand, and that transportation in the SH accounted for over 50% of the overall energy and GHG emissions over the 50-year lifespan. In contrast, excluding user transportation, the difference is less than 9%. Thus, transportation impacts were highly significant, especially for the suburban locations. This results for several reasons. First, suburban locations are associated with high transportation demand, most of which is met by car use with low occupancy rates. Second, in favorable (e.g., Mediterranean) climates like Lisbon's, use phase energy requirements of buildings are generally low. Third, the econometric model (Wiesmann et al. 2011) was based on statistical data and the average observed energy use, which can be significantly lower than estimates based on models and assumed thermal comfort levels (Ryan & Sanquist 2012).

On a per capita basis, the NRE ranged from 1.9 to 2.2 GJ person^{-1} year^{-1} for the construction phase, from 7.1 to 7.4 GJ person^{-1} year^{-1} for the use phase and from 3.5 to 15.3 GJ person^{-1} year^{-1} for transportation. As noted, previous studies have estimated a wide range of LC energy and GHG emissions for residences and user transportation, e.g., overall LC energy requirements (including all phases) vary from 26 to 249 GJ person^{-1} year^{-1}. This range can be explained by differences in climate, socio-economic and cultural context, urban design and topography, and by methodological differences among the studies, including lifespan assumptions (from 30 to 100 years), LC phases and components considered (e.g., urban infrastructure; Stephan et al. 2011; Stephan et al. 2013a), and the LCI approach used (e.g., EIO: Norman et al. 2006; and hybrid: Stephan et al. 2013a). Our results tend to be significantly lower than the other studies. In Toronto, Norman et al. (2006) predicted higher building construction (from 4.7 to 7.5 GJ person^{-1} year^{-1}) and much higher operational energy requirements (from 27.5 to 49.8 GJ person^{-1} year^{-1}), probably due to the much colder climate and higher socio-economic context; transportation requirements were somewhat higher, especially for the suburbs (from 7.9

to 28.8 GJ person^{-1} year^{-1}). The other studies also obtained higher results for multiple reasons: most studies considered a hybrid LCI approach to calculate embodied energy; total primary energy was calculated in most studies (accounting for renewable and non-renewable energy); only car transportation was considered (Stephan et al. 2011; Stephan et al. 2012; Stephan et al 2013b; Stephan & Stephan 2014; Treloar et al. 2000); several studies included urban infrastructure materials (Stephan et al. 2012; Stephan et al. 2013a, 2013b; Stephan & Stephan 2014); one study included a diversity of household activities (e.g., financial services, consumables and vacation: Treloar et al. 2000), and most studies used a higher ratio of building floor area/person (e.g., Treloar et al. 2000; Stephan et al. 2012; Stephan et al. 2013a, 2013b), which also increased construction and use phase requirements (on a per person basis).

Construction energy demands were significantly lower in our analysis than those reported in most of the studies cited in the literature review, although our results fell within the range of 1.5–15.1 GJ/m^2 found by Nässén et al. (2007). While this could be partially associated with climate and socio-cultural context, two methodological issues are likely to influence results significantly: (1) embodied NRE and GHG emissions were calculated using a process-based analysis; and (2) only the major construction materials were considered. Although process analysis is widely accepted as accurate and relevant when specific products are modeled and compared in a LC study, its finite system boundaries result in truncation errors (since processes outside boundaries are excluded) and thus may underestimate results, as noted above. Increasing the construction energy requirement and GHG emissions by 64% (as calculated in Nässén et al. 2007) would increase the overall LC results by 5–10%. In this case, construction would contribute more to the overall LC impacts (for the CA, it would account for 20–24%, instead of 13– 16%, and the SH would increase to 13–16%, instead of 8–11%); however, the main results and study conclusions would not be altered. Regarding construction materials, some finishing materials were excluded in our analysis because these were not expected to influence the results highly (Kellenberger & Althaus 2009). As shown in previous studies (Crawford 2014; Rauf & Crawford 2015), finishing materials have generally low embodied energy coefficients; however, due to their short service life, these materials can represent a significant share of the recurrent embodied energy (energy embodied in the materials used for retrofit and maintenance of the building). Crawford (2014) calculated the embodied energy of a house and found that finishes contributed to 31% of the recurrent embodied energy; most of the remainder was attributable to paints and carpets. Our buildings did not have carpets (floors were ceramic tiles and wood, which were accounted for) and paint was excluded. Our replacement rates may be lower than other studies because the retrofit scenario was based on a post-occupancy survey. The actual replacement rate of materials is likely to be lower than rates based on material service life.

Primary energy conversion factors also affect results. We considered NRE, because there is no unified approach to account for renewable sources (Molenbroek et al. 2011). This methodological choice should mainly affect electricity conversion. While we considered a factor of 1.76 $MJ_{prim\ fossil}/MJ_{fin}$, previous studies often have used total primary energy, which had considerably higher factors (e.g., 3.4 MJ_{prim}/MJ_{fin} in Stephan et al. 2012; (Fuller & Crawford 2011; Treloar et al. 2000). The 2003–12 generation mix in Portugal had about 40% of renewable energy on average (Garcia et al. 2014), and the total primary energy factor would be 2.16 MJ_{prim}/MJ_{fin}, which represents a 16% increase over our use phase NRE results. The LC overall primary energy (including all phases) would be 10% and 7% higher, for CA and SH, respectively. The relative contributions would also slightly change (CA: 12%, 66% and 22% for construction, use and user transportation phases, respectively, compared with 13%, 62% and 24%; SH: 8%, 38% and 54% compared with 8%, 25% and 57%). With the factor of 3.4 MJ_{prim}/MJ_{fin} used in the Australian studies where the share of renewables is considerably lower, our building use phase energy results would be about 65% higher. In this case, use phase results per capita, 14.9–15.5 GJ $person^{-1}$ $year^{-1}$ would be comparable with Fuller & Crawford (2011), 9 - 25 GJ $person^{-1}$ $year^{-1}$).

User transportation results also were lower than in previous studies. While most previous studies modeled only car travel (Table 1), we considered the local commuting mode mix of commuters and public transport, which decreased NRE and GHG emissions, although car use still accounted for most (74%) of the NRE and GHG emissions of user transportation. Another methodological choice that likely affected results was that only commuting was considered (round trips from residence to workplace on workdays 255 per year, about 5000 km per person for the suburban location (and less for alternative scenarios). Previous studies estimated the total household travel demand and included all types of trips, e.g., for work, shopping and leisure. In addition, geographic, cultural and socio-economic context is expected to increase travel demand and the associated energy use in countries such as Australia and Canada, where fleet efficiency is lower than in EU countries. For example, an analysis of transportation showed 6202 passenger-km per capita by car in western European cities, half of that in Australia and New Zealand, and energy use per private passenger vehicle was 3.3 MJ/VKT in Western Europe compared with 5.0 MJ/VKT in Canada (Kenworthy 2003). Lastly, user transportation could be allocated differently, e.g., commuting could be attributed to both residential and office buildings. While system boundaries and allocations depend on the study approach and purpose, all previous LCA studies integrating transportation (Table 3.1) allocated transportation to residential buildings.

The results demonstrate the significance of transportation demands and the influence of building location. In cases, the effect of location and the associated transportation demand exceeds the LC impacts of the building itself. The sensitivity analysis illustrates that the same house could be associated with 50% less energy and GHG emissions if placed in the city center, rather than at a suburban location. Like most southern European cities, Lisbon has been subject to extensive dispersion, e.g., in 1960, the Lisbon municipality had 800 thousand inhabitants and over 52% of the population of the metropolitan area, while in 2011 it had about 550 thousand and less than 20% of the 2.8 million inhabitants of the metropolitan area (Tenedório 2002; INE 2012). In general, the expansion of cities in southern Europe followed different trends and patterns from those in the north, a factor that has not received much attention in urban sustainability research. In recent decades, urban areas have become mono-centric territories, characterized by a notorious lack of planning, a decreasing share of public transportation, and increasing car ownership and use (Catalán et al. 2008). Strategies that consider the transport infrastructure might more effectively improve urban sustainability than simply addressing building efficiency, especially in areas with mild climates. Strategies should consider housing and workplace/school locations, transportation mode choices, and efficient use of transportation modes (e.g., reducing car use and increasing vehicle occupancy). Moreover, such urban planning and transportation strategies have additional benefits, e.g., improved air quality, reduced traffic and congestion, and less infrastructure devoted to roads and parking. Improving overall building efficiency in urban settings requires coordinated planning, decisions and actions from many stakeholders across multiple jurisdictions.

The scope and detail of the analysis involved several simplifications. The building construction model considered only the main elements, and materials transportation was very roughly approximated. We did not consider urban infrastructure or transportation requirements other than commuting (shopping and other essential travel will further increase the transportation share of impacts). We compared two dwellings with similar floor area and occupancy. Future research might account for the actual per capita floor area for different locations (based on statistical data), improving the realism of space demands. Lastly, pollution, noise and other environmental impacts were not considered.

Despite these limitations, the analysis provides a compelling demonstration of the significance of transportation in the overall LC impact of buildings. It emphasizes the need to consider building location in urban planning strategies and building environmental assessments. Thus, for more sustainable development, urban planning and design must include the impacts resulting from urban-scale decisions and strategies, including urban design, building location, transportation demand, and infrastructure. Since a diverse set of factors influence environmental performance, site specific assessments are needed to identify the most effective improvement opportunities.

3.5. Conclusions

This chapter compared the LC energy and GHG emissions of an inner city apartment and a suburban semidetached house, addressing construction, use phase and user commuter requirements over a 50-year lifespan. The suburban house had 75 to 100% higher results, mostly due to transportation (without transportation this difference would be only of 16%). Transportation requirements, which accounted for over 50% of the overall LC impacts for residence locations in suburban areas, were dominated by car use. Cars were not only the dominant transportation mode, but also had the highest NRE demand and GHG emissions due to low environmental performance and occupancy rate. The results clearly demonstrate the significance of transportation demand and, consequently, of a building's location.

Development strategies that consider the transport infrastructure might more effectively improve urban sustainability than simply addressing building efficiency, especially in areas with mild climates. Planning strategies and policies should consider housing and workplace/school locations, transportation mode choices, and efficient use of transportation modes (e.g., reducing car use and increasing vehicle occupancy). Moreover, such urban planning and transportation strategies have additional benefits, e.g., improved air quality, reduced traffic and congestion, and less infrastructure devoted to roads and parking. Improving overall building efficiency in urban settings requires coordinated planning, decisions and actions from many stakeholders across multiple jurisdictions.

Building on the significance of transportation in this chapter's results, the next chapter expands on the environmental and health impacts of commuting.

4 A comparative LCA of commuting modes addressing $PM_{2.5}$ intake and health effects

A comprehensive LCA of commuting alternatives is conducted that compares six urban transportation modes (car, bus, train, subway, motorcycle and bicycle) for eight impact indicators. Fine particulate matter ($PM_{2.5}$) emissions and health impacts are incorporated in the assessment using intake fractions that differentiate between urban and non-urban emissions, and an effect factor that combines exposure, dose response and severity factors. The potential benefits of different strategies for reducing environmental impacts are illustrated.[3]

Section 4.1 provides the introduction, including background and motivation for the study. Section 4.2 describes the materials and methods, including the life-cycle model. Section 4.3 presents the results and discussion, and section. Drawing on the study outcomes, section 4.4 summarizes the conclusions.

4.1. Introduction

The transportation sector plays a crucial role in targets and policies for reducing energy consumption and greenhouse gas (GHG) emissions. Currently, this sector accounts for 32% of global primary energy consumption in the EU-28, and a higher percentage, 40%, in Portugal (EU 2014). Current transportation demand trends suggest the potential for sizable increases in both vehicle ownership and fossil fuel demand, with significant and adverse implications for energy supply security, climate and urban air quality (WBCSD 2004; Litman & Burwell 2006; Black & Sato 2007; Flamm 2009; Hawkins et al. 2012). The need for research that assesses the environmental impacts of transportation and policies that promote sustainable and healthy mobility is obvious.

A life-cycle (LC) perspective provides insight on the environmental impacts associated with different processes and phases across the life-cycle and its use of consistent metrics is essential

[3] Significant portions of this chapter are from Bastos, J., Marques, P., Batterman, S. & Freire, F. (under review) Environmental impacts of commuting modes in Lisbon: a life-cycle assessment addressing particulate matter impacts on health.

for comparing alternative transportation modes (Bauer et al. 2015; François et al. 2017). However, most previous LC studies of transportation modes have compared alternative technologies within the same mode or compared a limited number of modes, and most have focused on one or two environmental indicators, generally energy use and GHG emissions (Bauer et al. 2015). A broader set of environmental indicators is needed to identify and avoid unintended trade-offs among mitigation strategies (Chester, Pincetl et al. 2013; Bauer et al. 2015; Meng et al. 2017). This includes the need to better address the impacts of pollutant emissions, such as fine particulate matter ($PM_{2.5}$), on health in transportation LCAs, which require spatially differentiated characterization factors to properly address exposure (JRC 2011; UNEP/SETAC 2016), as discussed in the following section. While previous LC studies have addressed several aspects of transportation systems in Europe (e.g., Girardi et al. 2015, Sánchez et al. 2013), none has compared alternative transportation modes.

This chapter provides a comparative LCA of six transportation mode choices in an urban setting. The study is comprehensive and novel in its use of a wide set of environmental indicators, differentiated intake fractions for urban and non-urban emissions, and an effect factor to estimate health impacts associated with $PM_{2.5}$, a key air pollutant. To illustrate the application of the model, strategies that can lower environmental and health impacts associated with commuting are compared in a scenario analysis.

4.1.1. Life-cycle studies of urban transportation

Many LCA studies of land-based transportation have been performed in the last decades. Most environmental assessments focused on personal transportation and explored new or alternative solutions (e.g., alternative technologies, fuels, or eco-design strategies) to reduce impacts (Del Pero et al. 2015).

Due to the strong dependency of road transportation on fossil fuels and associated emissions, previous LC studies have assessed and compared potential technologies and fuel alternatives, such as biofuels, electric or hydrogen vehicles (e.g., Bartolozzi et al. 2013; Bauer et al. 2015). Hawkins et al. (2012) performed a literature review on environmental impacts of hybrid and electric vehicles, supporting that LCA was the preferred tool for comparing environmental impacts of transportation impacts. A large body of literature has provided comparative LCAs of internal combustion engine vehicles (ICEV) and electric vehicles (EV), namely hybrid electric vehicles (HEV) and battery electric vehicles (BEV), highlighting that the impacts of vehicles are heavily dependent on the use phase. Although EVs perform better than conventional ICEVs, the relative performance of electricity-based vehicles is strongly affected by the electricity mix used. In fact, if high-efficient ICEVs are considered, EVs do not present such advantages. Hawkins et al. (2012) added that the typical/average size of conventional ICEVs is generally larger than HEVs and EVs, which is likely to influence results.

Less research is available on the LCA of railway transportation. In Europe, Del Pero et al. (2015) performed an LCA of a heavy metro train in the area of Rome, including material acquisition, manufacturing, use and end-of-life, and showed that the use phase had the largest contribution. The study considered the vehicle-kilometer traveled (VKT) as functional unit, and assumed passenger loads based on UNI EN 15663 (80% of seats occupied and 2.3/m^2 standing). Impacts were calculated with CML2001 for abiotic depletion, acidification, eutrophication, eco-toxicity, GHG emissions, human toxicity, ozone layer depletion, photochemical ozone creation and terrestrial eco-toxicity. The results indicated that potential approaches to increase environmental performance of the heavy metro train would be: reducing mass, increasing efficiency during operation, and increasing the recyclability rate. Recently, an increasing number of environmental produc declarations (EPD) has been published for railway transportation (trams, metro, regional and intercity trains (e.g., AnsaldoBreda 2011; Bombardier Transportation 2011). Electric rail transport is generally considered an efficient urban transportation alternative; however, its environmental performance is highly dependent on the electricity generation mix and vehicle occupancy (Del Pero et al. 2015; Chester et al. 2012).

Regarding multi-mode comparative analyses, Stodolsky et al. (1998) compared, at an early stage, rail and road modes for freight transportation. However, few other studies were found in the literature, besides the extensive work by Chester and colleagues on the LCA of transportation, developed in the last decade. Chester (2008) developed a comprehensive LC inventory for passenger transportation in the US. In 2009, Chester & Horvath (2009) calculated the LC energy and GHG emissions for buses, trains and airplanes in the US, including the supply chain and production of vehicles, infrastructure and fuel. The authors concluded that vehicle occupancy strongly affected the relative performance of these modes. In 2010, Chester & Horvath (2010) conducted a LCA for a high speed rail (HSR) connecting four cities in the USA, comparing it with heavy rail transit, car and airplane traveling. This study emphasized again the influence of occupancy, and the best performance of railway transportation, when higher occupancies were considered. In another study, Chester et al. (2010) developed an energy and emissions inventory for three metropolitan areas in the US. (San Francisco, Chicago and New York City), comparing automobile, diesel rail, electric rail and ferry, and including impacts associated with vehicle insurance, parking construction and maintenance. Energy and emissions were calculated for passenger-mile traveled (PMT) and vehicle-mile traveled (VMT), to provide comparability. Cars had the largest impact, accounting for over 85% of the regional energy and emissions, and the authors found that a LC perspective is highly significant in this context, as the overall environmental impacts of a transport service were up to 20 times those of vehicle operation. The authors also considered healthcare and GHG monetized externalities to evaluate the societal costs of passenger transportation. Chester & Horvath (2012) compared fuel-efficient and electric cars, with HSR and airplane traveling, to explore potential advances in technology, in California. Chester, Pincetl et al. (2013) performed a comparative LCA for the new rapid bus and transit light rail lanes in Los Angeles, considering energy consumption, GHG emissions and criteria air pollutants (including potential for smog and respiratory impacts), using both attributional and consequential LCA. The LCA included vehicle manufacture and maintenance, infrastructure construction and operation, and energy production components, but also vehicle and infrastructure insurance. Particulate matter and ozone were considered with impact characterization factors from TRACI model (Bare et al. 2002) to assess respiratory and smog stressors. The authors highlighted the importance of considering environmental impacts beyond geopolitical boundaries; however, local (urban) and remote emissions were added with no distinction in their potential effects, which depend on exposure. The LC water requirements of petrol cars, urban electric and regional diesel trains were compared with an input-ouput approach, in Melbourne, Australia (Stephan & Crawford 2016). Recently, daily environmental impacts of the overall urban transportation in Lyon, France, were assessed in a study that integrated LCA and a land use and transport interactions (LUTI) model (François et al. 2017). The analysis addressed nine impact indicators, including PM emissions in g PM_{10}-eq inhabitant^{-1} day^{-1}, which used the same characterization for exhaust emissions and emissions occurring in other LC phases (e.g., vehicle manufacture).

Most previous LCAs of transportation modes compared alternative technologies within the same mode or a very limited number of modes, and focused on one or two environmental indicators (mainly energy use and GHG emissions). However, including a broader set of environmental indicators is crucial to identify and avoid unintended trade-offs in mitigation strategies (Chester, Pincetl et al. 2013; François et al. 2017). In addition, no studies were found providing a comprehensive LCA of alternative urban transportation modes in Europe and the integration of $PM_{2.5}$ in life-cycle impact assessment (LCIA) needs to be improved. This chapter presents a comprehensive multi-mode LCA comparing alternative transport modes for urban travel in the region of Lisbon, considering a broader range of impacts and integrating the impacts of $PM_{2.5}$ emissions, and their effects on health.

4.1.2. Addressing health effects associated with $PM_{2.5}$ in transportation LCA

Transportation has diverse and important environmental impacts. At the global scale, it contributes significantly to dependence on fossil fuels, global warming and environmental degradation (Chester 2008; Woodcock et al. 2007). At the local scale, it causes a considerable public health burden (Künzli et al. 2000), including morbidity and mortality associated with exposure to traffic-related air pollutants, as well as road-traffic injuries and impacts associated with physical activity, noise, and stress (Woodcock et al. 2007; Chen et al. 2008; Stevenson et al. 2016). Particulate matter (PM) is considered one of the most significant air pollutants, causing or contributing to a large share of adverse health effects associated with pollution (US EPA 2009a; Lim et al. 2012; WHO 2013; Hänninen et al. 2014; Apte et al. 2015). PM exposure results from both primary emissions, and secondary PM that is formed in the atmosphere by the reaction of precursor pollutants (Fantke et al. 2015). PM is classified by size, with the most common classifications being respirable particles (PM_{10}), fine particles ($PM_{2.5}$), and ultrafine particles (UFP), which have aerodynamic diameters below 10, 2.5 and 0.1 µm, respectively (WHO 2003).

Spatially differentiated factors are needed to calculate the exposures and health effects from $PM_{2.5}$ emissions (Finnveden et al. 2009; JRC 2011; UNEP/SETAC 2016), especially in transportation applications, where vehicle emissions in urban settings are mostly released at ground-level, with potentially much greater impacts than emissions occurring in rural settings or released from tall stacks (Humbert et al. 2011). For assessments of $PM_{2.5}$-related health effects, UNEP/SETAC recommends that analyses proceed from emissions to concentrations and then to exposure-responses (Fantke et al. 2015). Ideally, such assessments should be spatially-resolved to account for the distribution of emission sources and the locations of exposed and vulnerable individuals (addressed later, in chapter 5), especially since concentrations of traffic-related air pollutants display substantial intra-urban variation and steep concentration gradients (Brauer et al. 2000; Fischer et al. 2000; Wilson et al. 2005; Batterman et al. 2010; Baldwin et al. 2015; Rodriguez Roman & Ritchie 2017). However, detailed and accurate data on emissions and population distribution are often unavailable at fine spatial scales (UNEP/SETAC 2016), and predicting concentrations, exposures and affected populations in a realistic manner can be challenging. The use of exposure characterization factors for archetype environments provides an alternative approach for addressing these factors and calculating reliable estimates of population exposure to $PM_{2.5}$ emission sources.

