

Health effects
caused by primary fine particulate matter (PM2.5) emitted from buses
in Helsinki Metropolitan Area, Finland

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Abstract

Fine particle ($PM_{2.5}$) emissions from traffic have been associated with premature mortality. The current work compares $PM_{2.5}$ induced mortality in alternative public bus-transportation strategies as being considered by the Helsinki Metropolitan Area Council, Finland. The current bus fleet and transportation volume is compared to four alternative hypothetical bus fleet strategies for the year 2020: (i) the current bus fleet for 2020 traffic volume; (ii) modern diesel buses without particle traps, (iii) diesel buses with particle traps, and (iv) buses using natural gas engines.

Average population $PM_{2.5}$ exposure level attributable to the bus emissions was determined for the 1996-97 situation using $PM_{2.5}$ exposure measurements including elemental composition from the EXPOLIS-Helsinki study and similar element-based source apportionment of ambient $PM_{2.5}$ concentrations observed in the ULTRA-study. Average population exposure to particles originating from the bus traffic in the year 2020 is assumed to be proportional to the bus emissions in each strategy. Associated mortality was calculated using dose-response relationships from two large cohort studies on $PM_{2.5}$ mortality from the U.S.

Estimated number of deaths per year (90% confidence intervals in parenthesis) associated with primary $PM_{2.5}$ emissions from buses in Helsinki Metropolitan Area in 2020 were 18 (0-55), 9 (0-27), 4 (0-14) and 3 (0-8) for the strategies (i) – (iv), respectively. The relative differences in the associated mortalities for the alternative strategies are substantial, but the number of deaths in the lowest alternative, the gas buses, is only marginally lower than what would be achieved by diesel engines equipped with particle trap technology. The dose-response relationship and the emission factors were identified as the main sources of uncertainty in the model.

Keywords: Risk assessment, public transport, compressed natural gas

Introduction

Much has been done to prevent health effects of air pollution by reducing emissions from critical sources. Despite major progress in emission controls¹⁻³, current urban air pollution still causes mortality and morbidity all over the world⁴⁻⁶. In particular, ambient particulate matter has been associated with adverse health effects even at the prevailing, relatively low, urban air concentrations. In addition to solid material, the ambient particles contain also volatile and liquid components and their chemical composition is very heterogeneous.

Adverse health effects have been seen for many particle size fractions, both in short-term (daily variation) and long-term studies⁴⁻⁶. In many epidemiological studies, the most potent effect has been linked to fine particulate matter (PM_{2.5}) (e.g.^{7,8}). There is some evidence for the existence of differences in toxic properties between particles from different sources^{9,10} but the mechanisms causing adverse health effects as well as the critical particle components are still unclear.

Particle emissions from many individual sources have been reduced; especially from sources related to energy production and industry^{1,3}. Recently, attention has focused on traffic-generated particulate matter. This includes both tail-pipe exhausts and particles from tyres and brakes. The need to develop low-emission vehicles has led to a number of improvements in engine design, fuel composition and particle trapping systems. Compressed natural gas (CNG) engines have been one of the new technologies introduced to lower emissions. At the same time, traditional diesel engine manufacturers have developed particle-trapping systems. As a consequence, authorities and decisions-makers now face the opportunity of choosing bus technologies from several options. To support the decision-making process, the Finnish Ministry of Social Affairs and Health commissioned a risk analysis to compare the health effects of particulate matter emissions from alternative bus technologies for use in the Helsinki Metropolitan Area.

To provide a proper answer to the authorities, a probabilistic risk analysis model was devised. It was based on the 'current fleet' –strategy, in which the year 2020 bus traffic is operated using a similar bus fleet as that in use in 1999, including buses with different types of diesel (some with particle traps) and gas engines¹¹. For comparison, three hypothetical bus strategies were defined: 'modern diesel', 'diesel

with particle trap' and 'natural gas bus' (CNG). In each of the hypothetical strategies it was assumed that all of the buses in the fleet would use the same kind of engine, fuel and particle traps. The aims of the this risk analysis are: (i) to provide an estimate of the statistical mortality due to fine particle emissions from buses in the study area, (ii) to compare PM_{2.5} emissions and health effects for different engine, fuel and particle traps, and (iii) to relate the exposure by buses to those caused by other fine particle sources. The possible mortality associated with non-particle air pollution was not accounted for, as it has been minimal compared to the effects of fine particles in previous studies^{12,13}. Economical factors were not included in our analysis.

