Aalto University
School of Science
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Emissions of Residential Wood Combustion in Urban and Rural Areas of Finland

Master's Thesis
Espoo, May 23 2012

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ABSTRACT OF MASTER'S THESIS

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Particulate matter (PM) is a significant threat of air pollution to human health in Finland and Europe. Residential wood combustion is a major cause of PM emissions. Therefore, the control of PM emissions is one of the most important challenges related to air quality.

The goal of this thesis was to identify the characteristics of Finnish residential wood combustion, study the $PM_{2.5}$ emissions and the population exposure they cause from different residential area types, and assess the emission reduction options for the future.

The total $PM_{2.5}$ emissions from residential wood heating was 8230 Mg a^{-1}, which amounted to 26% of the total emissions in Finland in 2005. Supplementary wood heating, i.e. stoves and masonry heaters, caused 70% of these. Non-urban areas were responsible for 57% of the total emissions.

Supplementary heating caused 89% of the total $PM_{2.5}$ exposure from RWC, with 80% of the total exposure coming from urban areas. In total, RWC was estimated to have caused around 200 premature deaths in 2005. From population exposure point of view, supplementary wood heating with stoves and masonry heaters was much more significant than primary heating with boilers.

Since the RWC is increasing in the future, the reduction of the emissions is important for the public health. The reduction measures should be targeted at supplementary wood heating in urban areas, as it comprises the majority of the population exposure. A viable way to reduce the emissions in the short term could be to affect the operational practices through an information campaign or legislation measures. A slower method is the renewal of the appliances with newer models with lower emissions.

Keywords:	Residential wood combustion, masonry heater, sauna stove, emissions, particulate matter, population weighted concentration, population exposure
Language:	English

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Hiukkaset on merkittävä terveysuhka Suomessa ja Euroopassa. Puun pienpoltto on suuri hiukkaspäästölähde. Hiukkaspäästöjen hallinta onkin yksi tärkeimmistä ilmanlaadun haasteista.

Tämän diplomityön tarkoitus oli määrittää Suomalaisen puun pienpolton ominaispiirteet, tutkia pienpolton pienhiukkaspäästöjä ja niiden aiheuttamaa väestöaltistusta eri asuinaluetyypeillä ja arvioida päästövähennyskeinoja tulevaisuudessa.

Kokonaispienhiukkaspäästöt puun pienpoltosta olivat 8230 Mg a^{-1}, joka oli 26% Suomen kokonaispäästöistä vuonna 2005. Lisälämmitys, eli varaavat takat ja uunit, aiheuttivat 70% päästöistä. Maaseutuasutus vastasi 57% kokonaispäästöistä.

Lisälämmitys aiheutti 89% puun pienpoltosta johtuneesta kokonaispienhiukkasväestöaltistuksesta, ja 80% altistuksesta johtui kaupunkialueiden päästöistä. Puun pienpolton arvioitiin aiheuttaneen 200 ennenaikaista kuolemaa vuonna 2005. Väestöaltistuksen näkökulmasta lisälämmitys varaavilla takoilla ja uuneilla oli huomattavasti merkittävämpää kuin päälämmitys boilereilla.

Koska puun pienpoltto on lisääntymässä tulevaisuudessa, päästöjen vähentäminen on tärkeää kansanterveyden kannalta. Vähennystoimet tulisi kohdistaa kaupunkialueiden lisälämmitykseen, sillä se aiheuttaa suurimman osan väestöaltistuksesta. Toteuttamiskelpoinen päästövähennyskeino lyhyellä aikavälillä voisi olla laitteiden käyttötapoihin vaikuttaminen valistuskampanjoilla tai lakisäädäntötoimilla. Hitaampi keino on laitteiden korvaaminen uusilla, vähäpäästöisillä malleilla.

