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EMISSION MODELING IN THE ASSESSMENT OF PM2.5 FROM TRAFFIC AND RESIDENTIAL WOOD COMBUSTION

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ABSTRACT

Fine particulate matter (PM) concentrations in ambient air have been cause severe health effects. The highest primary fine PM emissions in Finland are caused by sources with low emission height, such as traffic and residential combustion, which enable immediate exposure near the source. The Finnish Regional Emission Scenario (FRES) model calculates primary PM emissions in several size classes with 1×1 km^2 spatial resolution. In this study the importance of PM2.5 emissions from road traffic and residential wood combustion, and their relation to population densities in Finland were studied. The statistical comparison involved no atmospheric dispersion modelling, instead relative differences between sectors in potential human exposure were estimated. Preliminary results show that more population lives adjacent to PM2.5 emissions from road traffic than from residential wood combustion. The national total emissions from road traffic are lower. However, when only effects near the source are considered, PM2.5 emissions from road traffic are relatively more important to human health. The study gave new insight on the relative importance of the two main sectors on the potential PM2.5 exposure near the emission source. The results supplement PM health risk information that will be obtained from spatially more coarse regional PM assessment modeling.

Key Words: Traffic, Wood Combustion, Emission, Fine Particles, Population

1. INTRODUCTION

The ambient particulate matter (PM) has been associated with multiple adverse health effects worldwide. The adverse health effects have been seen for both short-term (daily variations) and long-term (chronic) studies (e.g. Stieb et al. 2003, Hoek et al 2002). The most consistently association has been found between ambient PM and increased cardiopulmonary mortality, lung cancer mortality and reduced lung function (WHO 2003).

The average $PM_{2.5}$ concentrations in Finland are relatively low compared to those in central and southern Europe. The highest concentrations occur in southern Finland, where for the Helsinki metropolitan area Pakkanen et al. (2001) have measured fine particle concentrations during October 1996 – May 1997 both at an urban and at a regional background site, and the average concentrations were 11.8 and 8.4 $\mu g/m^3$, respectively. It was estimated that at the background site, less than 10% of the measured concentration originates from local sources. For the urban site, long range transported (LRT) contribution to $PM_{2.5}$ concentration was estimated at 60-63% (Ojanen et al., 1998). These contributions are supported by computational values of Karppinen et al (2005), who estimate the LRT contribution to the measured $PM_{2.5}$ concentration in urban air in Helsinki at 64-76%.

The highest primary fine PM emissions in Finland originate from residential wood combustion and road traffic. Based on national emission statistics submitted to the Convention on Long-range Transboundary Air Pollution (CLRTAP) and its secretariat at the United Nations Economic Commission for Europe (UNECE) (Finnish Environment Institute 2005) residential wood combustion emitted 15.9 Gg(PM2.5) a^{-1} in 2003, which contributed 41% of the Finnish total PM2.5 emissions. However, emission factor estimates are under revision at the moment. Road traffic caused 3.0 Gg(PM2.5) a^{-1} direct exhaust and 1.7 Gg(PM2.5) a^{-1} indirect dust emissions. Emissions from these sources are released into the atmosphere from low height, and therefore enable immediate exposure near the source.

The health effects of fine particulate matter have recently been recognized in Europe and North America. International and national air quality guidelines set limits to daily and annual average concentrations, but the emissions are currently not included in the protocols of the Convention on Long-range Transboundary Air Pollution (LRTAP) or in the national emission ceilings directive (NECD) of the European Union (EU). Intensive research work is onging to address potential health effects (UNECE 2004a) and methods to include them in integrated assessment models (IAM). The possibilities for their inclusion in forthcoming emission reduction agreements under LRTAP Convention or within EU work, including the Clean Air for Europe (CAFE) thematic strategy, is being explored (UNECE 2004b). These assessments mostly cover transboundary air pollution, however, recently there have been attempts to link regional concentrations to urban background levels. The first results indicated that the relative share of local PM emission was generally below 50% and highly variable. However, need for studies focusing on primary PM effects at fine spatial resolution has been identified (Forsberg et al. 2005).