An emissions-based assessment using the intake fraction (iF) approach has been recommended for LCAs examining the potential health impacts associated with air pollution (JRC 2011; Fantke et al. 2015; UNEP/SETAC 2016), providing a simpler alternative than using dispersion models. The iF, defined as the fraction of emissions inhaled by the total exposed population (Apte et al. 2012), depends on locations of emission sources and populations, geography, and pollutant fate and exposure factors. Fate describes the behavior of the pollutant, including its distribution, dilution, reaction, dispersion and deposition in the environment; these factors depend on the pollutant (e.g., particle size, residence time) and meteorology (e.g., wind velocity and mixing height). Exposure is the dose of pollutants inhaled by an individual (or population), i.e., the amount of PM that enters the respiratory system. Exposure depends on the indoor and outdoor pollutant concentrations, breathing rates, and physical and chemical properties of the pollutant, e.g., size, chemical composition and solubility (Humbert et al. 2011; Hodas et al. 2016). Several approaches with different levels of complexity and data requirements can be used to estimate iFs for traffic-related air pollutants, including air quality dispersion models, one-compartment or “box” models, and empirically-determined emission-concentration or “roll-back” relationships (Stevens et al. 2007). Default iF values have been derived for PM, a set of source heights and archetypal environments (e.g., urban, rural or remote), based on the literature and USEtox (Humbert et al. 2011). iF values for urban or densely populated areas are higher than those for rural areas, reflecting the number of persons exposed.

Intake estimates may be combined with exposure-response and severity data to estimate health impacts. The approach in the Global Burden of Disease (GBD) studies (Lim et al. 2012) has been recommended (JRC 2011; Fantke et al. 2015; UNEP/SETAC 2016). In this approach, the disease burden from different outcomes is summarized into a single metric: disability-adjusted life years (DALYs), representing the sum of years of life lost (YLL) due to premature mortality and years lived with disability (YLD; Lim et al. 2012; Murray et al. 2012; Martenies et al. 2015). For exposure to ambient $PM_{2.5}$, the GBD studies consider: lower respiratory infections; trachea, bronchus and lung cancers; ischemic heart disease (IHD); cerebrovascular disease; and chronic obstructive pulmonary disease (COPD). Gronlund et al. (2015) combined these asoects if the GBD methodology (Lim et al. 2012) with the iF approach (Humbert et al. 2011) to develop characterization factors that summarize the burden of disease attributable to $PM_{2.5}$ emissions. Outcomes considered were not weighted by age or discounted in time in order to avoid debatable differentiation in the valuations (Hänninen et al. 2014; Martenies et al. 2015).

The few LCA studies of transportation that have addressed PM, considered emissions across the life-cycle, without differentiating the substantially greater impacts per mass of PM emissions in urban settings (e.g., vehicle exhaust emissions) from emissions in remote areas (e.g., electricity production, oil refining and other upstream processes; Chester, Pincetl et al. 2013; Cooney et al. 2013; Bauer et al. 2015; Ercan & Tatari 2015; François et al. 2017). While such analyses can represent inventory emissions or the potential for PM formation, they do not accurately estimate the potential for human health impacts from PM emissions (Hauschild & Huijbregts 2015).

4.1.3. Reducing environmental impacts of commuting

Strategies to reduce impacts associated with transportation, mostly focusing on energy use, emissions and congestion, have been implemented since the 1970s (Porter et al. 2013). In the last decades, many actions have been identified and explored in the literature, both addressing technological developments and travel behavior. One potential strategy is reducing the need to travel, e.g., reducing the number of trips by telecommuting, increasing vehicle occupancy and trip chaining (Woodcock et al. 2007). Other actions can reduce travel-related emissions and impacts without changing the modal mix and travel demand, e.g., efficiency increase due to improved engine design, emission controls and renewable energy sources. This includes electric and hybrid vehicles, identified as a major short-term opportunity to improve fleet performance by reducing use-phase local emissions, although the entire life-cycle should be considered to assess the overall energy and resource use (Woodcock et al. 2007).

Banister (2008) identified four types of actions for reducing the environmental impacts of transportation: (1) reducing the need to travel; (2) transport policy measures; (3) land-use policy measures and (4) technological innovation. These strategies can reduce energy demand and environmental impacts and also improve access and equity (Woodcock et al. 2007). Porter et al. (2013) provided an extensive review and discussion on the underlying issues and effectiveness of specific actions to address and change travel behavior, namely focusing on travel reduction and efficient driving strategies (including road and parking pricing, transit improvements, telework, real-time traffic and parking information, speed limit reduction, etc.).

This chapter includes a scenario analysis based on the strategies identified by Banister (2008), to assess their potential improvements in terms of environmental impacts associated with work travel. Work travel (commuting) represents a major portion of urban travel demand, and figures prominently in the literature and urban policies for this reason (Strathman et al. 1994; Maat & Timmermans 2009; Hongwei Dong et al. 2016).

4.2. Materials and methods

4.2.1. Mapping data on land transportation in Lisbon

We examined commuting between a suburban location in metropolitan Lisbon and a workplace in the city center. Like most southern European cities, Lisbon has a mono-centric structure: jobs are concentrated in the central area and residences in primarily peripheral and suburban areas. The selected residence location was in the parish (an administrative subdivision) that had the largest distance from the city center with both subway and train access. This parish (*Venda Nova, Amadora*) has a population of 8400 people and a population density of 7059 habitants/km^2, and 64% of its population commutes to other municipalities, mostly to Lisbon (INE 2012).

Distances for each transport mode (road, overground railway, subway) were measured using a base map of Lisbon from OpenStreetMap© database (OSM 2017), accessibility and transportation data from municipal planning documents (CML 2012), and open-source GIS software QGIS (QGIS 2016). The residence area was defined by the parish boundaries, and the workplace area by a polygon that excluded primarily residential areas (based on the share of exclusively residential buildings, which ranges from 6 to 93% across parishes (INE 2012)). Centroids of both areas were calculated and adjusted to the nearest point or station for each transport network considered, and distances were measured between the adjusted centroids. For example, the nearest rail station to each centroid was considered, and the distance between stations was used to calculate commuting impacts per trip.

Six transportation modes were considered: car, bus, subway, overground train, motorcycle and bicycle. These modes, along with walking (not included due to its low range and lack of environmental impacts), cover 98% of commuters in the greater Lisbon area (INE 2012). Each analysis used a single commuting mode (as compared to mixed modalities) because the Census data provided only the main commuting mode. Commuting distances varied slightly by mode: bicycling, car and motorcycle modes used the road distance (8.16 km); the bus distance was increased by 10% to account for a less direct route (8.98 km); and the rail and subway distances were based on the infrastructure network (8.19 and 8.31 km, respectively). For transport occupancy, bicycle and motorcycle were considered for 1 person, car occupancy was 1.5 based on statistical data (INE, 2012), and bus, subway and train occupancies were 20, 130 and 205, respectively, based on the transport provider (Carris, 2012; Comboios de Portugal, 2012; Metropolitano de Lisboa 2012). Because of their significantly different emissions, diesel and petrol vehicles were disaggregated, as well as 2- and 4-stroke motorcycles.

To aid interpretation and comparability of results, annual commuting impacts were estimated for the total population in the greater Lisbon area, using an average trip distance of 8.5 km. In 2011, the commuting population in greater Lisbon area was 1.224 million, of which 84% used one of the six modes considered (INE 2012).

4.2.2. Life-cycle model

The system boundary was defined considering seven LC phases (Figure 4.1): vehicle manufacturing, vehicle operation (including fuel production), vehicle maintenance, vehicle end-of-life, infrastructure construction, infrastructure maintenance, and infrastructure end-of-life.

The use phase for cars, buses and motorcycles, which dominate the mode mix, as well as the emissions and environmental impacts associated with transportation, was considered in detail. Emissions of vehicles within a mode can vary widely, which can strongly influence results. After reviewing the national vehicle stock, including the technology distribution and mileage split (Ntziachristos et al. 2008), variability was addressed by selecting several technologies within each vehicle category for 2013 (the most recent year available). Table 4.1 lists the vehicle types and technologies modeled of the use phase of road modes, and respective share covered in the fleet.

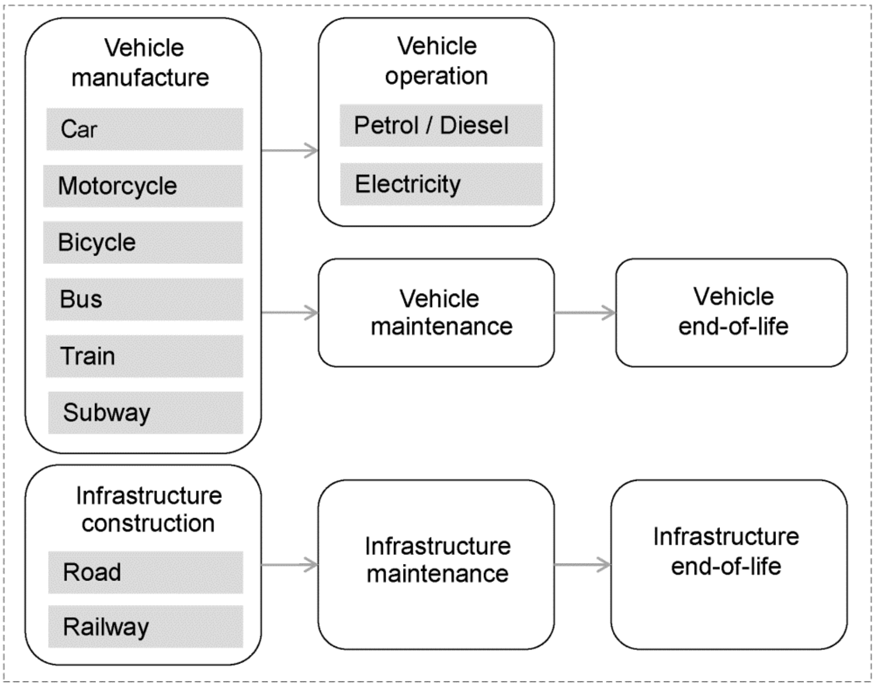

Figure 4.1 System boundary applied to the six transport modes.

Exhaust emissions during the use phase for each vehicle type and technology were calculated using the EMEP/EEA Inventory Guidebook (EEA 2013; Pastramas et al. 2014), which recommends three tiers that depend on the study objective and the level of detail available. Tiers 1 and 2 use simplified models that apply default values for some variables. A tier 3 model was used for exhaust emissions (Gkatzoflias et al. 2010) and non-exhaust emissions (Ntziachristos & Boulter 2009), as implemented in COPERT 4 software, which calculates emissions based on technical and activity data, e.g., the number and mileage of vehicles in the fleet. We express results as an average across the vehicle type/technology, weighted by the 2013 vehicle-kilometer travelled (VKT), together with a threshold that represents the variability within the fleet. An urban driving cycle was selected for buses (25 km/h). Cars and motorcycles used an 80/20% split between urban (25 km/h) and highway (105 km/h) cycles. Exhaust emissions (including hot and cold-starts) were calculated according to each mode's trip length for the Portuguese climate (which affects the fraction of trips driven with a cold engine).

For electric modes (rail, subway, electric car), the average Portuguese electricity mix between 2010 and 2014 was used (Garcia et al. 2014; Marques et al. 2015). This 5-year period helps to stabilize fluctuations experienced in single years. Since the Lisbon subway is mainly underground, tunnels were modeled following Mailbach et al. (1999).

Table 4.1 Road transportation use phase: vehicle types and technologies modeled and share (%) covered in the Portuguese fleet (2013)

Mode	Vehicle type/technology	EURO standards[a]	Share (%)
Car	Gasoline, 0.8 – 1.4 l	1 to 5	80
	Diesel, 1.5 – 2.0 l	1 to 5	80
Bus	Standard urban, diesel, 15 – 18 t	I to V	90
Motorcycle	2-stroke, <50 cc	pre-EURO and I to III	100
	4-stroke, <250 cc, 250 – 750 cc, > 750cc	pre-EURO and I to III	100

[a] The different notation used (Arabic and roman numerals) for passenger cars, buses and two-wheelers follows the legislation nomenclature.

Background data were taken from the *ecoinvent* database (Spielmann et al. 2007). Vehicle manufacture considered European conditions. Most new vehicles in Portugal are imported (ACAP 2013). Road and rail infrastructure requirements were allocated to each transport mode using a static approach that considered the annual use of infrastructure by each mode based on person-kilometer traveled (PKT), space needs and weight of vehicles, as well as fleet characteristics (Spielmann et al. 2007). For bicycling, vehicle manufacturing, maintenance, end-of-life and infrastructure construction were included, but infrastructure maintenance was omitted given bicycles' minimal contribution to road damage (Chester 2008), as were the (negligible) operational requirements. While the energy required by bicycle users is associated with increased food requirements, the assumption that cyclists' food intake rates exceed those of non-cyclists is arguable (Cherry et al. 2009); moreover, an estimate of environmental impacts associated with the possible increase in food intake would be highly uncertain. While building and maintaining separate routes for bicycles could impose significant infrastructure impacts, the present analysis uses a static approach, based on the current situation in Lisbon, which has few dedicated cycling routes. Similarly, infrastructure maintenance was omitted for motorcycles. Transport-related services, e.g., insurance, were not considered.

4.2.3. Life-cycle impact assessment

Table 4.2 lists the environmental categories and indicators considered, which were selected based on the impacts and metrics associated with transportation and recommendations for LCIA (European Commission 2012, 2013; EU 2016a; Litman 2016), and recommendations for life-cycle impact assessment (LCIA; JRC 2011). In brief, primary non-renewable energy (NRE) was based on the Cumulative Energy Demand method (Hischier & Weidema 2010); GHG used the IPCC 2007 method for a 100-year time horizon (IPCC 2007); freshwater eutrophication (FE) and marine eutrophication (ME) were calculated using the EUTREND model, in ReCiPe (Goedkoop et al. 2009); and acidification (AC) and terrestrial eutrophication (TE) used the Accumulated

Exceedance model (Seppälä et al. 2006). The models and characterization factors for GHG, FE, ME, AC and TE (and the use of intake fractions (iFs) for $PM_{2.5}$ described below), follow recommendations for the European context (JRC 2011). While other impacts, such as human and ecological toxicity, might be relevant in comparative LCAs of transportation modes, the recommended characterization methods have significant limitations and large uncertainties; these should be addressed in future research (JRC 2011).

Table 4.2 Life-cycle impact assessment (LCIA) categories and indicators

Impact categories/indicators		Description	Units
NRE	Non-renewable energy	Primary non-renewable fossil energy requirements	MJ_{prim}
GHG	Global warming	Emission of greenhouse gases	kg CO_2 eq
AC	Acidification	Evaluation of land acidifying substances	molc H^+ eq
TE	Terrestrial eutrophication	Evaluation of land eutrophying substances	molc N eq
FE	Freshwater eutrophication	Fraction of nutrients reaching freshwater end compartment (P)	kg P eq
ME	Marine eutrophication	Fraction of nutrients reaching marine end compartment (N)	kg N eq
$PM_{2.5}$ intake	Fine particulate matter intake	Intake of particulate matter <2.5 µm	mg $PM_{2.5}$
$PM_{2.5}$ health	Health effects from human exposure to $PM_{2.5}$	Health effects (cardiopulmonary disease and lung cancer) associated with $PM_{2.5}$ exposure	DALY

$PM_{2.5}$ intake was calculated as the product of emissions and the iF. For ground-level urban emissions, the iF was assumed to be 44 mg PM inhaled per kg PM emitted (44 ppm), a “global” value applicable to a urban outdoor ground-level emission sources (Humbert et al. 2011), which is similar to the 30 ppm estimated by Apte et al. (2012), for ground-level emissions in European cities. Ground-level urban emissions included the operational phase of internal combustion vehicles, diesel equipment emissions during infrastructure construction (rail and road modes), and road maintenance. ($PM_{2.5}$ emissions for infrastructure maintenance were not differentiated in rail modes nor in infrastructure disposal because these stages accounted for less than 3% of $PM_{2.5}$ across the LC.) Other LC stages, including electricity production and vehicle manufacture, used an iF of 2.6 ppm, which represents a global emissions-weighted iF for rural settings (Humbert et al. 2011). Secondary $PM_{2.5}$ used iFs of 0.89, 0.18 and 1.70 for SO_2, NO_x and NH_3 precursor urban emissions, respectively, and 0.79, 0.17 and 1.70 for SO_2, NO_x and NH_3 precursor non-urban emissions (Humbert et al. 2011).

Health effects associated with $PM_{2.5}$ emissions were estimated from the intake using an effect factor (EF) that combines exposure, dose response and severity factors (Gronlund et al. 2015). The EF was set to 78 DALY per kg of $PM_{2.5}$ inhaled; this value includes cardiopulmonary disease and lung cancer (Gronlund et al. 2015). In this analysis, exposure refers to the overall population exposure to emissions associated with the different transportation modes assessed; considerations on the individual exposures of the users (e.g., on-road exposures) are beyond the scope of this research.

The functional unit considered was the commuting of one individual for one year, which accounts for differences in route length between modes. We consider one round trip on 235 days per year (i.e., work year of 255 work days, including 20 vacation days). Results per person-kilometer traveled (PKT) are presented in the supplementary materials (appendix III).

4.2.4. Scenario analysis

A scenario analysis illustrates the application and potential trade-offs of alternative strategies that can reduce the environmental and health impacts associated with transportation and improve access and equity (Banister 2008; Woodcock et al. 2007): reducing the need to travel; transport policy measures; land-use policy measures; and technological innovation. Three modes were selected, as reference base cases: commuting by car (diesel), motorcycle (2-stroke) and bus. For each mode, the annual commuting demand (as calculated in the six mode comparative assessment) was compared with five alternatives: (1) teleworking 1 day per week; (2) increasing occupancy (from 1.5 to 2.0, 1.0 to 1.5 and 20 to 30 people/vehicle for cars, motorcycles and buses, respectively); (3) reducing travel distance by 30%; (4) using alternative energy sources (electric vehicles were considered for the three modes, considering electricity supply mix for Portugal between 2010 and 2014); and (5) shifting mode (from car to bus, motorcycle to subway and bus to bicycle). Impacts, calculated for each scenario and category, were compared to the base case.

4.3. Results

4.3.1. Comparison of commuting modes

Figure 4.2 summarizes results, showing means of the current technology mix as bars and ranges as whiskers (for buses, cars and motorcycles). Cars had the largest impacts for non-renewable energy (NRE), greenhouse gas emissions (GHG), acidification (AC) and freshwater eutrophication (FE); motorcycles and public transit modes (bus, train and subway) had intermediate impacts (12-84% lower than cars for NRE, GHG, AC and FE); and bicycles had the lowest impacts all categories (87-97% lower than cars). The operation phase dominated results, accounting for over 59% of the impacts in all categories except for FE. Vehicle manufacture was significant for cars and motorcycles, while infrastructure construction represented generally larger shares for the public transport modes, especially subways.

NRE and GHG emissions were strongly correlated. For these categories, cars (diesel and petrol) generally had the highest impacts (10680-11330 MJ and 720-795 kg CO_2 eq), followed by motorcycles (5419-7640 MJ and 373-532 kg CO_2 eq), and public transit modes (2704-3993 MJ and 210-266 kg CO_2 eq). As noted, the operation phase dominated results (e.g., 70-77% for subways, 88-90% for 4-stroke motorcycles), but two other LC phases were important for NRE and GHG impacts: vehicle manufacture (5-10% in private modes) and infrastructure construction (5-22% for public transit modes).

For AC, cars had the largest impacts (2.7-3.2 molc H^+ eq), closely followed by motorcycles and buses (1.4-2.0 molc H^+ eq); rail modes had significantly lower impacts (0.7-0.8 molc H^+ eq). Operation accounted for 65-90% of LC impacts, although vehicle manufacture was significant in individual modes (11-21% for cars and motorcycles). For TE, diesel vehicles had significant larger impacts (11.1 and 7.8 molc N eq for diesel car and bus, respectively); petrol cars and motorcycles had intermediate impacts (4.0-5.9 molc N eq) and rail modes were significantly lower (2.4-2.6 molc N eq). Operation dominated the results in all modes, accounting for 70-94% of the LC (except for bicycles). For FE, cars and rail modes had the largest impacts (101-105 and 92-93 g P eq, respectively); motorcycles, buses and bicycles had much lower impacts (13-33 g P eq). The operational phase was only dominant for rail modes (76-84% of the overall LC for train and subway); vehicle manufacture had large contributions for internal combustion modes (29-56%). For ME, diesel vehicles had the largest impacts (1027 and 718 g N eq for car and bus, respectively), followed by gasoline cars and motorcycles (364-534 g N eq) and transit modes (225-243 g N eq).