Asiasanat:	Puun pienpoltto, päästöt, varaava takka, puukiuas, hiukkaset, altistuminen
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Espoo, May 23 2012

Ville-Veikko Paunu

Abbreviations and Acronyms

CAFE	European Commission's Clean Air for Europe program
CMH	Conventional masonry heater
EC	Elemental carbon
FRES	Finnish Regional Emission Scenario model
MMH	Modern masonry heater
NC	Normal combustion
NMVOC	Non-methane volatile organic compounds
OC	Organic carbon
OGC	Organic gaseous compounds
PAH	Polyaromatic hydrocarbons
PM	Particulate matter
PM_x	Particulate matter, with particles which aerodynamical diameter is smaller than x μm
POM	Particulate organic matter
PWC	Population Weighted Concentration
RWC	Residential wood combustion
SC	Smouldering combustion
SVOC	Semivolatile organic compounds
VOC	Volatile organic compounds
VVOC	Very volatile organic compounds
YKR	Urban structure monitoring system

Contents

Chapter 1

Introduction

Residential wood combustion (RWC) is a major cause of particulate matter (PM) emissions. PM is a major threat of air pollution to human health in Finland and Europe (Pekkanen, 2010). Therefore, the control of PM emissions is one of the most important challenges related to air quality.

There are numerous different types of RWC appliances, all of which have their own combustion and emissions characteristics. Devices that are the main heating means for a house are used with continuous fuel feeding. Other devices, which are used for supplementary heating or aesthetic effect, are loaded batch per batch. This affects the controllability of the combustion and, therefore, the emissions. Another important factor for the emissions is operational practices. If optimal combustion conditions are not met, the emissions can be multifold.

The particulate emissions from wood combustion have adverse health effects. They cause premature deaths and restricted activity days, all of which have serious social and economic impacts. Particle emissions from RWC have typically strong local effects, so it is essential to know which kind of surroundings the particles are emitted in. For example, high emissions in a scarcely populated area might mean lower population exposure than smaller emission in a dense residential area.

Residential wood combustion comprises 25% of anthropogenic $PM_{2.5}$ emissions in Finland (Karvosenoja et al., 2008). Wood combustion usually acts as a supplementary heat source in single-family houses. In Helsinki Metropolitan Area most of the wood is used in masonry heaters for supplementary heating and in sauna stoves (Gröndahl et al., 2011). Primary wood heating, i.e. houses that are primarily heated with wood, is less common in urban than rural areas. Supplementary heating is often intermittent, and batch combustion is the most common type. Primary heating, on the other hand, is usually continuous, and is often done by continuous combustion.

This thesis addresses the following questions: (1) what are the characteristics of Finnish residential wood combustion and how do they affect the particulate emissions? (2) How do RWC particulate emissions and population exposure differ in different types of areas in Finland? The study was outlined to concentrate on fine particles, as they are the most important emissions from RWC. In order to answer these questions, a literature review on RWC emissions measurements was conducted. The PM emissions and population exposure were modelled using the Finnish Regional Emissions Scenario (FRES) model. The emissions and exposures from different residential area types were compared. Possible reduction options and their allocation was discussed.

Chapter 2

Emissions of Finnish Residential Wood Combustion Appliances

The most important emissions from residential wood combustion (RWC) are particulates. Particulate matter (PM), especially fine particles, have adverse health effects on humans. They also have effects on the climate. Most particulates have a cooling effect, as they reflect sunlight. However, black carbon, which is an important component of particles from RWC, absorbs sunlight and acts as a warming agent. The research of the health effects has longer history than of the climate effects. This thesis concentrates on the health effects on humans.

2.1 Health Effects of Particulate Matter

The health effects of particulate matter have been studied for decades. The conclusion in the last decade has been, that the particulate matter emissions are the most significant cause of negative health effects that come from the environment (Salonen and Pennanen, 2006). The most harmful are particles with diameter that is smaller than 2.5 μm ($PM_{2.5}$). These so called fine particles penetrate deep into the lungs and the smallest can even enter bloodstream.

Exposure to particulate matter cause numerous health effects. These range from almost unnoticeable symptoms to serious diseases and death. Milder symptoms are sore nose and throat. More serious and long-lasting diseases include coronary disease, chronic obstructive pulmonary disease, asthma and lung cancer, which develop slowly. Premature deaths are usually caused by long-term exposure.