Finnish regional IAM of PM is in development in KOPRA project (www.fmi.fi/research_air/air_47.html). The aim is to integrate PM emission and reduction cost estimates, atmospheric modeling and health risk modeling. The emissions at 1×1 km^2 resolution and the reduction costs of technical control measures are estimated in the Finnish Regional Emission Scenario (FRES) model. Atmospheric dispersion modeling have been done with the modelling system SILAM (Sofiev & Siljamo, 2003) for two grids: 30 km resolution was selected for the European computations based on official EMEP emission data and 5 km grid cell size was taken for Finnish regional simulations. In both runs, the output of meteorological model HIRLAM was used for the complete year 2000 with 30 km meteorological data resolution. The health risk model combines the emission and dispersion data with the background population data. The dose-response semi-model will be used to estimate the relationship between particles and associated health effects in Finland.

This study concentrated on primary PM2.5 emissions from residential wood combustion and road traffic in 1×1 km^2 grid, and their relation to population densities in adjacent grid cells. Emission calculation and spatial allocation are presented. Population densities are weighted with emissions for various distances from emission grid cells, and relative human exposure potential is estimated. The study involved no atmospheric dispersion modelling.

2. METHODOLOGY

2.1 Emission calculation and spatial allocation

The Finnish Regional Emission Scenario (FRES) model is a part of the integrated assessment model (IAM) system of air pollution (Johansson et al. 2001) which have lately been extended to PM (www.fmi.fi/research_air/air_47.html). FRES consists of coherent emission calculation from all anthropogenic sources with spatial allocation of emissions. The pollutants include primary particles in different sizes and the main precursor gases of secondary PM. Large energy production and industrial plants are described as point sources. Area emissions are calculated at country level, and spatially allocated at two steps to municipality and 1×1 km^2 level. A more detailed model description can be found from Karvosenoja and Johansson (2003a).

A summary on the spatial dimensions of road traffic and residential wood combustion is presented in Table 1. For road traffic the spatial allocation to municipality level is based on fuel consumption values from the Finnish road traffic emission calculation system LIISA (Mäkelä *et al.* 2002). Allocation to 1×1km^2 level was carried out based on national SLICES land use element (Mikkola et al. 1999). SLICES is a combination of different national land use GIS databases, of which output is raster databases in 10×10 and 25×25 m^2. The allocation that is based on road surface area does not take into account traffic volumes in different roads. As a basis for the municipality allocation of domestic combustion of different fuels, degree-day weighted gross-floor areas of different types of buildings from national building and dwelling register were used. Building and dwelling register is a part of SLICES and includes all Finnish buildings with information on e.g. gross floor area, heating types and fuels, building date and resident. Degree-day weighting was assumed to represent the differences in room heating needs in different parts of the country. Gross-floor area data from building and dwelling register were used also in 1×1km^2 level allocation of domestic wood combustion. Population data used in this study was also based on building and dwelling register. The spatial allocation of area emissions is described in detail in Karvosenoja et al. (this issue).

Table 1. The spatial allocation of road traffic and residential wood combustion in FRES model

Main sectors at country level; Number of subsectors and fuels	Sectors at municipality level; Data used as a basis for emission allocation	Sectors at $1 \times 1km^2$ level; GIS data used as a basis for emission allocation from municipality to 1x1km2 level
Road traffic; 10 subsectors, 3 fuels	Light-duty gasoline exhaust; Annual LD gasoline consumption[1]	Road traffic; Road surface area[2]
	Light-duty diesel exhaust; Annual LD diesel consumption[1]	
	Heavy-duty diesel exhaust; Annual HD diesel consumption[1]	
	Light-duty dust emissions; Annual LD fuel consumption[1]	
	Heavy-duty dust emissions; Annual HD diesel consumption[1]	
Residential combustion; 16 subsectors, 3 fuels	Wood comb. in boiler-heated residential buildings (RsB); Degree-day weighted gross-floor area (DWGA) of wood boiler -heated RsB[3]	Wood comb. in wood-heated RsB; Gross-floor area (GA) of wood-heated RsB[3]
	Wood comb. in stove-heated RsB; DWGA of wood stove -heated RsB[3]	
	Wood comb. in RsB heated primarily with other than wood; DWGA of detached RsB built after 1979 and heated primarily with other than wood[3]	Wood comb. in RsB heated primarily with other than wood; GA of detached RsB built after 1979 and heated primarily with other than wood[3]
	Wood comb. in recreational buildings (RcB); DWGA of all RcB[3]	Wood comb. in RcB; GA of all RcB[3]