Material and methods

Overview of the model

Helsinki Metropolitan Area Council has devised an alternative transportation system development scenarios for the year 2020¹⁴. We selected the public transportation intensive scenario for this work, because it was devised as an optimistic but plausible scenario about bus transportation growth. Population exposure to fine particles from bus traffic tail-pipe emissions was estimated for the year 1997 based on the *EXPOLIS*-Helsinki study¹⁵. The exposures from the alternative bus strategies for the year 2020 were then calculated by multiplying the year 1997 exposures by the growth estimate of public transportation (60 %)¹⁴ assuming that bus transportation also would expand at the same rate. Exposure differences between the bus strategies were assumed to be proportional to the estimated marginal tail-pipe emissions from the buses. This assumption was based on the hypothesis that primary fine particle emissions and exposure are related linearly over short distances. Emissions from street dust, brakes or tyres were assumed to be similar for all bus types and were not especially accounted for. Other factors, including population time activity, city structure and population density were assumed to remain constant. Additional mortality caused by fine particle emissions in each bus strategy was estimated by combining the exposure estimates and the cardio-pulmonary and lung-cancer background mortality in the target population with the dose-response functions obtained from two U.S. cohort studies^{16,17}.

Parameter uncertainty was propagated through the model by Monte Carlo simulation. Importance analyses were used to see how uncertainty in the input variables would affect the model outputs: rank-

order correlations between the input variables and the model outputs were calculated. The variables and uncertainty distributions included in the importance analysis are summarized in table 1. The whole model was implemented using Analytica™ version 3 (Lumina Decision Systems, Inc., CA) Monte Carlo simulation program and run with 10000 iterations.

Emission model

There were about 1200 buses in Helsinki Metropolitan Area in 1999, of which 49% met EURO II standard, 24% EURO I standard and 13% EURO 0 standard¹¹. The rest of the buses (about 14%) were either EURO II standard buses with particle traps or natural gas buses. The EURO standard particle emissions limits are listed in table 2. The year 1999 bus fleet composition was used as the 'current fleet' for the year 2020 to which the alternative bus strategies were compared.

The three alternative strategies ('modern diesel', 'diesel with particle trap' and 'natural gas bus' (CNG)) were defined to model differences between the bus types. In the 'Modern diesel' strategy buses are equipped with diesel engines and are assumed to meet the EURO III standard. The EURO III standard was enforced in 2000 and thus it is reasonable to assume that majority of the diesel buses will meet this standard in 2020. In the 'Diesel with particle trap' strategy, buses are EURO II standard diesel engine buses equipped with continuously regenerating particle traps (CRT)¹¹. There are also other types of trapping technologies on the market, but the CRT technology is already used in some of the buses in the current fleet and was therefore selected to represent this alternative technology. Ultra-low sulphur diesel fuel is currently used in road traffic in Finland. No changes were assumed in diesel fuel type.. Buses in the 'natural gas bus' strategy are so called third generation gas buses fueled with compressed natural gas.

Relative emission factor (R_{ef} in table 1) for the alternative bus strategies in relation to the 'current fleet' were 0.50, 0.25 and 0.14 for the 'modern diesel', 'diesel with particle trap' and 'natural gas bus' strategies, respectively¹¹. Relative emissions of the 'diesel with particle trap' and 'natural gas bus' strategies were expressed in the analysis using triangular uncertainty functions (table 1). The triangular uncertainty functions were based on particle emission studies and author judgment¹⁸⁻²⁰. Only studies including both of these engine technologies were considered. In these studies, the variability of

emissions from particle trap buses was higher than those from natural gas buses, and thus a larger emission uncertainty was used for ‘diesel with particle trap’ strategy. Uncertainties in the ‘Modern diesel’ strategy emissions were not modelled since the emissions are based on the maximum emissions allowed by the EURO III standard.