The health effects of particle matter depend on how deep into the system

Table 2.1: Some harmful effects of different particle sizes. (Salonen and Pennanen, 2006)

Particle size	**Short-term exposure**	**Long-term exposure**
Coarse particles (2.5-10 μm)	Asthma and chronic obstructive pulmonary disease worsen Respiratory infections	Chronic obstructive pulmonary disease?
Fine particles (<2.5 μm)	Asthma and chronic obstructive pulmonary disease worsen Coronary disease and cerebrovascular disease worsen Respiratory infections	Chronic obstructive pulmonary disease Atherosclerosis intensifies Lifetime shorten Asthma? Allergy?
Ultrafine particles (<0.1 μm)	Asthma worsen Coronary disease worsen	No research results

they are able to penetrate. Harmful effects associated with different particle sizes are presented in Table 2.1. Most of the biggest (>10 μm) particles are filtered in the upper respiratory track. Particles smaller than that find their way easier to the lungs and alveoli. From there, the ultrafine particles (<0.1 μm, $PM_{0.1}$) can penetrate the bloodstream and further into the heart and other organs. Particles cause damage to the cells, and the type of damage depends on the structure of the cell. Possible effects include increased cell death rate and increased inflammation risk, as well as physical damages, such as ruptures and blockage in veins. (Salonen and Pennanen, 2006)

European Commission's Clean Air for Europe (CAFE) program estimated, that fine particles caused 350 000 premature deaths within the population of EU (450 million) in 2000, shortening the lifespans on average by 8.1 months (CAFE, 2005). However, for certain groups with health problems the shortening can be even 10 years. In addition, fine particles increased medical and hospital treatments, and caused restricted activity days for tens of millions. The annual economical costs were estimated to be 270-280 billion euros. In Finland, which was the cleanest country in the report, it was estimated, that fine particles caused 1300 premature deaths and 600 chronic bronchitises in the year 2000. On top of that, tens of thousands of people suffering from cardiovascular diseases and also children were estimated to suffer from restricted activity and increased medication use. Economic losses were assessed to have been 1-2.9 million euros annually. It is notable that the

CAFE mainly estimated the effect of long-range transportation pollution. On the PILTTI project (Ahtiniemi et al., 2010) it was estimated that RWC and traffic emissions from Finland causes around a thousand premature deaths.

No threshold exposure for health effects have been found. Even exposure to small concentration for a long time can have significant consequences, since the amount of several particle types and, therefore, the effect in the system is cumulative. This means that they worsen chronic cardiovascular diseases all the time. Furthermore, high daily doses are believed to affect even after one to two months after the exposure.

2.2 Finnish Residential Wood Heating Appliances

Typical Finnish RWC applications are masonry heaters and different stove types. Finland has approximately 2.2 million small-scale wood-burning devices and, in addition, 1.5 million wood-burning sauna stoves (Tissari et al., 2008a). In 2009, Finnish residential buildings used 51.1 PJ of wood fuel (Statistics Finland, 2010). On average, the annual RWC activity increase during the 2000s has been 4% (figure 2.1) (Statistics Finland, 2010).

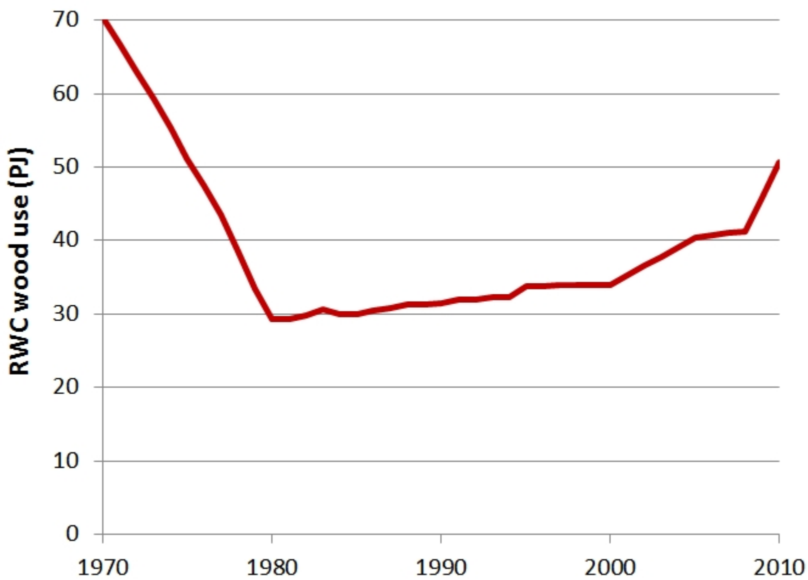


Figure 2.1: Fuel wood use in residential buildings in Finland. (Statistics Finland, 2010)

