Journal of Transport & Health 22 (2021) 101084

Contents lists available at ScienceDirect

Journal of Transport & Health

journal homepage: www.elsevier.com/locate/jth

A health economic assessment of air pollution effects under climate neutral vehicle fleet scenarios in Stockholm, Sweden

Hedi Katre Kriit[a,*], Johan Nilsson Sommar[a], Bertil Forsberg[a], Stefan Åström[b], Mikael Svensson[c], Christer Johansson[d,e]

[a] Department of Public Health and Clinical Medicine, Section of Sustainable Health, Umeå University, Umeå, Sweden
[b] IVL Swedish Environmental Research Institute, P.O. Box 53021, SE-400 14, Gothenburg, Sweden
[c] School of Public Health and Community Medicine, Institute of Medicine, University of Gothenburg, Sweden
[d] Atmospheric Science Unit, Department of Environmental Science and Analytical Chemistry, Stockholm University, 11418 Stockholm, Sweden
[e] Environment and Health Administration, SLB, Box 8136, 104 20 Stockholm, Sweden

A R T I C L E I N F O

Keywords:
Non-exhaust
Wear particles
Road dust
$PM_{2.5}$
PM_{10}
CO_2
Electric vehicles
Exhaust
Mortality
Morbidity
QALY
Costs

A B S T R A C T

Introduction: Electric vehicles (EVs) are heavily promoted as beneficial for climate and health. In most studies, it is assumed that EVs contribution to urban air pollution is zero due to no tailpipe emissions, ignoring the contribution of non-exhaust particles (brake, tire and road wear), which are unregulated in EU. This study of Stockholm, Sweden, aims to 1) assess how a future vehicle fleet impacts concentrations of particles of size less than 2.5 μm ($PM_{2.5}$) and evaluate the expected health outcomes economically and 2) compare this with CO2 savings.
Methods: Source specific dispersion models of exhaust and non-exhaust $PM_{2.5}$ was used to estimate the population weighted concentrations. Thereafter exposure differences within a business as usual (BAU2035) and a fossil free fuel (FFF2035) scenario were used to assess expected health and economic impacts. The assessment considered both exhaust and non-exhaust emissions, considering the vehicle weight and the proportion of vehicles using studded winter tires. Health economic costs were retrieved from the literature and societal willingness to pay was used to value quality-adjusted life-years lost due to morbidity and mortality.
Results: The mean population weighted exhaust $PM_{2.5}$ concentration decreased 0.012 μg/m^3 (39%) in FFF2035 as compared to BAU2035. Assuming 50% higher road and tire wear $PM_{2.5}$ emission because of higher weight among EVs and 30% less brake wear emissions, the estimated decrease in wear particle exposures were 0.152 (22%) and 0.014 μg/m^3 (1.9%) for 0 and 30% use on studded winter tires, respectively. The resulting health economic costs were estimated to €217M and €32M, respectively. An increase by 0.079 μg/m^3 (11%) was however estimated for 50% use of studded winter tires, corresponding to an €89M increase in health costs.
Conclusion: Considering both exhaust and wear generated particles, it is not straight forward that an increase of EVs will decrease the negative health impacts.

1. Introduction

The United Nations (UN) Sustainable Development Goal framework (SDGs) aim to achieve climate neutrality and decrease the

* Corresponding author.
E-mail address: hedi.kriit@umu.se (H.K. Kriit).

https://doi.org/10.1016/j.jth.2021.101084
Received 2 June 2020; Received in revised form 28 April 2021; Accepted 9 May 2021

hazardous health effects without compromising the global economy (United Nations, 2015). It is acknowledged that activities proposed to achieve one goal might conflict with another within the SDGs framework. This has however rarely been considered when analyzing effects of climate policy proposals. For example, it is almost automatically assumed that solution to achieve climate neutrality in the transport sector (e.g. low emissions zones (LEZ) etc.), by default reduces population exposure to hazardous air pollution levels. However, depending on the vehicle fleet composition and the pace of the shift to fossil free fuel, the future emissions composition may vary in time and more variables need to be accounted for in order to investigate the possible effects on urban air pollution levels and human health. Therefore, scientifically updated *(state of the art)* health impact assessments are needed to illustrate the possible health impacts and economic effects associated with policies aiming to achieve a sustainable urban transport.

In Europe the transport sector is responsible for a large economic turnover accounting for 5% of the unions gross domestic product (GDP) (European Union Science Hub, 2019). But, road transport is also responsible for 27% of greenhouse gas emissions on the continent, and hence a major driver of global warming (European Environmental Agency, 2018). Therefore, the transport sector is under pressure to become climate neutral before year 2045. The Swedish government has an ambitious goal to reduce greenhouse gas emissions from the transport sector by 70% compared with year 2010 (Ministry of Environment, 2017). Different scenarios that aim to guide a change in the transport fleet are developed and incentives to support the development towards set goals created.

One of the main proposed and heavily lobbied solutions within the transport sector is changing the existing fossil fuel dependent transport fleet to electric vehicles (EVs). The marketed positive effects of EVs are: reduced greenhouse gas emissions, reduced oil consumption and improvement in air pollution and noise levels in urban environments. A reduction in air pollution is expected due to zero tailpipe (exhaust) emissions. However, ignoring the impact of non-exhaust particles emissions, can lead to distorted conclusions that may mislead future policies.

Lately, due to stricter regulations, the tailpipe emissions have been reduced in Europe (European Environmental Agency, 2019). The urban background concentrations of particle matter of size 10 μm or less (PM_{10}) in cities like Stockholm have however not shown any significant decreasing trend during the last 10–15 years (Olstrup et al., 2018). Traffic generated emissions of non-exhaust particles (e.g. break, tire and road wear) have not been regulated, but constitute a large proportion of particular matter of 2.5 μm or less ($PM_{2.5}$) from traffic (Department for Environment, Food and Rural Affairs, 2019). The UK Air Quality Expert Group also concludes that non-exhaust particles will have implications the development of future air quality. A recent OECD rapport predicts an increase of $PM_{2.5}$ emissions with increased proportion of EVs (Farrow and Oueslati, 2020).

EVs are associated with a reduction in break wear emissions due to the regenerative braking system, since it does not rely on friction (Department for Environment, Food and Rural Affairs Defra and Air Quality Expert Group, 2019). However, tire and road wear emissions are expected to increase due to the heavier weight of EVs. In areas where brake wear constitutes the main proportion of non-exhaust emissions there might be a net decrease in emissions with more EVs on the roads. A recent review of scientific papers on the environmental aspects of EVs by Requia et al. (2018a) concluded that the overall benefits of EVs for reducing greenhouse gas emissions and human exposure depends on various factors, such as, the type of EVs, the source of energy generation, driving conditions, charging patterns, availability of charging infrastructure, government policies, and the climate of the region. Jacobson et al. (2005) found that EVs charged by renewable energy could save 3700 to 6400 lives annually in the US. Also, Christopher et al. (2014) show that the environmental health impacts may decrease by 50% if EVs are powered by natural gas, wind, water, or solar energy depending on the emissions from energy sources and the region (climate), however may increase by 80% compared to using conventional gasoline if EVs are powered with electricity from corn ethanol or coal as energy source. Assessments of air quality considering the change in formation of secondary PM and ozone as a result of higher proportion of EVs demonstrate environmental (Electric Power Research Institute, 2007; Nopmongcol et al., 2017; Schnell et al., 2019) and health (Christopher et al., 2014) benefits. According to Schnell et al. (2019) the average summer ozone abundances was expected to decrease with a larger proportion of EVs in the vehicle fleet in the U.S., regardless of the electricity source. However, reduced NOx emissions will decrease titration of ozone by nitrogen monoxide (NO), leading to increased levels of ozone in cities (Brinkman et al., 2010; Olstrup et al., 2018). Most studies have calculated impacts on peak ozone concentrations and $PM_{2.5}$, but without considering non-exhaust particle emission from vehicles as relevant for health impacts (Alhajeri et al., 2011; Brinkman et al., 2010; Nopmongcol et al., 2017; Thompson et al., 2011).