Exposure model

Annual average population exposure to bus-emitted PM_{2.5} in the Helsinki Metropolitan Area was estimated using two alternative exposure models as described below. The results from the two models were combined using Bernoulli distribution function and author judgement (table 1).

The first model is based on the EXPOLIS-Helsinki study, in which the observed average exposure to total PM_{2.5} in this area was 10.7 µgm⁻³ in 1996-97²². The average exposure was apportioned to source categories using elemental compositions. Using chemical reconstruction the particle masses from (i) long-range transported inorganic compounds, (ii) resuspended soil minerals, (iii) detergents and (iv) salts were estimated and subtracted from the observed total PM_{2.5} exposures. The remainder, consisting of local and long-range transported primary combustion particles, primary and secondary organic particles, and particles from tyre wear etc., was called “Combustion and other particulate matter” (CoPM) in the original work²². The average personal exposure to CoPM (3.5 µgm⁻³) was then used as the starting point for our top-down exposure model. Top-down scaling of the exposure according to relative weight factors is reasonable for a city like Helsinki, because there are virtually no local heating emissions except for those emitted from a few large combustion plants which are all equipped with high stacks. Thus traffic is the dominating ground level source of combustion particles.

To separate the exposure fraction attributable to the local bus emissions from the other CoPM sources, we used the following equation:

$$E_{\text{bus}} = (E_{\text{c}} - E_{\text{lrt}}) f_{\text{tr}} f_{\text{bus}} \quad (1)$$

where E_{bus} denotes average PM_{2.5} exposure to primary particles from local buses, E_{c} denotes exposure to total exposure to combustion originating particles (3.5 µgm⁻³), E_{lrt} denotes average exposure to long-

range transported primary combustion particles, f_{tr} denotes the ratio of local road-traffic particles to all local combustion particles, and f_{bus} denotes the ratio of local bus-derived primary CoPM to all local road-traffic-derived CoPM.

ApSimon et al.²¹ have estimated the concentration of combustion-based long-range transport (E_{lrr}) to be $2 \mu\text{gm}^{-3}$ in Helsinki. The uncertainty of the estimates was added using author judgement (table 1). Based on the EXPOLIS results on the ratio of exposure and outdoor concentrations²², we assumed that the average exposure to long-range transported $\text{PM}_{2.5}$ (E_{lrr}) would be 70 % of the corresponding outdoor concentration. To estimate the ratio of local road traffic exposure to local CoPM (f_{tr}) we used the following equation:

$$f_i = (\mathbf{Em}_i * \mathbf{wf}_i) / \sum(\mathbf{Em}_i * \mathbf{wf}_i) \quad (2)$$

where Em_i denotes the CoPM emissions in the Helsinki area (tn/year), wf_i denotes the relative weight factors to CoPM-emissions. Index i denotes different CoPM sources in Helsinki (table 3). Weight factor (wf) was an estimates of the the impact of a unit emission on the average population exposure in the area; and the impact was related to that of surface sources. Fraction of emissions, weight factors, and contributions to exposure are presented for different sources in Table 3.

As an alternative approach, exposure was calculated also based on ULTRA-study results reported by Vallius et al.²³. In the ULTRA-study, the contribution of the local traffic emissions was analysed by using so called absolute principal component analysis and multivariate linear regression based on both particle and gaseous air pollutant concentrations²³. Furthermore, sampling methods differ between the studies. In the EXPOLIS-study, sampling was based on individual measurements at the residences of the participants around the city and in the ULTRA-study sampling was based on a fixed monitoring site. In the ULTRA-study, the average local traffic generated ambient $\text{PM}_{2.5}$ concentration was estimated to be $2.5 \mu\text{gm}^{-3}$ between the years 1996-1999. The corresponding average exposure was estimated by using the outdoor concentration to personal CoPM exposure ratio (99%) obtained in the EXPOLIS-results²². Based on the results from a software estimating road traffic exhaust emissions²⁴, and assuming identical intake fractions for buses and total road traffic, the ratio of bus exposure to total

road traffic exposure (f_{bus}) was estimated to be 0.17. The same ratio was used in both exposure sub-models. Uncertainty of the ratio (f_{bus}) was estimated by using author judgement (table 1).