Figure 2.2 shows the residential wood fuel use by combustion appliance type in 2008 in Finland. The biggest portion of wood was used in masonry heaters and ovens, with log boilers second and sauna stoves third. While the number of log boilers is smaller than masonry heaters and sauna stoves, they are used for primary heating. Therefore, the wood use per appliance is higher.

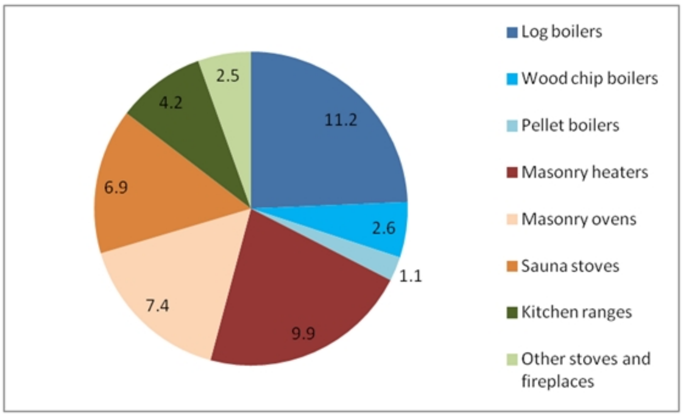


Figure 2.2: Fuel wood per appliance type use in residential buildings in Finland in 2008 (based on (Torvelainen, 2009)).

The small-scale wood combustion appliances and burning habits in Finland differ significantly compared to other countries. In many places in Central Europe, for example, the aim is to produce heat slowly over a long period of time, which means that the combustion is slow and the emissions high, if heat accumulator is not used. In Finland, the combustion rate is high and the operating time is short. (Tissari et al., 2008a)

2.2.1 Masonry Heaters

Tissari et al. (2008b) defines masonry heaters as "enclosed combustion appliances made of masonry products, a combination of masonry products or soapstone". They have a high mass (typically 800-3000 kg, but even up to 6000 kg), and usually an upright firebox, from which the exhaust gas flows to an upper combustion chamber, then down through the ducts and from the bottom or top of the heater in to the chimney. Figure 2.3 shows how the air is taken into the combustion chamber. The large mass of the heater efficiently stores the heat and releases it slowly into the surroundings (at the average rate of 1-3 kW) in a period ranging from 10 hours to two days. Most new detached houses in Finland are equipped with a masonry heater.

Figure 2.3: A modern masonry heater. The arrows indicate the intake air. (Alakangas et al., 2008b)

The combustion process of masonry heaters differs from stoves and conventional fireplaces. They have qualities that reduce the emissions. The surfaces of the firebox are hot and closed. The surfaces reflect heat back into the flame, creating the gas turbulence that complete combustion needs. Furthermore, secondary combustion is good in masonry heaters due to secondary combustion chamber, and efficiency is high because of the large mass. Compared to open fireplaces masonry heaters' air intake size is restricted and the operating temperature is higher. (Tissari, 2008; Tissari et al., 2008a)

Masonry heaters can be divided into conventional masonry heaters (CMH) and more developed modern masonry heaters (MMH). Modern masonry heaters have some advantages compared to conventional ones (Tissari, 2008). The primary airflow is controlled and there are small air inlet holes in the grate. These holes have several advantages: the secondary air is spread to surround the fuel batch, so mixing of air and combustion gases is increased; decreased air flow through the grate decreases coarse particle ejection into the flue gas; the overall excess air is reduced and thus the combustion temperature is higher, which reduces incomplete combustion during pyrolysis, and the emissions are lower. Also, when the secondary air is preheated it may decrease alkali metal compound release and improve secondary combustion.