Diesel cars and 2-stroke motorcycles resulted in the highest $PM_{2.5}$ intake (average 6.1 and 9.3 mg person^{-1} year^{-1}, respectively). Other modes had results below 2.9 mg person^{-1} year^{-1}. Technology and fleet variability strongly influenced results, especially for 2-stroke motorcycles, diesel cars and buses. The operation phase dominated $PM_{2.5}$ intake, accounting for 59 (subway) to 97% (2-stroke motorcycle) of the total. Emissions in other phases had mostly negligible contributions due to their smaller LC share (private modes) and the low iF assumed for emissions in non-urban areas. Health effects (as DALYs) show the same trends as $PM_{2.5}$ intake since a constant effect factor (EF) of 78 DALYs/kg $PM_{inhaled}$ was applied to the intake.

Unsurprisingly, the bicycling commuting mode had the lowest burdens in all categories. Impacts for this mode were dominated by the manufacturing phase (73 to 88% of the impacts); other phases had small impacts given cycling's negligible maintenance and infrastructure requirements.

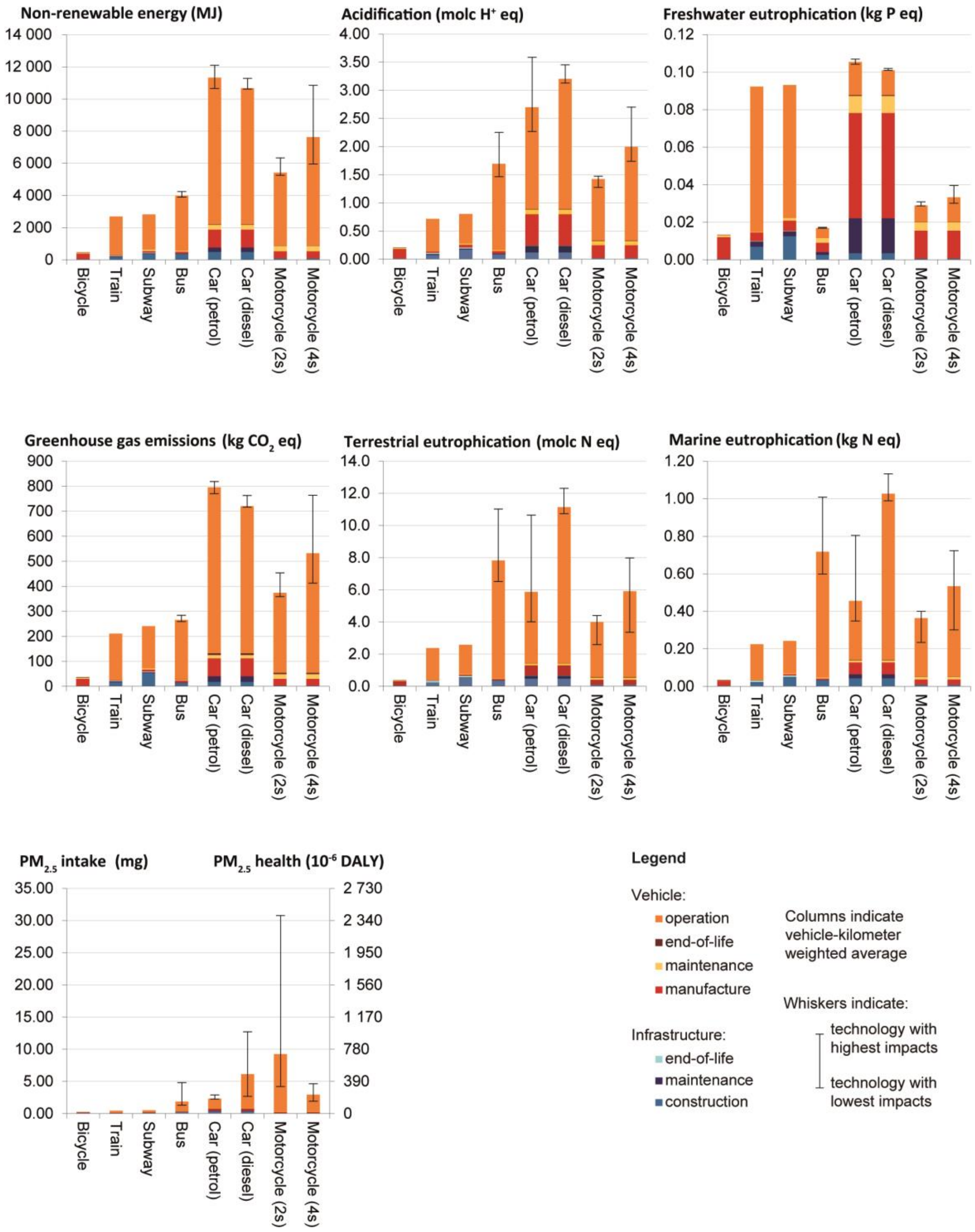

Figure 4.2 Life-cycle impacts of commuting per person along a year. Bars show mean across mix of technologies (2013); whiskers show range across the technologies.

4.3.2. Effect of spatial differentiation

The health impact analysis separated urban and non-urban intake fractions (iFs), which recognizes some of the spatial differences in the LCIA's exposure assessment. Figure 4.3 contrasts differences between spatially differentiated and undifferentiated approaches. With a single iF of 15 ppm for all primary $PM_{2.5}$ emissions, and 0.89, 0.18 and 1.7 ppm for secondary $PM_{2.5}$ from SO_2, NO_x and NH_3, respectively (Humbert et al. 2011), intake and health impacts increase by 105 to 122% for subway, train and bicycle, while impacts of the internal combustion modes drop by 47 to 61% for diesel cars, buses and motorcycles, and by 20% for petrol cars. From the health perspective, using a single or global iF strongly discourages electric-powered transit modes since electricity fuel cycle emissions have the same intake, exposure and health impacts as emissions in urban areas. These results demonstrate the need to distinguish urban and non-urban emissions and health impacts of $PM_{2.5}$.

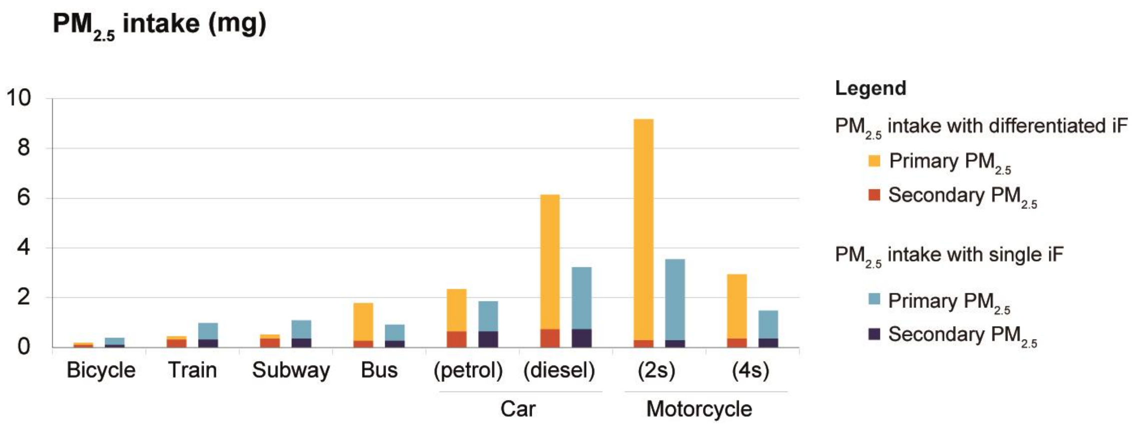

Figure 4.3 Life-cycle $PM_{2.5}$ intake per person-year considering differentiated iFs separating urban and non-urban emissions, and a single (global) iF.

4.3.3. Commuting impacts in Lisbon

Annual commuting impacts in the greater Lisbon area are presented in Table 4.3, for the population using the current mode mix. Cars account for over 84% of the overall commuting impacts, a result of the large share (65%) of commuters using cars and the high impacts associated with this mode. Our estimate of 604 kt CO_2 eq per year for commuting represents only 10% of emissions in the Lisbon area (based on a 2009 estimate of 6.37 million t CO_2 eq that includes transport, buildings, e.g., heating, cooling, lighting and cooking, and commercial and industrial activity; APA 2011). Using a regional input-output model, GHG emissions estimates in four European cities (Malmö, Sofia, Barcelona and Freiburg) ranged between 1.6 and 2.3 t CO_2 per capita (Creutzig et al. 2012). Our estimate of about 0.5 t per capita include only commuting (commercial, freight travel, and non-commutes are excluded). While our estimates for Lisbon seem reasonable, this comparison suggests a need to consider vehicle ownership and use, mode choice and all travel behaviors in transportation impact assessments.

Table 4.3 Total commuting impacts for greater Lisbon and share (%) by mode. Based on 8.5 km per trip, a commuting population of 1.023 million people and the main mode mix (2011). Number of persons using each mode is shown on second row.

	Total 1 022 785 p		Bicycle 1 818 (%)	Train 94 438 (%)	Subway 63 067 (%)	Bus 190 416 (%)	Car 663 942 (%)	Motorcycle 9 104 (%)
NRE	8 773 974	GJ	0.01	3.0	2.1	8.2	86.0	0.7
GHG	604 699	t CO_2 eq	0.01	3.4	2.6	7.9	85.4	0.7
AC	2 538	10^3 molc H^+ eq	0.02	2.8	2.0	12.0	82.5	0.6
TE	8 290	10^3 molc N eq	0.01	2.8	2.0	17.0	77.6	0.6
FE	59 365	kg P eq	0.03	10.1	6.7	3.4	79.4	0.3
ME	743 438	kg N eq	0.01	3.0	2.1	17.4	76.9	0.6
$PM_{2.5}$ intake	3 778	g	0.01	1.2	0.9	8.5	88.0	1.5
$PM_{2.5}$ health	295	DALY	0.01	1.2	0.9	8.5	88.0	1.5

4.3.4. Scenario analysis

Table 4.4 summarizes the potential benefits of five strategies intended to reduce commuting impacts. All strategies were beneficial in each LC category except electric vehicles, which significantly increased FE impacts compared to the reference cases, as seen earlier. The mode shift strategy also increased FE impacts in the motorcycle reference case because the alternative was an electric mode (subway); otherwise, mode shift resulted in some of the largest benefits among the strategies considered.

Strategies that reduced travel demand (teleworking, increased occupancy, and reduced distance) proportionally decreased impacts in all categories (e.g., 20% for teleworking one day per week). In contrast, strategies that shifted modes or used alternative technologies had impacts that varied by category and complex trade-offs between categories, e.g., shifting from motorcycle to subway reduced NRE, GHG, AC, TE and ME by more than 47% (per person-year), but FE impacts almost tripled. For the car reference case, shifting to bus was the most beneficial with respect to NRE, GHG, AC and FE, but other strategies (especially electric vehicles) offered larger reductions in TE, ME, $PM_{2.5}$ intake and DALYs.

Table 4.4 Analysis of alternate scenarios showing environmental impacts of commuting of 1 person for 1 year for (1) car (diesel), (2) 2-stroke motorcycle and (3) bus base cases, and five alternative strategies for reducing environmental impacts. Percent of the base case shown in parentheses. The reference case for cars uses the diesel and petrol weighted average; the reference case for motorcycles uses the 2- and 4-stroke weighted average.

	NRE		GHG		AC		TE		FE		ME		$PM_{2.5}$ intake	
	MJ		kg CO_2 eq		molc H^+ eq		molc N eq		g P eq		g N eq		mg	
(1) Car														
Base case	10 906		746		2.71		8.18		103		827		6.14	
Telecommuting	8 725	(80)	597	(80)	2.17	(80)	6.55	(80)	82	(80)	662	(80)	4.91	(80)
Occupancy increase	8180	(75)	560	(75)	2.03	(75)	6.14	(75)	77	(75)	620	(75)	4.60	(75)
Mode shift	3 993	(37)	266	(36)	1.69	(63)	7.83	(96)	17	(17)	718	(87)	1.89	(31)
Distance reduction	7 635	(70)	522	(70)	1.90	(70)	5.73	(70)	72	(70)	579	(70)	4.30	(70)
Other technology	5 677	(52)	394	(53)	2.11	(78)	4.28	(52)	324	(316)	411	(50)	1.20	(20)
(2) Motorcycle														
Base case	6 641		461		1.74		5.06		31		458		5.78	
Telecommuting	5 312	(80)	369	(80)	1.39	(80)	4.04	(80)	25	(80)	366	(80)	4.63	(80)
Occupancy increase	4 427	(67)	307	(67)	1.16	(67)	3.37	(67)	21	(67)	305	(67)	3.85	(67)
Mode shift	2 827	(43)	240	(52)	0.81	(46)	2.58	(51)	93	(297)	243	(53)	0.48	(8)
Distance reduction	4 648	(70)	323	(70)	1.22	(70)	3.54	(70)	22	(70)	320	(70)	4.05	(70)
Other technology	1 964	(30)	135	(29)	0.87	(50)	1.60	(32)	122	(389)	151	(33)	1.36	(23)
(3) Bus														
Base case	3 993		266		1.69		7.83		17		718		1.89	
Telecommuting	3 194	(80)	213	(80)	1.35	(80)	6.26	(80)	14	(80)	574	(80)	1.51	(80)
Occupancy increase	2 662	(67)	177	(67)	1.13	(67)	5.22	(67)	11	(67)	479	(67)	1.26	(67)
Mode shift	474	(12)	36	(14)	0.20	(12)	0.37	(5)	1	(8)	35	(5)	0.17	(9)
Distance reduction	2 795	(70)	186	(70)	1.19	(70)	5.48	(70)	12	(70)	503	(70)	1.32	(70)
Other technology	3 014	(75)	214	(80)	0.94	(55)	2.60	(33)	130	(766)	245	(34)	0.45	(24)

4.4. Discussion

4.4.1. NRE and GHG

NRE and GHG are the most commonly used impact categories in previous transportation LCAs. Often, the other impact categories are not described, which limits the comparisons that can be made with the literature. For cars, bus and motorcycles, our results were within 13% of values obtained in a comprehensive LCA of urban transportation modes in the USA using 2003 to 2007 data (Chester 2008), but considerably lower (25-63%) for trains, probably due to the low burdens associated with the Portuguese electricity mix, as well as different vehicle characteristics and occupancy assumptions. Our results for diesel buses were significantly lower (55-68%) than the values using a hybrid LC model in the US (Ercan & Tatari 2015); a difference that may in part be due to the lower service life and hybrid model considered in their study. Our results were in line with recent results for cars in Europe (Girardi et al. 2015). Compared to diesel buses in Spain (Sánchez et al. 2013), our results for NRE and GHG emissions associated with diesel buses were 18 and 25% lower, respectively. For bicycles, our energy requirements were 30% lower than a comparison of electric and conventional bicycles (Engelmoer 2012); but our GHG emissions were 30% higher than those estimated in a Japanese study (Shibahara et al. 2013), excluding food requirements. A comparative multi-mode assessment for China that included vehicle manufacture and use, had significantly higher CO_2 emissions associated with internal combustion vehicles (30 to 174% higher than ours), likely due to higher emission rates associated with the fleet, but lower CO_2 emissions for bicycles (34% lower than our GHG estimate; Cherry et al. 2009). Despite the variability associated with vehicles, transport modes and methodological choices (including use stage, infrastructure allocation, and vehicle manufacture), our NRE and GHG results are generally comparable to the recent literature. (Additional inter-study comparisons are provided in the supplementary materials in Appendix III.)

4.4.2. PM emissions and health impacts

Many factors affect PM emissions associated with commuting and transportation in general. For example, PM exhaust emissions depend on driving cycle, fuel properties, engine design, vehicle technology, age and fleet mix; non-exhaust emissions depend on pavement types, silt loading and precipitation frequency; and PM emissions for electricity depend on the fuel cycle and generation mix, e.g., hydro versus coal, as well as the emission control equipment used on fossil fuel power plants. Many of the factors affecting PM emissions vary significantly from place-to-place and over time as technology is adapted and used, which accounts for some of the differences in the literature, as well as the variability demonstrated in Figure 4.2. Our results emphasize $PM_{2.5}$ due to its health significance (Hooftman et al. 2016). In comparison to PM_{10}, a larger share of $PM_{2.5}$ is due to direct exhaust emissions and secondary formation (Putaud et al. 2010).

The few LCA studies in the literature that have examined PM emissions have used different approaches. Cooney et al. (2013) estimated use phase PM_{10} emissions of 0.2 and 0.4 g per VKT for electric and diesel buses, respectively. We obtained similar results for diesel buses (fleet average exhaust emissions of 0.3 g PM_{10}/VKT; range from 0.2 to 0.6 g for EUR V and EUR I, respectively), though our emission factor for electric buses was higher, about 0.3 g PM_{10}/VKT (excluding road construction, maintenance and disposal), probably due to the electricity mix and vehicle characteristics. Ercan & Tatari (2015) estimated life-cycle PM_{10} emissions of 0.7 g per VKT for diesel buses; the higher results are partly associated with a shorter service life (714 500 km compared to our 1 000 000 km) and with the hybrid model used. Chester, Pincetl et al. (2013) compared bus rapid transit (BRT) and light rail transit (LRT) with car trips using a "respiratory stressor" metric in $PM_{2.5}$ equivalents, and estimated: 68 and 24 mg $PM_{2.5}$ eq/PKT for car and bus rapid transit, respectively (after conversion from mg $PM_{2.5}$ eq per person-mile traveled); this study also noted the significance of emissions outside of cities, but characterization factors for emissions in urban and non-urban areas were not differentiated. Our emission factors were lower (32 and 10 mg $PM_{2.5}$/PKT for cars and buses, respectively), possibly because their stressor metric represents an upper limit of impacts that could occur (rather than actual impacts), and because recent controls have lowered emissions of diesel engines. In a comparative LCA for a wide range of car technologies (Bauer et al. 2015), the potential PM formation associated with internal combustion vehicles (diesel and gasoline) was about 35-40% lower than for electric cars, mainly due to emissions from coal power plans in the electricity supply chain. The authors highlight the uncertainty associated with the inventories and characterization factors used for PM formation (and with human toxicity and acidification). Recently, Hooftman et al. (2016) obtained PM emission factors of 0.16-0.26, 0.04-0.05 and 0.025 g PM_{10} eq/VKT, for diesel, petrol and electric cars, respectively, in Belgium; these estimates included exhaust emissions, tire and brake wear, and road abrasion). Our results for diesel and petrol cars were comparable (0.07 and 0.05 g PM_{10}/VKT, respectively), but our estimate for electric cars was much higher (0.1 g PM_{10}/VKT).

We calculated the health impacts associated with $PM_{2.5}$ using archetype iFs and an effect factor in DALYs. A Lisbon-specific iF estimate might differ from the one used, because of population density and meteorological parameters (addressed later, in chapter 5). The effect factor considered included two health outcomes, cardiopulmonary disease and lung cancer, which account for much of the $PM_{2.5}$-associated health burden (Gronlund et al. 2015). This EF was calculated for $PM_{2.5}$ and the North American population. A previous estimate for PM_{10} and Europe (van Zelm et al. 2008) would yield somewhat larger (by 19%) impacts (Gronlund et al. 2015). Including all-cause mortality with an EF of 110 DALY/kg $PM_{2.5}$ inhaled (Gronlund et al.

2015), would increase health impacts by 41%; however, these estimates have greater uncertainty.

In urban settings, $PM_{2.5}$ drives health impact estimates, causing substantially more DALYs than other pollutants (Anenberg et al. 2010; WHO 2016). However, health impacts are also caused by other transport-related air pollutants, including NO_2, O_3 and CO. Thus, the DALY estimates in the present study likely underestimate the true health impact. While the health impacts associated with exposure to multiple pollutants can be quantified, the assessment methods require population-specific information (e.g., health status), have considerable uncertainty, and can produce different results. To date, there is no consensus regarding an adequate approach for use in LCIA (Finnveden et al. 2009; JRC 2011; Pizzol et al. 2011; UNEP/SETAC 2016).

The health impact analysis is focused on potential effects of commuting-related emissions on the overall population. Assessing the exposures and impacts of subgroups, e.g., cyclists and other commuters is beyong our scope. Similarly, the use of an archetype iF for ground-level outdoor emissions in urban settings does not encompass effects of different microenvironments (e.g., tunnels) in $PM_{2.5}$ dispersion and consequent concentrations, neither the exposures occurring in trains or subways. The health impact analysis did not consider noise. Although ISO 14040 recommends its integration into transportation LCAs, no framework yet is recommended for this stressor. Other public health aspects not considered include safety and benefits, such as increased physical activity associated with cycling and walking (Woodcock et al. 2007).

4.4.3. Main strengths and limitations

The present study has important strengths. We present one of the first LCAs that quantified potential health impacts associated with $PM_{2.5}$ emissions from commuting, and the results showed large differences between commuting modes. Second, we perform a comprehensive LCA of a person's commuting activity that compared six alternative transport modes, using a consistent system boundary and a wide set of environmental categories. Third, we demonstrate the importance of applying characterization factors specific to emission sources and their locations, which should improve the accuracy of health impact estimates. Lastly, using scenario analysis, we summarize the impacts and trade-offs of different strategies that can lower environmental and health impacts of commuting. The model and findings are robust and comparable to much of the literature, and the methods can be used for different data sets and contexts. Detailed and case-specific analyses may be required to support decision-making.