Although EVs have no tailpipe emissions, through tires and brakes, they still contribute to non-exhaust emissions together with road wear and suspension of road dust. The quantity of non-exhaust particles emitted from the roads is dependent of the vehicle weight. EVs are heavier than internal combustion engine vehicles due to their heavy batteries, and will thereby cause higher tire and road wear emissions. The link between vehicle weight and non-exhaust PM_{10} has been known for a while (Carslaw, 2006). Simons (2013) demonstrated the relationship between vehicle weight and non-exhaust particles and presented emission factors for brake, tire and road wear per kilogram of vehicle weight.

PM_{10} and $PM_{2.5}$ are the most commonly used measures of exposure in epidemiological studies to estimate the negative health effects of air pollution in the population. The health effect of ultrafine particles has in two consequent systematic reviews of epidemiological studies been found inconclusive (HEI, 2013; Ohlwein et al., 2019). Therefore, ultrafine particles are not used in health impact assessments (Segersson et al., 2017). Most serious health effects of particles, e.g. shortening of life, are mainly attributed to the fine ($PM_{2.5}$) fraction of PM_{10}, according to projects coordinated by WHO (Heroux et al., 2015).

In this study we assess how a fossil fuel free future vehicle fleet impacts on $PM_{2.5}$ emissions, through changes in both exhaust and non-exhaust emissions (brake, tire and road wear), and thereby evaluate the resulting changes in health economic costs. In addition, the resulting reduction in CO2 emissions was evaluated.

2. Materials and methods

2.1. $PM_{2.5}$ emissions and dispersion

The relation between local traffic emissions and exposure was estimated based on an atmospheric dispersion model and a diagnostic wind model (Segersson et al., 2017). Detailed particle exposure and impact assessment was performed for the Greater Stockholm area (model domain 35 km × 35 km) at a grid cell resolution of 100 m × 100 m. A climatology of 360 weather cases was used to obtain annual mean population exposure concentrations for $PM_{2.5}$ for every grid cell (for details see Segersson et al., 2017). Emissions outside this study domain were not considered.

Road networks were described with a high level of detail. Traffic volumes of heavy and light duty vehicles (with more than 1000 vehicles per day) were based on a combination of measurements and modelling. The vehicle fleet composition, i.e. share of petrol, diesel, ethanol and gas fueled vehicles of different Euro standards as well as electric and hybrid vehicles, was derived from the national vehicle registry. Emissions factors for PM-exhaust for different vehicle types, speeds and driving conditions for the years 2015 and 2035 were calculated based on HBEFA version 4.1 (The Handbook Emission Factors for Road Transport, 2019). Non-exhaust emissions include brake, tire and road wear, and have been estimated from a combination of measurements. The road wear emissions factor depends on the studded tire proportion in the vehicle fleet (Denby et al., 2013). Studded tires were assumed to be used during the winter season (November–April). The fraction of non-exhaust $PM_{2.5}$ was assumed to be 30% of non-exhaust PM_{10} (for further details, see Segersson et al. (2017)).

2.2. Future vehicle fleet scenario

The different vehicle fleet emissions scenarios are described in Table 1. Details on the vehicle fleet composition and emissions in the scenarios are provided in Supplement file 1 (See Table S1.1). We apply the same total vehicle transport kilometers in the scenarios and the same proportion of different vehicle types (busses, heavy good vehicles (HGV), light goods vehicle (LGV) and passenger cars (PC)). The proportion of EVs is much higher in the fossil fuel free (FFF2035) scenario: 15%, 3%, 53%, 43% for buses, HGV, LGV and PC in FFF2035 compared to 0%, 0%, 3% and 7% for the same vehicle types in business-as-usual (BAU2035).

Within in the FFF2035 two different assumptions were used to illustrate the weight contribution to total $PM_{2.5}$ concentrations: 1) a *FFF2035-0* calculation, which assumes no change in tire and road wear for EVs compared to conventional vehicles, but a decrease in brake wear by 30% for EVs and; 2) an *alternative FFF2035-1* calculation assuming 50% higher tire and road wear due to EV's higher weight in comparison to conventional vehicles, as well as a 30% decrease in brake wear among EVs. The proportion of studded winter tires among passenger cars was varied (0%, 30% and 50%) in order to demonstrate the impact on $PM_{2.5}$ concentrations. Currently, the use of studded winter tires is forbidden on 3 roads in central Stockholm, but despite this the proportion of studded tires during winter is around 25% on these roads and 30–40% along most other inner-city roads. On highways outside the inner city the proportion is around 50–60% (Brydolf et al., 2019). In other cities in Sweden the proportion of studded winter tires varies from around 40% in the south to more than 90% in the north. In central Oslo the proportion is less than 10% and in other Norwegian cities up to around 50% (Vegvesen, 2018).

Table 1
Future vehicle fleet scenarios. See Supplement file 1. Table S1.1 for details on vehicle compositions in scenarios.

Scenario	Vehicle fleet composition
Business as usual 2035 (BAU2035)	The Swedish road traffic administration projection of the vehicle fleet 2035, where both fossil and fossil free vehicles ares present.
Fossil fuel free (FFF2035)	Fossil free vehicle fleet in Stockholm 2035 according to traffic experts at the Swedish Transport Administration and the City of Stockholm.

Table 2
Change in particle emissions factors for brake, tire and road wear among EVs compared to conventional vehicles. HGV = Heavy Goods Vehicles, LCV = Light Goods Vehicles, PC = Passenger Cars.

Vehicle type	Material	FFF2035-0	FFF2035-1
		Change (%) compared to non-electric vehicle	Change (%) compared to non-electric vehicle
Bus	Brake	-30	-30
Bus	Tire	0	+50
Bus	Road surface	0	+50
HGV	Brake	-30	-30
HGV	Tire	0	+50
HGV	Road surface	0	+50
LCV	Brake	-30	-30
LCV	Tire	0	+50
LCV	Road surface	0	+50
PC	Brake	-30	-30
PC	Tire	0	+50
PC	Road surface	0	+50

Table 3
Emissions factors for CO_2 in BAU2035. PC = Passenger car, HGV = Heavy Goods Vehicles, LCV = Light Goods Vehicles.

Vehicle category	Power technology	gCO_2/km
Bus	ethanol	0
Bus	electricity	0
Bus	diesel	0
Bus	CNG	93
HGV	petrol	287
HGV	diesel	457
LCV	petrol	119
LCV	electricity	0
LCV	diesel	119
PC	petrol	122
PC	electricity	0
PC	diesel	89
PC	flex-fuel E85	135
PC	biofuel CNG/petrol	10

2.3. *Non-exhaust emissions comparing EVs with conventional vehicles*

The EVs impact on non-exhaust emissions was based on results by Simons (2013), where they compared non-exhaust particle emissions between large (2000 kg) and medium (1600 kg) sized vehicles. The 25% higher weight resulted in a 50% increase of $PM_{2.5}$ non-exhaust emissions. An approximate 25% higher weight among EVs compared with their conventional counterparts was reported by Timmers and Achten (2016). According to vehicle specifications for the most sold EVs in Sweden, their weight ranged between approximately 1500-2500 kg (Bil Sweden, 2020). Brake wear was assumed to be 30% lower among EVs due to re-generative breaking. These percentual changes in non-exhaust emissions were used also for other vehicle types, and shown in Table 2.

2.4. *Tailpipe CO_2 emissions*

The fossil fuel tailpipe CO_2 emissions are based on HBEFA 3.1 for both year 2015 and year 2035. The increased proportion of biofuels in diesel, gas and petrol was accounted for in the BAU2035 scenario (as reported to the Swedish Government by the Swedish Transport Administration 2019, Håkan Johansson, personal communication). The resulting emissions factors for CO2 are shown in Table 3. The tailpipe CO2 emissions were assumed to be zero for the EVs. For comparison with other studies on CO_2 abatement costs and social costs of carbon, we also calculate the 'offset' CO2 cost required for the FFF2035 scenario to imply benefits for society.

2.5. *Population weighted $PM_{2.5}$ exposure*

For both scenarios, a population weighted annual mean concentration of $PM_{2.5}$ was calculated as the sum of all products of population times annual mean concentrations in each grid cell (100 m × 100 m) divided by the total population in the model domain. Since the same climatology was used in both scenarios the difference in exposure between the scenarios was due to the difference in emissions.