Dose-Response model

A dose-response model was built to describe the slope of the dose-response function and the plausibility of the $\text{PM}_{2.5}$ health effect. Multiple health outcomes have been detected in epidemiological studies in relation to $\text{PM}_{2.5}$, but in this study we considered mortality due to long-term exposure. Morbidity effects, such as lung function reduction and lower respiratory symptoms⁶, were not included. Mortality effects were estimated to dominate the effect. Although the inclusion of morbidity effects would have increased the total effect, the information would not have been critical for bus option comparisons. There are three large epidemiological cohort studies related to chronic $\text{PM}_{2.5}$ exposure, of which two have linked outdoor $\text{PM}_{2.5}$ concentration to mortality^{16,17} and one has associated the nearness to a major road with mortality²⁷. Mortality estimate from the third study contained many confounding factors related to mortality (e.g. road noise), and it was therefore not included. We estimated the concentration-response coefficient by drawing values with equal probability from the result distributions reported in the first two studies^{16,17} (table 1).

The plausibility of the estimated health effects was included in the dose-response model using author judgement. Plausibility was defined as the probability that the observed dose-response relationship actually represents a causal association. We assumed that the probability for $\text{PM}_{2.5}$ being the true cause of the effects is 70%, 90% and 10% for cardiopulmonary, lung cancer and all other mortality, respectively. The plausibility for cancer was higher because there are known carcinogens in PM, while it is more controversial, what the true agent is causing cardiovascular effects in the air pollution mix.²⁸

There are also studies on toxicity differences between $\text{PM}_{2.5}$ particles from different sources^{9,10}. The major issue in this analysis was the possible difference between the general ambient air particles and the particles generated by different bus types. Since no such toxicological comparisons were available, we did not model the possible differences in the toxicity in this analysis. We assumed no threshold in the dose-response relationship, because there is no evidence for a threshold for $\text{PM}_{2.5}$ ^{6,29} and because it is unlikely that any putative threshold would affect the studied bus strategies differently.

Mortality assessment

The additional mortality (ΔM) associated with the $PM_{2.5}$ bus emissions was estimated by using the equation:

$$\Delta M = M(e^{\beta \Delta E} - 1) \quad (3)$$

where β is the dose-response coefficient, ΔE change in exposure, and M background mortality.

Background mortality was calculated from APHEA2-project data³⁰. Background cardiopulmonary (*International Classification of Disease* (ICD-10) codes: I11-I70 and J15-J47), lung cancer (C34), and total mortalities (<R) were 3338, 317, and 6541 deaths per year, respectively, in Helsinki Metropolitan Area in 1996. We assumed that the population in Helsinki metropolitan area will be 1 million in 2020 (about 970 000 in year 1999).

Results

The mortality due to primary fine particles from buses ranged from 3 to 18 cases per year in the different bus strategies (table 4). Of the examined options, 'Diesel with particle trap' and 'natural gas bus' strategies showed similar reductions in mortality while in the 'Modern diesel' strategy, the mortality remained at a higher level. All three alternative strategies clearly reduced mortality compared to the 'Current fleet' case. These results indicate that a change in the bus fleet composition could reduce bus-induced mortality in Helsinki and that the largest reduction would be achieved by using either natural gas or particle trap buses. Exposure estimates for traffic- and bus-related primary $PM_{2.5}$ were (90% confidence intervals in parentheses) 1.8 (1.5-2.4) $\mu g m^{-3}$ and 0.3 (0.2-0.5) $\mu g m^{-3}$, respectively, in 1997. The average bus-related primary $PM_{2.5}$ exposure was estimated to be 0.5 (0.3-0.8) $\mu g m^{-3}$, in 2020 if there were no changes in the bus fleet composition. The estimated cardiopulmonary mortality was approximately 9 times larger than lung cancer mortality due to the higher background mortality rate.