A number of assumptions and simplifications were required. We focused on a small set of simplified trips and scenarios that may not be representative of commuting patterns in Lisbon or other cities. Data on travel demand at the city scale would allow more insight regarding the impacts of urban transport and the strategies that might mitigate such impacts. Our analysis focused on single commuting trips. Although recognized to be a considerable simplification of travel behavior, many studies and policies focus on work travel because it is a large share of urban travel, often aggravated by congestion; and multi-purpose trips are complex and uncertain to model (Hongwei Dong et al. 2016). The multi-mode comparison provides environmental impacts per person-year, and results per person-kilometer traveled (PKT) are provided in supplementary materials (appendix III). Additional impact categories and indicators could be considered in future analyses, such as human- and eco-toxicity.

Another limitation is the static nature of the LC approach. Data were selected for a reference period and technological evolution and other changes were not considered. For internal combustion vehicles, technical evolution will increase efficiency and decrease emissions. For electric vehicles, environmental performance is mainly determined by the electricity supply mix and battery production. At present, electric vehicles (EV) penetration in the Portuguese fleet is limited (Garcia et al. 2015), thus, EVs were not considered in the comparative assessment. In part, this exclusion can be justified given the time needed for significant penetration of the market and the difficulty in predicting technologies and behavioral changes, e.g., the possible shift to larger and more powerful vehicles. Lastly, the electricity supply mix in Portugal has already a high share (52%) of renewable sources (about twice the average of the EU; EU 2016a).

4.5. Conclusions

This chapter provided a comparative LCA of six commuting modes that evaluated a wide set of environmental impacts, including $PM_{2.5}$ intake and the associated health effects. It demonstrates the importance of applying spatially differentiated characterization factors to address exposure to traffic-related air pollutants, and it highlights the need for comprehensive approaches to avoid problem shifting in transportation-related strategies. Differentiated $PM_{2.5}$ exposures associated with emissions in urban and non-urban settings, using two iFs resulted in higher impacts for internal combustion vehicles, but lower impacts for electricity-powered vehicles, compared to the use of a single iF for all emissions. This should more accurately represent actual impacts. In addition, we show important trade-offs between modes, e.g., internal combustion personal vehicles had larger impacts than other modes in most categories, but electricity-powered modes (transit and personal vehicles) had generally higher freshwater eutrophication impacts. While technological innovation can significantly lower impacts on a distance or use basis (e.g., PM emissions per kilometer traveled), other strategies are needed to control travel demand and promote lower emission modes (e.g., transit, bicycle and walking). Policies that aim to improve environmental performance of urban transportation systems and air quality within urban areas should consider both local and regional impacts, and potential trade-offs. While the results and conclusions of the Lisbon analysis may apply elsewhere, site-specific and detailed assessments are recommended and practical for supporting policy making to encourage more sustainable urban development.

Supplementary materials

Appendix III presents brief background on the topic, the transport mode mix for Lisbon (2011), additional information on the life-cycle model, namely a map with the residence and workplace locations, detailed results per person-kilometer traveled (PKT) for all modes and impact categories, and inter-study comparison of results.

5 Intake fraction estimates for on-road $PM_{2.5}$ emissions: exploring spatial variation of emissions and population distribution

The intake fraction (iF) expresses population exposure resulting from pollutant emissions. City-wide iFs estimated using simple one-compartment models, used in a number of previous studies, have significant uncertainties and do not capture the intra-urban variation in exposure that is important for estimating health effects associated with traffic-related air pollutants. This chapter presents a novel and efficient approach for developing spatially-resolved iF estimates using dispersion modeling for near-road exposures, which accounts for the spatial and temporal variation in meteorology, emissions and the population spatial distribution. Using the new approach, iF estimates for traffic-related primary fine particulate matter ($PM_{2.5}$) in Lisbon are developed, and results are compared to those from a one-compartment model.[4]

Section 5.1 draws the introduction and 5.2 the background on city-wide one-compartment models and spatially-resolved iF estimates. Section 5.3 describes the materials and methods for calculating spatially-resolved and the one-compartment model iF estimates. Section 5.4 presents the results and discussion, including sensitivity analyses and comparison with literature iF estimates, and section 5.5 summarizes the conclusions.

5.1. Introduction

Exposure to airborne particulate matter (PM) has been associated with severe health impacts, including reduced life expectancy, respiratory and cardiovascular morbidity (e.g., aggravation of asthma, respiratory problems and increased hospital admissions), and cardio-pulmonary and lung cancer mortality (WHO 2003, 2013, 2016; Lim et al. 2012). In Europe, over 90% of urban dwellers are currently exposed to fine particulate matter ($PM_{2.5}$) concentrations above the World Health Organization (WHO) guideline of 10 μg/m^3 (Shneider et al. 2014), and $PM_{2.5}$ has been estimated to cause over 300 000 premature deaths annually (Watkiss et al.

[4] Significant portions of this chapter are from Bastos, J., Milando, C., Freire, F. & Batterman, S. (under review) Intake fraction estimates for on-road fine particulate matter ($PM_{2.5}$) emissions: exploring spatial variation of emissions and population distribution in Lisbon, Portugal.

2005). Road transportation is one of the main $PM_{2.5}$ sources in urban areas, and locations near major roads have been associated with high exposures (Tainio et al. 2014; Karagulian et al. 2015).

The intake fraction (iF) expresses the fraction of a pollutant emitted from one or more sources that is inhaled by a defined population (Bennett et al. 2002; Marshall & Nazaroff 2006; Stevens et al. 2007). iF estimates can be useful in many situations, including comparative analyses of emission reduction strategies and health effects characterizations in life-cycle assessments, as used in the previous chapter (Stevens et al. 2005; Marshall & Nazaroff 2006; JRC 2011; Fantke et al. 2015). City-wide iFs have been estimated for a number of cities (e.g., Stevens et al. 2007; Apte et al. 2012). These iF estimates can have significant uncertainty (Marshall et al. 2003; Stevens et al. 2007; Apte et al. 2012), and they do not incorporate the spatial (or intra-urban) variation that may be important for predicting exposures and health impacts (Marshall et al. 2005; Greco, Wilson, Hanna, et al. 2007; Tainio et al. 2009, 2014). Such variation is especially important for traffic-related air pollutants (Greco, Wilson, Hanna, et al. 2007; Tainio et al. 2014).

This chapter presents a novel approach for determining iFs that account for intra-urban variation of exposures. Spatially-resolved iF estimates are calculated by combining dispersion modeling and geographic information systems (GIS) in an efficient and scalable approach that accounts for the spatial and temporal variation in meteorological conditions, pollutant emissions and population distributions. We present an application for primary $PM_{2.5}$ road emissions for the Lisbon metropolitan area. Results are compared with a one-compartment model iF estimate and the earlier literature.

5.2. Background

5.2.1. iFs for traffic-related air pollution

Several methods have been used to estimate iFs for traffic-related air pollutants, including one-compartment and air quality dispersion models. One-compartment or “box” models provide a simple approach, using just a few parameters to account for key factors; however, these models are highly simplified and accuracy is limited (Marshall et al. 2003; Stevens et al. 2007). Air quality dispersion models, discussed later, provide a more sophisticated approach that can predict spatially- and temporally-resolved concentration estimates; however, such models are complex and require extensive input data (e.g., emissions, meteorology and road configuration), and computational requirements can be large (Stevens et al. 2007).

For example, city-wide iFs for motor-vehicle primary and secondary $PM_{2.5}$ emissions in Mexico City calculated using these methods (including a steady-state box model, a dynamic box model, a regression model that considered populations at different distances from the city center, an inventory based estimate using a source apportionment of monitored $PM_{2.5}$ levels, and a photochemical dispersion model) yielded iFs that varied by a factor of five (23 to 120 ppm; central estimate of 60 ppm; Stevens et al. 2007).

5.2.2. City-scale iFs using one-compartment models

One-compartment models assume that pollutant emissions are dispersed into a single fully-mixed compartment, representing a highly simplified approach with few data requirements. These models can provide some insight on the influence of several variables affecting population exposure to non-reactive air pollutants, specifically, the model domain or area, population, dilution rates (i.e., the product of wind speed and mixing height) and breathing rates (Stevens et al. 2007). Table 5.1 summarizes results and parameters in five studies examining city-scale applications. In an early study exploring sensitivity to a wide set of parameters in archetype environments (Lai et al. 2000), the urban area ranged from 100 to 10 000 km^2 and population from 0.6 to 60 million people, giving dilution rates from 300 to 3000 m^2/s and iFs from 4.4 to 440 ppm. A study focusing on working-age population in Helsinki estimated a low iF of 7 ppm (Loh et al. 2009), largely due to the low population density (663 p/km^2) that resulted from the age group selection (25 - 55 years old, which corresponded to 46% of the metropolitan area population). Other studies have used population densities from 1712 to 8330 p/km^2, dilution rates from 270 to 610 m^2/s and breathing rates from 13 to 20 m^3/d (Stevens et al. 2007; Humbert et al. 2011; Apte et al. 2012). City-specific population and meteorology for Mexico City gave a very high iF of 120 ppm (Stevens et al. 2007). A study covering 3646 cities worldwide gave a population-weighted average iF of 39 ppm (range of 0.6 to 260 ppm; Apte et al. 2012). Small, medium and large cities (populations of 0.1 – 0.6, 0.6 – 3.0, and over 3 million inhabitants) had population-weighted average iFs of 15, 35 and 65 ppm, respectively (Apte et al. 2012). Humbert et al. (2011) separated emission sources by height (ground-level, low- and high-stack), based on literature-derived data, and estimated an average iF of 26 ppm for an archetypal urban area, about half of that for ground level sources (44 ppm), applied in the previous chapter. To better compare the literature results, we calculated the iF on a per person basis, referred to as $iF_{personal}$ (Table 5.1). This statistic shows a tighter range (9 to 22 parts per trillion (ppt)), excluding the parametric sensitivity analysis performed by Lai et al. (2000).

City-wide iF estimates using one-compartment models have several limitations. These estimates strongly depend on the spatial domain selected, e.g., increasing the domain size can dilute emissions into a larger volume, lowering the iF, while restricting the domain to high density land uses will inflate the iF (Lamancusa et al. 2017). Other limitations include the steady-state nature of box models (e.g., constant emissions across the day), and the lack of spatial resolution (Stevens et al. 2007; Apte et al. 2012).

Table 5.1 Summary of iF estimates and associated parameters for urban areas using one-compartment steady-state models from 5 studies. $iF_{personal}$ (last column) facilitates comparison between studies.

	Domain	Area (km^2)	Width (km)	Population (10^3 p)	Population density (p/km^2)	Breathing rate (m^3/d)	Dilution rate (m^2/s)	iF (ppm)	$iF_{personal}$ (ppt)
Stevens et al. (2007)	Mexico city metro area	5022	71	8600	1712	20.0	270	120	13.9
Apte et al. (2012)	3646 cities worldwide	610[a]	24.7[a]	4200[a]	6885[a]	14.5[a]	540[a]	39[a]	9.2[a]
Humbert et al. (2011)	Urban area archetype	240	15.5	1992	8300	13.0	610	44/26[b]	22.1/13.1[b]
Loh et al. (2009)	Helsinki metro area	745	27.3	494[c]	663	19.9	600	7	14.1
Lai et al. (2000)	Urban area archetype	100	10	600	6000	18.7	300-3000	4.4-44	7.3-73.3
Lai et al. (2000)	Urban area archetype	900	30	5400	6000	18.7	300-3000	13-130	2.4-24.1
Lai et al. (2000)	Urban area archetype	10000	100	60000	6000	18.7	300-3000	44-440	0.7-7.3

[a] population weighted mean values
[b] ground level iF/emission-height-weighted iF
[c] working-age population

5.2.3. Spatially-resolved iF estimates

Air quality dispersion models allow spatially- and temporally-resolved predictions of concentrations, which can be used to estimate iFs that vary by region, season and emission source. Table 5.2 summarizes methods and results in five prior studies that used dispersion models to consider spatial variability in iF estimates. In the USA, iFs for mobile source primary and secondary $PM_{2.5}$ emissions were estimated for 3080 counties using the Climatological Regional Dispersion Model (CRDM) (Greco, Wilson, Spengler, et al. 2007). iF estimates ranged from 0.12 to 25 ppm (mean of 1.6 ppm); and 50% of the total intake occurred within a median distance of 150 km for primary $PM_{2.5}$, and within 390 to 740 km for secondary $PM_{2.5}$. The authors concluded that long range dispersion models with coarse spatial resolution can be used to evaluate exposure to traffic-related primary $PM_{2.5}$ emissions in rural or remote areas and for secondary $PM_{2.5}$, but higher resolution is needed for traffic-related primary $PM_{2.5}$ in dense urban areas, as much of the total intake occurs near the source. In Boston, Massachusetts, intra-urban variability (due to population distribution) in iFs for primary $PM_{2.5}$ road emissions was explored using the CAL3QHCR short-range dispersion model for populations up to 5 km from the road (Greco, Wilson, Hanna, et al. 2007). This analysis, which assumed the same emission rates across the 23 398 road segments, gave a mean iF of 12 ppm (hourly values from 0.8 to 53 ppm).

Regional and seasonal variations in iFs for primary and secondary PM emissions across the USA were calculated using source apportionments and the Comprehensive Air quality Model with Extensions (CAMx) regional air quality model (Lamancusa et al. 2017). The analysis considered spatially differentiated emissions and population densities to estimate exposures in 25 regions (10 cities with over 60 000 people, 6 smaller cities with population density over 286 p/km^2, 3 rural regions with lower population density and 3 pristine regions in national parks). Over 75% of the intake for urban emissions occurred within 50 km (mean: 22 km) from the source for primary $PM_{2.5}$.

Table 5.2 Overview of previous iF estimates using spatially-resolved approaches: summary of main model characteristics, methods and results. CRDM – Climatological Regional Disperion Model; CAMx – Comprehensive Air quality Model with Extensions; EXPAND – Exposure to Air Pollution, especially to Nitrogen Dioxide and particulate matter; CAR-FMI - Contaminants in the Air from a Road - Finish Meteorological Institute; iD 75 - distance required for the cumulative iF to each 75% of its total.

Study	Domain	Pollutants	Sources	Methods	iFs/main results
Greco, Wilson, Hanna et al. 2007	Boston, USA	primary $PM_{2.5}$	traffic	CAL3QHCR line-source short-range dispersion model	0.8 to 53 ppm (mean: 12)
Greco, Wilson, Spengler et al. 2007	USA (3080 counties)	primary and secondary $PM_{2.5}$	mobile	CRDM dispersion model	primary $PM_{2.5}$: 0.12 to 25 ppm (mean: 1.6) Secondary $PM_{2.5}$: 0.001 – 10 ppm
Loh et al., 2009	Helsinki, Finland	Benzene	traffic	EXPAND model combined with CAR-FMI Gaussian finite line source model	mean iF 10 ppm
Tainio et al., 2014	Warsaw, Poland	7 pollutants, including PM	3 066 sources; 14 categories: mobile, area, high point and other point	CALPUFF dispersion model	mean iF PM from mobile sources: 51 ppm; intraurban variability: 4 to 100 ppm
Lamancusa et al., 2017	USA	primary and secondary PM	25 source regions (including 10 cities)	CAMx regional air quality model	iD75 in most cities was <50 km (except sulfates); primary $PM_{2.5}$ from urban areas: mean iD75 = 22 km, mean iF = 26 ppm)

Two recent studies in Europe developed spatially-resolved iFs. In Helsinki, an approach combining dispersion modeling and spatial and temporal information on population activity patterns for a working-age population (494 000 people) using 100 x 100 m cells highlighted the dependence of iFs on population density (Loh et al. 2009). The overall iF was 10 ppm, between a 7 ppm box model estimate and a 39 ppm estimate based on personal monitoring data (n=129 persons). In Warsaw, iFs for 3066 emission sources in various source categories, e.g., mobile, area and point, were estimated using CALPUFF and local meteorological data (Tainio et al. 2014). Mean emission-weighted iFs ranged from 0.013 to 51 ppm across the source categories. For primary $PM_{2.5}$ emissions from mobile sources, the emission-weighted average iF was 44 ppm (range from 4 to 100 ppm).

Both the iF studies and the broader literature show that iF estimates for traffic-related primary $PM_{2.5}$ emissions are substantially higher in urban areas, and that a significant share of exposure occurs in the near-road environment (Zhang & Batterman 2010; Greco, Wilson, Spengler, et al. 2007; Lamancusa et al. 2017). Consequently, city-wide iF estimates (and dispersion-based models with coarse resolution) do not represent the spatial variation in exposure to traffic-related air pollutants, which is governed by population density, the distribution of emissions and meteorological conditions (Marshall & Nazaroff 2006; Greco, Wilson, Hanna, et al. 2007; Zhang & Batterman 2010). These factors may require site-specific analyses using dispersion or other types of spatially-resolved models to produce the concentration field, follower by the use of GIS to address spatial variations in population density and other factors affecting exposure (Vienneau et al. 2009; Reyna et al. 2015). While dispersion models can support health impact estimates, their application has been limited due to their complexity, time and resource requirements (Milando et al. 2016).

5.3. Materials and methods

Spatially-resolved iF estimates were derived by combining several tools and datasets and modeling emissions, concentrations and the resulting exposures. In brief, traffic activity, fleet data and emission factor models were used to calculate road emissions; road configuration and local meteorological data were used in dispersion modeling to predict concentrations in specific buffers along roads; and demographic and activity data were used to estimate population intake. The iF at hour *t* is:

$$iF_t = \sum_{i=1}^{n} \frac{P_{i,t} \times Q_{i,t} \times C_{i,t}}{E_{i,t}} \tag{5.1}$$

where $P_{i,t}$ = population, $Q_{i,t}$ = breathing rate per person (m^3/h), $C_{i,t}$ = mean hourly ambient concentration ($\mu g/m^3$), and $E_{i,t}$ = road emissions ($\mu g/h$) for hour *t* and location *i.* To account for the temporal variation in emissions (traffic activity), meteorological conditions and population activity patterns, four periods over the day were considered (INE 2011; Brito 2012): (1) night-time (8 pm to 6 am, 10 hours), (2) morning commute (6 to 10 am, 4 hours), (3) day-time (10 am to 4 pm, 6 hours) and (4) evening commute (4 to 8 pm, 4 hours). The population, breathing rate, emissions and concentrations vary for these periods, as described next.

5.3.1. Study site

The Lisbon metropolitan area (38° 24' 32'' - 39° 03' 52'' N, 08° 29' 27'' - 09° 30' 01'' W) is located on the western coast of Portugal. In 2011, it included 18 municipalities, 211 parishes and 34 937 statistical subsections (here referred to as "census blocks"), with a total population of 2 821 876 and an area of 3 002 km^2 (940 p/km^2); Lisbon city had 547 733 people and 85 km^2 (6 448 p/km^2) (INE 2012). Elevations across the metropolitan area range from 0 to 528 m, and 0 to 228 m in Lisbon city (INE 2012). Winds occur mostly from the north and northwest, in particular during the *nortada*, which occurs on 45% of summer and spring days, with wind speeds above 5 m/s (Lopes et al. 2013). Meteorological data (NOAA 2012, 2016) was obtained for the Lisboa/Gago Coutinho station (coordinates 38.7667, -9.1333; station reference POM0008579), for 2001-2002, which is at the Lisbon airport, near the city center. This period was selected based on completeness and consistency of observations. Census data used the geographic coordinate system GCS ETRS 1989 and the projected coordinate system ETRS 1989 TM06 Portugal (INE, 2012). These coordinate systems were maintained for all map data in the study (data using other systems was converted).

5.3.2. Spatially-resolved iF estimates

Road network and buffers

The road network for Lisbon metropolitan area included highways and main roads with publically available traffic data (IMTT 2016; TIS-CML 2015; APA 2015). The spatial configuration of the road network was based on OpenStreetMap© road data (OSM 2017). A total of 601 km of road length, broken down into 181 road segments were represented in GIS (see supplementary materials in appendix IV). To calculate near-road exposures, four buffers were defined on each side of the road's centerline at distances of 10 to 50, 50 to 100, 100 to 200 and 200 to 500 m. At road crossings, buffers were intersected to account for emissions from the two roads. For areas within 500 m of three or more major roads, only two roads were considered; this applied to few areas.

Road emissions

Road emissions were estimated by combining daily mean traffic volumes, disaggregated by road segment, with temporal traffic activity profiles and fleet emission factors. Table 5.3 summarizes the fleet composition, annual mileage and emissions, by the main vehicle groups. Daily mean traffic volumes were obtained from public reports (IMTT 2016; TIS-CML 2015; APA 2015). Simplified but separate hourly temporal profiles of passenger and commercial traffic activity were developed (Figure 5.1), based on the daily variation of traffic flows in Lisbon (Brito

2012), resembling profiles developed elsewhere (Batterman 2015; Batterman et al. 2015; Roh et al. 2016). Based on annual km-traveled, passenger vehicles accounted for 73% of the traffic; commercial vehicles accounted for 27% (Ntziachristos et al. 2008). Temporal profiles considered only weekdays, because the day-time population distribution was based on commuting patterns (as described below). Weekend patterns would differ, e.g., morning commuting period is absent, and most traffic occurs in the afternoon and evening peiods (see supplementary materials in appendix IV).