The population data was based on the coordinates for the home addresses of the total population, categorized into the age classes: 0–1 years, from 1 to 5 years and every five-year interval up to 85 years, and then a final class >85 years.

2.6. Adverse health outcomes and calculation of incident cases

The adverse health impacts due to air pollution exposure were estimated based on the Swedish ASEK report presenting the rationale, principle and values recommended by the Swedish Transport Administration to be used in social cost-benefit analyses (CBA) (Söderqvist et al., 2019). The following health outcomes attributable to $PM_{2.5}$ exposure was included in the health assessment: mortality from long-term exposure, stroke, myocardial infarction, diabetes type II, chronic obstructive pulmonary disease (COPD), childhood asthma, preterm birth and restricted activity days (RADs). The incident cases (Inc) of each adverse health outcome that was attributable to particle exposure was calculated as follows (Equation (1)):

Equation 1:

$$Inc = ERF\ x\ C\ x\ P\ x\ f$$

Where, the exposure-response function (ERF) is the linear increase in risk per concentration as typically assumed (Heroux et al., 2015), C is the population weighted concentration, and P is the number of individuals (calculated in groups defined by age and sex) and f is the baseline incidence. A part from preterm birth and childhood asthma, ERFs are only applied to individuals $\geq$30 years of age.

2.7. Health economic evaluation

2.7.1. Valuation of mortality

The valuation of mortality was based on a calculation of the number of QALYs lost, which was subsequently multiplied by a value per QALY to get the total economic value of the mortality. The current study applied 250k Euro (€) per QALY based on a Swedish population-based study (Persson and Olofsson, 2018). The total number of QALYs lost due to long-term $PM_{2.5}$ exposure was adjusted by the average age-specific health-related quality of life estimates among the general population in Sweden (Burström et al., 2001).

2.7.2. Economic costs of morbidity outcomes

A literature search using the *Pubmed* search engine was conducted to obtain health economic costs per case of all health outcomes. Search terms and its combinations were based on a previous study estimating the costs of air pollution for the National Health Service (NHS) in England (Pimpin et al., 2018). The search combinations were adjusted to primarily identify cost data with the most relevance for a Swedish policy setting (for more details see Supplement file S2. Table S2.1). The following inclusion criteria were applied: a) published between year 2005 and 2020; b) using register data and conducted in Sweden or in other European countries; c) outcome defined by ICD-code; d) costs represent the health outcome in question; e) report both formal health care costs and/or costs due to production loss.

Total health economic costs (TC) for each morbidity outcome were calculated as the sum of Net Present Value costs (NPV) per category (Equation (2)). The categories considered were formal health care costs, costs due to production loss and the value of a lost QALY. The TC for each cost category, calculated based on the duration of the morbidity outcome (Svensson, 2019). The calculation of NPV used a discount rate of 3.5% (Söderqvist et al., 2019) and was reported in € at the year 2017 value ($€_{2017}$). For detailed information of the TC calculation for included health outcomes see Supplement file S2.2.

Equation 2:

$$NPV_{cat} = \sum_{t=t_{inc}}^{T} \frac{c_{cat,t}}{(1+r)^{(t-t_{inc})}}$$

Where:

NPV = Net Present Value
t_{inc} = life-year when incidence occur
T = years of life expectancy
r = annual discount rate
$c_{cat,t}$ = cost category or sub-category

2.7.3. Valuation of loss of quality-adjusted life-years (QALYs)

To account for the costs associated with the premature mortality and morbidity, the total number of QALYs lost for each health outcome was calculated and subsequently multiplied by a value per QALY. For premature mortality, the lost QALYs was calculated as the expected number of lost life-years adjusted by health-related quality of life during the lost-life years. For morbidity, the lost QALYs was calculated as the reduction in health-related quality of life ("utility score") for each health outcome multiplied by the duration of the outcome. Data on health-related quality of life decrements for each health outcome category were based on a literature search by

Table 4
Health economic costs for each health outcome and the expected number of quality-adjusted life-years lost based on the average duration of disease.

Health outcome	Years of life lost/ Duration of a disease	Reference	Value of a QALY/Cost per case (€$_{2017}$)	Included costs	Reference	Number of QALYs lost	Reference
Mortality	11	Söderqvist et al. (2019)	250 000	WTP	(Persson and Olofsson, 2018)	9.7	Burström et al. (2001)
Stroke	10	Assumption	260 000	DMC + PL Informal care	Lekander et al. (2017) Persson et al. (2017)	2.85	Ghatnekar et al. (2013)
MI	1	Assumption	21 000	DMC + PL	Mourad et al. (2013)	0.09	Gencer et al. (2016)
COPD	19	Assumption	11 000	DMC[a]	Ställberg et al. (2018)	5.51	Arne et al. (2009)
Diabetes type II	9	Gudbjörnsdottir et al. (2017)	49 000	DMC + PL	Janssen et al. (2020)	1.5	Neumann et al. (2014)
Childhood asthma	17	Assumption	14 000	DMC + PL	Ferreira De Magalhães et al. (2017)	1.7	Bergfors et al. (2015)
Adult asthma	66	Assumption	166 000		Jansson et al. (2007)	9.24	Jansson et al. (2019)
Preterm birth	84	Assumption	103 000	DMC + PL	Mangham et al. (2009)	NA	NA
Restricted Activity Days	NA	NA	210	WTP + PL[b]	(Statistics Sweden Medellöner i Sverige, 2017; Ready et al., 2004)	NA	NA

[a] The average onset of COPD is estimated to be at 65 years of age, and therefore no productivity loss was estimated.
[b] PL – productivity loss as average daily salary in Sweden year 2017.

applying the same criteria as mentioned above and in addition: f) utility weights were based on validated questionnaires such as EQ-5D or SF-36 (See Supplement file S2. Table S2.3).

2.7.4. Calculating the health cost per scenario

To estimate the total health economic costs due to changes in population $PM_{2.5}$ exposure, each outcome was multiplied with the average cost per outcome. For intangible costs, cases were multiplied with the average loss of QALYs and then multiplied with the societal willingness to pay for a QALY (Table 4). The total cost per health outcome was then added to the value of lost QALYs. Lastly, the total health economic cost was calculated as the sum of costs for each outcome.

2.7.5. Uncertainty of exposure-response function and cost estimates

To account for uncertainties, Monte Carlo simulations including uncertainties related to both relative risks and cost estimates for all health outcomes were conducted. For the epidemiological ERFs the standard deviation for log estimates of relative risks was extracted from the original studies. For costs estimates, if the original study did not report any uncertainty estimates, a ±20% range for the estimated costs was applied (Briggs et al., 2012). Uncertainty estimates were missing for the following health outcome costs: mortality, stroke, myocardial infarction and diabetes type II. A gamma-distribution was applied to describe the distribution of cost estimates. One thousand random iterations were generated, from which basic 95% confidence intervals were calculated.

2.7.6. Uncertainty of emissions

Sensitivity analyses including a wide range of emission factors were performed by considering: a) an increase of the proportion of personal EVs; b) an additionally increased road and tire wear among EVs; c) a decreased break wear emission from EVs due to regenerative braking (where tire and road wear were assumed to be 50% higher).

3. Results

3.1. Population weighted concentrations of $PM_{2.5}$

The population weighted average exhaust $PM_{2.5}$ concentration was estimated to decrease by 39% (0.012 μg/m^3) in a FFF2035 scenario compared to BAU2035 (Table 5).

The population exposure of wear generated $PM_{2.5}$ was dependent on the proportion of vehicles using studded winter tires and on the assumptions regarding the tire and road wear of EVs compared to conventional vehicles. Assuming no increase in tire and road wear but 30% decrease in brake wear with EVs compared to conventional vehicles (Table 5. FFF2035-0), the overall $PM_{2.5}$ exposure decreases in a FFF2035 scenario compared to BAU2035 even with 50% use of studded winter tires. If the tire and road wear was 50% higher for EVs there was an increase in total $PM_{2.5}$ exposure when the studded winter tire proportion was 50%, but decrease for a 30% and 0% studded winter tire proportion.

Table 5
Population weighted average $PM_{2.5}$ concentration differences (μg/m^3) and resulting health economic costs comparing FFF 2035 with BAU 2035 for different proportions of studded winter tires.

