

A simple concept for GIS-based estimation of population exposure to primary fine particles from vehicular traffic and domestic wood combustion

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We introduce a simple concept called Population-based Time-Average Inhalation (PTAI) intended to evaluate the relative effect of emission sources on population exposure. The PTAI is defined as the sum of the individual time-average inhalations at a particular distance from an air pollution source. We demonstrate this concept by evaluating the PTAI's for primary fine particulate matter emission from vehicular traffic and domestic wood combustion in Finland. The PTAI's for these two pollution categories were evaluated by combining the emission volume and location data with population location data and with the time-average inhalation per person. The PTAI for the vehicular traffic was significantly higher than that for domestic wood combustion. The result suggests that there is a difference in the exposure potency (per emission volume) between these two sources. The PTAI differences between domestic wood combustion sub-sectors imply that there is a significant difference also in exposure potency between different sub-sectors.

Introduction

The evaluation of exposure to inert air pollutants depends in general on the location of the emission source and the population, the emission volumes, as well as the atmospheric dispersion of the pollutants. The exposure to ambient-air fine-particulate matter (PM_{2.5}) originating from

certain specific emission sources can be estimated using atmospheric dispersion models (e.g. SILAM *see* <http://silam.fmi.fi/>, CALPUFF *see* Bennett *et al.* 2002a). The regional-scale dispersion models often include physical and chemical transformation and removal of particles from the atmosphere by deposition. These dispersion models require a large amount of meteorologi-

cal data and substantial expertise. Therefore, concepts have been developed to incorporate dispersion and exposure information in the integrated analyses such as risk assessment which do not need to incorporate the actual calculations. Intake fraction (iF) is the fraction of the emission of an agent that is eventually contained in the exposure, i.e., is inhaled or ingested by an individual in the target population (Bennett *et al.* 2002b). The intake fraction concept has been used in a number of ambient air fine particulate matter studies, with varying spatial resolution and emission sources, since it presents the exposure information in a relatively simple manner (e.g. Levy *et al.* 2003, Marshall and Behrentz 2003, Greco *et al.* 2007).

Outdoor air PM_{2.5} has been associated with multiple adverse health effects. These effects have been demonstrated in both short-term (daily variations) and long-term (chronic) studies (e.g. Hoek *et al.* 2002, Stieb *et al.* 2003, Pope and Dockery 2006). The most consistent association has been found between outdoor air PM_{2.5} and increased cardiopulmonary mortality, lung cancer mortality and reduced lung function (WHO 2003).

The most comprehensive assessments of the health effects at the population level of PM_{2.5} (such as the CAFE, Clean Air For Europe, Holland *et al.* 2005) have been based on annual average concentrations modelled for large areas (e.g. EC 2005). The computed concentrations are therefore mainly determined by regional and long-range transport. The contributions of local PM_{2.5} emissions on urban concentrations in European cities are characteristically less than 50%; however, this fraction is both spatially and temporally highly variable in urban areas. For example, Forsberg *et al.* (2005) have emphasized the need for the evaluation of health effects of primary PM_{2.5} with a more detailed (urban or local scale) spatial resolution. This applies especially to near ground level emission sources, such as domestic wood combustion and vehicular traffic. Vehicular traffic emissions are mostly released near population clusters. In contrast, domestic wood combustion occurs to a large extent in rural areas, but it is also a significant pollution source in small towns and suburban residential districts (Glasius *et al.* 2006, Tissari

et al. 2007) and in cities (e.g., in Oslo, Kukkonen *et al.* 2005). In fact, it has been calculated that, domestic wood combustion and vehicular traffic represents the largest sources of primary PM_{2.5} in Finland, with 7.6 and 5.8 Gg a⁻¹ in 2000, respectively (Karvosenoja *et al.* 2008). Therefore, there is a good reason to study the influence of these two source categories on the health of the population.

The spatial characteristics of the population and dispersion near to the emission sources are two important factors to be considered when estimating exposure to primary fine particles. The spatial and temporal characteristics of a population may vary substantially from one area to another. These variables are usually relatively well known, at least with respect to the location of residences and workplaces. In many cases the population data can also be obtained from public sources. In contrast, variables affecting the transport of airborne pollutants are rarely as well known or as widely available, and estimation of exposure often requires the use of complex atmospheric models. Therefore, it would seem worthwhile to develop also simpler concepts to estimate the intake fractions.

In this study, we introduce a simple concept called Population-based Time-Average Inhalation (PTAI), with which we evaluate the relative contribution of emission sources to the population exposure to primary PM_{2.5}. The PTAI is defined to be the sum of the individual time-average inhalations at a certain distance from air pollution source (unit m³ s⁻¹). The PTAI concept has been derived from the intake fraction concept. The details are explained in the Material and methods. Our specific aims are (i) to define the PTAI concept as part of intake fraction (iF) and to evaluate the relative influence of local sources on GIS-based population exposure, (ii) to compare relative influence of fine particles from vehicular traffic and domestic wood combustion on exposure, and (iii) to compare the relative contribution of domestic wood combustion sub-categories derived fine particles on exposure. The study does not contain any atmospheric dispersion modelling. The concept presented here is useful for evaluating the relative influence on population exposure of near ground level emissions of primary, inert pollutants.