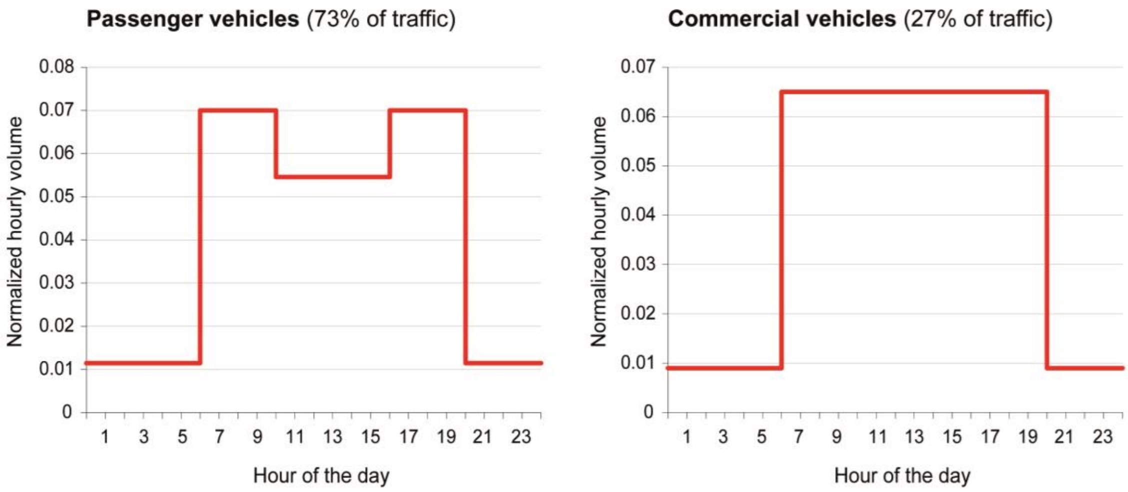

Figure 5.1 Temporal profile of traffic volumes for passenger and commercial vehicles.

Emission factors for the 2013 Portuguese fleet (Ntziachristos et al. 2008) were estimated using the EMEP/EEA Inventory guidebook (EEA 2016), which provides models for exhaust emissions, gasoline evaporation and road tire and brake wear. We used the tier 3 model implemented in COPERT 5 software (Ntziachristos et al. 2009) to calculate exhaust and non-exhaust emissions based on technical and fleet activity data, e.g., vehicle technologies and mileage split. (Tiers 1 and 2 use simplified models that apply default values for many variables.) The meteorological and speed data used are provided in the supplementary materials in appendix IV.

Table 5.3 Summary of the national fleet composition and annual mileage (2013; Ntziachristos et al. 2008), and emission factors (modeled).

	Stock (10^3 vehicles)	Fleet annual mileage (10^6 km)	Emission factors (mg/km)	
			National road	Highway
Passenger vehicles	5 192	56 512	24	22
Passenger cars	4 675	54 553	23	21
Mopeds and motorcycles	502	1 542	29	28
Buses	15	417	145	110
Commercial vehicles	1 354	20 818	50	64
Light commercial vehicles	1 219	17 511	43	62
Heavy duty trucks	135	3 307	90	75

Near-road exposure concentrations

Ambient concentrations from traffic-related emissions were estimated in each buffer using an efficient dispersion modeling approach. First, we determined the mean orientation of each road segment using GIS, and then each segment was classified into one of four directions (north – south, N-S; east – west, E-W; southwest - northeast, SW-NE; and southeast – northwest, SE – SW). Then, the Research LINE-source model (RLINE), a line source model specifically developed for traffic emissions and near-road environment (Snyder et al. 2013), was used to calculate emission-to-concentration (or "transfer") coefficients for each road direction and distances up to 500 m from the road, using a line of receptors placed perpendicularly to the road segment at 5 m intervals (15 to 500 m from the road) on both sides (196 receptors). The road segment was modeled as a 3 km long linear source with unit emissions (1 g m^{-1} s^{-1}). The receptors height was 2.5 m. Concentrations were predicted using RLINE (numerical option for calculation) and local (2001-2002) hourly surface and upper air meteorological data (NOAA 2012, 2016) processed by the AERMET surface meteorological processor (Cimorelli et al. 2004, 2005). A total of 14 954 hours of Lisbon meteorological data was modeled, representing about 85% of the hours in the 2-year period. (By day period, data completeness ranged from 78 to 93%.) RLINE predictions were then averaged within each of four buffers (10 to 50, 50 to 100, 100 to 200 and 200 to 500 m), and divided by the emission rate, providing the hourly transfer coefficients. The coefficients were averaged by day period, segment direction and buffer distance, resulting in 128 transfer coefficients (4 road directions × 4 buffers × 2 sides of road × 4 day periods). Lastly, concentrations were determined as the product of the road segment emission rates and the transfer coefficients at the desired location, considering the distance and direction from the road.

This approach estimates concentrations for areas within 500 m of major roads using only four archetype segments and geographic data. The approach is computationally efficient, an important consideration given the number of road segments, receptors and hours needed to develop long-term estimates in urban areas. Sensitivity analyses were performed to evaluate how the buffer size, the road orientation estimate, and the assumed segment length influenced results.

Background concentration

The RLINE dispersion modeling accounted for emissions on the larger roads and predicted concentrations to a distance of 500 m from the road. To account for emissions roads beyond 500 m, a one-compartment model was used with the assumption that the “far-field” environment is well-mixed. This provided an hourly “background” concentration for the metropolitan area, calculated as:

$$C_t = \frac{E_t}{V_t} \tag{5.2}$$

where C_t = mean hourly concentration (µg/m^3) at hour t, E_t = total road emissions (µg/h) and V_t = ventilation rate (m^3/h) for hour *t* (see section 5.3.3 for further details on the ventilation rate). Hourly results were averaged for the four day periods into a 24-hour mean. The background concentration estimate, assumed to be uniform across the modeled domain, “double counted” contributions from the 3 km segment used to model the near-road concentration at each buffer, however, this contribution was negligible, less than 1% of the background estimate. (Mean emission rates of the 3 km segment ranged from 50 to 338 g/h for night and commuting periods, respectively, while the network total emission rates ranged from 7 700 to 47 900 g/h.) Double counting could be eliminated using site-specific background estimates that removed the emissions from the 3 km segments modeled using RLINE.

Population distribution

Block-level demographic data was used to map the population distribution into “cells”, which are defined as the intersection of census blocks and buffers described. The population in each cell was calculated for day- and night-time periods to account for both workers (many of whom work in the urban core) and residents (who tend to be more dispersed in suburban areas). The overall population of most of the 18 Lisbon area municipalities decreases during the day (up to 28% depending on municipality), while the Lisbon city population increases (by 56%; INE 2012). Census data were assumed to represent the night-time distribution of residents, and day-time population distribution was based on origin and destination of commuting trips at the municipality level (INE 2012). Because information describing population shifts at finer resolution was unavailable, a relatively simple mapping procedure was used to allocate the day-time population to census blocks. The day-time population was assumed to include the share of residents remaining in the block (those who did not work or study and those who worked or studied at home), and workers and students in the municipality. The population working or studying in each municipality was allocated to blocks based on the number of buildings of various types in the block weighted by a factor intended to reflect the number of workers: exclusively residential buildings, mainly residential buildings, and mainly non-residential buildings were assigned weights of 1, 3 and 6, respectively (INE 2012). The population density was assumed to be uniform within each census block after excluding a 10 m buffer along the road axis.

Breathing rates

Age- and gender-specific breathing rates were estimated for day- and night-time periods using the metropolitan Lisbon demographic data (Table 5.4; US EPA 2011, 2009b). Day-time rates, which averaged 0.80 m^3/h (moderate activity), were assumed for daytime and both commuting periods (6 am to 8 pm); the night-time breathing rate was 0.34 m^3/h (passive activity).

Table 5.4 Population age and gender distribution in Lisbon metropolitan area and breathing rates (based on INE 2012; US EPA 2011, 2009b)

		Breathing rates (m^3/d)		
Group	Population	Night-time (10 h)	Day-time[a] (14 h)	24-hour mean
Children				
0-14	437 881	6.38	15.26	11.56
Men				
15-24	148 856	9.56	23.04	17.42
25-64	748 913	9.64	22.91	17.38
>65	213 260	9.02	19.12	14.91
Women				
15-24	146 187	7.96	19.13	14.48
25-64	826 197	7.55	18.64	14.02
>65	300 582	6.75	14.71	11.39
Total	2 821 876	8.08	19.12	14.52

[a] includes commuting periods

5.3.3. City-wide iF estimate: one-compartment model

For comparison with the spatially-resolved estimate, a city-scale single iF was calculated using a one-compartment model:

$$iF = \frac{P \times Q_b}{V} \tag{5.3}$$

where P = population, Q_b = breathing rate per person (m^3/s), and V = ventilation rate (m^3/s). The population was based on census data (INE 2012), and the breathing rate used the average shown in Table 5.4 (14.5 m^3/d). The ventilation rate V (m^3/s) was calculated as:

$$V = \mu \times H \times W \tag{5.4}$$

where μ = mean wind speed (m/s), H = mixing height (m), and W = width in crosswind direction (m). Hourly mixing heights for the mechanically generated boundary layer were calculated using AERMOD (Cimorelli et al. 2004, 2005) from twice daily radiosonde observations for 2001 and 2002 (NOAA 2016). Hourly surface observations of wind speed for the same period (NOAA 2012) were extrapolated to the hourly mixing height using a power law coefficient of 0.32 and a cut-off height of 200 m, above which the wind speed was assumed constant (Apte et al. 2012). A total of 16 026 hours had valid data or about 90% of the hours in the 2-year period (89 to 95% by day period). Surface wind speeds below 0.5 m/s and mixing heights below 50 m were excluded. The wind speed at 10 m height averaged 3.6 m/s (range: 0.5 to 13 m/s), and the mixing height averaged 1317 m (range: 61 to 4000 m). The dilution rate was calculated as the harmonic mean of the hourly product of μ and H (Stevens et al. 2007; Apte et al. 2012). Width W was considered to be 55 km, the square root of the area, assuming a square domain. iF estimates were calculated with average daily parameters, and also using emissions and ventilation parameters specific to the four time periods used in the spatially-resolved approach.

5.4. Results and discussion

5.4.1. Spatially-resolved iF estimates: dispersion-based model

Table 5.5 summarizes predicted concentrations and iF estimates for the population living or working within 500 m from roads. 24-hour mean $PM_{2.5}$ concentrations averaged 0.65 µg/m^3 across the 33 230 cells (range: 0.36 µg/m^3 at night to 1.76 µg/m^3 during the morning commuting period); cells near large roads with high traffic volumes had much higher concentrations. Concentrations at night and during day-time periods were much lower than those during commuting periods (Figure 5.2), reflecting the patterns of emission and ventilation rates. Emission rates are low at night (7.7 kg/h for the modeled road network) and increase during commuting periods (47.9 kg/h), while mid-day emissions are just slightly lower (41.6 kg/h). However, dispersion increases considerably during the daytime periods, which tends to lower concentrations. This diurnal pattern has the effect of lowering iF estimates compared to the use of constant emissions. The variation depends on both the daily emission profile and prevailing meteorological conditions, e.g., areas with frequent and strong nocturnal inversions during the morning will show greater differences between morning and evening commuting periods, and areas with shorter and earlier evening commuting periods may have yet lower concentrations in the evening.

Table 5.5 Summary of spatially-resolved analysis showing cell size, population, predicted concentrations and iFsfor cells within 500 m of major roads. Based on 33 230 cells. Concentrations and intake rates include RLINE and background estimates.

	Units	Mean	St.dev.	Min	P10	P90	Max
Cell area	m^2	14 816	51 237	0	199	29 934	2 629 723
Night population density	p/km^2	7 681	11 704	0	0	25 156	113 804
Day population density	p/km^2	6 586	10 109	0	0	18 906	168 464
Night population	p	32	64	0	0	94	865
Day population	p	27	51	0	0	75	901
Mean hourly concentration	$\mu g/m^3$						
Night		0.363	0.332	0.066	0.104	0.767	3.680
Morning commuting		1.756	1.696	0.331	0.493	3.740	19.068
Day		0.589	0.963	0.045	0.063	1.534	12.121
Evening commuting		0.972	1.250	0.100	0.165	2.344	15.090
24-hour mean intake	µg/d	257	636	0	0	655	15 715
iF	ppt	362	895	0	0	922	22 121
$iF_{personal}$	ppt	18	22	1	4	41	325

Figure 5.3 demonstrates the development of iF estimates for a small section of Lisbon containing several road segments. RLINE concentrations in the near-road buffers (panel *a*) are applied to the number of individuals in the cells (panel *b*, average of day and night-time populations by cell), giving $PM_{2.5}$ intake for the near-road population (panel *c*). This variation, which is fairly typical, shows the dependence of intake on population distributions, especially in the near-road environment. While concentrations along roads have fairly homogeneous patterns with some nuances (e.g., concentrations may be higher on one side of a road due to prevailing winds), the population distribution can be highly heterogeneous. Since intake is the product of concentrations and population (neglecting the variation in breathing rates and other factors), intake tends to be very heterogeneous. Thus, cells with high population density can have high

intake, even in buffers distant from the road or near low traffic roads. Such results show the importance of local data, in particular, the size of the population near busy roads.

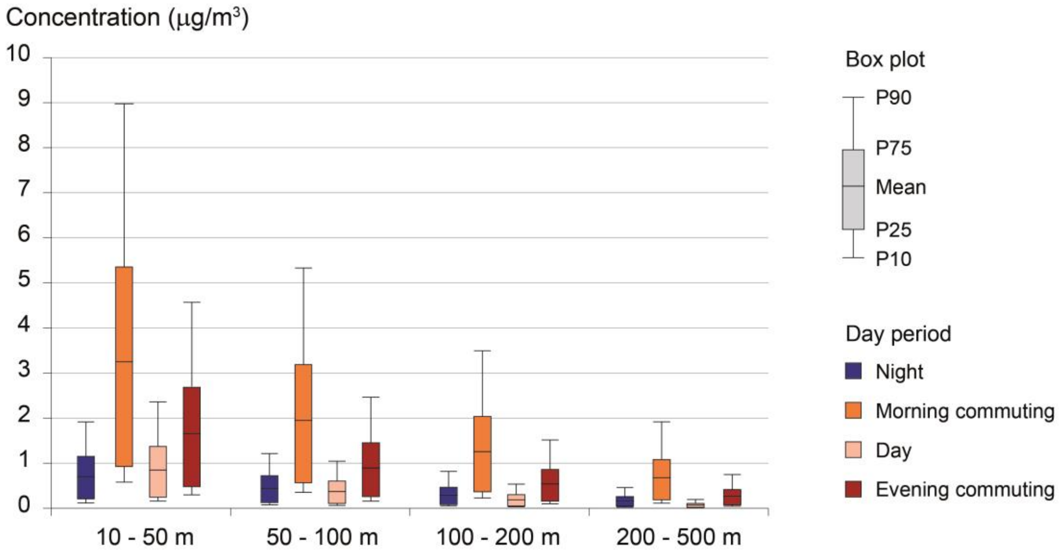

Figure 5.2 Near-road mean $PM_{2.5}$ concentrations for four buffers and four day periods. Plots show mean, 10^{th}, 25^{th} 75^{th} and 90^{th} percentile concentrations.

Table 5.5 depicts the population living or working near roads. In Lisbon, about 5% were within 50 m, 8% within 50 to 100 m, 21% within 100 to 200 m and 65% within 200 to 500 m. These buffers corresponded to 9, 11, 21 and 58% of the total buffer area, respectively (491 km^2, equal to 16% of the Lisbon metropolitan area). About 11% of cells were vacant, resulting in no intake (i.e., iF = 0). Diurnal population shifts decreased the iF by about 13% compared to the use of residence (census) locations alone, reflecting that about 1.047 million people live within 500 m of major roads, but only an estimated 0.893 million are within 500 m during the day.

The overall iF estimate for the metropolitan Lisbon is 16.4 ppm, which includes the near-road exposure iF contribution of 12.0 ppm (using night- and day-time populations of 1.05 and 0.89 million, respectively), and the far-field exposure iF contribution of 4.3 ppm (population of 1.77 and 1.93 million beyond 500 m of the modeled road network in night- and day-time, respectively). Thus, most (74%) intake occurs within 500 m of major roads. This near-road region represents 16% of the study area and contains 31 to 38% of the population.

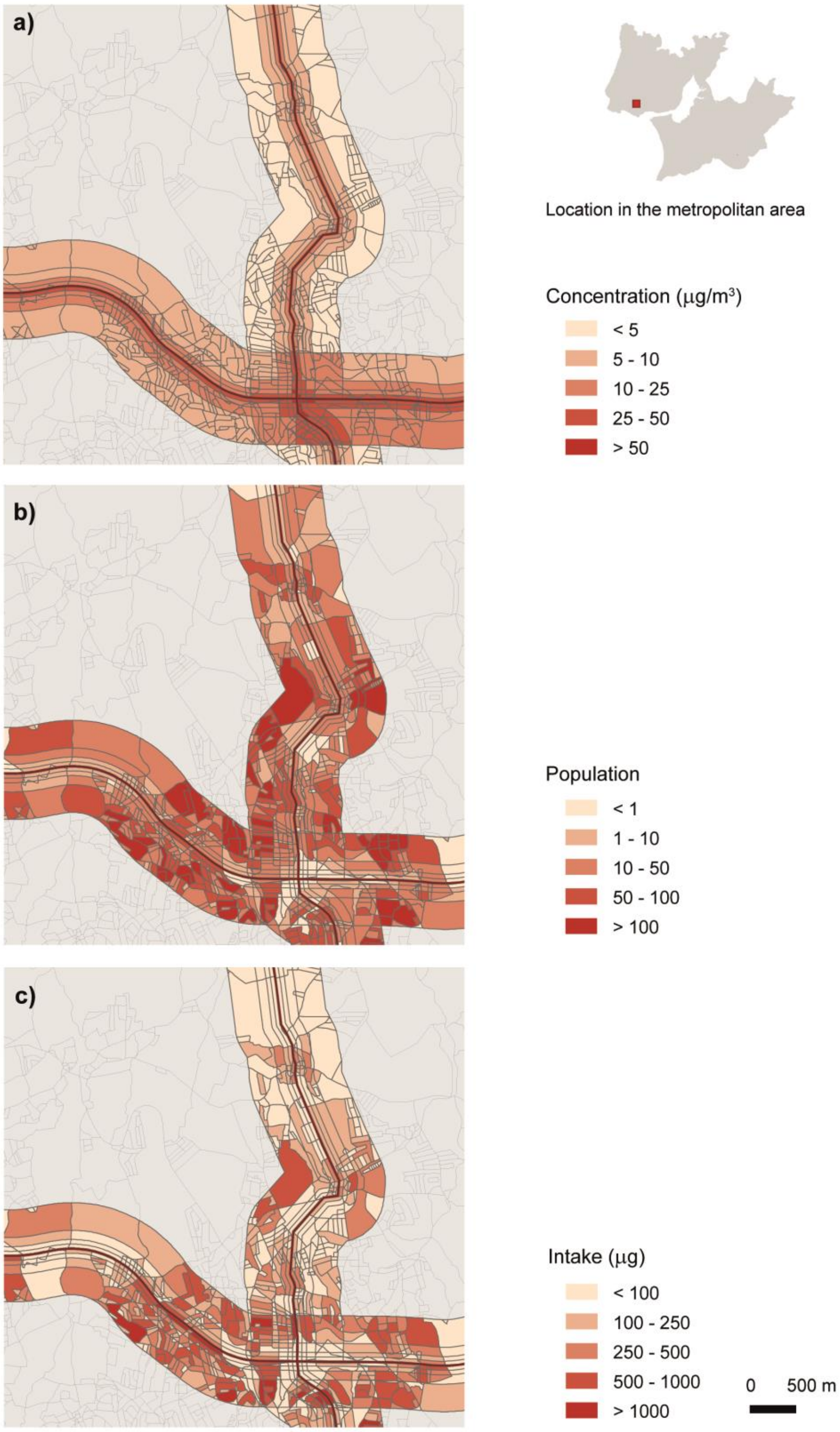

Figure 5.3 Example of development of spatially-resolved iFs in Lisbon metropolitan area, showing 24-hour mean (a) ambient concentrations, (b) near-road population and (c) intake. Includes near-road and background concentrations.

5.4.2. City-scale iF estimate: one-compartment model

The iFs calculated using the one-compartment models are shown in Table 5.6. With the steady-state model, the iF estimates for Lisbon (9.3 ppm) and the larger metropolitan area (8.1 ppm) are very similar, although on a per person basis there is a 6-fold difference. Compared to Lisbon city, the ventilation rate is about 6 times higher across the metropolitan Lisbon area (corresponding to the width difference), but the population is 5 times greater. These factors offset each other in the iF calculation. These estimates do not incorporate the temporal varion in the emission profiles.

The one-compartment model with hourly variation in the dilution rates and emissions provided an iF estimate of 6.5 ppm, slightly lower than the steady-state one-compartment model of 8.1 ppm. (The former estimate used the same breathing rates, emission and dilution rates, disaggregated by four day periods, as in the spatially-resolved iF estimates.) At night, breathing rates and emissions are low, which tends to offset the lower night-time dilution rates. During the day, emissions increase considerably (together with breathing rates), however, dilution rates also increase, due to faster winds and greater mixing height.