Difference between FFF2035 and BAU2035 (negative values means decrease)					
Proportion of vehicles using studded winter tires	FFF-BAU case	$PM_{2.5}$-exhaust μg/m^3 (%)	$PM_{2.5}$-wear μg/m^3 (%)	Net total μg/m^3 (%)	Annual economic costs as central estimates €$_{2017}$
0%	FFF2035–0	-0.012 (-39%)	-0.167 (-24%)	-0.178 (-24%)	-237 000 000
	FFF2035–1 Increased road and tire wear	-0.012 (-39%)	-0.152 (-22%)	-0.163 (-22%)	-217 000 000
30%	FFF2035–0	-0.012 (-39%)	-0.075 (-10%)	-0.086 (-12%)	-114 000 000
	FFF2035–1 Increased road and tire wear	-0.012 (-39%)	-0.014 (-1.9%)	-0.024 (-3.3%)	-32 000 000
50%	FFF2035–0	-0.012 (-39%)	-0.013 (-1.8%)	-0.025 (-3.3%)	-33 000 000
	FFF2035–1 Increased road and tire wear	-0.012 (-39%)	0.079 (+11%)	0.067 (+8.7%)	89 000 000

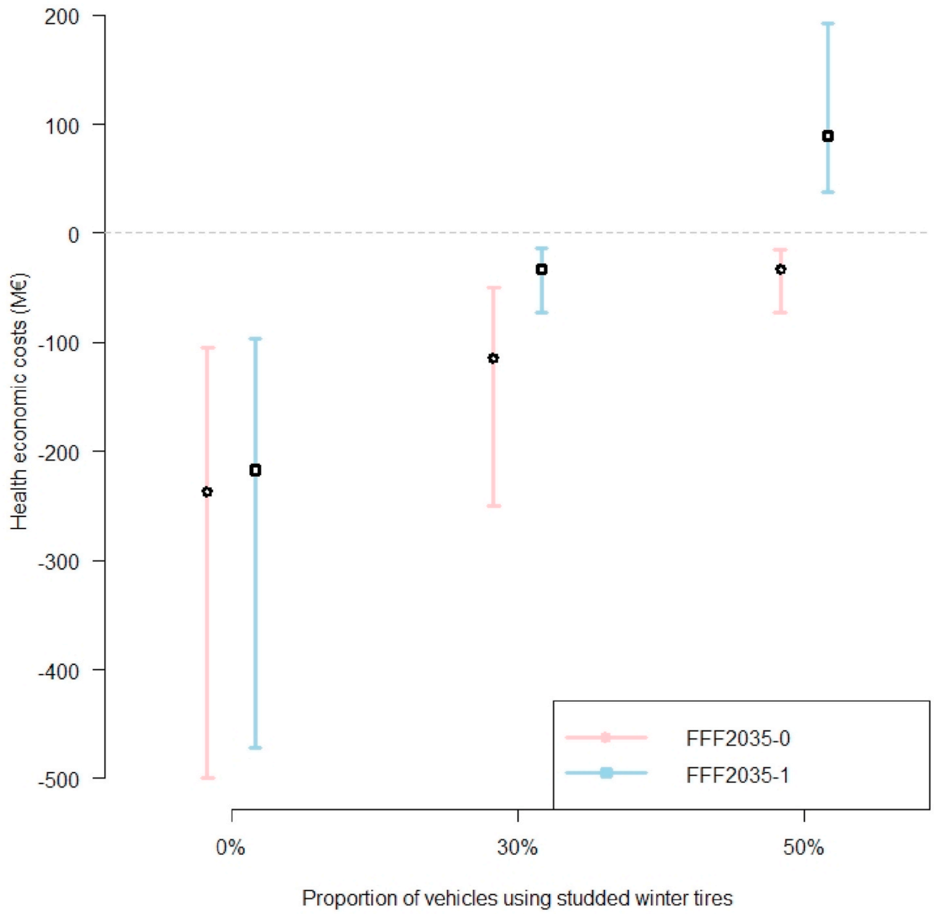

Fig. 1. Health economic costs differences between vehicle fleets in FFF2035 and BAU2035 scenarios in relation to the proportion of vehicles using studded winter tires. Results were presented for: FFF2035-0 assuming no impact of EVs weight on road and tire wear; and FFF2035-1 assuming that road and tire wear increases by a factor 1.5 among EVs. Error bars indicate 95% confidence intervals from Monte Carlo simulations.

3.2. Economic costs associated with changes in total $PM_{2.5}$ exposure

The total $PM_{2.5}$ exposure resulting adverse impact on human health was estimated to correspond to economic costs of about €1 billion (95% CI €460M-2.2 billion) given a BAU2035 scenario, and the differences of costs between FFF2035 and BAU2035 was estimated to be €89M higher with 50% use of studded winter tires, but estimated to be €32M and €217M lower with 30% or no use of studded winter tires, respectively (Fig. 1.). The resulting economic cost estimated due to road and tire wear increase by 50% for EVs was €20M given 0% winter studded tires. However, when assuming 50% use of studded winter tires, health economic costs was

Table 6
Population average $PM_{2.5}$ concentration differences and estimated health economic costs attributable to the extra weight among EVs.

Proportion of passenger vehicles using studded winter tires	Difference between net total $PM_{2.5}$ concentration (μg/m^3) ("FFF2035-1"– "FFF2035-0")	Health economic cost estimate applicable to EVs (€$_{2017}$)
0%	0.015	20 000 000
30%	0.062	81 000 000
50%	0.092	122 000 000

estimated to increase by €122M (Table 6).

3.3. Greenhouse gas emissions

In year 2015 the tailpipe emissions of CO2 was 1.5 million ton yearly and was estimated to increase to 2.3 million ton by 2035 given a BAU2035 scenario. In the FFF2035 scenario, the emissions was estimated to be 0.7 million ton lower compared to BAU2035, thereby not entirely offsetting the increase in emissions between the years 2015 and 2035. The annual health economic costs of the effects on $PM_{2.5}$ emissions thus corresponded to range between €-350 and €+132 per ton CO2 reduction.

3.4. Sensitivity analysis (Supplement file 3. Sensitivity analysis)

3.4.1. Proportion of EVs

The difference in the net total $PM_{2.5}$ average concentration between a FFF2035 and BAU2035 scenario decreases with an increasing proportion of EVs in the future vehicle fleet, irrespective of the proportion of vehicles using studded winter tires (See Table S3.1.1). If only 1% of the vehicle fleet was assumed to be EVs, and 50% of the vehicles used studded winter tires, the health economic costs of a FFF2035 scenario was estimated to increase by 14% (corresponding to €103M) compared with BAU2035 (See Table S3.1.2). If 30% or 0% of studded winter tires were assumed, the health economic costs were estimated to decrease by 14.5% (corresponding to €37M) and 12.5% (corresponding to €241M), respectively.

3.4.2. Road and tire wear among EVs

The impact of increasing the multiplication factor for road and tire wear increased with an increased use of winter studded tires (See Table S3.2.1). When assuming an increase of 60% road and tire wear and 50% use of studded winter tires the health economic costs were estimated to increase by 21% (corresponding to €113M) in a FFF2035 compared to BAU2035 scenario. By assuming an increase of 60% road and tire wear a range of 9% (corresponding to €-238M) reduction to 1.5% (corresponding to €-214M) increase in health economic cost was estimated assuming 0% studded winter tires (See Table S3.2.2).

3.4.3. Varying the decrease of brake wear for EVs

The EVs regenerative braking system reduces the emissions of brake wear particles. If the brake wear particles emission factors in Table 2 were to be reduced by 70%, a 25% reduction in net total $PM_{2.5}$ was estimated given 0% use of studded winter tires (See Table S3.3.1). With 50% use of studded winter tires the net total $PM_{2.5}$ concentration was expected to increase in a FFF2035 scenario. For 0% use of studded winter tires, the health economic costs were estimated to decrease for all brake wear factors considered. For 50% use of studded winter tires, the range of change in health economic costs were estimated between a 26% decrease (corresponding to €65M) and a 13.5% increase (corresponding to €101M), compared to the main analysis (See Table S3.3.2).