Material and methods

In this section, first, the theoretical background of the PTAI is presented, where we derive the PTAI from the intake fraction concept defined by Bennett *et al.* 2002b. Second, we estimate the vehicular traffic and domestic wood combustion primary fine particulate matter emissions in Finland to a resolution of $1 \times 1 \text{ km}^2$. Third, we calculate the spatial allocation of the population. These spatially allocated datasets are then used to estimate PTAI's for the population near to the emission sources (less than 16 km).

Restructuring the intake fraction to atmospheric and population-based components

The approximate relative differences between the total (temporally averaged) exposures caused by various pollution source categories can be estimated, assuming that the average release heights of these source categories are sufficiently similar. The evaluation of exposures for these source categories also has to be conducted during the same period, in order to include the same variation in the relevant meteorological factors. Secondary pollution cannot be evaluated with this same concept, and the meteorological conditions must be assumed to be identical in all parts of the country.

In the case of inhalable air pollutants the intake fraction is defined as

$$\text{iF} = (\text{TAI}/Q) \times \sum (c_i \times P_i), \quad (1)$$

where c (in g m^{-3}) is the concentration of the pollutant from that source in the inhalation zone of the population, P is the size of the population under study, Q (g s^{-1}) is the emission rate and TAI is the time-average inhalation per person ($\text{m}^3 \text{ s}^{-1}$). Different locations are indexed by i . Intake fraction is a dimensionless variable. Some previous authors, e.g. Greco *et al.* (2007), used analogous study methods.

Intake fraction is a useful aggregated variable, e.g., being used in the analysis of health effects and risks. The average concentration for population exposure can be defined as $Q \times \text{iF}/(P \times \text{TAI})$, where P is the number of the total popu-

lation under consideration. If a linear non-threshold exposure-response function can be assumed, the total health effect H in the population can be calculated simply as

$$\begin{aligned} H &= Q \times \text{iF}/(P \times \text{TAI}) \times \beta \times P \times \text{BG} \\ &= Q \times \text{iF} \times \beta \times \text{BG}/\text{TAI}, \end{aligned} \quad (2)$$

where β is the slope of the exposure-response line and BG is the background incidence rate of the health effect of concern.

Here, we aim at separating the complicated atmospheric part of the intake fraction from the population part. First, the intake fraction can be rewritten as

$$\begin{aligned} \text{iF} &= \text{TAI}/Q \times \sum (c_i \times P_i) \\ &= \sum (c_i/Q) \times (P_i \times \text{TAI}), \\ &= \sum (S_i \times \text{PTAI}_i) \end{aligned} \quad (3)$$

where two new variables were defined. PTAI is the Population-based Time-Average Inhalation, i.e. simply the sum of the individual time-average inhalations at location i ($\text{m}^3 \text{ s}^{-1}$). S is the volume-average time (s m^{-3}) for unit emission of $\text{PM}_{2.5}$ mass (g). The concentration of a pollutant at a particular location i at a particular time t is a function of the emissions, atmospheric transport and chemistry, and the environmental conditions. If we use X to denote all other variables except for the emissions, we can write

$$c_{i,t} = f(Q_t, X_{i,t}). \quad (4)$$

If one assumes that c_i (integrated over a reasonable period) is linearly proportional to Q (also integrated over the same time period),

$$c_i = Q \times f(X_i) \leftrightarrow f(X_i) = c_i/Q = S_i. \quad (5)$$

The determination of S_i requires atmospheric dispersion modelling. After S_i has been computed, it is a straightforward task to summarise the two-dimensional location index i into a simpler, one-dimensional index d , the distance from the source. S_d would be a very practical variable to use, since it is pollutant location-independent, and the new equation for intake fraction, although almost identical with Eq. 3, is much simpler to use and compute:

$$iF = \sum (S_i \times PTAI_i) \approx \sum (S_d \times PTAI_d) \quad (6)$$

Equation 6 is the theoretical basis for the methodological development in this study. We will assume for the moment that S_d is a reasonable summary variable for describing complicated atmospheric processes, and focus on $PTAI_d$. In addition, we need to assume that (i) S_i and $PTAI_i$ are not correlated at a given d , and (ii) $f(X_i)$ agrees reasonably well with the atmospheric processes in a new location. It is not at all obvious, how far from the original season, geographical location, or wind pattern S_d still remains a useful and reliable summary measure. The validity of these two assumptions will be a topic of further studies.

We calculated $PTAI$'s for $PM_{2.5}$ for two major emission sources in Finland, namely domestic wood combustion and vehicular traffic. The daily inhalation volume was defined as a normative value of $20 \text{ m}^3 \text{ d}^{-1}$. This inhalation volume has been used in a number of intake fraction studies and is based on a past recommendation from EPA (see e.g. EPA 1997).