In masonry heaters, the most problematic issues from the emission point of view are too fast pyrolysis and too high combustion rate compared to the air intake. Since air intake size is restricted, there may be an overall lack of available oxygen. Gasification rate can be controlled by the primary air supply, fuel moisture content and log and batch size. When combustion conditions are good, fine particle emissions are mostly composed of the vaporised ash forming elements from the wood. (Tissari, 2008; Tissari et al., 2008b)

2.2.2 Sauna Stoves

Sauna stoves are very common in Finland. They are problematic from the emission point of view. The structure is simplistic and combustion process undeveloped. A picture of a wood burning sauna stove is presented in Figure 2.4.

Figure 2.4: A Harvia sauna stove. (Harvia, 2012a)

Because of a small firebox and no secondary combustion, the efficiency of sauna stoves is low. Only half of the heat released is stored in the stones in the stove. Heating need in a sauna room is temporarily high, so sauna stoves operate with a high combustion rate. Furthermore, the air supply is insufficient in relation to the high gasification rate, and this results in incomplete combustion. To reduce the emissions, the combustion technique should be developed (for example, secondary air supply would lower emissions) or secondary removal techniques used. (Tissari, 2008)

2.2.3 Residential Wood Boilers

Wood log boilers are used mainly in rural areas as primary heating devices. In Finland, these are typically updraught (over-fire) boilers, which have higher emissions than other log boiler types, and are sometimes used without a heat storage tank. When a boiler is used without a heat storage tank, they are often used with low thermal output using smouldering combustion, and this causes high emissions. Other common boiler type is multi-fuel boilers, which can burn oil, wood or pellets. They are sometimes used primarily for wood log combustion with an updraught technique, and without a heat storage tank. Finnish manufacturers have also some crossdraught boilers, which are more developed and have lower emissions. However, these often lack control techniques that Central European similar boilers have, and the combustion process is more dependent on the user. Figure 2.5 shows schematics of updraught and downdraught boilers. (Tissari et al., 2005)

Wood chip boilers are mainly used to heat agricultural buildings and big single buildings, such as schools in rural areas. They operate similarly to side feed pellet burners, but with larger burner screws in order to allow the use of larger fuel particles. Most typical new appliances are 100-200 kW devices for agricultural buildings. The smaller scale boilers are almost exclusively stoker burners. In these, the burner can be mounted partially inside the firebox of the boiler, and partially outside, and only the hot flue gases are led to the boiler. (Tissari et al., 2005)

Pellet boilers (figure 2.6) are still rare in Finland. Most common pellet appliances follow the so called Swedish model. An oil burner of a former oil boiler is replaced with a pellet burner, and a pellet storage is built next to the boiler, from which the fuel is fed with a screw. Also the wood chip burners can be used to burn pellets. (Tissari et al., 2005)

