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

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Abstract

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 non-urban 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.

Keywords

Environmental impacts; Transport; Primary energy; Greenhouse gas; Fine particulate matter (PM_{2.5})

1. Introduction

The transportation sector plays a crucial role in the targets and policies aimed at reducing energy consumption and greenhouse gas (GHG) emissions. Currently, this sector accounts for 32% of global primary energy consumption in the 28 EU countries, and a higher percentage, 40%, in Portugal (European Union, 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 (Flamm, 2009; Litman and Burwell, 2006; World Business Council for Sustainable

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Supporting information

Brief literature review on the topic; transport mode mix for Lisbon 2011; additional information on the life-cycle model, namely a map with the residence and workplace locations and details on the Portuguese fleet data; detailed results per person-kilometer traveled (PKT) for all modes and impact categories; and inter-study comparison of results.

Development, 2004; Hawkins et al., 2012; Black & Sato, 2007). There is a critical need for research that assesses the environmental impacts of transportation and policies that promote sustainable and healthy mobility.

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 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 and more refined set of environmental indicators is needed to identify and avoid unintended trade-offs among mitigation strategies (Chester et al. 2013; Bauer et al. 2015; Meng et al. 2017). Impacts of pollutant emissions, including fine particulate matter (PM_{2.5}), on health in transportation life-cycle assessments (LCA) should be better addressed. Such assessments require spatially differentiated characterization factors to properly assess health effects, in particular differentiating urban and non-urban environments (Joint Research Centre, 2011; Fantke et al., 2017), as discussed in the following section (and in the supplementary materials). While 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 urban transportation modes.

This study provides a comparative LCA of six transportation mode choices in an urban setting. The study is novel in using a wide set of environmental indicators, spatially-differentiated intake fractions, and an effect factor that estimates 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.

1.1 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 (Lim et al., 2012; Hänninen et al., 2014; World Health Organization, 2003, 2013; US Environmental Protection Agency, 2009). 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 (PM₁₀), fine particles (PM_{2.5}), and ultrafine particles (UFP), which have aerodynamic diameters below 10, 2.5 and 0.1 µm, respectively (World Health Organization, 2003).

Spatially differentiated factors are needed to calculate the exposures and impacts of PM_{2.5} emissions (Joint Research Centre, 2011; Finnveden et al., 2009; UNEP/SETAC 2016, Fantke et al. 2017), especially in transportation applications where vehicle emissions in urban settings are mostly released at ground level with potentially greater impacts than emissions occurring in rural settings or those 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 would be spatially resolved to account for the distribution of emission sources and the locations of exposed and vulnerable individuals, especially since concentrations of traffic-related air pollutants display substantial intra-urban variation and steep concentration gradients (Baldwin et al., 2015; Batterman et al., 2010; Wilson et al., 2005; Brauer et al., 2000; Fischer et al., 2000; Rodriguez-Roman and Ritchie, 2016). However, detailed and accurate data on emissions, population demographics and health status 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 for addressing the spatial variation of population exposure to PM_{2.5}.

An emissions-based assessment using the intake fraction (iF) approach has been recommended for LCAs examining potential health impacts associated with air pollution (Fantke et al., 2015; Joint Research Centre, 2011), 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). 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). 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). Default iF values have been derived for PM and 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 has been recommended (Fantke et al., 2015; Joint Research Centre, 2011). 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; ischaemic heart disease (IHD); cerebrovascular disease; and chronic obstructive pulmonary disease (COPD). Gronlund et al. (2015) combined these aspects of 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 (Martenies et al., 2015; Hänninen et al., 2014).

The few LCA studies that have addressed both transportation and PM-related health effects have considered PM emissions across the life-cycle (LC), but have not differentiated 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 et al., 2013; Cooney et al., 2013; Bauer et al., 2015; Ercan & Tatari, 2015; François et al., 2017). While such analyses can represent emission inventories or the potential for PM formation, they do not accurately estimate the potential for human health impacts from PM emissions (Hauschild and Huijbregts, 2015).

2. Materials and methods

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/km², and 64% of its population commutes to other municipalities, mostly to Lisbon (INE, 2013).

Distances for each transport mode (road, railway, subway) were measured using a base map of Lisbon from OpenStreetMaps database (OSM), 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 share of exclusively residential buildings, which ranges from 6 to 93% across parishes (INE, 2013)). 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 selected, and the distance between stations was considered.

Six transportation modes were considered: car, bus, subway, 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, 2013). 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, 2013), and bus, subway and train occupancies were 20, 130 and 205, respectively, based on the transport provider (Carris, 2012; Metropolitano de Lisboa, 2012; Comboios de Portugal, 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 1.023 million used one of the six modes considered (INE, 2013).