$PTAI$, indexed by the distance between source and receptor ($PTAI_d$), was calculated for distances 0–16 km at a resolution of one kilometre. The selection of maximum 16 km was based on the assumption that after that distance, the emission from near the ground level releases can no longer be distinguished from background sources in any meteorological conditions. $PTAI_d$ s were first calculated for all cells in the grid with non-zero emissions. Cell-specific $PTAI_d$ s were then averaged using emissions as weights. This resulted in source-specific $PTAI_d$ s for Finnish wood combustion and vehicular traffic emissions.

Without information about S_d , this concept can only be used for the evaluation of the relative population exposures originating from various pollution source categories, not for computing absolute exposures.

The evaluation of emissions and their spatial allocation

The spatially resolved emission estimates of this study are based on the Finnish Regional

Emission Scenario (FRES) model (Karvosenoja 2008). The spatial allocation of vehicular traffic and domestic wood combustion emissions in FRES is described here; that of other area emissions in Karvosenoja *et al.* (2005). Data sources for $PM_{2.5}$ emission calculation are presented in Karvosenoja *et al.* (2008).

The emissions are described at country, municipality and $1 \times 1 \text{ km}^2$ levels. Emissions (EM) are calculated at the country level from annual activity data (A) and emission factors (EF). For each area sources:

$$EM_{p,t} = A_t \times EF_p, \quad (7)$$

where p refers to the pollutant and t to time. Country-level emissions are allocated to all of the 448 municipalities in Finland using the respective relative fractions (RF) of municipality level allocation data (B) in 25 aggregated emission sources. Area source emission in a municipality m is:

$$EM_{m,p,t} = EM_{p,t} \times RF_{m,t}, \quad (8)$$

where

$$RF_{m,t} = B_{m,t} / \sum_m B_{m,t}. \quad (9)$$

Municipality level emissions are further allocated to $1 \times 1 \text{ km}^2$ grid. Emission in a grid cell n is:

$$EM_{m,n,p,t} = EM_{m,p,t} \times RF_{m,n,t}, \quad (10)$$

where

$$RF_{m,n,t} = D_{m,n,t} / \sum_n D_{m,n,t}, \quad (11)$$

where D is geographical allocation information at $1 \times 1 \text{ km}^2$ level. A summary on the spatial dimensions of vehicular traffic and domestic wood combustion is presented in Table 1. For vehicular traffic, the spatial allocation to municipality level is based on fuel consumption values from the national vehicular traffic emission calculation system LIISA (Mäkelä *et al.* 2002). The differentiation to the $1 \times 1 \text{ km}^2$ level was carried out based on the national SLICES land use elements (Mikkola *et al.* 1999). SLICES is

a combination of different national land use GIS databases; the output is grid-based databases for 10×10 and 25×25 m². The allocation was based on road surface area and does not take into account vehicular traffic volumes in different road segments.

Domestic wood combustion is here defined as wood heating in residential and recreational houses. The identification of wood-heated residential and recreational houses was performed using the national building and dwelling register that is a part of SLICES (Mikkola *et al.* 1999). This encompass all Finnish buildings and includes information on building type, primary heating media, gross floor area, inauguration date and number of residents. However, 36% of the wood combustion does take place as supplementary heating in primarily electricity-heated and oil-heated houses. The building and

dwelling register does not contain information on supplementary heating devices. Instead, the identification of supplementary wood heated buildings was done using the time of construction date, with the assumption that detached houses commissioned in 1980 or later can use supplementary wood heating, whereas this is not the case for older houses. This assumption is based on the fact that the installation of supplementary wood heaters increased sharply during the 1980s. It is known that approximately 70%–80% of detached houses built after the 1970s are equipped with supplementary wood burning applications (Sevola *et al.* 2003).

The relative differences in wood combustion volumes between individual wood-heated buildings can be assumed to be proportional to room heating need and efficiency of the combustion appliance. The room heating need depends on

Table 1. The spatial allocation of vehicular traffic and domestic wood combustion in the FRES model. The allocation was done first from National level to municipality level and then from municipality level to 1×1 km² level. The vehicular traffic and domestic wood combustion emissions categories were divided into 10 and 22 sub-sectors, respectively. LD = Light-duty vehicles (passenger cars, vans, motorcycles); HD = Heavy-duty vehicles (trucks, busses, other heavy-duty); RsB = residential buildings; Rch = recreational houses; DW = heating-degree-day-weighted; GA = gross floor area.

Main sectors at country level; number of sub-sectors	Sectors at municipality level; (data used as a basis for emission allocation)	Sectors at 1×1 km ² level; (GIS data used as a basis for emission allocation)
Road traffic; 10 sub-sectors	Light-duty (LD) gasoline exhaust (Annual LD gasoline consumption ¹) LD diesel exhaust (Annual LD diesel consumption ¹) Heavy-duty (HD) diesel exhaust (Annual HD diesel consumption ¹) LD non-exhaust emissions (Annual LD gasoline + diesel consumption ¹) HD non-exhaust emissions (Annual HD diesel consumption ¹)	Vehicular traffic (Road surface area ²)
Domestic wood combustion; 22 sub-sectors	Primarily wood-heated residential buildings (RsB) (Heating-Degree-day ³ weighted gross-floor area (DWGA) of wood-heated RsB ⁴) Supplementary wood-heated RsB (DWGA of detached RsB commissioned after 1979 and heated primarily in ways other than wood ⁴) Wood-heated recreational houses (Rch) (DWGA of wood-heated Rch ⁴)	Primarily wood-heated RsB (Gross-floor area (GA) of wood-heated RsB ⁴) Supplementary wood-heated RsB (GA of detached RsB commissioned after 1979 and heated primarily in ways other than wood ⁴) Wood-heated Rch (GA of wood-heated Rch ⁴)

¹ LIISA (Mäkelä *et al.* 2002). ² SLICES land use element (Mikkola *et al.* 1999). ³ Finnish Meteorological Institute.