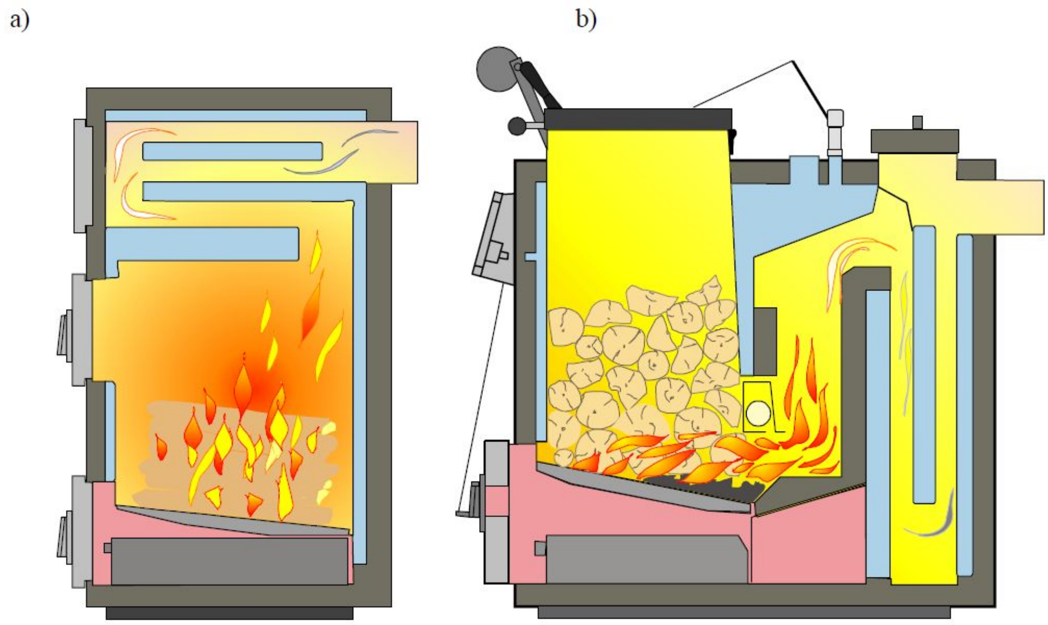


Figure 2.5: A schematic of (a) an updraught and (b) a downdraught boiler. (Tissari et al., 2005)

Figure 2.6: A pellet boiler. (Pellettipojat, 2012)

2.2.4 Other Appliances in Finland

In addition to masonry heaters, there is a number of masonry cookstoves (figure 2.7) in Finland, which often have a wood burning oven and a masonry heater in the same structure. These appliances can be used for both cooking and heating. Furthermore, there are some wood burning cookstoves in older buildings. There are also some open fireplaces in Finland. These appliances don't have air intake control, and the emissions are often high.

Figure 2.7: A masonry cookstove. (Alakangas et al., 2008b)

Iron stoves (other than sauna stoves) are not as common in Finland than in, for example, Central Europe. Iron stoves (figure 2.8) don't store the heat like masonry heaters. Therefore, if constant heating is needed, smouldering combustion might be used. This causes increase in the emissions.

Pellet burners (fig 2.9), just like pellet boilers, are still rare in Finland. In these appliances, the pellets can be fed manually or automatically. Especially with automatic fuel feeding, the combustion process can be controlled well. This allows optimal combustion conditions to be used over a long time, and emissions to be low.

One quite common device in Finnish summer cottages is a water boiler. In these boilers, the water is heated for domestic use, and used directly from the boiler (instead of supplying it into a water pipe). The devices are rather undeveloped, and combustion might be poor, causing high emissions.

Figure 2.8: An iron stove. (Lämpömaa, 2012)

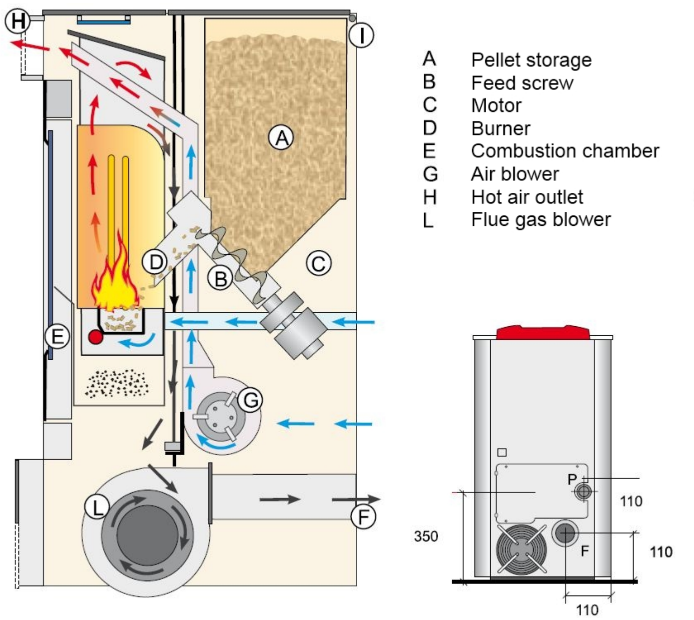


Figure 2.9: A schematic of a pellet burner. (Alakangas et al., 2008b)

2.3 Emissions of Air Pollutants

The $PM_{2.5}$ emissions from RWC are highly variable. Three key factors affecting the emissions in RWC can be identified: appliance type, fuel quality, and burning practices and conditions. A more complete list of factors affecting the emissions of RWC is presented in Table 2.2.