2.2 Life-cycle model

The system boundary was defined considering seven LC phases (Figure 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). Exhaust emissions during the use phase for each vehicle type and technology were calculated using the EMEP/EEA Inventory Guidebook (Nielsen, 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).

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 & Auto Informa, 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 increased 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 presently has few dedicated cycling routes. Similarly, infrastructure maintenance was omitted for motorcycles. Transport-related services, e.g., insurance, were not considered. The model was developed in SimaPro, a widely used LCA tool (Speck et al. 2015; Herrmann & Moltesen, 2015).

2.3 Life-cycle impact assessment

Table 1 lists the environmental categories and indicators considered, which were selected based on the impacts and metrics associated with transportation (European Commission, 2012, 2013; Litman, 2016) and recommendations for life-cycle impact assessment (LCIA; Joint Research Centre, 2011). Estimates of primary non-renewable energy (NRE) were based on the Cumulative Energy Demand method (Hischier et al., 2010); GHG used the IPCC method and a 100-year time horizon (Solomon et al., 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 for PM_{2.5} described below) follow recommendations for the European context (Joint Research Centre, 2011). Other impact categories, including human toxicity and eco-toxicity, are also potentially important for comparative assessment of transportation alternatives; however, currently recommended characterization methods for these indicators need improvements and should be used with caution (Joint Research Centre, 2011), and thus these categories have been excluded from the present analysis.

PM_{2.5} intake was calculated as the product of emissions and the intake fraction (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 an 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 SO₂, NO_x and NH₃ precursor urban emissions, respectively, and 0.79, 0.17 and 1.70 for SO₂, NO_x and NH₃ 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 DALYs per kg of PM_{2.5} inhaled, a value that includes cardiopulmonary disease and lung cancer (Gronlund et al., 2015).

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.

2.4 Scenario analysis

A scenario analysis illustrates the application and potential trade-offs of different 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.

3. Results

3.1 Comparison of commuting modes

Figure 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.

Non-renewable energy (NRE) and greenhouse gas (GHG) emissions were strongly correlated. For these categories, cars (diesel and petrol) generally had the highest impacts (10680-11330 MJ and 720-795 kg CO₂ eq), followed by motorcycles (5419-7640 MJ and 373-532 kg CO₂ eq), and public transit modes (2704-3993 MJ and 210-266 kg CO₂ 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 acidification (AC), cars had the largest impacts (2.7-3.2 molc H⁺ eq), closely followed by motorcycles and bus (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 terrestrial eutrophication (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 freshwater eutrophication (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 (76-84% of the overall LC); vehicle manufacture had large contributions for internal combustion modes (29-56%). For marine eutrophication (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 (225-243 g N eq).

Diesel cars and 2-stroke motorcycles resulted in the highest PM_{2.5} intake, averaging 6.1 and 9.3 mg/person-year, respectively; other modes had intakes below 2.9 mg/person-year. 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 of 78 DALYs/kg PM_{inhaled} was applied.

Unsurprisingly, the bicycling commuting mode had the lowest burden in each category. 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. However, cyclists can experience increased exposure to air pollutants while commuting (Ramos et al., 2016). An analysis of exposures and impacts on specific groups (e.g., cyclists) is beyond our present scope.

3.2 Effect of spatial differentiation

The health impact analysis separated urban and non-urban intake factors to address some of the spatial differences in the LCIA's exposure assessment. Figure 3 contrasts differences between spatially differentiated and undifferentiated approaches. Using 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

SO₂, NO_x and NH₃, 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}.

3.3 Commuting impacts in Lisbon

Table 2 shows annual commuting impacts in the greater Lisbon area for each mode with 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 CO₂ eq per year for commuting represents only 10% of emissions in the Lisbon area (based on a 2009 estimate of 6.37 million t CO₂ eq that includes transport, buildings (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 CO₂ per capita (Creutzit et al., 2012). Our estimate of about 0.5 t per capita includes only commuting; commercial, freight travel, and non-commuting travel 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 inventory and impact assessments.

3.4 Scenario analysis

Table 3 summarizes potential benefits of five strategies intended to reduce commuting impacts. All strategies were beneficial in each LC category except electric vehicles, which significantly increased freshwater eutrophication (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 across 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.

4. Discussion

4.1. NRE and GHG

NRE and GHG are the most commonly used impact categories in previous transportation LCAs. Often, other impact categories are not described, which limits the comparisons that

can be made with the literature. For cars, buses 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 USA (Ercan & Tatari, 2015), a difference that may in part be due to the lower service life and the hybrid model considered in their study. Our results were in line with recent results for cars in Europe (Girardi et al., 2015). Compared to a study 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 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 CO₂ emissions associated with internal combustion vehicles (30 to 174% higher than ours), likely due to higher emission rates associated with the fleet, but lower CO₂ 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. In particular, assumptions regarding service life and occupancy may contribute to a wide range of results for transit modes. (Additional inter-study comparisons are provided in the supplementary materials.)