⁴ Building and dwelling register (Mikkola *et al.* 1999).

room volume, energy efficiency of the building and heating degree day factor. The heating degree day factor is related to the heating need resulting from different ambient temperatures in different parts of the country. The spatial allocation of domestic wood combustion was performed using gross floor areas of wood-heated buildings as a surrogate for room volume, and heating degree day factor of different municipalities. For country-wide assessment, there was no information available on the energy efficiencies of individual buildings or combustion appliances.

Evaluation of the spatial distribution of the population

In this study, we have addressed only the population at residential locations, but not the population in workplaces, in vehicular traffic or at other locations. Population data used in this study was based on the locations of permanent residents from building and dwelling register (Mikkola *et al.* 1999).

The use of permanent resident locations as a basis for population data does not, however, yield a realistic estimate of exposed population for the emissions from recreational houses (e.g. summer cottages), since very few recreational houses are registered for permanent resident use. Therefore, we performed a sensitivity calculation by adding one person to each of the 467 790 Finnish recreational houses. From the population exposure point of view, this was considered as approximate average number of people staying in recreational houses. This is a result of the assumption that when there is a family of three persons staying in a recreational house, the probability that there are similar families in neighbouring recreational houses (i.e. receptors) at the same time is 33%.

Results

The spatially allocated emissions and population are illustrated as 10×10 km² maps for the whole Finland, and as 1×1 km² maps for southwestern Finland, where the majority of the population is located (Figs. 1 and 2). Vehicular traffic emissions mostly originate from the largest towns

and the major roads connecting these towns. The largest towns are situated in southern and southwestern Finland, and along the coast of the Baltic Sea. Domestic wood combustion emissions are more evenly distributed throughout the whole of the southern and central Finland. Most of the wood heated residential buildings and recreational houses are located outside the major cities (Fig. 2). However, supplementary wood heating takes place to a considerable extent also in residential areas of the major cities.

PTAI's were calculated for the two emission source categories considered, and for the three sub-sectors of domestic wood combustion (Fig. 3). The PTAIs for vehicular traffic are approximately four to eight times higher than those for domestic wood combustion. One notable exception was supplementary wood heating for residential buildings, which occurs primarily in suburban areas, whereas other domestic wood combustion occurs mainly in rural areas. The additional sensitivity analysis, where recreational houses were assumed to have one additional inhabitant, had only a negligible effect on the results. Nonetheless, when all sub-sectors of domestic wood combustion were combined, the PTAI's were less than a half of those of vehicular traffic.

Country level PM_{2.5} emission calculation of vehicular traffic and domestic wood combustion in 2000 are presented in Karvosenoja *et al.* (2008). National total emissions from domestic wood combustion (7.6 Gg a⁻¹) are higher than those from vehicular traffic (5.8 Gg a⁻¹). With respect to the sub-sectors of domestic wood combustion, the largest emission source is primary wood heating in residential buildings, the total emission of which is 4.0 Gg a⁻¹. Supplementary wood heating and recreational houses account for 2.2 and 1.3 Gg a⁻¹, respectively. With respect to the vehicular traffic, the majority of emissions originate from diesel exhausts, 3.8 Gg a⁻¹. Vehicular non-exhaust dust and resuspension contribute a total of 1.6 Gg a⁻¹.

Discussion

In this study, we introduced the concept of Population-based Time-Average Inhalation and then applied that concept to the primary fine par-