Although the appliance type gives an idea of the emissions of combustion, there are differences between the same type of appliances, and especially appliance sub-types. For example, conventional masonry heaters have typically higher emissions than modern ones. The main fuel characteristics affecting emissions are moisture content, ash content, and wood species. For burning practices, emissions are influenced by several factors, such as log size, batch size, arrangement of logs, and lighting. Most important factors in the burning conditions are high combustion temperature, sufficient amount of combustion air supply, and adequate mixing of combustion air and flue gas. These conditions allow complete combustion, which reduces emissions. (Tissari, 2008)

Even though appliance type has important effect on the emissions, combustion practices can have even higher effect (Paunu et al., 2012). Poor combustion practices, for example smouldering combustion or wet wood, can significantly increase the emissions of any device. Therefore, all appliances should be used with close to optimal parameters to achieve low emissions.

2.3.1 Wood as Fuel

Most wood species growing in Finland are suitable for fuel use. In residential combustion used wood is usually in the form of logs. Also wood chips, pellets, briquettes and hog fuel are used.

Fresh wood has typically 40-60% moisture content. In residential use, wood is usually dried before use. The desired moisture content for wood logs is around 15-20%. Dry pellets might have moisture content of 6%. For wood chips, the moisture content might be even up to 60%. Wood has a high content of volatiles, 80-90%. Therefore, it is a long-flame fuel and needs a large combustion chamber. About 99% of the dry matter of wood is carbon, hydrogen and oxygen. Nitrogen content is typically under 0.5%, and sulphur low, less than 0.05%. The ash content of stemwood excluding the bark is typically under 0.5%, or under 2% for coniferous trees. Bark has higher ash content, from 1.6-3%. The sulphur ash content in general is lower than in other solid fuels. The mineral content is usually less than 0.5%, main compounds being calcium (Ca), potassium (K), magnesium (Mg), manganese

Table 2.2: Factors affecting emissions of residential wood combustion. (Tissari et al., 2005)

Factor	**Characteristic affecting emissions**	**Effect**
Fuel	Moisture	Lowers combustion temperature
	Ash content	Increases particulate emissions
	Amount of gasifying substances	Pyrolysis control more difficult, flame needs a lot of space
	Log size	In continuous combustion affects how constant the combustion is In batch combustion affects firing and gasification rate
Appliance	Firebox size, shape and materials	Affects draft conditions and combustion temperature
	Flue gas outlet dimensions	Affects draft conditions
	Air supply	Affects the amount and mixing of combustion air
Smokestack	Stack height, size and shape	Affects draft conditions
Combustion conditions	Natural draft Affects combustion control	
	Flue gas residence time	Affects emission burnout
	Combustion temperature	Affects emission burnout
	Air supply and mixing	Affects flue gas and air mixing
Flue gas after-treatment	After-treatment appliances	Affects emission quantity, and appliance operation and use
Operating conditions	Combustion rate	Affects gasification rate and amount of combustion gases
	Fuel loading rate	Affects combustion rate and momentary power
	Control devices	Affects fuel, air and appliance power control
Appliance user	Operation habits, garbage burning	Affects several combustion factors

(Mn), sulphur (S), chlorine (Cl), phosphorus (P), iron (Fe), aluminium (Al), and zinc (Zn). (Alakangas, 2005; Tissari, 2008)

Typical heating values of Finnish wood logs are between 14 and 15 MJ kg^{-1} for moisture content of 20% (Alakangas et al., 2008a). For wood pellets with moisture content of 10% heating value is around 17 MJ kg^{-1} (Alakangas, 2005). The water content has clear impact on the heating value; the drier the wood the higher the heating value. Compared to other solid fuels, the heating value of wood is low, and wood also needs more storage space.