1) Mäkelä et al. 2002; 2) SLICES (Mikkola et al. 1999); 3) Building and dwelling register (Mikkola et al. 1999)

The emission calculation of road traffic and residential wood combustion at country level are presented in Tables 2 and 3. Traffic exhaust emission factors and reduction efficiencies are based on the RAINS model of IIASA (Klimont et al. 2002). The emission factors of traffic induced dust are based on international literature reported in Karvosenoja et al. (2002). Annual activity (i.e. fuel consumption) and technology utilization rate (i.e. age profile of vehicle fleet) values in year 2000 are based on the Finnish road traffic emission calculation system LIISA (Mäkelä *et al.* 2002).

Residential wood combustion activities (i.e. wood combustion amounts in different types of combustion appliances) are based on several questionnaire studies (Sevola et al. 2003, Tuomi 1990) and expert estimates (*e.g.* S. Tuomi, Finnish Work Efficiency Institute, 28.8.2003). The emission factors are based on the results of a Nordic project (Sternhufvud et al. 2004) and recent Finnish measurements (Raunemaa et al. in press).

2.2 Emission weighted population

For the two studied sectors s, residential wood combustion and road traffic, emission weighted population EWP values were calculated for various distance zones from the emission sources. EWPs describe sector specific emission weighted average populations at various distance zones over all Finnish emission grid cells. For emission grid cells n (i.e. grid cells where emission $EM_n > 0$), population grid cells p (i.e. grid cells where population $POP_p > 0$) in various distances adjacent to emission grid cells are explored. EWPs are calculated for each distance zone d from 0 to 16 km in 1 km steps (see Figure 1).

$$EWP(s,d) = \frac{\sum_{n}(EM_n(s) \cdot \sum_{p} POP_p(d))}{\sum_{n} EM_n(s)} \quad (1)$$

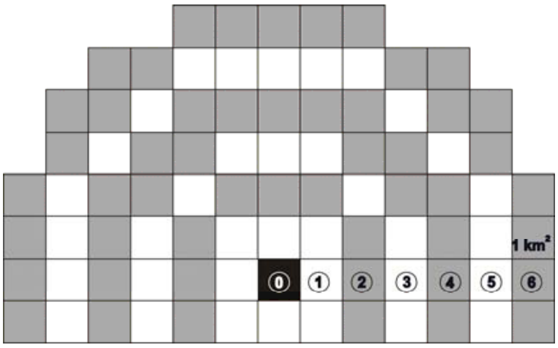

Figure 1. The schematic presentation of distance zones (circled numbers) from emission grid cell (black). The distance was from 0 to 16 km in the calculations.

2.3 Average exposure concentration

Without dispersion modeling it is not possible to quantitatively assess human exposure to emissions. However, relative differences between sectors in potential human exposure can be estimated. If the dilution and transport of emissions in the atmosphere (here described with dilution function DF) is assumed equal to all geographical directions and in all parts of Finland, PM concentration caused by a primary PM emission source at certain distance from the emission source is relative to emission quantity. For one emission grid cell n', average exposure concentration EC at a particular distance from the source can be calculated:

$$EC^{n'}(s,d) = EM_n^{\,n'}(s) \cdot DF(s,d) \cdot \sum_{p} POP_p^{\,n'}(d) \quad (2)$$

Since DF is assumed constant, average exposure concentration over all Finnish grid cells is:

$$EC(s,d) = DF(s,d) \cdot \sum_{n}(EM_n(s) \cdot \sum_{p} POP_p(d)) = DF(s,d) \cdot \sum_{n} EM_n(s) \cdot EWP(s,d) \quad (3)$$

i.e. relative to EWP multiplied by the total Finnish emission of the sector. This is a useful simplification for sector comparison, although the full utility can only be obtained if dilution functions are estimated as well.