4. Discussion

To the best of our knowledge this is the first health economic assessment estimating the impacts of a future fossil free vehicle fleet scenario that in addition considers changes in both exhaust and non-exhaust emissions. The findings demonstrate that non-exhaust particles dominate the net total $PM_{2.5}$ emissions from road traffic and becomes even more dominating with increased proportion of EVs. Important determinants for the amount of non-exhaust particles are the proportion of vehicles using studded winter tires. The increase of EVs in the FFF2035 scenario reduced the net total $PM_{2.5}$ concentrations, but because of the higher weight an increase in net total $PM_{2.5}$ was estimated if 50% of the passenger cars use studded winter tires. Therefore, a reduction of health economic costs for a FFF2035 scenario is estimated, but an increase in health economic costs when assuming 50% use of winter studded tires. The health economic costs attributable to increased use of EVs ranged between €20–122M depending on the assumption of increased wear among EVs due to the extra weight compared to conventional vehicles and the proportion of vehicles using winter studded tires.

The results of this study emphasise that the assumption that EVs by default will result in better air quality in urban environments and reduced socioeconomic cost of air pollution is much more complex. In Canada Requia et al. (2018b) estimated that with a 10% proportion of EVs in the vehicle fleet the $PM_{2.5}$ concentration would decrease by 46%, which is twice as large a decrease in relation to our study FFF2035-0 scenario. Similarly, Maesano et al. (2020) estimate that a hypothetical traffic scenario where 100% personal cars were EVs in Paris would reduce the PM_{10} concentration by 30%. Both studies are limited in their estimation of the non-exhaust contribution to total PM. Requia et al. 2018b do not account for road wear or EVs weight impact, and Maesano et al. (2020) accounted for secondary particles but not for tire wear or EVs weight impact on non-exhaust particles. In addition, both studies ignored the effect of studded winter tires, although both countries permit the use of such tires. However, these $PM_{2.5}$ concentrations are not directly comparable to ours, since these are annual average differences over the geographical areas, whereas this study presents population weighted averages. Our results highlight the importance to include road wear, EVs weight and the use of studded winter tires in the analysis when estimating the impact on net total PM concentration.

A study done in England to estimate the health economic benefits of implementing LEZs, included premature mortality and morbidity outcomes (coronary event, preterm birth, low-birth weight and childhood asthma) (Lomas et al., 2016). LEZs was estimated to reduce preterm mortality corresponding to 8.4 QALYs and morbidity effects to 3.5 QALYs. Valued monetarily, a reduction in health economic costs by £168 000 and £78 000 was estimated, respectively. The study highlighted the need to consider and assess the uncertainty of a wider range of stochastic variables, such as source-specific dispersion modelling, estimated reduction in emissions and inclusion of health effects and valuation of health outcomes (Lomas et al., 2016). Considering these additional factors in the evaluation of a FFF2035 scenario compared with BAU2035, we estimate 9.7 QALYs lost due to mortality and 20.9 QALYs lost due to morbidity effects. The health economic costs are however not directly comparable between studies since we used a ten times higher QALY valuation and since we include life-time costs related to morbidity outcomes. Our valuation of 20.9 QALYs are considerably higher than the estimate from Lomas et al. of 3.5 QALYs, due to the novel inclusion of long-term health outcomes compared to only short-term health effects attributable to annual $PM_{2.5}$ concentration.

LEZs allow EVs due to their zero tail-pipe emissions, and Lomas et al. (2016) estimate that LEZs zones would be a cost-effective intervention from health perspective due to improve air quality. The interventions that imply increase use of EVs would in their health economic assessments benefit from considering differences in road wear compared to conventional vehicles. The health economic costs due to the higher weight among EVs was in our study estimated to €20M, assuming no use of winter studded tires. Our assessment assumed that the vehicle weight has a linear relationship with non-exhaust particles based on Simons (2013). There are however no such measurement studies specifically for EVs.

The year 2035 CO2 emissions for a FFF2035 scenario was estimated to be 680 000 ton lower than in the BAU2035 scenario. Given 50% use of studded winter tires the health economic costs attributable to $PM_{2.5}$ exposure would imply that economic costs of CO2 should be larger than €131 per ton CO2 in order for the FFF2035 scenario to have benefits compered to CO2 costs. Note that this estimate was even before considering the implementation costs of a FFF2035 scenario. As a comparison, Nordhaus (2017) indicates social costs of carbon to range between €26–146 per ton CO2 in a baseline scenario and €311 per ton CO2 in a scenario that goes on a trajectory with temperature peaking at 2.5° (the highest possible model ambition, given the model constraints). On the other hand, if there would be no use of studded tires, the $PM_{2.5}$ emissions reductions associated with the use of EVs would add benefits for society corresponding to up to €320 per ton CO2.

Since the proportion of studded winter tires contribution to $PM_{2.5}$ concentration is very dominant and permitted in the Nordic countries, where governments also heavily subsidy the introduction of EVs, the cost-benefit analysis calculation should include the health economic costs that occur due to increased non-exhaust exposure to be able to fully assess the consequences of EVs from societal perspective. As a comparative example, depending on the proportion of winter studded tires, the Swedish subsidy of €0.55 per kg CO2 for a small EV (Lindberg and Kellberg, 2020) would additionally increase by €0.13 per kg CO2 given the difference in health economic costs between a FFF2035 and BAU2035 scenario, assuming 50% of use of winter studded tires, which is the current proportion used in Stockholm (Brydolf et al., 2019). However, in a setting with no use of studded winter tires, a health economic cost reduction is estimated by €0.32 per kg CO2. Also, driving with studded winter tires have been shown to increase the CO2 emissions due to increased fuel consumption compared to the use of friction tires (Carlsson et al., 1995; Zubeck et al., 2004) This has not been accounted for in the current study; and therefore, cost estimates per ton CO2 could be somewhat lower for scenarios with studded winter tires. The use of studded winter tires is thereby an important issue to consider both from a health and climate perspective.

In this study, our estimation of the road traffic impact on $PM_{2.5}$ and CO2 was based on kilometers driven, both for end-of-pipe emissions and the impact on non-exhaust particles. In addition to these impacts, the emissions of $PM_{2.5}$ and CO2 related to production of new EVs and the energy source used to charge EVs have previously been estimated (Schnell et al., 2019). Net negative emissions of CO2 during the vehicle lifetime might occur in Sweden, since electricity is produced mainly in hydro- and nuclear power stations, whereas the same assumption may not hold in Eastern Europe or US where much of the energy is derived from coal (Requia et al. 2018a; Schnell et al., 2019). From a life-cycle perspective the EVs have shown positive effect with regards to curbing CO2 emissions compared to conventional vehicles. Although, previous life-cycle studies estimating the impact from road traffic $PM_{2.5}$ only consider exhaust emissions (Luk et al., 2015). This study however, shows the importance of including the non-exhaust particles.

4.1. Strengths and limitations

Our novel use of the QALY measure in valuating mortality and morbidity due to air pollution exposure is an improvement of environmental health economic impact assessments. Using the QALY measure facilitates comparison with other health interventions from a societal perspective. However, it is important to acknowledge that the QALY value varies between sectors, where for instance the Swedish Dental and Pharmacy Agency uses a roughly five times lower QALY value to establish cost-effectiveness valuations of drugs and interventions in the health care sector. The appropriate threshold value reference point depends on the perspective of the decision-makers. Within the health care sector, the interventions are disease specific and individual based assuming a fixed budget with marginal cost-effectiveness. If a societal perspective is applied a flexible budget is often assumed and therefore relevant for survey for the willingness-to-pay for a QALY (Brouwer et al., 2019). The later approach follows the welfare theory and the consumer sovereignty principle and reflects e.g. how environmental and transport sector decides what is affordable. As discussed by Olofsson et al. (2019) the budgets are rarely completely flexible in reality and therefor the higher QALY value based on willingness-to-pay should be considered as one important step towards improved decision-making process, but not sufficient.

Previous air pollution valuations and policy documents have used the Value of Statistical Life or Value of a Life Year to value mortality (Hurley et al., 2005), which has been a debated praxis in the literature. Moreover, most of the cost-of-illness studies used as the bases for estimating costs attributable to air pollution have not accounted for the valuation of QALYs lost due to morbidity outcomes (Pervin et al., 2008), thus underestimating the costs. This study attempts to include as many cost components as possible e.g. including also informal care for stroke survivals, which has been recognized as a limitation in previous assessments (Pervin et al., 2008).