The uncertainties in the mortality estimates are high and include a zero-effect possibility. Uncertainties in two variables, the plausibility of the cardiopulmonary mortality and the emission factor, had the

largest impacts on the final effect estimates (figure 1). For the exposure estimates, the main variables affecting the uncertainty were the ratio of bus exposure to the total road traffic exposure and the mean exposure to road traffic PM_{2.5}. The importance of other variables on the final mortality results was low.

Discussion

This risk analysis compares the effects of three alternative bus technologies to bus-induced mortality in Helsinki. The main results were: (i) bus-induced mortality could be reduced by changing the bus fleet composition from the current one and (ii) particle trap buses and natural gas buses would result in similar mortality reductions, while there was a clear benefit associated with either of these compared to the traditional diesel bus. The average exposure due to bus traffic in Helsinki Metropolitan Area was estimated to be 3% of the average total fine particle exposure ($10.7 \mu\text{g m}^{-3}$) whereas exposure due to local traffic (including buses) and long-range transported combustion particles contributes about 17% to the total exposure.

Cohen et al.¹⁸ conducted a cost-effectiveness analysis of alternative fuels used in urban public transportation buses in the U.S. The health effects attributable to exposures to the various air pollutants (PM_{2.5}, SO₂, NO₂ and CO₂) emitted by the buses, were quantified using quality adjusted life years (QALY). The main results of that study were similar when compared to the results of our study, but the magnitude of the health effects was estimated to be a larger in this study. While Cohen et al. estimated the near source dispersion (15 km within the emission) with an exponential decay function fitted to CAL3QHCD-model, our exposure estimates are based on chemical analyses of particle samples around the city. It seems that the exposure model used in this analysis gives higher exposure estimates than that reported in the Cohen et al. study even though the population density was higher in the Cohen et al. study (5000 inhabitant/km² compared to about 770 inhabitant/km²). The exposure model used in this analysis estimates higher exposure levels. This is probably because our approach captures better the exposure very near the source. This highlights the importance of any reduction in the PM emissions from the buses.

The current analysis concentrated on primary fine particles. The other pollutants related to bus emissions, such as secondary particles, ozone and NO_x, have also been estimated to be harmful to

human health⁶. The magnitude of health effects associated to ozone and NO_x are, however, substantially lower than those associated with fine particles. For example Hutchinson et al.³¹ noted that the health effect reduction was dominated by the buses particle emission reduction. Gaseous emissions are also precursors for secondary particles. Cohen et al. found that NO_x emissions made an important contribution to mortality following long-range transport and generation of secondary particles. In Helsinki, the contribution of secondary particles to all health effects would probably be lower than that observed by Cohen et al., because the exposure near the source was estimated to be larger. In addition, most exposure to secondary particles would occur in surrounding areas, which are sparsely populated around Helsinki. Furthermore, the observed health effects associated with secondary particles are more uncertain than those due to primary particles (e.g. Schlesinger et al.³²). Although the emissions of NO_x's are high and vary between the strategies¹¹, it is unlikely that they would substantially change the main results observed in this study.

Upstream emissions from feedstock extraction and fuel production activities were not considered here as they occur mostly outside the current study area and thus were not within its scope. In addition, Cohen et al.¹⁸ noted that the risk associated with the upstream emissions is small compared to the direct emissions and that the risk did not vary substantially between the possible strategies.

Emission estimates for the different strategies were based on the literature. In the particle trap and gas bus strategies, the uncertainties of the emissions were included in the model as an uncertainty distribution. In our literature review, the average particle emission from the gas buses were lower and less variable than those of particle trap buses^{11,19,20}. Similar results have been observed with ultrafine particle emissions³³. Cohen et al.¹⁸ estimated, however, that the particle emissions from particle trap buses are lower than those from gas buses. Factors such as vehicle aging and maintenance status may also have large effects on the emissions, but it is not clear to what extent this would lead to any difference between the strategy options. The composition of the diesel fuel was similar in all strategies because the major changes in the diesel fuel compositions have already been achieved (e.g. reduction of sulphur content¹¹). Uncertainty due to fuel composition is thus small.