Table 5.6 Parameters and results for the one-compartment iF steady-state and emission-weighted models (with hourly variation of dilution rates and emissions)

Domain	Area	Width	Population	Population density	Breathing rate	Dilution rate	iF	iF personal
	(km^2)	(km)	(million p)	(p/km^2)	(m^3/d)	(m^2/s)	(ppm)	(ppt)
Steady-state model								
Lisbon metropolitan area	3 002	55	2.82	940	16.0	1 069	8.1	2.9
Lisbon city	85	9	0.55	6444	16.0	1 069	9.3	17.1
Temporally-disaggregated model								
Lisbon metropolitan area								
Night-time	3 002	55	2.82	940	8.1	672	7.2	2.5
Morning commuting	3 002	55	2.82	940	19.1	808	14.1	5.0
Day-time	3 002	55	2.82	940	19.1	4 969	2.3	0.8
Evening commuting	3 002	55	2.82	940	19.1	2 763	4.1	1.5
24-hour mean	3 002	55	2.82	940	14.5	2 118	6.5	2.3

5.4.3. Comparison of iF estimates

The one-compartment estimates for metropolitan Lisbon (steady-state model: 8.1 ppm, temporally-disaggregated model: 6.5 ppm) are less than half that estimated using the spatially-resolved approach (16.4 ppm), and only slightly more than the 4.3 ppm estimate for the far-field population (1.86 million people, 66% of the overall population). Near-road exposure accounted for 74% of the spatially-resolved iF (population of 0.96 million living within 500 m from major roads, 34% of the overall population, 16% of the domain area) and far-field exposure accounted for 26% (population of 1.86 million living beyond 500 m of the modeled road network).

The iF values for Lisbon are at the low end of the literature range for urban areas (section 5.2), a result of Lisbon's relatively low population density and high dilution rate. In earlier work examining many cities, Apte et al. (2012) estimated an iF of 13.5 ppm for Lisbon (the study published mean results for cities aggregated by world region; data for Lisbon was obtained by private communication), which applied to a population of 1.9 million people in a 222 km^2, giving a population density of 8 748 person/km^2. In contrast, we considered the entire metropolitan area with a population of 2.8 million and an area of 3 002 km^2, giving a population density of only 940 (Table 5.6). On a per person basis, our $iF_{personal}$ estimate is 2.9 ppt, less than half of the earlier estimate (7.0 ppt).

As noted earlier, iFs based on one-compartment models are sensitive to the domain selected. The domain should correspond to the scope and goal of the study. In this study, iF estimates considered exposure to traffic-related air pollutants across the Lisbon metropolitan area, and administrative borders were used to define the domain. In contrast, Apte et al. (2012) based the land area and city population on a dataset derived from satellite mapping of built land cover, and obtained a population density that exceeded those in any of the 18 municipalities that comprise metropolitan Lisbon (139 p/km^2 in *Alcochete* to 7 389 p/km^2 in *Amadora*; INE 2012). Satellite-based estimates differ significantly from administrative boundaries, e.g., parks, forested and water areas may be excluded within an urban area (Schneider et al. 2009; Angel et al. 2010). Excluding such areas, which increases the population density with the effect of increasing the iF, may be appropriate for traffic-related emissions since exposure and intake occurs near the source. However, results can depend on the scale and urban form, and the similarity between our dispersion modeling-based iF and Apte et al. (2012) results may be fortuitous. Two urban settlements with the same area and population but different urban forms (e.g., road configuration and population distribution) could have significantly different iFs when spatially-resolved, but very similar iF estimates using a one-compartment model.

The high dilution rate in Lisbon is reflected in the one-compartment model iF estimate. Mixing heights in Lisbon generally exceed values used in one-compartment modeling (Table 5.1). Our 2001-2002 data shows mean and median mixing heights of 1317 and 1144, respectively, which are consistent with previous Lisbon 1999-2000 monthly-averaged midday mixing height estimates (843 to 1465 m), based on radiosonde data (Baklanov et al. 2005), and also with the mean height (1219 m) of the first inversion layer derived from twice daily radiosonde observations for Lisbon in 2001-2002 (NOAA 2016). Our dilution rate (harmonic mean: 1069 m^2/s) is also comparable to the 1130 m^2/s calculated by Apte et al. (2012) for Lisbon using the NASA MERRA database (Rienecker et al. 2011).

5.4.4. Sensitivity analyses

Several sensitivity analyses were completed for key modeling parameters. First, analyses using four road orientations (range ±22.50º), presented previously, were compared with the use of eight road orientations (range ±11.25º). The latter modeling increased 24-h mean concentrations by an average of 3 to 6% across the four day periods, and increased the overall near-road iF by 11% (12.0 to 13.4 ppm). This suggests that RLINE modeling may require additional road orientations. The effect of the number of orientations will depend on the road configuration and meteorological variables (especially wind direction), but handling additional orientations in the modeling framework is not difficult.

Second, to examine the effect of buffer size, we modeled an east-west road in western Lisbon (highway A5; 25.6 km, 11 segments), and compared iF estimates for the four buffers used previously (cut-offs at 50, 100, 200 and 500 m) with both two larger buffers (200 and 500 m), and eight smaller buffers (25, 50, 75, 100, 150, 200, 350 and 500 m). These three sets of buffers used 1 808, 1 340 and 2 875 cells, respectively. Compared to the 4-buffer case, the 2-buffer model increased the iF by 17% while the 8-buffer case decreased the iF by 25%. While results might differ from other roads and areas for Lisbon, these demonstrate the sensitivity to buffer size and the need for highly-resolved analyses near major roads. While a larger number of buffers is potentially more accurate, we do not necessarily know where individuals reside or work with high precision. Further investigation on buffer size is warranted.

A third sensitivity analysis examined how the segment road length used by RLINE affected results. Compared to the 3 km segment used, a 1 km segment decreased mean concentrations by 16 to 17% (across 33 230 cells and for four day periods), and the overall near-road iF (including background concentration) decreased by 20%. With a 10 km segment, mean concentrations increased by 6-7% and the iF increased by 8%. The 3 km segment length represents a compromise with respect to modeling actual geometry that captures the bulk of the near-road impacts.

These analyses suggest that potential errors due to discretizing roads by direction and using a standard length can cause errors on the order of 10%, and that utilizing a limited number of buffers may cause errors on the order of 20%. Due to data gaps and the computational challenge, we did not model the actual road network or use a highly resolved receptor grid to calculate iFs, which might serve as a reference case. The sensitivity analyses help to identify possible error sources and support our major conclusion, that is, the importance of near-road exposures.

5.4.5. Computational considerations

The RLINE dispersion modeling for Lisbon required four 2-year runs for each road direction and the 196 receptors; these runs were completed within 10 hours on a PC workstation. In contrast, modeling metropolitan area with the 601 km road network would require 1.2 million receptors using a 50 m grid or 192 000 receptors for the near-road (within 500 m) area, presenting an enormous computation burden.

5.4.6. Main strengths and limitations

This chapter presents an innovative and efficient approach for estimating iFs that account for near-road exposure to traffic-related air pollutants using local and spatially-resolved emission and demographic data. The approach represents the small-scale spatial variation in concentrations and population density, which is important for intake and impact estimates for traffic-related air pollutants since most exposure occurs near major roads. In addition, the approach does not have the scale or domain sensitivity observed in simple one-compartment models. Thus, the suggested approach appears to be more accurate and robust than one-compartment models and coarse resolution dispersion models which have strong scale dependencies and which do not account for spatial heterogeneity.

In addition to estimating the overall intake and iF, the spatially-resolved approach identifies areas where intake and exposure rates are high. This information can inform and support decision-making in transportation and urban planning. For example, it can be used to map and prioritize areas and roads for traffic management policies (e.g., tolls, reduced speeds, time or fleet restrictions on specific roads or zones); inform road infrastructure and transit planning to account for health impacts, and in a development context, target areas with lower exposures for urbanization and densification. More generally, spatially-resolved analyses are required to assess exposure and health disparities, and to evaluate and resolve potential environmental injustice issues in which vulnerable populations (e.g., by race/ethnicity, or socio-economic level) are associated with higher exposures and adverse health consequences (Hajat et al. 2015; Pratt et al. 2015; Kravitz-Wirtz et al. 2016).

Finally, a third important strength is the computation efficiency of the approach. The use of RLINE, considered to be a state-of-the-science dispersion model for near-road applications (Snyder et al. 2013), involves considerable computation, and large-scale applications of this model are difficult. We modeled a large area by pre-computing transfer coefficients for different road alignments. Factors that might be considered to increase the accuracy include additional road-angles, segment lengths, smaller buffers, and terrain and road features (e.g., road grade).

Our analysis required a number of simplifications and assumptions, mainly due to the lack of publicly available data. The road network was incomplete, traffic-activity profiles were simplified, the day-time population was approximated, national fleet data and emission factors were used, and congestion was not considered. Only major streets were modeled. While these account for the bulk of emissions, the use of a more detailed and complete road network could improve accuracy. Accuracy could be improved using link-specific emissions inventory data, Lisbon-specific fleet information, more detailed population activity patterns, and the inclusion of on-road exposure. An improved emissions inventory might include spatially-resolved traffic-activity data (e.g., vehicle type, age, speeds, etc.). Despite the simplifications and assumptions made in the present analysis, our iF estimates appear to be more accurate than those based on simple box models.

Dispersion modeling was simplified, in part to address data gaps and to increase computational efficiency. The RLINE model, like other line source models, incompletely addresses influences of the urban form, such as effects from narrow roads, street canyons, channeling of winds, and other factors that can influence dispersion (Tang & Wang 2007). The accuracy of dispersion models is highly dependent on the quality and relevance of meteorological data (Stevens et al. 2007), and the single meteorological site used might not be representative of the entire metropolitan area. In addition, mixing height and wind profile data were based on only two radiosonde measurements per day. We examined only traffic-related emissions and primary $PM_{2.5}$; modeling secondary PM could add significant complexity (Lamancusa et al. 2017) although iF estimates may not increase considerably (Greco, Wilson, Spengler, et al. 2007). Lastly, our background estimates (attributable to far-field traffic emissions) used a simple box model, which does not account for possible gradients in the far-field, and it double counted emissions from the road segment used to estimate near-road concentrations; however, this was shown to have negligible (<1%) impact. Using four archetype road segments and discretizing dispersion modeling outputs into only four buffers significantly increased efficiency; however, the use of additional and smaller (narrower) buffers could improve accuracy.

Additional limitations reflect the use of ambient concentrations as a measure of personal exposure. We did not account for time-activity data and outdoor-indoor penetration of pollutants (Dons et al. 2011; Tainio et al. 2014). On-road exposures and traffic congestion were not considered. Urban scale iF estimates have yet to account for these types of effects.

5.5. Conclusions

We present a novel approach for estimating spatially-resolved iFs for traffic-related air pollutants that featured a computationally-efficient dispersion model approach that accounts near-road exposure and the population distribution. The overall iF estimate for the Lisbon metropolitan area, 16.4 ppm for $PM_{2.5}$, is nearly twice that based on one-compartment models for the same area. Most (74%) of the intake occurs within 500 m of major roads (16% of the domain) by a subset of the population (32 to 37%). Unlike the simplified one-compartment models, the suggested approach accounts for the temporal and spatial variability of emissions, accounts for population shifts over the day, identifies areas and populations that are highly exposed, and it appears robust with respect to the spatial domain considered.

The suggested approach can increase the accuracy of exposure and health effect estimates associated with traffic-related air pollutants. It demonstrates the importance of using local and spatially-resolved data for exposure and health impact estimates. The results can inform and support policies and decision-making addressing urban air quality and public health, particularly, transportation-related planning that can prioritize areas to decrease or displace emissions, such as road fees, time or fleet restrictions on specific roads or zones and truck routing. It can also inform urban planning in a development context, to target areas with lower exposures for urbanization and densification.

Supplementary materials

Appendix IV presents further details on emission factors modeling (namely road traffic speeds and meteorological data used) and traffic volume profiles for Lisbon municipality.

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6 Conclusions

6.1. Key findings

This dissertation examined several important linkages between urban form, the environment and health, and addressed critical issues in the application of life-cycle assessment (LCA) to urban areas. The linkages provide insight on the effects of different urban form characteristics on environmental and health impacts associated with residential builings and transportation, and the approaches can support decision-making and help urban planners on the design and implementation of strategies toward sustainable development. Chapters 2 to 5 examined different aspects of these linkages using case studies for the Lisbon area, each of which used a variety of models and datasets. Most of these datasets are readily available for other cities, and the approaches and models can be applied to other settings. Many of the findings presented in chapters 2 to 5 appear generalizable, as discussed later in this chapter. Overall, this research extended and improved the application of LCA to urban systems, by considering a wide variety of impacts, addressing several key methodological issues, identifying possible improvement opportunities and unintended trade-offs, and avoiding problem-shifting.

This concluding chapter summarizes and discusses the key findings of the dissertation, framed by the four research questions presented in section 1.2. Contributions of the research are then discussed, followed by suggestions for future research.

1) How do modeling and parameter choices, such as the selection of functional units, affect the LC energy and GHG emissions of residential buildings?

Chapter 2 presented a life-cycle (LC) comparative analysis of energy and greenhouse gas (GHG) emissions for three residential building designs, which accounted for building construction, retrofit and use phases, across a 75-year lifespan. The building use phase was dominant, accounting for over 75% of the total energy requirements and GHG emissions. The results highlight the importance of the functional unit when comparing among different buildings. Results expressed on the basis of built area or occupancy showed opposite trends, e.g., larger buildings had higher energy and GHG emissions per person, but lower energy and GHG emissions per square meter.

LC studies of buildings present many methodological differences, which can make inter-study comparison difficult. The analysis showed that the selection of the functional unit, building lifespan and final-to-primary energy conversion factor can significantly affect results. The selection of functional units depends on the goal and scope of the LC study. With an area-based functional unit, larger dwellings have lower energy requirements and lower GHG emissions for the same occupancy. In contrast, an occupancy-based functional unit (often used in studies at the urban scale) can overlook the building's performance, e.g., high occupancy could compensate for poor environmental performance. Thus, to provide comprehensive and consistent insight on a building's environmental impacts, the use of both occupancy- and area-based functional units is recommended.

A second recommendation is that building LC analyses use *primary energy* requirements, rather than final energy use, and disaggregate non-renewable energy demand. This incorporates not only final energy consumption, but also the (upstream) energy used to produce and deliver it. This is particularly relevant in the building use phase, since it dominates LC impacts and since different types of energy can be used, including gas and electricity. Final-to-primary energy conversion factors can vary significantly according to the type of energy used, e.g., the non-renewable energy demand and environmental impacts per kWh of electricity used strongly depends on the electricity mix.

LC studies of buildings are associated with considerable variability and uncertainty. The wide range of building designs, construction processes, technologies (e.g., heating and cooling systems and electricity generation mix), climates, socio-economic aspects and user behaviors contribute to the high variability. Uncertainty is in large part associated with the long lifespan of buildings. Technological evolution in such long periods is difficult if not impossible to anticipate and implications in terms of environmental impacts are equally difficult to predict. It is important to understand the main aspects driving environmental impacts associated with the built environment and consider how these can change in the future. Scenario analyses with a wide range of factors is one approach to try to anticipate potential and likely outcomes.

The research highlights the importance of addressing dwelling sizes and occupancy in urban development strategies. In-dwelling energy consumption depends mainly on the number of persons per household, floor area per capita and user behavior. The number of persons per household has decreased due to aging and social changes (1 in 3 households in the EU-28 has only one person), and floor area per capita has increased (Næss 2001; O'Broin 2007; UN-Habitat 2016). Urban planning and policies could control the growth of the building stock, in particular, focusing on the renovation of existing buildings, promoting the construction and renovation of smaller building typologies, and limiting construction of large houses. Urban planning can also promote the use of renewable energy technologies and shared equipment to reduce environmental impacts associated with urban areas; however, these strategies, which depend on new or additional infrastructure to reduce environmental impacts, should be supported by environmental and sustainability assessments to evaluate their potential benefits, costs and implications.

2) What is the potential significance of user commuting requirements in the LC energy and GHG emissions of residential buildings?

In chapter 3, the LC energy demand and GHG emissions of an inner city apartment and a suburban semidetached house were compared in an analysis that considered construction, use phase and user commuting requirements. Energy demand and GHG emissions for the suburban house were 75 to 100% higher than for the city apartment, mostly due to transportation requirements. Without transportation this difference would be less than 16%. Transportation demand for the suburban house accounted for over 50% of the overall energy and GHG emissions over the 50-year lifespan. When comparing different residence locations, suburban areas were associated with 50 to 115% higher commuting energy and GHG emissions than central areas. Transportation impacts were dominated by car use, which accounted for over 74% of the commuting energy demand and GHG emissions in all parishes. In addition to being the dominant transportation mode, cars also had the highest non-renewable energy demand and GHG emissions due to the higher impact per person-kilometer traveled (partially due to low occupancy rates).

Chapter 3 results demonstrate the significance of transportation demand and the influence of location in the LC impacts associated with residential buildings. For residences located far from workplaces, e.g., in outlying suburban areas, LC impacts from transportation can exceed those of the building itself. This suggests that strategies that consider and improve the transport infrastructure and services might improve urban sustainability more than simply addressing building efficiency. This is particularly likely in areas with mild climates, such as Lisbon, since building heating and cooling requirements (or insulation embodied requirements) are likely to be lower (Nemry et al. 2010). Thus, the evaluation of strategies to improve urban sustainability should consider both housing and workplace/school locations and transportation mode choices, as well as improving the efficiency of transportation modes (e.g., reducing car use and increasing vehicle occupancy). Moreover, the coordination of urban planning and transportation strategies can confer additional benefits, including improved air quality, reduced traffic and congestion, and less infrastructure devoted to roads and parking.

The results of the LC analysis combining housing and transportation are important. Analyses that exclude one or the other have can overlook shifts in burdens and impacts, leading to potentially poor decisions. For example, a low energy building in a car-oriented neighborhood can result in much higher environmental impacts than a conventional building in a location with high accessibility (e.g., walkability and transit systems). This can lead to sub-optimal results when solutions are implemented that were intended to promote sustainable development. (Chapter 3 considered energy demand and GHG emissions for the commuting mode mix in Lisbon; this was further expanded in chapter 4 which provided a detailed comparative assessment of commuting modes for a wide set of environmental and health impacts, as described below.)

In practice, the analysis reinforces the environmental benefits of compact development. Many urban areas in the developed world are still observing the effects of the car-oriented planning that governed urban development in the last decades (Jones 2014). Urban planning strategies need to effectively control peri-urbanization, in particular for car-oriented developments, promote infill development, renovation of the building stock and adequate housing and transportation solutions. As in many other European cities, the resident population decreased in the municipality of Lisbon in recent decades (EU 2016b), and the existing building stock is incompletely occupied.

To realize the gains made possible by simultaneously considering and improving building and transportation efficiency in urban settings, coordinated planning, decisions and actions from many stakeholders across multiple jurisdictions are required. There is an evident need for a multi-sectorial and multi-scale planning and governance to efficiently manage urban development at a metropolitan level, and to overcome administrative borders and potentially conflicting objectives of different municipalities and addressing the interplays between local, regional and global scales (Bulkeley & Betsill 2005; Bulkeley & Betsill 2013). Activities across different governmental groups and departments (e.g., transportation, environment, accessibility and housing) should be brought together, since integrated strategies might offer the best improvement opportunities.

3) How should the potential environmental and health effects associated with alternative commuting modes be assessed and compared?

Chapter 4 provided a comparative LCA of six commuting modes for a wide set of environmental impacts, including $PM_{2.5}$ intake and the associated health effects. The analysis shows that cars had the largest impacts for non-renewable energy, GHG emissions, acidification and freshwater eutrophication; motorcycles and public transit modes (bus, train and subway) had intermediate impacts in these categories (12-84% lower than cars); and bicycles had the lowest impacts across categories (87-97% lower than cars). The operation phase dominated results, accounting for over 59% of the impacts in all categories except for freshwater eutrophication.

The study demonstrates the importance of applying characterization factors to address exposure to air pollutants that are specific to the emission source type and its location, which should improve the accuracy of health impact estimates. In particular, the analysis differentiated $PM_{2.5}$ exposures resulting from emissions in urban and non-urban settings, using two intake fractions (iF). Compared to the use of a single iF for all emissions, the differentiation resulted in increased health impacts for internal combustion vehicles (e.g., cars, buses and motorcycles), and decreased impacts for electricity-powered vehicles (e.g., subway and train). From the health perspective, using a single or global iF strongly discourages electric-powered transit modes since electricity fuel cycle emissions are assumed to have the same intake, exposure and health impacts as those in urban areas. Building on this comparison, it was recommended that $PM_{2.5}$ emissions and health impacts are spatially differentiated. The use of appropriate characterization factors that can represent the higher concentrations, exposures and intake of vehicle emissions in urban areas should improve the accuracy of health impact estimates.