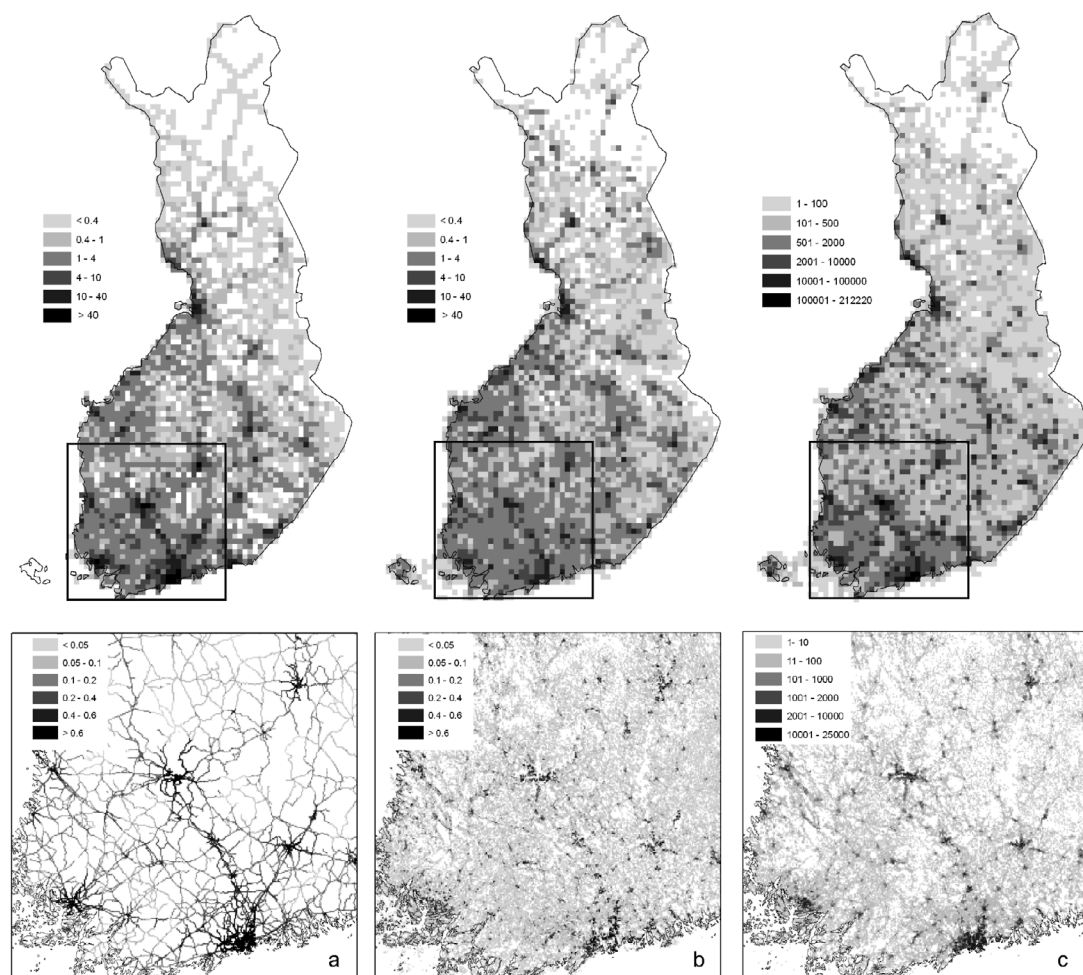


Fig. 1. Finnish PM_{2.5} emissions in 2000 (Mg a⁻¹) originating from (a) vehicular traffic and (b) domestic wood combustion. Panel c shows the location of the population in 2000. The upper and lower panels present the results in the whole of the country using a resolution of 10 × 10 km², and in southern Finland using 1 × 1 km².

ticulate matter emissions from vehicular traffic and domestic wood combustion in Finland. The PM_{2.5} emissions were allocated from the country level calculations, and disaggregated into municipal level and 1 × 1 km² spatial resolutions by using the available spatial databases. The PTAI's were calculated for these low emission sources based on the allocated emission and population data. The results indicated that, on average, the relative exposure to vehicular traffic emissions is probably larger than that for domestic wood combustion. With respect to the domestic wood combustion sub-sectors, the relative exposure for residential buildings supplementary wood heating was highest and showed a similar relation-

ship relation to the population as vehicular traffic emissions did.

Country-level emission uncertainties have been studied in detailed by Karvosenoja *et al.* (2008). Uncertainties were found to be considerable for domestic wood combustion (the limits of the 95% confidence interval equals mean value minus 28% to the mean value plus 37%) and vehicular non-exhaust emissions (minus 38%, plus 52%). Vehicular traffic exhaust emissions have been estimated to be less uncertain, ±7%. In the following section, we discuss in qualitative terms the additional uncertainties caused by top-down spatial emission allocation and the resulting consequences on calculated PTAI values.