The exposures were estimated by using a top-down method, calculating the fraction of bus related fine particles from a previously measured particle concentration. This approach has two advantages. First, modeling of the bus exposures from total exposure to primary combustion generated particles observed in two field studies sets a reliable upper boundary to the exposure estimates and, thus, limits the uncertainty of the estimated exposures. Secondly, this approach makes it possible to use a fairly simple model that makes re-calculations and controlling of the model easier compared to a bottom-up model that would require more detailed information e.g. on the relationship between emissions, dispersion and population time activity.

A simple model includes, of course, simplifying assumptions. The exposure difference between the strategies was based on the assumption on linearity between the marginal emissions and the marginal exposure. The same assumption has also been used by others¹⁸. Non-linearity could occur due to e.g. changes in atmospheric chemistry, but this is unlikely in the case of small changes in total emissions. Certain input values, such as the average CoPM exposure, the long-term transported concentration of combustion particles, and the bus emissions, were presented in the literature without uncertainty information. This was accounted for in the analysis by estimating the corresponding uncertainties using author judgement. The importance of uncertainties in the model were estimated using importance analysis (a rank correlation between an individual input value and the model output). The analysis showed that even relatively large uncertainties in the input values of the exposure model did not change the main results. Our model estimated only the average long-term population exposure and did not try to assess which individual are most severely affected by PM_{2.5} emissions from buses. In the future, questions related to specific groups of interest, such as different age groups and people with previous disease history, should be studied in more detail.

The causal relationship between fine particulate matter exposures and health effects is still uncertain to some extent. This question is critical, because emission abatement would be meaningless without a causal relationship. To study quantitatively the possibility of a non-causal association, a plausibility variable was included in the model. Even at the plausibility level set to 90%, the plausibility of the relationship remained the single most important source of uncertainty. The use of ambient dose-response slope from the U.S. studies instead of the unknown exposure-response slope of traffic emitted

primary PM_{2.5} mass in Helsinki, is a source of error and uncertainty in the model. The composition of the PM_{2.5} mixtures, the exposure patterns and the exposed populations all differ. This may well have produced an unknown bias of the estimated mortalities but it does not affect the rank order of the alternative bus fleets. No threshold value was included in the model even though the average particle level in Helsinki is at the lower level of the epidemiological studies^{16,17}. It is possible, that the PM_{2.5} exposure would be reduced below a threshold in a fraction of the population. However, given the small change due to bus options, this fraction must be very small. In addition, some people may be below the threshold already before any action. The health effect estimate would be reduced by the proportion of such people in the whole population. It would have been difficult to estimate such an individual threshold..

An important source of non-quantified uncertainty in the model seems to be the lack of information related to differences of toxicity between PM_{2.5} from diesel vs. gas powered buses. There are several studies related to the toxicity of diesel exhausts³⁴⁻³⁷, but no equivalent information is available for gas bus-derived particles. There is insufficient information on particle toxicity differences to separate health effects from different combustion sources from each other. In the future, it will be necessary to identify those particle characteristics that are causally related to health effects if we are to correctly estimate the health effects evoked by different sources and to avoid consequent errors in the risk management optimizations.

Conclusions

Estimated excess mortality caused by the alternative bus fleets in 2020 in Helsinki Metropolitan Area varied between 3 and 18 deaths, indicating that levels of PM and the corresponding health effects can be affected to some extent by changing bus types. The difference in the excess mortality between natural gas buses and the presentdiesel engines with proper trapping systems is not large. Thus it is questionable whether the costs and alternative risks associated with the more complicated fuel storage and delivery systems for compressed natural gas would be covered by the marginal health benefits from the lower PM emissions. Emissions from buses in Helsinki are only a small fraction of the total traffic and other combustion particle emissions in the area and thus there are also possibilities for

acquiring similar or larger reductions in ambient PM concentrations and corresponding reductions in health effects.

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Model availability

The risk model described in this article is downloadable from the internet address

<http://www.ktl.fi/risk/>.

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Table 1: Variables included in the uncertainty analysis.