The analysis highlights the need for comprehensive approaches for the environmental and health impact assessment to adequately evaluate impacts and avoid problem shifting in transportation-related strategies. Strategies that reduce travel demand (e.g., teleworking, greater vehicle occupancy and shorter trip distances) decreased impacts across all categories. In contrast, strategies that shifted modes or used alternative technologies had impacts that varied by category and also showed trade-offs between categories, e.g., shifting from motorcycle to subway reduced energy demand, GHG emissions, acidification, terrestrial and marine eutrophication by more than 47%, but freshwater eutrophication impact almost tripled. A holistic perspective should be adopted in order to identify and optimize potential opportunities for reducing environmental and health impacts associated with $PM_{2.5}$, and avoid possible trade-offs between the outcomes of different strategies. While the conclusions of the Lisbon assessment may apply elsewhere, site-specific assessments are helpful or possibly essential for supporting policies that promote sustainable urban development.

While technological innovation can significantly reduce impacts of motorized transportation per km traveled, structural strategies are needed to control travel demand and promote mobility with reduced environmental impacts (e.g., transit, bicycle and walking), and urban planning and design play a key role. Significant technological improvements on the energy conversion and emission rates of internal combustion vehicles have been achieved in recent decades, but vehicle ownership and travel demand continue to increase, both in developing and in developed regions of the world, partially offsetting the potential environmental benefits (Rode et al. 2014; UN-Habitat 2016). Currently, the imminent electrification of private transportation offers an exceptional opportunity to mitigate urban air pollution and noise. However, such technological change can result in increased GHG emissions if the electricity mix is fossil-fuel based, and in problem shifting, e.g., exacerbating resource depletion and freshwater eutrophication problems (Hawkins et al. 2013). Moreover, technological shifts might maintain or even increase inequity in terms of accessibility as they often focus on private motorized transportation. Compact transit-oriented development can help increase accessibility by reducing origin-to-destination distances and dependence on motorized transport, which is particularly relevant for increasing accessibility of lower socio-economic groups. Lastly, urban development strategies that increase active travel can have additional health benefits (Woodcock et al. 2009).

Stricter policies, such as restrictions on parking and older vehicles in inner cities are also important to reduce car use and associated environmental impacts; however, the implementation and effectiveness of such policies is limited by the low public acceptance (Wang et al. 2014). Environmental and health assessments that can quantify the costs and burdens associated with car use, and the potential benefits from these strategies, can contribute to increasing public awareness, engagement and acceptability (Drews & van den Bergh 2016). Moreover, there are potential benefits of integrating climate change and local air pollution management strategies, as they both focus on the reduction of fossil fuel use and emissions, and improved public health is one of the strongest motivators of behavioral change (Petrovic et al. 2014).

4) How can exposure and health effects associated with traffic-related air pollution be addressed within an LCA framework?

Building on the finding in chapter 4 that separate urban and non-urban iF values yield significant differences in estimating exposures and health impacts from transport emissions, chapter 5 introduced an approach for calculating spatially-resolved iF estimates for traffic-related air pollutants. This work combined dispersion modeling and GIS to account for the spatial distribution of emissions and population, and it considered local meteorological data. In comparison to the one-compartment models that provide a single city-wide iF estimates (or essentially the two-compartment model used in chapter 5 for urban and non-urban settings), the new approach accounts for the spatial distribution of emissions and population, which improves the accuracy of estimated exposures and health impacts, and which provides insight on intra-urban variations that can be relevant for planning and policy decisions. For $PM_{2.5}$ in Lisbon metropolitan area, the spatially-resolved approach yielded an overall iF estimate of 16.4 ppm. For major highways, near-road exposure (within 500 m) resulted in an iF estimate of 12.0 ppm (population of about 1 million people within 16% of the domain area) and far-field exposure (1.8 million persons) resulted in an iF of 4.3 ppm. In comparison, a one-compartment model yielded an iF estimate of 8.1 parts per million (ppm), less than half of the updated value.

Chapter 5 demonstrates a simplified dispersion modeling approach using four archetype road segments to estimate emission-to-concentration transfer coefficients, which are combined with spatially-resolved emission and population data. It provides a practical solution for estimating near-road concentrations and intake that significantly reduces the resource and time requirements associated with conventional dispersion modeling. The datasets used are publicly available and the approach can easily be replicated and applied to other cities. However, more complete and disaggregated data, in particular for emissions and population spatial and temporal distribution, could improve the results.

The new approach can improve the evaluation of health effects associated with traffic-related air pollutants by increasing the spatial resolution of iF estimates needed to match population patterns. This also allows the identification of critical areas of exposure and vulnerable populations that might be subject to higher exposures. The approach can inform and support decision-making by identifying improvement opportunities and priorities for policies that focus on the most exposed or affected areas. Spatially-resolved approaches enable a comparative analysis of strategies that may reduce exposures, e.g., decreasing or displacing emissions, for example, with road fees or zoning restrictions. The approach can inform urban planning and land use decisions related to urban regeneration, densification and traffic routing.

6.2. Research contribution

The research makes methodological contributions with respect to extending and improving the application of LCA to urban areas, and it provides insight on several key linkages between the urban form and environmental and health impacts.

On a methodological level, several improvements were made to environmental assessments of urban areas, residential buildings and transportation. Drawing on the models developed and evaluated for four Lisbon-based applications, we provided recommendations for the evaluation of environmental and health impacts associated with urban areas. This research showed the importance of applying a LC perspective in environmental assessment of urban areas (in chapters 2 to 4), the varying results obtained using different impact indicators and metrics (in chapters 2 to 4), the need to adequately characterize resource use and emissions to estimate potential environmental and health impacts (in chapters 2 to 5), the significance of considering a larger system boundary and implications at the urban scale (beyond direct impacts of isolated urban components; in chapter 3), and the importance of spatially-resolved analyses, which are needed to address local environmental and health impacts, as well as environmental justice (in chapter 5). The advances contribute to more comprehensive and robust assessments for environmental, air quality and health risk, which can better inform and support decision-makers in comparing alternative strategies, identifying improvement opportunities and avoiding problem shifting.

While the applications in this dissertation were based in Lisbon, the models and methodological approaches appear to be robust as results were comparable to much of the literature. The approaches and methods can be used to address similar problems in other cities and different settings, given that appropriate datasets are available. As mentioned, the approaches use publicly available statistical data, most of it available on international datasets (at global or European level).

The objective results of this study provide important insight on the linkages between urban form and environmental and health impacts. The applications showed that the larger dwelling floor area per capita can significantly increase environmental impacts per capita, and the type of energy used and electricity mix can affect impacts associated with the use phase of residential buildings, which dominates their LC energy and GHG emissions. While residential buildings contribute to a significant share of energy demand and GHG emissions associated with urban areas, user transportation requirements can exceed those of a building if it is located in car-dependent (suburban) locations. The environmental impacts of different urban transportation modes vary significantly; however, shifts in mode and technology can have trade-offs, e.g., electricity-powered transportation modes can contribute to important reductions in energy demand, GHG emissions and health effects due to traffic-related air pollution, but exacerbate freshwater eutrophication problems. Environmental and health impacts occur at different geographic scales, and can have significant spatial variations, which should be addressed to target critical areas and social inequities.

The results and findings appear to be generalizable to other urban areas and settings. The Lisbon area and the problems examined in the dissertation are representative of those currently occurring in many industrialized regions across the world. Urbanization, demography and mobility trends in Lisbon resemble those in many other cities, in particular in Southern and Western Europe (UN-Habitat 2016). Europe has a dense network of relatively compact cities, with population densities typically between those observed in North America and Asian cities. With 2.8 million inhabitants, Lisbon falls within the range of cities where most urban populations in Europe reside (0.25 to 5 million inhabitants) (UN-Habitat 2016). Most European countries (EU-15) observed a slow growth of the urban population share in the last 50 years, which occurred almost entirely in suburban areas and clusters in the outskirts of urban centers (UN-Habitat 2016). The expansion of urban areas has contributed to an increase of environmental impacts per capita (UN-Habitat 2016). Environmental sustainability in Lisbon, and other Portuguese cities, has benefited from EU regulations since the 1970s (Fenger 2009; Guedes et al. 2009). While technological developments and emission regulations have increased energy efficiency in cars, reducing energy demand by vehicle km traveled by 9% in the EU, overall GHG emissions decreased by only 5% between 2000 and 2013, in part due to an increase of vehicle-kilometer travelled (VKT) of 7% in the same period. Despite many common trends and characteristics, the complexity and diversity of urban forms across European cities alone is noteworthy (Schwarz 2010), and site-specific analyses are recommended.

The research highlighted the potential of urban planning and design for improving environmental performance and shaping more sustainable cities. The studies suggest the benefit of *integrating* urban planning and design, architecture and transportation planning with environmental and health goals. The complexity of the linkages between urban form and environmental performance of urban areas, as well as the many actors involved and long time needed to change the built environment can all be obstacles (Schaffer & Vollmer 2010). We need system-level strategies that integrate different sectors and scales to reduce environmental impacts and improve public health and well-being, with particular attention to inequities. Urban planning can play a significant role in overcoming fragmentation in governance and public policy decision-making (UN-Habitat 2009). In some metropolitan areas, including Lisbon and Oporto, offices to coordinate strategies of different municipalities and convene multiple departments were created (to overcome the fragmentation of policy-making into different groups and administrative borders). However, their power has been strongly limited by budget, staff, legal and administrative aspects (UN-Habitat 2016).

In real-life applications, urban planners, architects, policy-makers and other stakeholders need objective frameworks, tools and data to support decision-making, in particular to address environmental and health impacts (Schaffer & Vollmer 2010; Ramaswami et al. 2016). We need adaptive planning strategies and policies, supported by scientific evidence (Schaffer & Vollmer 2010). The lack of appropriate frameworks and tools, together with inadequate financial and political support, have been suggested as limitations for urban planning strategies implementation, monitoring and evaluation (Holden 2006; Ramaswami et al. 2016). LCA and other industrial ecology tools can be particularly useful in this context, and should be implemented in practice. Environmental impact assessment based studies should be required and adequately performed in urban planning and architectural practice. Lastly, urban planning must be driven by community needs, and public engagement is vital for successful implementation of sustainable urban development strategies (Schaffer & Vollmer 2010). While urban living and compact development can contribute to more energy efficient practices and reduced environmental impacts, this is by no means guaranteed: no form or development can be sustainable if it is not attractive and inhabited by vital, healthy and sustainable communities.

6.3. Future work

Drawing on the applications, findings and limitations discussed in chapters 2 to 5, several aspects could be improved in future work. Validating the models and evaluating uncertainty of our findings is challenging, but it would be a valuable contribution. Application to other cities could demonstrate the significance and generalizability of the findings. More complete and disaggregated data would significantly improve the models and results as it would reduce the need for simplifications and assumptions that contribute to uncertainty. To improve the assessment of health effects, additional issues could be addressed, such as noise and pollutants other than $PM_{2.5}$.

A number of further studies are identified that would be useful to inform and support sustainable urban development, based on the limitations identified in chapters 2 to 5. First, studies extending the model to other types of buildings (e.g., office and commercial buildings), non-working travel demand and other transportation systems (e.g., freight and ships) could provide more insight on environmental impacts associated with urban areas. Second, a wider range of environmental impacts developing dynamic approaches, with sensitivity to technological evolution and behavioral changes would be useful, in particular in supporting decision-makers. Third, health impact estimates would provide valuable insight by integrating an intake-to-damage characterization and including local population demographic and health data. Such insight could help to redress how environmental and health impacts are distributed across the population (environmental justice).

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Appendix I: Abstracts of the four main publications

1. Life-cycle energy and greenhouse gas analysis of three building types in a residential area in Lisbon

Joana Bastos[a], Stuart A. Batterman[b], Fausto Freire[a]

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Residential buildings consume a large fraction of energy and thus represent a major opportunity for reducing energy requirements and greenhouse gas (GHG) emissions. This article presents a life-cycle energy and GHG analysis of three representative residential building types in a well-known area in Lisbon (*Bairro de Alvalade*). The life-cycle model focused on building construction, retrofit and use phases, applied an econometric model to estimate energy use in Portuguese households, and considered two functional units: per square meter per year and per person per year. Over the buildings' 75-year lifespan, the use phase accounted for most (69–83%) of the primary energy requirements and GHG emissions. Larger buildings have lower life-cycle energy requirements and GHG emissions on a square meter basis. On a per person basis, however, this pattern is reversed and larger buildings are associated with higher energy requirements and GHG emissions. Due to the considerable variability and uncertainty associated with life-cycle analyses of buildings, the use of both occupancy- and area-based functional units is recommended.

Bastos, J., Batterman, S., Freire, F. (2014) Life-cycle energy and greenhouse gas analysis of three building types in a residential area in Lisbon. *Energy and Buildings* 69: 344-353. http://doi.org/10.1016/j.enbuild.2013.11.010

2. Significance of mobility in the life-cycle assessment of buildings

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While most life-cycle assessments of buildings have focused on construction and use phases, the location of a building can significantly affect the transportation demand of its

inhabitants. The life-cycle energy and greenhouse gas (GHG) emissions of two representative buildings in Lisbon, Portugal, are compared: an apartment building in the city centre and a semidetached house in a suburban area. An integrated approach is used to conduct a life-cycle analysis that includes building construction, building use and user transportation. Sensitivity analyses are used to evaluate impacts for multiple locations. For the apartment, building use accounted for the largest share of energy and emissions (63–64%), while for the house, most (51–57%) of the energy and emissions were associated with user transportation. Energy and GHG emissions for suburban locations were significantly higher (by 55–115%) than those in the city-centre locations, largely due to individuals commuting by car. The analysis demonstrates the significance of transportation and highlights the importance of residence location in urban planning and environmental assessments. These results are likely to apply to other southern European cities that have expanded with significant growth in car ownership and use. To improve urban sustainability, development strategies should consider the transport infrastructure in addition to building efficiency.

Bastos, J., Batterman, S., Freire, F. (2016) Significance of mobility in the life-cycle assessment of buildings. *Building Research and Information* 44 (4): 376 – 393. http://dx.doi.org/10.1080/09613218.2016.1097407

3. Environmental impacts of commuting modes in Lisbon: a life-cycle assessment addressing particulate matter impacts on health

Joana Bastos[a], Pedro Marques[a], Stuart A. Batterman[b], Fausto Freire[a]
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[b] Department of Environmental Health Sciences, University of Michigan, USA

A life-cycle assessment of commuting alternatives is conducted that compares six transportation modes (car, bus, train, subway, motorcycle and bicycle) for eight impact indicators. Fine particulate matter ($PM_{2.5}$) emissions and health impacts are incorporated in the assessment using intake fractions that differentiate between urban and nonurban emissions, combined with an effect factor. The potential benefits of different strategies for reducing environmental impacts are illustrated. The results demonstrate the need for comprehensive approaches that avoid problem-shifting among transportation-related strategies. Policies aiming to improve the environmental performance of urban transportation should target strategies that decrease local emissions, life-cycle impacts and health effects.

Bastos, J., Marques, P., Batterman, S., Freire, F. (under review) Environmental impacts of commuting modes in Lisbon: a life-cycle assessment addressing particulate matter impacts on health.

4. Intake fraction estimates for on-road fine particulate matter ($PM_{2.5}$) emissions: exploring spatial variation of emissions and population distribution in Lisbon, Portugal

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The intake fraction (iF) expresses population exposure resulting from pollutant emissions. City-wide iFs estimated using simple one-compartment models, used in a number of previous studies, have significant uncertainties and do not capture intra-urban variation in exposure that is important for estimating health effects associated with traffic-related air pollutants. We present a novel and efficient approach for developing spatially-resolved iF estimates using dispersion modeling for near-road exposures, which accounts for the spatial and temporal variation in meteorology, emissions and the population living and working near major roads. iF estimates are developed for primary emissions of traffic-related fine particulate matter ($PM_{2.5}$) in Lisbon, Portugal, which are compared to those from a one-compartment model. Both methods use local meteorological and population data to represent exposures for a total of 2.8 million people. The new method produces an overall iF value of 16.4 ppm for the Lisbon metropolitan area, over twice that of the one-compartment model (8.1 ppm). Most of the exposure (12.0 ppm) occurs for the subset of the population (1 million people) living or working within 500 m of large roads. The iF for the remainder of the population (1.8 million people) is only 4.3 ppm. The spatially-resolved iF accounts for high concentration areas, which can be densely populated and account for much or most of the exposure from traffic-related emissions. The new method is computationally efficient and can improve estimates of exposure and health impacts occurring in urban areas, leading to more effective urban and transportation planning decisions to mitigate impacts.

Bastos, J., Milando, C., Freire, F., Batterman, S., (under review) Intake fraction estimates for on-road fine particulate matter ($PM_{2.5}$) emissions: exploring spatial variation of emissions and population distribution in Lisbon, Portugal.

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Appendix II: Full list of publications

1. Articles in internationally reviewed scientific journals

Main articles

Bastos, J., Batterman, S., Freire, F. (2014) Life-cycle energy and greenhouse gas analysis of three building types in a residential area in Lisbon. *Energy and Buildings* 69: 344-353. http://doi.org/10.1016/j.enbuild.2013.11.010

JCR® impact factor (2015): 2.973

Bastos, J., Batterman, S., Freire, F. (2016) Significance of mobility in the life-cycle assessment of buildings. *Building Research and Information* 44 (4): 376-393. http://dx.doi.org/10.1080/09613218.2016.1097407

JCR® impact factor (2015): 2.196

Bastos, J., Marques, P., Batterman, S., Freire, F. (under review) Environmental impacts of commuting in Lisbon: a life-cycle assessment addressing particulate matter impacts on health.

Bastos, J., Milando, C., Freire, F., Batterman, S. (under review) Intake fraction estimates for on-road fine particulate matter emissions: exploring spatial variation of emissions and population distribution in Lisbon, Portugal.

Other articles related with the research

Soares, N., Bastos, J., Pereira, L., Soares, A., Amaral, A., Asadi, E., Rodrigues, E., Lamas, F., Monteiro, H., Lopes, M., Gaspar, A. (2017) A review on current advances in the energy and environmental performance of buildings towards a more sustainable built environment. *Renewable and Sustainable Energy Reviews* 77: 845-860. http://dx.doi.org/10.1016/j.rser.2017.04.027

JCR® impact factor (2015): 6.798

Soares, N., Martins, A.G., Carvalho, A.L., Caldeira, C., Du, C., Castanheira, É., Rodrigues, E., Oliveira, G., Pereira, G.I., Bastos, J., Ferreira, J.P., Ribeiro, L.A., Figueiredo, N., Šahović, N., Miguel, P., Garcia, R. (under review) The challeng.ing paradigm of interrelated energy systems towards a more sustainable future

2. Communications in international conferences and symposia

Papers in conference proceedings

Bastos, B., Batterman, S., Freire, F., (2015). "Life-cycle assessment of a city apartment and a suburban house: addressing the issue of occupancy". *Energy for Sustainability 2015 – Designing for People and the Planet, May 14-15*, Coimbra, Portugal.

Bastos, J., Batterman, S., and Freire, F. (2014) "How does residential location influence the environmental life-cycle impacts of a household?", *40th IAHS Conference, Sustainable Housing Construction, December 16-19*, Funchal, Portugal.

Bastos, J., Batterman, S., Freire, F. (2013). "Comparative life-cycle analysis of residential typologies in Lisbon, Portugal: an apartment building and two semi-detached houses*". Energy for Sustainability 2013, Sustainable Cities: Designing for People and the Planet, 8-10 September*, Coimbra, Portugal.

Bastos, J., Freire, F., Bandeirinha, J. (2012). "Urban form potential in sustainable development: a framework for the environmental assessment of urban patterns", *2ª Conferência do PNUM. Morfologia Urbana nos Países Lusófonos. PNUM 2012: Urban Morphology in Portuguese Speaking Countries. ISCTE 5 -6 July*, Lisbon, Portugal.

Oral communications

Bastos, J., Milando, C., Freire, F., Batterman, S. (2017) "Exploring intra-urban variability of intake fractions for $PM_{2.5}$: an application to the life-cycle assessment of road transportation in Lisbon". *2017 International Conference on Urban Health – Health Equity: The new urban agenda and sustainable development goals, September 26-29*, Coimbra, Portugal.

Bastos, J., Milando, C., Freire, F., Batterman, S. (2017) "Exploring intra-urban variability of intake fractions for $PM_{2.5}$: an application to the life-cycle assessment of road transport in Lisbon, Portugal". *2017 Joint Conference ISIE and ISSST, Science in Support of Sustainable and Resilient Communities, June 25-29*, Chicago, IL, USA.

Bastos, J., Marques, P., Batterman, S., Freire, F. (2017) "Comparative life-cycle assessment of transportation modes for commuting in Lisbon". *Energy for Sustainability 2017 – Designing cities & communities for the future, University of Coimbra, February 8-10*, Funchal, Portugal.

Bastos, J.; Marques, P.; Batterman, S.; Freire, F. (2016). "Life-cycle assessment of urban transportation modes". *CEM2016 – Mechanical Engineering Conference, June 1-3*, Porto, Portugal.

Ribau, J. P., Silva, C.M., Bastos, J., Castanheira, É., Malça, J., Freire, F. (2016). "Biodiesel production and use in the urban transport sector in Portugal." *CEM2016 – Mechanical Engineering Conference, June 1-3*, Porto, Portugal.

Bastos, J., Batterman, S., Freire, F. (2015). "Integrating user transportation in the life-cycle assessment of buildings". *2nd Discussion Forum on Industrial Ecology and Life-Cycle Management, 5-6 March*, Coimbra, Portugal.