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 2. Our results emphasize PM_{2.5} due to its health significance (Hoofman et al., 2016). In comparison to PM₁₀, 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 PM₁₀ 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 PM₁₀/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 PM₁₀/VKT (excluding road construction, maintenance and disposal), probably due to the electricity mix and vehicle characteristics. Ercan and Tatari (2015) estimated life-cycle PM₁₀ emissions of 0.7 g per VKT for diesel buses; the higher results are partially associated with a shorter service life (714 500 km, compared to our 1 000 000 km) and the hybrid model

implemented. Chester 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 emissions 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 plants in the electricity supply chain. The authors highlight the uncertainty associated with the human toxicity, acidification and PM formation indicators. 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. The effect factor considered included two commonly used 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 estimated for $PM_{2.5}$ and the North American population. Effect factor estimates for other locations and health endpoints can yield different and larger impacts, e.g., an estimate for PM_{10} and Europe (van Zelm et al., 2008) would increase impacts by 19% (Gronlund et al., 2015), while including all-cause mortality (EF of 110 DALY/kg $PM_{2.5}$ inhaled) would increase health impacts by 41% (Gronlund et al., 2015). A Lisbon-specific iF estimate might differ from the one used, reflecting differences in population density, meteorological parameters, pollutant mixture, population susceptibility, and other factors, however, the derivation of a Lisbon-specific EF was not feasible given the data available or within the scope of the paper.

In urban settings, $PM_{2.5}$ drives health impact estimates, causing substantially more DALYs than other pollutants (World Health Organization, 2016; Anenberg et al., 2010). However, health impacts are also caused by other transport-related air pollutants, including NO_2 , O_3 and CO. Thus, 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; Joint Research Centre, 2011; Pizzol et al., 2011). 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 beyond our scope. Similarly, the use of an archetype iF for ground-level outdoor emissions does not represent exposures occurring in trains, subways, while cycling or other microenvironments.

The health impact analysis also 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).

4.3 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 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 associated with 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 site-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 (Dong et al., 2015). The multi-mode comparison provides environmental impacts per year, and impacts per person-kilometer traveled are provided in supplemental materials. Trip length was based on transport provider documents, GIS measured distances and the assumption that a bus route would be 10% longer than the same trip by car. 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. At present, EV penetration in the Portuguese fleet is limited (Garcia et al., 2015), thus, EV vehicles 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 (50%) of renewable sources (over twice the average of the EU; European Commission, 2012).

5. Conclusion

This comparative LCA of six commuting modes 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 resulting from emissions in urban and non-urban settings using two intake fractions (iF) 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, and 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 among different strategies. While the results and conclusions of the Lisbon analysis may apply elsewhere, site-specific and detailed assessments are both recommended and practical for supporting policy making to encourage more sustainable urban development.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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Figure 1.
System boundary applied to the six transport modes.

Figure 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.

Figure 3. Life-cycle PM_{2.5} intake per person-year considering differentiated iFs separating urban and non-urban emissions, and a single (global) iF.

Table 1.

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	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

Table 2.

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.

	Total 1 022 785 p		Bicycle 1 818 p (%)	Train 94 438 p (%)	Subway 63 067 p (%)	Bus 190 416 p (%)	Car 663 942 p (%)	Motorcycle 9 104 p (%)
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

Table 3.

Analysis of 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	(%)	mole H ⁺ eq	(%)	mole 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	82	662	4.91	(80)
Occupancy increase	8180	(75)	560	(75)	2.03	(75)	6.14	77	620	4.60	(75)
Mode shift	3 993	(37)	266	(36)	1.69	(63)	7.83	17	718	1.89	(31)
Distance reduction	7 635	(70)	522	(70)	1.90	(70)	5.73	72	579	4.30	(70)
Other technology	5 677	(52)	394	(53)	2.11	(78)	4.28	324	411	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	25	366	4.63	(80)
Occupancy increase	4 427	(67)	307	(67)	1.16	(67)	3.37	21	305	3.85	(67)
Mode shift	2 827	(43)	240	(52)	0.81	(46)	2.58	93	243	0.48	(8)
Distance reduction	4 648	(70)	323	(70)	1.22	(70)	3.54	22	320	4.05	(70)
Other technology	1 964	(30)	135	(29)	0.87	(50)	1.60	122	151	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	14	574	1.51	(80)
Occupancy increase	2 662	(67)	177	(67)	1.13	(67)	5.22	11	479	1.26	(67)
Mode shift	474	(12)	36	(14)	0.20	(12)	0.37	1	35	0.17	(9)
Distance reduction	2 795	(70)	186	(70)	1.19	(70)	5.48	12	503	1.32	(70)
Other technology	3 014	(75)	214	(80)	0.94	(55)	2.60	130	245	0.45	(24)