Variable	Distribution uncertainty	Distribution parameters	Explanations and references
Relative emission factor (R_{ef}), 'diesel with particle trap'	Triangular ¹⁾	1.0 (0.6;1.4)	Mode ¹¹ min and max ¹⁸⁻²⁰ and AJ ²⁾ .
'Natural gas bus'	Triangular	1.0 (0.8;1.2)	
Concentration of combustion-based long-range transported PM _{2.5} (E_{lr})	Triangular	2.0 (1.0;2.5) μgm^{-3}	Mode ²¹ , min and max AJ.
Relative weight factor for road traffic emissions (wf_{tr})	Triangular	2.0 (1.0;3.0)	AJ.
Exposure to road traffic PM _{2.5}	Bernoulli ³⁾	$P = 0.7$ for 1.8 μgm^{-3} , $P = 0.3$ for 2.4 μgm^{-3} .	1.8 μgm^{-3} from EXPOLIS ¹⁵ and Koistinen et al. ²² , 2.4 from Vallius et al. ²³ . Probabilities AJ.
The fraction of bus exposure of total road traffic exposure (f_{bus})	Triangular	0.17 (0.1;0.25)	Mode ²⁴ min and max AJ.
Dose response coefficient (β) for cardiopulmonary mortality	Mixed ⁴⁾	1.014 (0.0053-0.0254)	Relative increase of mortality per 1 μgm^{-3} increase of outdoor PM _{2.5} concentration. Values were drawn with equal probability from the two distributions reported in ^{16,17}
lung cancer mortality	Mixed	1.016 (-0.0009-0.0364)	
all other mortality	Mixed	1.002 (-0.0073-0.0102)	
Plausibility ⁴⁾ of cardiopulmonary mortality	Bernoulli	$P = 0.7$ yes, $P = 0.3$ no	AJ
lung cancer mortality	Bernoulli	$P = 0.9$ yes, $P = 0.1$ no	
all other mortality	Bernoulli	$P = 0.1$ yes, $P = 0.9$ no	

1) Parameters for triangular distribution in form mode (min;max).

2) AJ = Author judgment

3) Bernoulli (Binomial) binary **probability distribution with probabilities (P, 1-P)**.

4) Combination of two normally distributed variables (95 confidence intervals).

5) Plausibility = "Is the observed effect due to true causal connection?"

Table 2: PM-emission standards for particles set or planned by EU for new heavy-duty vehicles and years of coming into force²⁵.

Standard	Directive	Year	Max emissions (g/kWh)
EURO-0	88/77	1988	-
EURO-I	91/542	1992	0.36
EURO-II	91/542	1996	0.15
EURO-III	98/69/EU	2000	0.10
EURO-IV	Planned	2005	0.02
EURO-V	Planned	2008	0.02

Table 3: Apportionment of the exposure to combustion particles ($3.5 \mu\text{g m}^{-3}$ CoPM as defined in the text) was based on relative emissions from different local sources (road traffic Mäkelä 2002²⁴ other YTV 1998²⁶) and weight factors estimated by author judgment.

Source (i)	Local emissions (E_{m_i})	Weight factor (wf_i)	Exposure ($\mu\text{g m}^{-3}$)
Energy production	62	0.1	0.2
Other point sources	2.9	1	0.1
Surface sources	3.9	1	0.1
Road traffic	29	2 ¹⁾	1.8
Harbour	2.3	1	0.1
Long-range transport	-	-	1.4
Total	100 %	-	3.5

1) Uncertainty modeled using triangular function with parameters listed in table 1.

Table 4: Estimated deaths per year (90% CI) associated with primary PM_{2.5} emissions from buses in Helsinki Metropolitan Area in 2020 for the different bus strategies.

Bus strategy	Cardiopulmonary mortality	Lung cancer mortality	Total mortality
'Current fleet'	15.9 (0 - 46.6)	2.2 (0 - 6.1)	18.1 (0 - 55.0)
'Modern diesel'	7.9 (0 - 23.0)	1.1 (0 - 3.0)	9.0 (0 - 27.0)
'Diesel with particle trap'	3.9 (0 - 12)	0.6 (0 - 1.6)	4.4 (0 - 14.1)
'Natural gas bus'	2.3 (0 - 6.8)	0.3 (0 - 0.9)	2.6 (0 - 8.0)

Figure 1: Importance analysis of the variables. Importance analysis was done by calculating rank-order correlations between the input variables and the model output across 10000 iterations.