3. RESULTS AND DISCUSSION

Country level PM2.5 emission calculation of road traffic and residential wood combustion in 2000 are presented, and FRES emissions compared to statistics (Finnish Environment Institute 2005) in Tables 2 and 3. Residential wood combustion country total emissions are higher than road traffic emissions. When FRES results are compared to statistics, the difference is large in residential wood combustion. Considerable uncertainties in the activities and especially emission factor estimates of residential wood combustion have been identified (Karvosenoja and Johansson 2003b). The emissions submitted to CLRTAP and its secretariat at UNECE (Finnish Environment Institute 2005) and earlier FRES estimates have been calculated using relatively old emission factor estimates that are based mainly on international measurements. Since that the emission factors have been revised in the light of recent Nordic (Sternhufvud et al. 2004) and Finnish (Raunemaa et al. in press) measurements. However, the uncertainties are still large because of the nature of residential wood combustion. Residential wood combustion appliances are relatively simple and poorly controllable, and emission factors are strongly dependent on user's habits.

Biggest uncertainties in traffic emission calculation lie on indirect dust emission factor estimates. So far there were very little Finnish road dust measurements available, and the current estimates used in FRES are based on international measurements (presented in Karvosenoja et al. 2002). The estimates are relatively conservative describing average emissions from clean paved roads. Indirect dust emissions may, however, be considerably higher, especially during spring time because of winter sanding dust resuspension. Statistics (Finnish Environment Institute 2005) do not include estimate on indirect traffic emissions for the year 2000. Indirect emissions in current statistics 2003 are calculated using the same emission factors than in the FRES model.

Table 2. PM2.5 emissions in 2000 from road traffic and comparison to statistics.

	Activity	Unabated PM2.5 emission factor	Control technology; utilization rate; reduction efficiency	PM2.5 emission (Gg a^{-1})
Motorcycles & mopeds, 2-stroke, exhaust	0.2 PJ a^{-1}	95 mg MJ^{-1}	EURO1, 9%, 30%	0.02
Passenger cars, vans & 4-stroke motorcycles, gasoline, exhaust	72 PJ a^{-1}	6.0 mg MJ^{-1}	EURO1, 16%, 45% EURO2, 31%, 45% EURO3, 4%, 82%	0.3
Passenger cars & vans, diesel, exhaust	32 PJ a^{-1}	111 mg MJ^{-1}	EURO1, 14%, 35% EURO2, 30%, 74% EURO3, 4%, 81%	2.4
Trucks, buses & other heavy duty, diesel, exhaust	45 PJ a^{-1}	58 mg MJ^{-1}	EURO1, 20%, 36% EURO2, 47%, 74% EURO3, 7%, 82%	1.4
Buses, natural gas, exhaust	0.05 PJ a^{-1}	1.8 mg MJ^{-1}	-	0.0001
Light-duty resuspension and tyre wear dust	43 10^9 km a^{-1}	18 mg km^{-1}	-	0.8
Light-duty break wear dust	43 10^9 km a^{-1}	2.8 mg km^{-1}	-	0.1
Heavy-duty resuspension and tyre wear dust	3.4 10^9 km a^{-1}	180 mg km^{-1}	-	0.6
Heavy-duty break wear dust	3.4 10^9 km a^{-1}	59 mg km^{-1}	-	0.06
Road traffic TOTAL	**149 PJ a^{-1} / 47 10^9 km a^{-1}**			**5.8**
Road traffic exhaust in statistics in 2000 (Finnish Environment Institute 2005)				3.6

Table 3. PM2.5 emissions in 2000 from residential wood combustion and comparison to statistics.