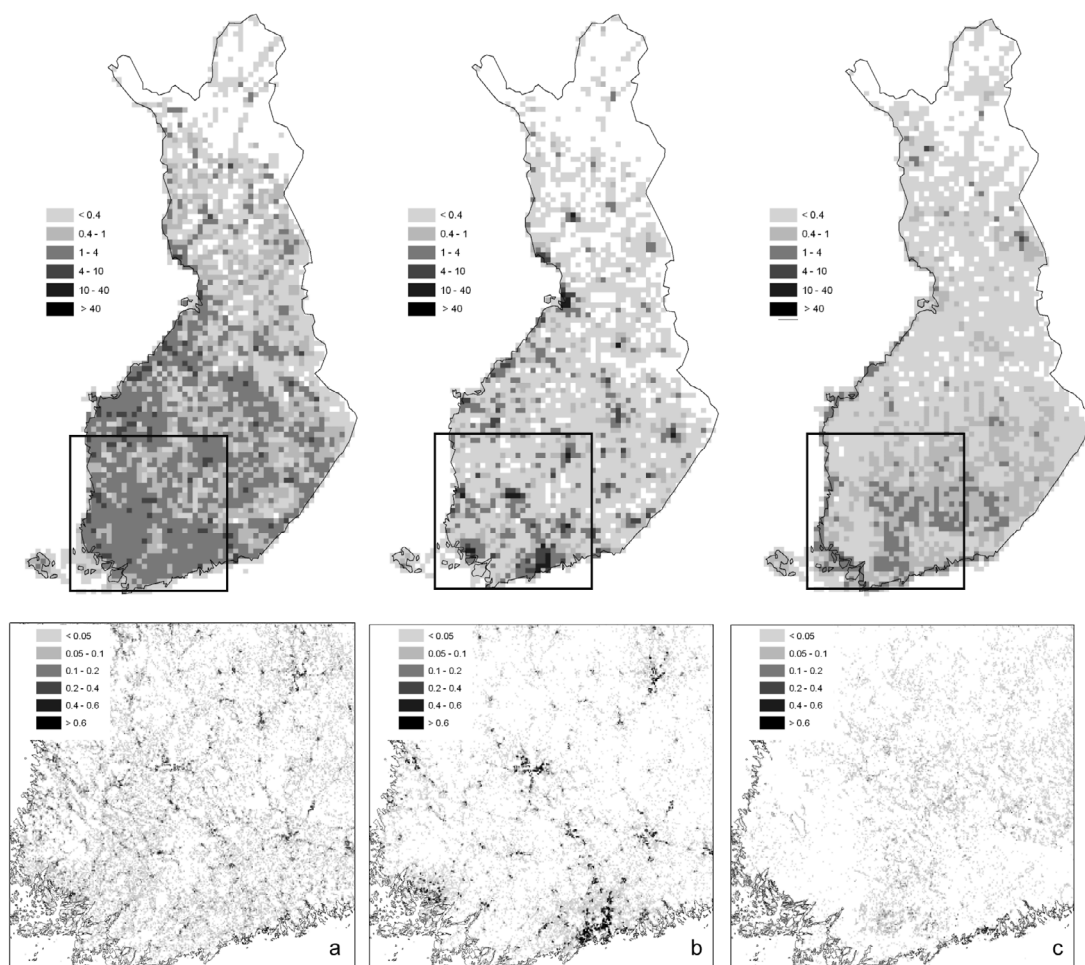


Fig. 2. Finnish PM_{2.5} emissions in 2000 (Mg a⁻¹) originating from (a) wood combustion as a primary heating source in residential buildings, (b) wood combustion as a supplementary heating source in residential buildings and (c) wood combustion in recreational houses. The upper and lower panels present the results in the whole of the country using a resolution of 10 × 10 km², and in southern Finland using 1 × 1 km².

In domestic wood combustion, the allocation is made using gross-floor area information from the building and dwelling register. The data quality of the building and dwelling register were analyzed and found to be relatively good. However, the building and dwelling register does not contain information on supplementary wood heating which was responsible for 29% of the total domestic wood combustion emissions. These emissions were allocated to the detached houses commissioned in 1980 or later because supplementary wood heaters have been mainly installed in these houses. However, there are approximately 20%–30% of the housing stock

from that period without heaters and also a considerable number of older buildings with wood heaters (Sevola *et al.* 2003). Therefore this part of the spatial allocation does contain a considerable degree of uncertainty. It may be assumed that the emissions were overestimated in southern Finland in some of the densely populated residential areas that are located relatively far from forested areas, and where the utilization of wood fuels is therefore less economical. As a result, the PTAI values on supplementary wood heating might be overestimated to some degree.

With respect to vehicular traffic, the municipality level spatial allocation in the FRES model

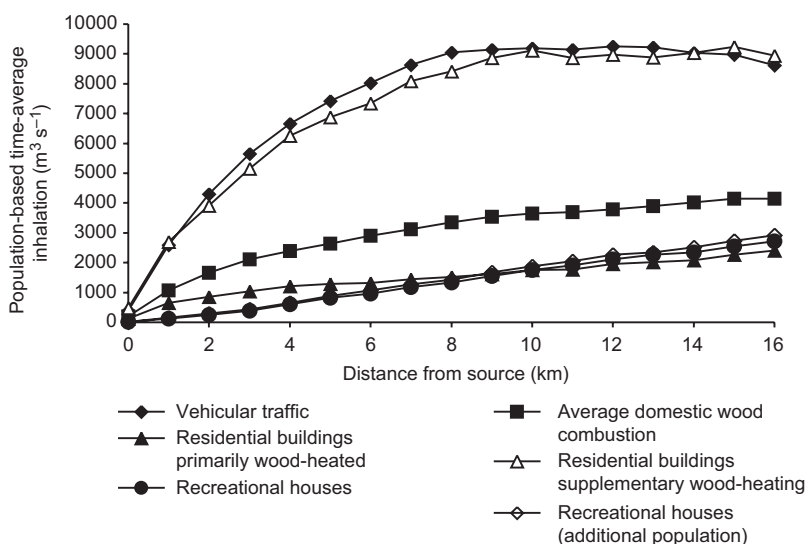


Fig. 3. Population-based time-average inhalation as a function of distance from the source. Numerical results are presented (i) for two source categories, vehicular traffic and domestic wood combustion, (ii) for three sub-categories of wood combustion, and (iii) for the additional sensitivity analyses (additional population) with respect to wood combustion in recreational houses. The additional sensitivity analysis was performed by adding one person to each recreational houses representing the situation where part of the population is in their summer cottages.