Regarding the choice of $PM_{2.5}$ as exposure measure for estimating health effects, an alternative assumption to instead use PM_{10} and the ERF for PM_{10}, would not result in much different results than with $PM_{2.5}$. This because the studies estimating both ERFs report 2–4 times higher increase in risk per mass concentration of $PM_{2.5}$ for major effects such as MI and stroke (Cesaroni et al., 2014; Stafoggia et al., 2014), and for mortality 6 times higher increase per mass concentration for $PM_{2.5}$ than for the coarse fraction ($PM_{2.5}$ -PM_{10}) (Lefler et al., 2019).

Moreover, the health economic assessments have mainly focused on short-term health effects based on the available epidemiological knowledge (Lomas et al., 2016), which underestimates the total societal costs due to $PM_{2.5}$ exposure. In this study an economic valuation of lifetime costs with regards to chronic morbidity outcomes was applied, which is consistent with the latest epidemiological research linking $PM_{2.5}$ exposure to chronic morbidity incidences (Thurston et al., 2017). A limitation estimating long term air pollution effects on chronic illness is lack of epidemiological studies assessing the lag-times between exposure and health outcomes.

An assumption was made that all wear generated particles (road, tire and brake wear) become airborne, i.e. we do not consider the differences in particle sizes of the wear generated particles and that some of the particle fraction will be deposited on road surface. Particles may be washed off the roads and never suspended into the air (Denby et al., 2013). We further acknowledge that the FFF2035 scenario considered, assuming 40% of personal vehicles to be EVs, is far from the currently 1% pure EVs and 4% electric hybrids in Sweden vehicle fleet (Bil Sweden, 2019). In terms of vehicle weights the consumer preference is moving towards heavier SUVs instead of smaller light weight vehicles, which would cause additionally higher exposure to non-exhaust particles as compared to our scenario.

We only consider health effects of local, primary particle emissions in the Greater Stockholm area, whereas reductions of vehicle emissions of NOx and hydrocarbons will also lead to reductions in the formation of secondary particles and ozone (Requia et al. 2018a; Schnell et al., 2019). The population affected by secondary particle and ozone exposure may be much larger, but stretches outside of our study domain. However, the epidemiological research demonstrates stronger health risks associated with exposure to local, primary pollutants compared to secondary particles and ozone (Jerrett et al., 2005; Turner et al., 2016). To further improve environmental health economic assessments of increased use of EVs, both primary and secondary $PM_{2.5}$ emissions could be included to assess the population exposure.

5. Conclusion

Many studies have included rather uncertain effects of formulation of secondary pollutants, whereas primary non-exhaust particles have not been previously discussed. This is the first study to examine health and economic effects associated with both exhaust and non-exhaust emissions with fine spatial resolution. The findings demonstrate that non-exhaust particles dominate the net total $PM_{2.5}$ emissions from road traffic and becomes even more dominating with increased proportion of EVs. Health economic assessments evaluating EVs impact on total $PM_{2.5}$ concentration should consider the effect on non-exhaust particles to be able to fully evaluate the potential societal impacts of EVs.

Author contributions

Hedi Katre Kriit: Conceptualization, Data curation, Software, Formal analysis, Investigation, Methodology, Project administration, Validation, Visualization, Roles/Writing - original draft, Writing - review & editing.

Johan Nilsson Sommar: Supervision, Visualization, Methodology, Validation, Roles/Writing - original draft, Writing - review & editing.

Bertil Forsberg: Conceptualization, Data curation, Formal analysis, Funding acquisition, Methodology, Project administration, Resources, Supervision, Validation, Roles/Writing - original draft, Writing - review & editing.

Mikael Svensson: Data curation, Formal analysis, Investigation, Methodology, Validation, Writing - review & editing.

Stefan Åström: Methodology, writing – original draft.

Christer Johansson: Conceptualization, Data curation, Software, Formal analysis, Investigation, Methodology, Supervision, Validation, Visualization, Roles/Writing - original draft, Writing - review & editing.

Funding

We thank the City of Stockholm, Stockholms Luft-och Bulleranalys, and The Medical Faculty at Umeå University (Strategic grant for PhD student position) for making this study possible. BF and SÅ were also financed by the NordForsk project, Nordic WelfAir (grant no. 75007).

Declaration of competing interest

None.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jth.2021.101084.

References

Alhajeri, N.S., McDonald-Buller, E.C., Allen, D.T., 2011. Comparisons of air quality impacts of fleet electrification and increased use of biofuels. Environ. Res. Lett. 6, 024011.

Arne, M., Janson, C., Janson, S., Boman, G., Lindqvist, U., Berne, C., Emtner, M., 2009. Physical activity and quality of life in subjects with chronic disease: chronic obstructive pulmonary disease compared with rheumatoid arthritis and diabetes mellitus, 2009 Scand. J. Prim. Health Care 27 (3), 141–147, 27.

Bergfors, S., Åström, M., Burström, K., Egmar, A.C., 2015. Measuring health-related quality of life with the EQ-5D-Y instrument in children and adolescents with asthma. Acta Paediatr. 104, 167–173.

Bil Sweden, 2019. Tidsserier nyregistreringsstatistik, fordon i trafik 31/12 2019 fördelade på drivmedel. http://www.bilsweden.se/statistik/tidsserier. (Accessed 21 October 2020).

Bil Sweden, 2020. Statistik. Available from: https://www.bilsweden.se/statistik/nybilskopare-personbilar. (Accessed 23 October 2020).

Briggs, AH., Weinstein, MC., Fenwick, EAL., Karnon, J., Sculpher, MJ., Paltiel, AD., 2012. Model Parameter Estimation and Uncertainty Analysis: A Report of the ISPOR-SMDM Modeling Good Research Practices Task Force Working Group–6. Med Decis Making 32 (5), 722–732.

Brinkman, G.L., Denholm, P., Hannigan, M.P., Milford, J.B., 2010. Effects of plug-in hybrid electric vehicles on ozone concentrations in Colorado. Environ. Sci. Technol. 44.

Brouwer, W., van Baal, P., van Exel, J., Versteegh, M., 2019. When is it too expensive? Cost-effectiveness thresholds and health care decision-making. Eur. J. Health Econ. 20, 175–180.

Burström, K., Johannesson, M., Diderichsen, F., 2001. Health-related quality of life by disease and socio-economic group in the general population in Sweden. Health Pol.(Amsterdam, Netherlands) 55, 51–69.

Brydolf, M., Elmgren, M., Norman, M., 2019. Share of Studded Winter Tires in Stockholm (only in Swedish). Environmental and Health Administration. Stockholms Luft- Och Bulleranalys(SLB). Box 8136, 10420, (Stockholm, Sweden).

Carlsson, A., Centrell, P., Öberg, G., 1995. Studded Tyres: Socio-economic Calculations.

Carslaw, D.C., 2006. New Directions: a heavy burden for heavy vehicles: increasing vehicle weight and air pollution. Atmos. Environ. 40, 1561–1562.

Cesaroni, G., Forastiere, F., Stafoggia, M., Andersen, Z.J., Badaloni, C., Beelen, R., Caracciolo, B., de Faire, U., Erbel, R., Eriksen, K.T., Fratiglioni, L., Galassi, C., Hampel, R., Heier, M., Hennig, F., Hilding, A., Hoffmann, B., Houthuijs, D., Jockel, K.H., Korek, M., Lanki, T., Leander, K., Magnusson, P.K., Migliore, E., Ostenson, C.G., Overvad, K., Pedersen, N.L., J, J.P., Penell, J., Pershagen, G., Pyko, A., Raaschou-Nielsen, O., Ranzi, A., Ricceri, F., Sacerdote, C., Salomaa, V., Swart, W., Turunen, A.W., Vineis, P., Weinmayr, G., Wolf, K., de Hoogh, K., Hoek, G., Brunekreef, B., Peters, A., 2014. Long term exposure to ambient air pollution and incidence of acute coronary events: prospective cohort study and meta-analysis in 11 European cohorts from the ESCAPE Project. BMJ: British Medical Journal 348, f7412.