	Activity (PJ a^{-1})	Unabated PM2.5 emission factor (mg MJ^{-1})	Control tech.; util rate; red. eff.	PM2.5 emission (Gg a^{-1})
Residential buildings, manual feed boilers with accumulator tank	5.4	80	-	0.4
Residential buildings, manual feed boilers without accumulator tank	2.7	700	-	1.9
Residential buildings, automatic feed wood chip boilers	1.4	80	-	0.1
Residential buildings, automatic feed pellet boilers	0.1	30	-	0.003
Residential buildings, iron stoves	0.3	1000	-	0.3
Residential buildings, other stoves and ovens[1]	24	100	-	2.4
Residential buildings, open fireplaces	0.4	800	-	0.3
Recreational buildings, iron stoves	0.8	1000	-	0.8
Recreational buildings, other stoves and ovens[1]	4.0	100	-	0.4
Recreational buildings, open fireplaces	0.3	800	-	0.2
Residential wood combustion TOTAL	**39**			**6.8**
Residential wood combustion in statistics in 2000 (Finnish Environment Institute 2005)				15

1) incl. masonry heaters, masonry ovens, kitchen ranges and sauna stoves

1×1 km^2 resolution PM2.5 emissions from road traffic and residential wood combustion, and population are presented as 10×10 km^2 maps for the whole Finland, and as 1×1 km^2 maps for south-western Finland (Figure 2). Traffic emissions occur in population centers in south-western Finland. Residential wood combustion emissions are relatively evenly distributed to the whole southern and central Finland.

Spatial top-down emission allocation may add uncertainty. In residential combustion allocation is done using building and dwelling register (Mikkola et al. 1999) that contains information on primary heating devices of all Finnish buildings. The building and dwelling register data has been analyzed for this part and is considered relatively certain. However, 25% of the residential wood combustion emissions is caused by wood use in electricity and oil heated houses as supplementary heat source. The building and dwelling register does not contain information on supplementary heating devices. In this study, supplementary wood heating was spatially allocated using the time of inauguration in the building and dwelling register, with the assumption that detached houses commissioned in 1980 or later use supplementary wood heating, and older houses do not. This assumption is based on the fact that vast majority of residential buildings built in 1980s, 90s and 2000s are equipped with supplementary wood heaters. This part of the spatial allocation retain considerable uncertainty, but it was the best practical basis considering data availability.

For road traffic the spatial allocation to municipality level in FRES model is based on national road traffic emission calculation system LIISA (Mäkelä *et al.* 2002) that is considered relatively certain. The allocation from municipality to $1 \times 1km^2$ level was based on road cover land use information in SLICES land use element (Mikkola et al. 1999). It does not include small private or residential roads. Furthermore, the allocation does not take into account traffic volumes in different roads. Therefore emission allocation inside municipalities should be considered uncertain.