was based on activity data in the national vehicular traffic emission calculation system LIISA (Mäkelä *et al.* 2002) that is considered to be relatively well defined. However, the data on urban vehicular traffic activity outside the Helsinki area, accounting for 29% of the total vehicular traffic activity, does contain considerable uncertainty (K. Mäkelä, VTT Technical Research Centre of Finland, pers. comm.). The allocation of 1×1 km² grid cells within the municipality was based on road cover land use information in the SLICES concept. This allocation did not take into account traffic volumes in different road segments. This leads to considerable uncertainty within the municipalities. The influence of this factor on PTAI values could not be estimated at this stage of the study. In the future, the Finnish national road database digiroad® (www.digiroad.fi) which contains locations of roads and streets and traffic volumes at road segments, will be incorporated into the FRES model.

Vehicular traffic emissions are released to the atmosphere from close to the ground; these range characteristically (from 0.1 to a few meters). The concentrations of PM_{2.5} commonly decline rapidly downwind from roads and streets on a scale of tens or hundreds of metres (e.g. Tiitta *et*

al. 2002). Close to the source the dispersion is often strongly influenced by buildings and other obstacles. The emission heights for domestic wood combustion emissions from the stacks of detached houses in Finland range typically from 5 to 15 m. The near-field dispersion may be substantially different from that of vehicular traffic. Atmospheric dilution therefore tends, on average, to be more efficient for domestic wood combustion, in comparison to vehicular traffic.

The population data used in this study were based on the building and dwelling register. The population statistic is relatively accurate, and the data was considered of good quality. However, the use of population data based on the locations of permanent residences leads to some uncertainty. The majority of the population works and conducts business outside the 1×1 km² grid cell of their place of residence. This probably leads to an underestimation of exposure in city centres and other urban areas, and thus an underestimation of the PTAI values to vehicular traffic. Deterministic exposure models have been developed that take into account commuting into individual cities, as well as the vehicular traffic and recreational activities within an urban agglomeration (e.g., Kousa *et al.* 2002). How-

ever, this kind of data is not available for the whole country.

Summing up the qualitative discussion above, it could be estimated that, the uncertainties and data gaps do not alter the order of magnitude of the calculated PTAI values. The PTAI results show that domestic wood heating, for the most part, had less potential to cause exposure (per emission) to primary PM_{2.5} in comparison with vehicular traffic. However, the high PTAI values of supplementary wood heating does indicate that regulatory measures to reduce wood combustion emissions should be investigated and enforced by the authorities in order to prevent the increase of emissions in suburban areas. This is especially important since the tendency to install supplementary wood heaters into new detached houses is expected to continue. There is one sign of a positive development; the Finnish government is preparing new emission and efficiency regulations for new domestic wood combustion stoves and heaters.

In conclusion, we have developed a concept to estimate intake fractions for PM_{2.5}, or for other inert air pollutants, in two parts that require different source data and computational needs. The other component, namely Population-based time-average inhalation, was defined and applied in this study for the primary fine particulate matter emissions from vehicular traffic and domestic wood combustion. The results showed that PTAI for vehicular traffic emissions is clearly higher than that originating from average domestic wood combustion emissions, thus meaning that traffic has the potential to cause a higher PM_{2.5} exposure per unit emission. The PTAI for supplementary wood heating was highest from domestic wood combustion sub-sectors, evidence that this sub-sector is most important per unit emission. The applicability of the concept of separating population and atmospheric parts of intake fraction needs to be further studied. Our future aim is to carry out dispersion modelling and produce dilution functions for vehicular traffic and domestic wood combustion at a resolution of 1 × 1 km². This will enable quantitative population exposure and health risk assessment.