Christopher, W.T., Jason, D.H., Julian, D.M., 2014. Life cycle air quality impacts of conventional and alternative light-duty transportation in the United States. Proc. Natl. Acad. Sci. Unit. States Am. 111, 18490.

Denby, B.R., Sundvor, I., Johansson, C., Pirjola, L., Ketzel, M., Norman, M., Kupiainen, K., Gustafsson, M., Blomqvist, G., Omstedt, G., 2013. A coupled road dust and surface moisture model to predict non-exhaust road traffic induced particle emissions (NORTRIP). Part 1: road dust loading and suspension modelling. Atmos. Environ. 77, 283–300.

Department for Environment, Food and Rural Affairs (Defra) Air Quality Expert Group, 2019. Non-Exahust Emissions from Road Traffic. Department of Environment, Food and Rural Affairs.

Electric Power Research Institute, 2007. Environmental Assessment of Plug-In Hybrid Electric Vehicles, vol. 2: United States Air Quality Analysis Based on AEO-2006 Assumptions for 2030.

European Environmental Agency, 2018. Greenhouse Gas Emissions from Transport in Europe. https://www.eea.europa.eu/data-and-maps/indicators/transport-emissions-of-greenhouse-gases/transport-emissions-of-greenhouse-gases-11. (Accessed 11 December 2019).

European Environmental Agency, 2019. Emissions of Air Pollutants from Transport. https://www.eea.europa.eu/data-and-maps/indicators/transport-emissions-of-air-pollutants-8/transport-emissions-of-air-pollutants-8. (Accessed 21 October 2020).

European Union Science Hub, 2019. Transport Sector Economic Analysis. European Comission. https://ec.europa.eu/jrc/en/research-topic/transport-sector-economic-analysis. (Accessed 11 December 2019).

Farrow, K., Oueslati, W., 2020. Non-exhaust Emissions from Road Transport: Causes, Consequences and Policy Responses. OECD.

Ferreira De Magalhães, M., Amaral, R., Pereira, A.M., Sá-Sousa, A., Azevedo, I., Azevedo, L.F., Fonseca, J.A., 2017. Cost of asthma in children: a nationwide, population-based, cost-of-illness study. Pediatr. Allergy Immunol. 28, 683–691.

Gencer, B., Rodondi, N., Auer, R., Nanchen, D., Räber, L., Klingenberg, R., Pletscher, M., Jüni, P., Windecker, S., Matter, C.M., Lüscher, T.F., Mach, F., Perneger, T.V., Girardin, F.R., 2016. Health Utility Indexes in Patients with Acute Coronary Syndromes, 3, e000419.

Ghatnekar, O., Eriksson, M., Glader, E.-L., 2013. Mapping health outcome measures from a stroke registry to EQ-5D weights. Health Qual. Life Outcome 11.

Gudbjörnsdottir, S., Linder, E., Svensson, A.-M., Samuelsson, P., Eliasson, B., Miftaraj, M., Olofsson Eeg, K., 2017. Årsrapport - 2017. Nationella Diabetesregistret.

Heroux, M.E., Anderson, H.R., Atkinson, R., Brunekreef, B., Cohen, A., Forastiere, F., Hurley, F., Katsouyanni, K., Krewski, D., Krzyzanowski, M., Kunzli, N., Mills, I., Querol, X., Ostro, B., Walton, H., 2015. Quantifying the health impacts of ambient air pollutants: recommendations of a WHO/Europe project. Int. J. Publ. Health 60, 619–627.

Hurley, F., Hunt, A., Cowie, H., Holland, M., Miller, B., Pye, S., Watkiss, P., 2005. Methodology paper (volume 2) for service contract for carrying out cost-benefit analysis of airquality related issues. In: Particular in the Clean Air for Europe (CAFE) Programme. European Commission DG Environment.

Jacobson, M.Z., Colella, W.G., Golden, D.M., 2005. Cleaning the air and improving health with hydrogen fuel-cell vehicles. Science (New York, N.Y.) 308, 1901.

Janssen, L.M.M., Hiligsmann, M., Elissen, A.M.J., Joore, M.A., Schaper, N.C., Bosma, J.H.A., Stehouwer, C.D.A., Sep, S.J.S., Koster, A., Schram, M.T., Evers, S.M.A.A., 2020. Burden of disease of type 2 diabetes mellitus: cost of illness and quality of life estimated using the Maastricht Study. Diabet. Med. J. Br. Diabet. Assoc. 37 (10), 1759–1765. https://doi.org/10.1111/dme.14285.

Jansson, S.-A., Rönmark, E., Forsberg, B., Löfgren, C., Lindberg, A., Lundbäck, B., 2007. The economic consequences of asthma among adults in Sweden. Respir. Med. 101, 2263–2270.

Jansson, S., Hedman, L., Stridsman, C., Axelsson, M., Kriit, H.K., Lindberg, A., Lundbäck, B., Rönmark, E., Bacman, H., 2019. Life-years Lost due to Asthma. ERS International Congress 2019, Madrid, Spain.

Jerrett, T.M., Burnett, A.R., Ma, B.R., Pope, E.C., Krewski, J.D., Newbold, J.K., Thurston, J.G., Shi, J.Y., Finkelstein, J.N., Calle, J.E., Thun, J.M., 2005. Spatial analysis of air pollution and mortality in Los Angeles. Epidemiology 16, 727–736.

Lefler, J.S., Higbee, J.D., Burnett, R.T., Ezzati, M., Coleman, N.C., Mann, D.D., Marhall, J.D., Bechel, M., Wang, Y., Robinson, A.L., Arden Pope, C., 2019. Air pollution and mortality in a large, representative U.S. cohort: multiple-pollutant analyses, and spatial and temporal decompositions. Environ. Health 18 (1), 101.

Lekander, I., Willers, C., von Euler, M., Lilja, M., Sunnerhagen, K.S., Pessah-Rasmussen, H., Borgstrom, F., 2017. Relationship between functional disability and costs one and two years post stroke. PLoS One 12, e0174861.

Lindberg, H., Kellberg, C., 2020. Statliga åtgärder För Fler miljöbilar. Riksrevisionen.

Lomas, J., Schmitt, L., Jones, S., McGeorge, M., Bates, E., Holland, M., Cooper, D., Crowther, R., Ashmore, M., Rojas-Rueda, D., Weatherly, H., Richardson, G., Bojke, L., 2016. A pharmacoeconomic approach to assessing the costs and benefits of air quality interventions that improve health: a case study. BMJ Open 6, e010686.

Luk, J.M., Saville, B.A., Maclean, H.L., 2015. Life cycle air emissions impacts and ownership costs of light-duty vehicles using natural gas as a primary energy source. Environ. Sci. Technol. 49, 5151.

Maesano, C.N., Morel, G., Matynia, A., Ratsombath, N., Bonnety, J., Legros, G., Da Costa, P., Prud'Homme, J., Annesi-Maesano, I., 2020. Impacts on human mortality due to reductions in PM10 concentrations through different traffic scenarios in Paris, France. Sci. Total Environ. 698.

Mangham, L.J., Petrou, S., Doyle, L.W., Draper, E.S., Marlow, N., 2009. The cost of preterm birth throughout childhood in England and Wales. Pediatrics 123, e312.

Ministry of Environment, 2017. The Climate Policy Framewrok. https://www.government.se/articles/2017/06/the-climate-policy-framework/. (Accessed 26 October 2020).

Mourad, G., Alwin, J., Strömberg, A., Jaarsma, T., 2013. Societal costs of non-cardiac chest pain compared with ischemic heart disease – a longitudinal study. BMC Health Serv. Res. 13.

Neumann, A., Schoffer, O., Norström, F., Norberg, M., Klug, S.J., Lindholm, L., 2014. Health-related quality of life for pre-diabetic states and type 2 diabetes mellitus: a cross-sectional study in Västerbotten Sweden. Health Qual. Life Outcome 12.

Nopmongcol, U., Grant, J., Knipping, E., Alexander, M., Schurhoff, R., Young, D., Jung, J., Shah, T., Yarwood, G., 2017. Air quality impacts of electrifying vehicles and equipment across the United States. Environ. Sci. Technol. 51, 2830–2837.