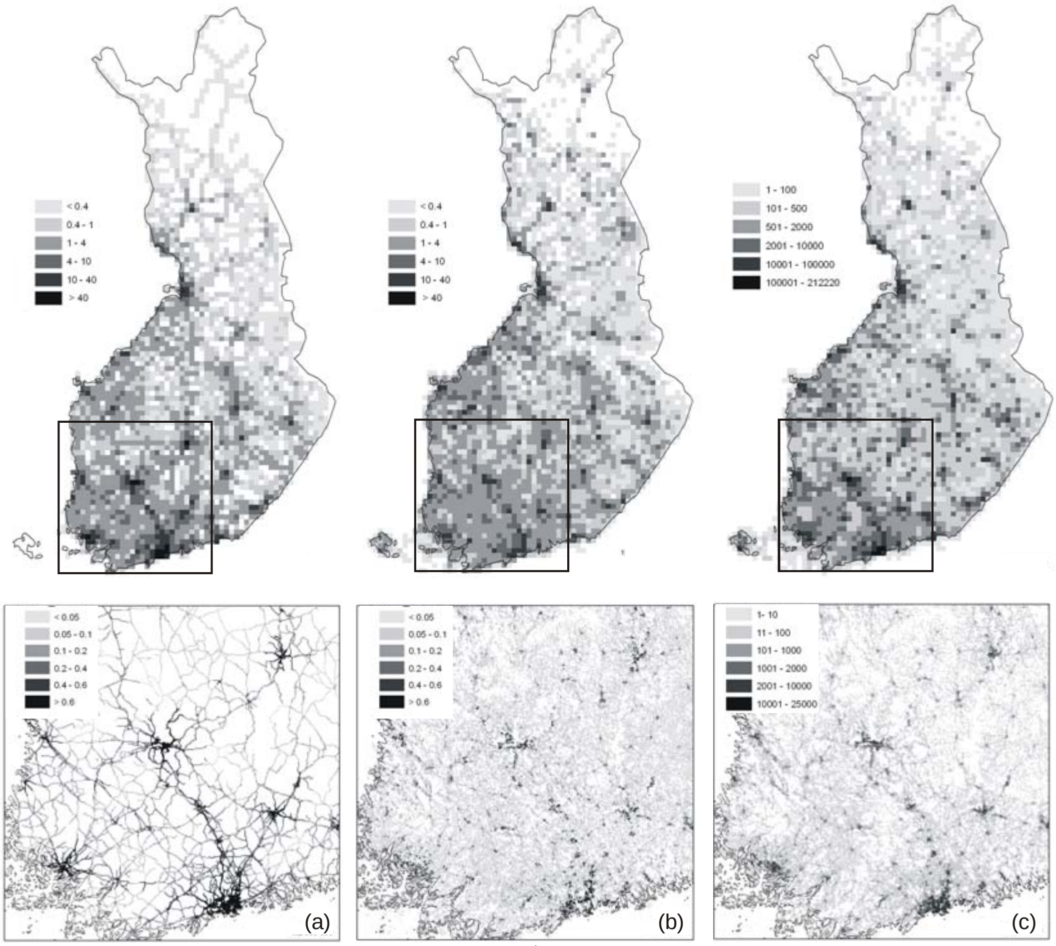

Figure 1. Finnish PM2.5 emissions (Mg a^{-1}) from (a) road traffic and (b) residential wood combustion in 2000 presented in 10 × 10 km^2 grid (upper) and 1 × 1 km^2 grid (lower). Population is presented in (c).

Table 5. Emission weighted population (EWP) and average exposure concentration divided by dilution function (EC/DF) in different distance zones. Inh = inhabitant.

Distance zone	EWP [inh.]	EWP per grid cell [inh km^{-2}]	EC/DF [inh Gg a^{-1}]	EC/DF per grid cell [inh Gg a^{-1} km^{-2}]	EWP [inh.]	EWP per grid cell [inh km^{-2}]	EC/DF [inh Gg a^{-1}]	EC/DF per grid cell [inh Gg a^{-1} km^{-2}]
	Road traffic				Residential wood combustion			
0	562	562	3260	3260	276	276	1877	1877
1	3361	420	19494	2437	1401	175	9527	1191
2	4163	347	24145	2012	1617	135	10996	916
3	4849	303	28124	1758	1817	114	12356	772
4	8514	266	49381	1543	3071	96	20883	653
5	6651	238	38576	1378	2376	85	16157	577
6	8552	214	49602	1240	3097	77	21060	526
7	7879	197	45698	1142	2864	72	19475	487
8	8664	181	50251	1047	3220	67	21896	456
9	11025	162	63945	940	4273	63	29056	427
10	8218	147	47664	851	3270	58	22236	397
11	9550	133	55390	769	3870	54	26316	366
12	8360	123	48488	713	3432	50	23338	343
13	9951	113	57716	656	4204	48	28587	325
14	9052	103	52502	597	4029	46	27397	311
15	8016	95	46493	553	3711	44	25235	300
16	9607	86	55721	498	276	276	31518	281

Table 5 presents the values of emission weighted population (EWP) and average exposure concentration divided by dilution function (EC/DF) in different distance zones for the two studied sectors. Dilution function was assumed constant in this study. EWP values for road traffic are more than twice as high as for residential wood combustion for all the distance zones, i.e. more population lives adjacent to PM2.5 emissions from road traffic. Road traffic emissions take for large extent place in urban areas, while residential wood combustion is more common in sparsely populated areas. Also EC/DF values are higher for road traffic than for residential wood combustion. This indicates that, when only effects near the source are considered, PM2.5 emissions from road traffic are relatively more important to human health.

5. CONCLUSIONS

This study provided a useful method to analyze the relative importance of two emission sectors with comparable emission heights without dispersion modeling. It indicated that road traffic is more important in potential health risks than residential wood combustion. The results in 1×1 km^2 grid supplement PM health risk information that will be obtained from the regional integrated assessment modeling project KOPRA, that will be carried out in 5×5 km^2 for Finland.

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