Bastos, J., Batterman, S., Freire, F. (2014). "Life-cycle assessment of the built environment: integrating housing and mobility requirements of a household in Lisbon". *Discussion Forum on Industrial Ecology and Life-Cycle Management. 7-8 April*. Coimbra. Portugal.

Posters

Bastos, J., Marques, P., Batterman, S., Freire, F. (2017) "Comparative life-cycle assessment of urban road transport in Lisbon, Portugal. *17th National Conference and Global Forum on Science, Policy and the Environment - Integrating environment and health, National Council for Science and Environment, January 24-26*, Washington D.C., USA.

Bastos, J., Batterman, S., Freire, F. (2017) "Exploring the linkages between environmental impacts and urban form: a life-cycle approach". *Michigan University-wide Sustainability and Environment (MUSE) conference and workshop, February 9-10*, University of Michigan, Ann Arbor, USA.

Bastos, J., Batterman, S., Freire, F. (2016). Linking urban design and sustainable development: an integrated life-cycle approach. *MITPortugal Annual Conference - 10 years Engineering a better future, 30th June*, University of Minho, Braga, Portugal.

Bastos, J., Marques, P., Batterman, S., Freire, F. (2016). Comparative life-cycle assessment of urban road transport: Application to commuting in Portugal. *UMTRI Transportation Safety Research Symposium, October 6,* University of Michigan, Ann Arbor, USA.

Bastos, J., Batterman, S., Freire, F. (2015). "Integrating user transportation in the life-cycle assessment of buildings", *ISIE 2015 – 8th Conference of the International Society for Industrial Ecology, July 7th to 10th*, University of Surrey, Guildford, UK.

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Appendix III: Supplementary materials to Chapter 4

Contents

1. Background: environmental impacts of urban travel

Urban transportation has a diversity of environmental impacts, both global and local. On a global scale, it contributes significantly to dependence on fossil fuels (representing a risk in energy security), global warming and environmental degradation (Chester 2008; Woodcock et al. 2007). At the local scale, transportation affects citizens' quality of life in social, economic and environmental ways (Albalate & Bel 2009), but its impact on health is a major concern. Health impacts associated with transportation in an urban context include urban air pollution, road-traffic injuries, physical activity, noise, and stress (Woodcock et al. 2007).

Air pollution associated with transportation is especially relevant in urban environments due to potentially high exposures and the consequent adverse effects on health (Chester 2008). The major local scale air pollutant emissions of concern associated with transportation include (Chester 2008; Nielsen 2013; Gorham 2002): (1) particulate matter (PM), which is generally classified by diameter as PM_{10} and $PM_{2.5}$ (µm), and comprises of a mixture of small particles and liquid droplets that can affect cardiovascular and respiratory systems and cause and aggravate asthma, bronchitis, diabetes, lung cancer, adverse birth outcomes, other diseases, and reduced life expectancy and mortality; (2) carbon monoxide (CO), which reduces the oxygen-carrying capacity of the circulatory system, causing hypoxia, brain problems and even asphyxiation if exposures are high; (3) nitrogen oxides (NO_x), which contribute to formation of ground-level ozone (O_3) and which itself aggravates asthma and other respiratory diseases; (4) volatile organic compounds (VOC), which include a large number of compounds that evaporate at normal temperatures and which can cause headaches, nausea, central nervous system problems, and cancer, among other health problems; VOCs also contribute to O_3. Other air pollutants associated with transportation include: (5) sulphur dioxide (SO_2), which adversely affects the pulmonary, respiratory and cardiovascular systems, contributes to acid deposition (along with NO_2) that causes multiple ecological impacts and forms secondary $PM_{2.5}$, and (6) lead (Pb), which can impede brain development and causes anemia and kidney damage. Motor vehicle fuels in Europe and most other developed countries now have most of the sulphur removed, and lead is no longer used as a fuel additive, thus concerns related to SO_2 and Pb emission have been greatly diminished.

2. The life-cycle model: supplementary figures

Figure A3.1 shows the 2011 transport mode mix for the Lisbon area and the residence parish (data refer to the main transport mode used).

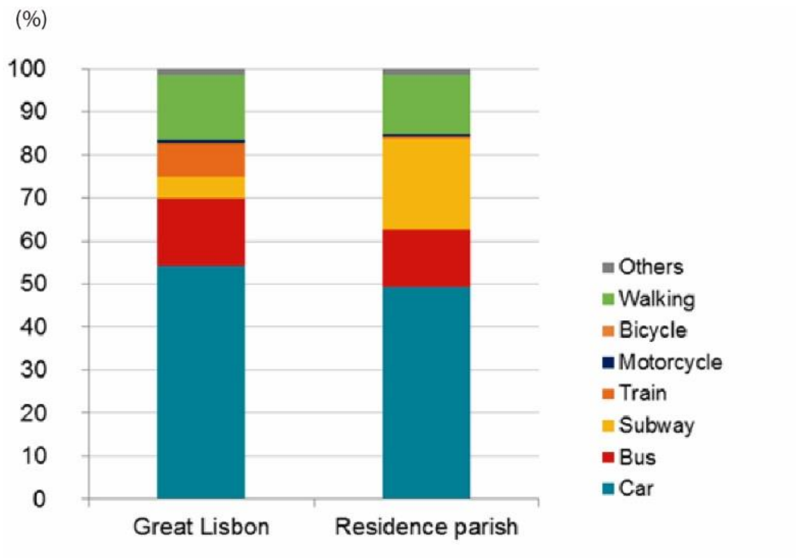

Figure A3.1 Commuting mode mix in the great Lisbon area and in the residence parish (INE, 2012)

Figure A3.2 shows the residence and workplace locations together with the main transportation infrastructure in Lisbon.

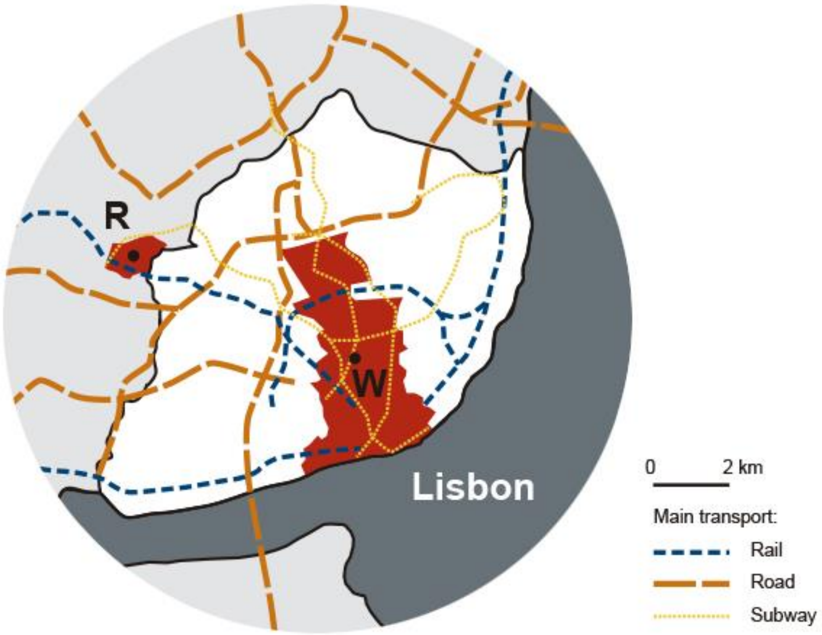

Figure A3.2 Map of Lisbon: residence and workplace locations

3. Detailed results per person-kilometer traveled (PKT)

Table A3.1 Detailed results per PKT: non-renewable primary energy (NRE), greenhouse gas emissions (GHG), acidification (AC), terrestrial eutrophication (TE), freshwater eutrophication (FE), marine eutrophication (ME), $PM_{2.5}$ intake and $PM_{2.5}$ health (continues on next page).

	NRE	GHG	AC	TE	FE	ME	$PM_{2.5}$ intake
	(MJ)	(g CO_2 eq)	(molc H^+ eq)	(molc N eq)	(kg P eq)	(kg N eq)	(10^{-6} mg)
Passenger car, petrol							
Infrastructure							
Construction	0.1287	4.7177	3.12E-05	1.26E-04	9.30E-07	1.15E-05	73.2185
Maintenance	0.0712	5.6973	3.09E-05	4.50E-05	4.84E-06	5.32E-06	5.4387
Disposal	0.0008	0.0500	4.27E-07	2.02E-06	3.73E-09	1.84E-07	0.0738
Vehicle							
Manufacture	0.2912	18.7675	1.47E-04	1.67E-04	1.46E-05	1.65E-05	20.1531
Maintenance	0.0789	3.5015	2.07E-05	2.92E-05	2.36E-06	2.99E-06	2.5877
Disposal	0.0096	1.8522	2.59E-06	6.46E-06	1.28E-07	6.15E-07	0.2575
Operation	2.3737	172.7286	4.71E-04	1.15E-03	4.59E-06	8.18E-05	340.8644

Table A3.1 (continuation). Detailed results per PKT: non-renewable primary energy (NRE), greenhouse gas emissions (GHG), acidification (AC), terrestrial eutrophication (TE), freshwater eutrophication (FE), marine eutrophication (ME), $PM_{2.5}$ intake and $PM_{2.5}$ health (continues on next page).

	NRE	**GHG**	**AC**	**TE**	**FE**	**ME**	**$PM_{2.5}$ intake**
	(MJ)	(g CO_2 eq)	(molc H^+ eq)	(molc N eq)	(kg P eq)	(kg N eq)	(10^{-6} mg)
Passenger car, diesel							
Infrastructure							
Construction	0.1287	4.7177	3.12E-05	1.26E-04	9.30E-07	1.15E-05	73.2185
Maintenance	0.0712	5.6973	3.09E-05	4.50E-05	4.84E-06	5.32E-06	5.4387
Disposal	0.0008	0.0500	4.27E-07	2.02E-06	3.73E-09	1.84E-07	0.0738
Vehicle							
Manufacture	0.2912	18.7675	1.47E-04	1.67E-04	1.46E-05	1.65E-05	20.1531
Maintenance	0.0789	3.5015	2.07E-05	2.92E-05	2.36E-06	2.99E-06	2.5877
Disposal	0.0096	1.8522	2.59E-06	6.46E-06	1.28E-07	6.15E-07	0.2575
Operation	2.2038	153.1905	6.03E-04	2.53E-03	3.44E-06	2.31E-04	1304.9558
Bus, diesel							
Infrastructure							
Construction	0.0886	3.2471	2.15E-05	8.70E-05	6.40E-07	7.92E-06	50.3948
Maintenance	0.0052	0.4123	2.24E-06	3.25E-06	3.50E-07	3.85E-07	0.3936
Disposal	0.0005	0.0344	2.94E-07	1.39E-06	2.57E-09	1.27E-07	0.0508
Vehicle							
Manufacture	0.0255	1.8369	1.03E-05	1.73E-05	1.16E-06	1.66E-06	2.7738
Maintenance	0.0150	1.0178	4.35E-06	6.29E-06	5.97E-07	1.13E-06	0.5654
Disposal	0.0000	0.0695	2.09E-08	9.50E-08	9.58E-10	1.02E-08	0.0016
Operation	0.8111	56.4211	3.62E-04	1.74E-03	1.27E-06	1.59E-04	301.7336
Motorcycle, 2-stroke, petrol							
Infrastructure							
Construction	0.0255	0.9350	6.18E-06	2.51E-05	1.84E-07	2.28E-06	14.5111
Maintenance	-	-	-	-	-	-	-
Disposal	0.0002	0.0099	8.46E-08	3.99E-07	7.39E-10	3.65E-08	0.0146
Vehicle							
Manufacture	0.1144	7.0038	5.92E-05	8.01E-05	3.86E-06	7.52E-06	7.2386
Maintenance	0.0886	4.8644	1.95E-05	3.72E-05	1.16E-06	3.50E-06	3.120162
Disposal	0.0023	1.3597	1.61E-06	4.34E-06	6.24E-08	4.30E-07	0.12437
Operation	1.1820	83.1799	2.85E-04	8.98E-04	2.29E-06	8.12E-05	2309.428
Motorcycle, 4-stroke, petrol							
Infrastructure							
Construction	0.0255	0.9350	6.18E-06	2.51E-05	1.84E-07	2.28E-06	14.51111
Maintenance	-	-	-	-	-	-	-
Disposal	0.0002	0.0099	8.46E-08	3.99E-07	7.39E-10	3.65E-08	0.014621
Vehicle							
Manufacture	0.1144	7.0038	5.92E-05	8.01E-05	3.86E-06	7.52E-06	7.23859
Maintenance	0.0886	4.8644	1.95E-05	3.72E-05	1.16E-06	3.50E-06	3.120162
Disposal	0.0023	1.3597	1.61E-06	4.34E-06	6.24E-08	4.30E-07	0.12437
Operation	1.7611	124.6334	4.34E-04	1.39E-03	3.41E-06	1.26E-04	647.9598

Table A3.1 (continuation). Detailed results per PKT: non-renewable fossil energy (NRE), greenhouse gas emissions (GHG), acidification (AC), terrestrial eutrophication (TE), freshwater eutrophication (FE), marine eutrophication (ME), $PM_{2.5}$ intake and $PM_{2.5}$ health.

	NRE	**GHG**	**AC**	**TE**	**FE**	**ME**	**$PM_{2.5}$ intake**
	(MJ)	(g CO_2 eq)	(molc H^+ eq)	(molc N eq)	(kg P eq)	(kg N eq)	(10^{-6} mg)
Bicycle							
Infrastructure							
Construction	0.0136	0.4987	3.30E-06	1.34E-05	9.83E-08	1.22E-06	7.739264
Maintenance	-	-	-	-	-	-	-
Disposal	0.0001	0.0053	4.51E-08	2.13E-07	3.94E-10	1.95E-08	0.007798
Vehicle							
Manufacture	0.0904	7.3778	4.49E-05	7.22E-05	3.05E-06	6.92E-06	10.36223
Maintenance	0.0191	1.2026	4.61E-06	8.96E-06	3.00E-07	8.69E-07	0.938188
Disposal	0.0004	0.3937	2.64E-07	1.22E-06	5.98E-09	1.19E-07	0.026684
Operation	-	-	-	-	-	-	-
Train, electricity							
Infrastructure							
Construction	0.0463	4.3070	1.93E-05	5.69E-05	1.83E-06	5.14E-06	8.606696
Maintenance	0.0099	0.7965	4.29E-06	6.06E-06	6.91E-07	6.86E-07	0.662349
Disposal	0.0124	0.7694	5.88E-06	2.77E-05	6.48E-08	2.53E-06	0.71231
Vehicle							
Manufacture	0.0115	0.7801	6.24E-06	8.76E-06	1.21E-06	8.28E-07	1.299714
Maintenance	0.0024	0.1476	7.5E-07	1.81E-06	4.72E-08	2.09E-07	0.149974
Disposal	0.0000	0.0039	1.88E-08	9.31E-08	2.23E-10	7.13E-08	0.002123
Operation	0.6201	47.8297	0.00015	0.000518	2.01E-05	4.89E-05	22.76584
Subway, electricity							
Infrastructure							
Construction	0.0943	13.6010	4.41E-05	0.000137	3.23E-06	1.22E-05	15.90824
Maintenance	0.0093	0.7505	4.05E-06	5.71E-06	6.51E-07	6.46E-07	0.624036
Disposal	0.0117	0.7249	5.54E-06	2.61E-05	6.1E-08	2.39E-06	0.671108
Vehicle							
Manufacture	0.0277	2.0761	1.24E-05	1.68E-05	1.43E-06	1.65E-06	2.229499
Maintenance	0.0218	1.2230	5.18E-06	9.39E-06	3.45E-07	9.81E-07	0.710091
Disposal	0.0000	0.0102	2.2E-08	1.01E-07	3.09E-10	2E-07	0.003308
Operation	0.5589	43.0828	0.000135	0.000467	1.81E-05	4.4E-05	20.51918

4. Inter-study comparison of results

Table A3.2 Non renewable energy (NRE) and greenhouse gas (GHG) results per person-kilometer traveled (PKT) in this and previous studies in the literature

	Car		Bus		Rail		Motorcycle		Bicycle	
	NRE (MJ)	GHG (g CO_2 eq)	NRE (MJ)	GHG (g CO_2 eq)	NRE (MJ)	GHG (g CO_2 eq)	NRE (MJ)	GHG (g CO_2 eq)	NRE (kJ)	GHG (g CO_2 eq)
Our study	2.8 - 3.0	188 - 207	0.9	63	0.7	55 - 61	1.4 - 2.0	97 - 138	124	9.5
Chester 2008	2.9	236.2	0.7 - 5.5[a]	53 - 422[a]	1.2 - 1.9[b]	81 - 143[b]	1.7	109	-	-
Girardi et al. 2015	3.1[c]	205[c]	-	-	-	-	-	-	-	-
Ercan & Tatari 2015	-	-	-	140 - 196[d]	-	-	-	-	-	-
Cherry et al. 2009[e]	-	268	-	173	-	-	-	153	-	6.26
Sánchez et al. 2013	-	-	1.1	84	-	-	-	-	-	-
Engelmoer 2012	-	-	-	-	-	-	-	-	176	-
Shibahara et al. 2013	-	-	-	-	-	-	-	-	-	20[f]

[a] the study provided peak and off-peak results, i.e., the authors distinguished a period of more intensive use and a low occupancy period
[b] the range covers result for five rail systems
[c] values adjusted to our vehicle service life and occupancy
[d] values derived from results per VKT assuming occupancy of 20 people
[e] values are for CO_2 only, associated with production and use phases, and adjusted to our vehicle service lives and occupancy
[e] operational requirements considered additional food needs of the rider

Appendix III References

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Appendix IV: Supplementary materials to Chapter 5

1. Traffic profiles

Figure A4.1 presents hourly traffic profiles for weekdays and weekends, showing that the most significant difference is in the morning commuting period. Data is for Lisbon municipality and was based on Brito (2012). Daily adjustment factors across the week vary between 1.04 and 1.11 for weekdays, and are 0.86 and 0.72 for Saturday and Sunday, respectively (Brito, 2012).

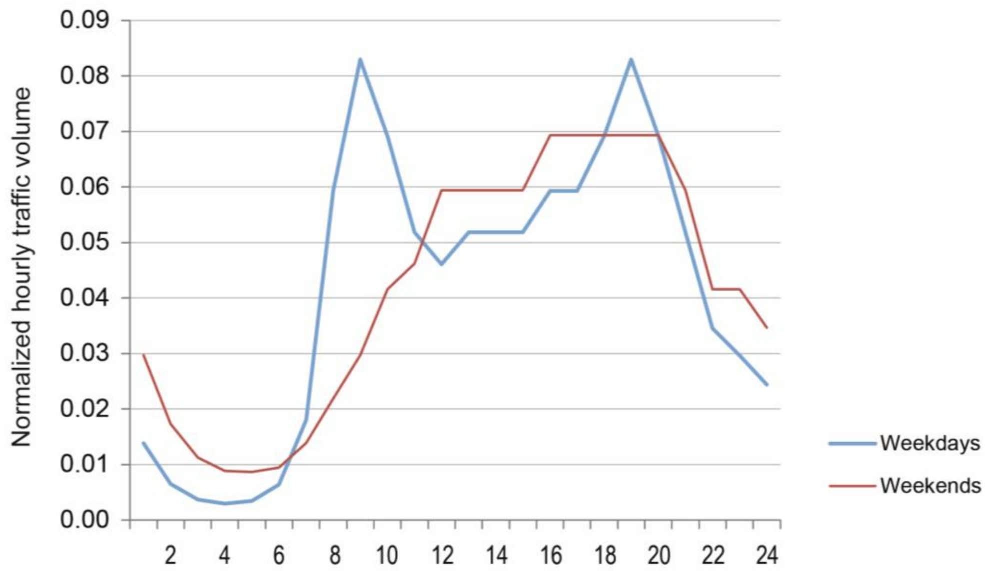

Figure A4.1 Normalized hourly variation of traffic for weekdays and weekends in Lisbon municipality for first level road network (all vehicles), based on Brito (2012)

2. Emission factors modeling

Vehicle emission factors for passenger and commercial vehicles were modeled in COPERT for two types of roads, highways and national roads, assuming average speeds of 90 and 60 km/h, respectively. Speed limits vary generally from 80 to 120 km/h for highways and 50 to 90 km/h for national roads (Road Traffic Code, approved by Decree-Law nr. 114/94, of May 3rd, and amended by Law nr. 72/2013, of September 3rd). The meteorological data considered in the model, presented in Table A4.1, was based on climate normals for Lisbon metropolitan area, 1971 – 2000 (IPMA, n.d.).

Table A4.1 Meteorological data used for vehicle emission factors, based on modeled historical climate normals for Lisbon metropolitan 1971 – 2000.

	Min. temperature (°C)	Max. temperature (°C)	Relative humidity (%)
January	6.7	13.1	83.7
February	6.7	13.9	80.9
March	7.6	15.6	77.8
April	8.9	17.2	75.7
May	10.7	20.0	72.5
June	13.6	24.1	67.1
July	15.9	28.0	62.7
August	16.4	28.5	62.2
September	14.9	25.5	65.5
October	12.2	20.5	75.5
November	9.5	16.4	81.0
December	7.4	13.6	83.6

Appendix IV References

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