Nordhaus, W.D., 2017. Revisiting the social cost of carbon. Proc. Natl Acad. Sci. U.S.A. 114, 1518 report.

Ohlwein, S., Kappeler, R., Kutlar Joss, M., Künzli, N., Hoffmann, B., 2019. Health effects of ultrafine particles: a systematic literature review update of epidemiological evidence. Int. J. Publ. Health 64, 547–559.

Olofsson, S., Gerdtham, U.G., Hultkrantz, L., Persson, U., 2019. Value of a QALY and VSI estimated with the chained approach. Eur. J. Health Econ. 20, 1063–1077.

Olstrup, H., Forsberg, B., Orru, H., Spanne, M., Nguyen, H., Molnár, P., Johansson, C., 2018. Trends in air pollutants and health impacts in three Swedish cities over the past three decades. Atmos. Chem. Phys. 18, 15705–15723.

Persson, U., Olofsson, S., 2018. Ett QALY är värt mer än två miljoner kronor (Läkartidningen).

Persson, J., Levin, L.-Å., Holmegaard, L., Redfors, P., Svensson, M., Jood, K., Jern, C., Blomstrand, C., Forsberg-Warleby, G., Forsberg-Wärleby, G., 2017. Long- term cost of spouses' informal support for dependent midlife stroke survivors. Brain Behav. 7.

Pervin, T., Gerdtham, U.-G., Lyttkens, C.H., 2008. Societal costs of air pollution-related health hazars: a review if methods and results. Cost Eff. Resour. Allocation: C/E 6, 19.

Pimpin, L., Retat, L., Fecht, D., de Preux, L., Sassi, F., Gulliver, J., Belloni, A., Ferguson, B., Corbould, E., Jaccard, A., Webber, L., 2018. Estimating the costs of air pollution to the National Health Service and social care: an assessment and forecast up to 2035.(Research Article). PLoS Med. 15, e1002602.

Ready, R., Navrud, S., Day, B., Dubourg, R., Machado, F., Mourato, S., Spanninks, F., Rodriquez, M., 2004. Benefit transfer in Europe: how reliable are transfers between countries? Environ. Resour. Econ. 29, 67–82.

Requia, W.J., Mohamed, M., Higgins, C.D., Arain, A., Ferguson, M., 2018a. How clean are electric vehicles? Evidence-based review of the effects of electric mobility on air pollutants, greenhouse gas emissions and human health. Atmos. Environ. 185, 64–77.

Requia, W., Arain, A., Koutrakis, P., Dalumpines, R., 2018b. Assessing particulate matter emissions from future electric mobility and potential risk for human health in Canadian metropolitan area. Air Quality, Atmos. Health 11, 1009–1019.

Schnell, J.L., Naik, V., Horowitz, L.W., Paulot, F., Ginoux, P., Zhao, M., Horton, D.E., 2019. Air quality impacts from the electrification of light-duty passenger vehicles in the United States. Atmos. Environ.

Segersson, D., Eneroth, K., Gidhagen, L., Johansson, C., Omstedt, G., Engström Nylén, A., Forsberg, B., Ryan, L., Anderson, C., 2017. Health impact of PM10, PM2.5 and black carbon exposure due to different source sectors in Stockholm, Gothenburg and Umea, Sweden. Int. J. Environ. Res. Publ. Health 14.

Simons, A., 2013. Road transport: new life cycle inventories for fossil-fuelled passenger cars and non-exhaust emissions in ecoinvent v3. Int. J. Life Cycle Assess. 21, 1299–1313.

Stafoggia, M., Cesaroni, G., Peters, A., Andersen, Z.J., Badaloni, C., Beelen, R., Caracciolo, B., Cyrys, J., de Faire, U., de Hoogh, K., Eriksen, K.T., Fratiglioni, L., Galassi, C., Gigante, B., Havulinna, A.S., Hennig, F., Hilding, A., Hoek, G., Hoffmann, B., Houthuijs, D., Korek, M., Lanki, T., Leander, K., Magnusson, P.K., Meisinger, C., Migliore, E., Overvad, K., Ostenson, C.G., Pedersen, N.L., Pekkanen, J., Penell, J., Pershagen, G., Pundt, N., Pyko, A., Raaschou Nielsen, O., Ranzi, A., Ricceri, F., Sacerdote, C., Swart, W.J., Turunen, A.W., Vineis, P., Weimar, C., Weinmayr, G., Wolf, K., Brunekreef, B., Forastiere, F., 2014. Long-term exposure to ambient air pollution and incidence of cerebrovascular events: results from 11 European cohorts within the escape project. Environ. Health Perspect. 122 (9), 919.

Statistics Sweden Medellöner i Sverige. 2017. URL https://www.scb.se/hitta-statistik/sverige-i-siffror/utbildning-jobb-och-pengar/medelloner-i-sverige/(accessed 26.10.2020).

Ställberg, B., Janson, C., Larsson, K., Johansson, G., Kostikas, K., Gruenberger, J.B., Gutzwiller, F.S., Jorgensen, L., Uhde, M., Lisspers, K., 2018. Real-world retrospective cohort study ARCTIC shows burden of comorbidities in Swedish COPD versus non-COPD patients. npj Prim. Care Respir. Med. 28.

Svensson, M., 2019. Hälsoekonomisk utvärdering: metod och tillämpningar, 1:1. Studentlitteratur AB, Lund.

Söderqvist, T.B.C., Kriit, H.K., Tidblad, J., Andersson, J., Jansson, S.-A., Svensson, M., Wallström, J., Andersson, C., Orru, H., Sommar, N.J., Forsberg, B., 2019. Underlag För Reviderade ASEK-värden För luftföroreningar: Slutrapport Från Projktet REVSEK (only in Swedish). Trafikverket, Trafikverket.

The Health Effects Institute (HEI), 2013. Understanding the Health Effects of Ambient Ultrafine Particles. In: HEI Perspectives, vol. 3. Health Effects Institute, Boston, MA.

The Handbook Emission Factors for Road Transport (HBEFA). Version HBEFA 4.1 from May 2019. http://www.hbefa.net/e/index.html. (Accessed 28 October 2020).

Thompson, T.M., King, C.W., Allen, D.T., Webber, M.E., 2011. Air quality impacts of plug-in hybrid electric vehicles in Texas: evaluating three battery charging scenarios. Environ. Res. Lett. 6, 024004.

Thurston, G.D., Kipen, H., Annesi-Maesano, I., Balmes, J., Brook, R.D., Cromar, K., De Matteis, S., Forastiere, F., Forsberg, B., Frampton, M.W., Grigg, J., Heederik, D., Kelly, F.J., Kuenzli, N., Laumbach, R., Peters, A., Rajagopalan, S.T., Rich, D., Ritz, B., Samet, J.M., Sandstrom, T., Sigsgaard, T., Sunyer, J., Brunekreef, B., 2017. A joint ERS/ATS policy statement: what constitutes an adverse health effect of air pollution? An analytical framework. Eur. Respir. J. 49.

Timmers, V.R.J.H., Achten, P.A.J., 2016. Non-exhaust PM emissions from electric vehicles. Atmos. Environ. 134, 10–17.

Turner, M.C., Jerrett, M., Pope, C.A., Krewski, D., Gapstur, S.M., Diver, W.R., Beckerman, B.S., Marshall, J.D., Su, J., Crouse, D.L., Burnett, R.T., 2016. Long-term ozone exposure and mortality in a large prospective study. Am. J. Respir. Crit. Care Med. 193, 1134.

United Nations, 2015. Transforming Our World: the 2030 Agenda for Sustainable Development. UN Publishing, New York.

Vegvesen, 2018. Piggfriandelen høyere enn noensinne. Oslo. https://www.vegvesen.no/om+statens+vegvesen/presse/nyheter/nasjonalt/piggfriandelen-hoyere-enn-noensinne. (Accessed 23 October 2020).

Zubeck, H., Aleshire, L., Harvey, S., Porhola, S., Larson, E., 2004. Socio-economic Effects of Studded Tire Use in Alaska. Institute of Socio-Economic Research, Univeristy of Alaska Anchorage.