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References

- Bennett M.J., Yansura M.E., Hornyik I.G., Nall J.M., Caniparoli D.G. & Ashmore C.G. 2002a. Evaluation of the CALPUFF Long-range transport screening technique by comparison to refined CALPUFF results for several power plants in both the eastern and western United States. In: *Proceedings of the Air and Waste Management Association's 95th Annual Conference, June 23–27, Baltimore, MD*, Paper #43454.
- Bennett D.H., McKone T.E., Evans J.S., Nazaroff W.W., Margni M.D., Jolliet O. & Smith K.R. 2002b. Defining intake fraction. *Environ Sci Technol.* 36: 207A–211A.
- EC (European Commission) 2005. *Thematic strategy on air pollution*. Communication from the Commission to the Council and the European Parliament. Brussels, 21 Sep. 2005 COM.
- EPA (U.S. Environmental Protection Agency) 1997. *Exposure factors handbook*. National Center for Environmental Assessment Office of Research and Development.
- Forsberg B., Hansson H.-C., Johansson C., Areskoug H., Persson K. & Järholm B. 2005. Comparative health impact assessment of local and regional particulate air pollutants in Scandinavia. *Ambio* 34: 11–19.
- Glasius M., Ketzel M., Wahlin P., Jensen B., Monster J., Berkowicz R. & Palmgren F. 2006. Impact of wood combustion on particle levels in a residential area in Denmark. *Atmos. Environ.* 40: 7115–7124.
- Greco S.L., Wilson A.M., Spengler J.D. & Levy J.I. 2007. Spatial patterns of mobile source particulate matter emissions-to-exposure relationships across the United States. *Atmos. Environ.* 41: 1011–1025.
- Hoek G., Brunekreef B., Goldbohm S., Fischer P., & van den Brandt P.A. 2002. Association between mortality and indicators of traffic-related air pollution in the Netherlands: a cohort study. *Lancet* 360: 1203–1209.
- Holland M., Hunt A., Hurley F., Navrud S. & Watkiss P. 2005. *Final methodology paper (volume 1) for service contract for carrying out cost-benefit analysis of air quality related issues, in particular in the clean air for*

- Europe (CAFE) programme. Didcot, UK AEA Technology Environment.
- Karvosenoja N. 2008. Emission scenario model for regional air pollution. *Monographs Boreal Env. Res.* 32: 1–55.
- Karvosenoja N., Porvari P., Raateland A., Kupiainen K. & Johansson M. 2005. The spatial allocation of air pollutants in Finnish regional emission model. In: Topcu S., Yardim M.F., Bayram A., Elbir T. & Kahya C. (eds.), *Proceedings of the 3rd Air Quality Management Conference, Istanbul 26–30 September*, Hava Kirlenmesi Arastirmalari, Izmir, Turkey, pp. 571–580.
- Karvosenoja N., Tainio M., Kupiainen K., Tuomisto J.T., Kukkonen J. & Johansson M. 2008. Evaluation of the emission uncertainties of PM_{2.5} originated from vehicular traffic and domestic wood combustion in Finland. *Boreal Env. Res.* 13: 465–474.
- Kousa A., Kukkonen J., Karppinen A., Aarnio P. & Koskentalo T. 2002. A model for evaluating the population exposure to ambient air pollution in an urban area. *Atmos. Environ.* 36: 2109–2119.
- Kukkonen J., Pohjola M., Sokhi R.S., Luhana L., Kitwiroon N., Fragkou L., Rantamaki M., Berge E., Odegaard V., Slordal L.H., Denby B. & Finardi S. 2005. Analysis and evaluation of selected local-scale PM₁₀ air pollution episodes in four European cities: Helsinki, London, Milan and Oslo. *Atmos. Environ.* 39: 2759–2773.
- Levy J.I., Wilson A.M., Evans J.S. & Spengler A.D. 2003. Estimation of primary and secondary particulate matter intake fractions for power plants in Georgia. *Environ. Sci. Technol.* 37: 5528–5536.
- Marshall J.D. & Behrentz E. 2005. Vehicle self-pollution intake fraction: children's exposure to school bus emissions. *Environ. Sci. Technol.* 39: 2559–2563.
- Mikkola A., Jaakkola O. & Sucksdorff Y. *The slices project: National classification of land use, land cover and soil, and the production of databases*. Finnish Environment 342, Helsinki, Finland, 1999. [In Finnish with English summary].
- Mäkelä K., Laurikko J. & Kanner H. 2002. *Road traffic exhaust gas emissions in Finland. LIISA 2001.1 calculation model*. VTT Research Notes 2177, Finland.
- Pope C.A. & Dockery D.W. 2006. Health effects of fine particulate air pollution: lines that connect. *J. Air Waste Manage. Assoc.* 56: 709–742.
- Sevola Y., Peltola A. & Moilanen J. 2003. Polttopuun käyttö pientaloissa 2000/2001. *Finnish Forest Research Institute Research Papers* 894.
- Stieb D.M., Judek S. & Burnett R.T. 2003. Pseudo-analysis of time-series studies of air pollution and mortality: Update in relation to the use of generalized additive models. *J. Air Waste Manage. Assoc.* 53: 258–261.
- Tiitta P., Raunemaa T., Tissari J., Yli-Tuomi T., Leskinen A., Kukkonen J., Härkönen J. & Karppinen A. 2002. Measurements and modelling of PM_{2.5} concentrations near a major road in Kuopio, Finland. *Atmos. Environ.* 36: 4057–4068.
- Tissari J., Salonen R.O., Vesterinen R. & Jokiniemi J. (eds.) 2007. *Puun pienpolton päästöt, ilmanlaatu ja terveys [Emissions from residential wood combustion, air quality and health]*. Kuopion yliopiston ympäristötieteen laitoksen monistesarja 2/2007. [In Finnish with English summary].
- WHO 2003. *Health aspects of air pollution with particulate matter, ozone and nitrogen dioxide, Report on a WHO Working Group*. Report on a WHO working group. Bonn, Germany, 13–15 January 2003.