2.3.2 Combustion Process

Combustion is defined as a reaction where fuel reacts with oxygen, and heat is produces by this chemical process. From fuel particle point of view the combustion process consist of several phases: drying and heating of fuel, pyrolysis, firing and combustion. The first three phases consume heat. The flaming combustion and the combustion of the residual char generate heat. With wood fuels, combustion reactions happen mostly between gaseous products. In the residual char combustion the reactions are between gases and carbon, which is on the surface of the solid char. (Tissari, 2008)

The first phase, the drying and pyrolysis, consists of the warming of the fuel particle to drying temperature and, thereafter, vaporization of majority of the water. When the moisture content of the fuel has dropped sufficiently, the temperature of the fuel increases and the vaporization of the volatile hydrocarbons starts. Pyrolysis contains many complex chemical reactions, that are parallel and sequential. The fuel constituents begin to hydrolyse, oxidise and dehydrate, while the large structures, such as cellulose, hemicellulose and lignin, begin to degrade. Several gaseous and liquid products are formed during pyrolysis, e.g. volatile organic compounds (VOCs), H_2O, CO_2, H_2 and CO. (Tissari, 2008)

Devolatilization of wood starts at 200°C, and the devolatilization rate increases fast above that. Most volatiles have vaporized at 400°C, and devolatilization rate drops quickly. Decompositions of hemicellulose, cellulose and lignin occur at 200-350°C, 250-450°C and 200-500°C, respectively. (Tissari, 2008)

The kindling of the combustion gases happen when the amount of heat produced is higher than the heat loss to the environment. Fuel particle size and moisture have a significant effect on the time it takes the kindling to begin. Moisture slows down the kindling, as the vaporization of water consumes energy and the water vapour cools the surface of the fuel particle. Normally the pyrolysis products burn around the fuel particle as a diffusion flame. This generates heat for other pyrolysis reactions. As the heat generation increases the fuel temperature rises. The combustion is accelerated until pyrolysis gas production is slowed down. During the pyrolysis, the proportional share of carbon compared to hydrogen increases, and the residual char combustion starts. (Tissari, 2008; Tissari et al., 2005)

The last stage of the combustion is the flameless combustion of residual char. In this phase, the combustion happens on the surface of the fuel particle. Biomass combustion has normally around 10-30% of residual char content by dry weight. However, 25-50% of the total energy from the combustion comes from this phase. This phase also lasts for the longest time, as the diffusion of oxygen to the char surface is slow. The reactions between the gases and the char surface can also happen inside the fuel particle. Therefore, the porosity of the fuel particle affects the duration of the combustion. (Tissari, 2008; Tissari et al., 2005)

2.3.3 Batch and Continuous Combustion

Combustion in an RWC appliance is either continuous or batch type, depending on the appliance. In batch combustion, the fuel is loaded in separate batches, and the combustion starts from the first phase for each batch. The emissions differ between batches and combustion phases. For the first batch, fuel quality and combustion and firing practices have the strongest effect on emissions. Usually the temperature of the firebox is low at the time of firing, i.e. the high temperature that is needed for combustion is absent. To lower emissions, firing should be done on top of the first batch. This forces released gases to go through the flames and they are at least partly combusted. The first batch burns slower than subsequent batches, and especially the firing phase is longer than in the subsequent batches, so the combustion gases have more time to combust. Therefore the emissions from the first batch are not necessarily higher than from the following ones. As the firebox heats up as the combustion progresses, the rate of pyrolysis increases, and this can cause an increase in the emissions if pyrolysis gases are not combusted. (Tissari et al., 2005)

In continuous combustion, the fuel is fed continuously, and possibly automatically, to the burning chamber. All combustion phases are in effect all the time, and happen in the fuel layer. The combustion process is stable, and controllability is better compared to batch combustion devices. Unstable combustion can occur in situations where the process if interfered, appliance use is intermittent, during cleaning, or when the appliance is used with partial load.

2.3.4 Wood Combustion Emissions

When the carbon in the wood is completely combusted, only carbon dioxide (CO_2) is formed. However, the combustion is often incomplete. Therefore, part of the carbon is released as carbon monoxide (CO) and hydrocarbons (C_xH_y). Hydrocarbons can be grouped according to their boiling point: very volatile, volatile, and semivolatile organic compounds, and particle phase compounds (VVOC, VOC, SVOC, and POM, particle organic matter). Often VOC means non-methane volatile organic compounds (NMVOC), from which methane has been excluded. In measurements, the term organic gaseous compounds (OGC) is also used. Hydrocarbons can also be grouped according to the functional group. From these, polyaromatic hydrocarbons (PAH) are the most important ones, as many of them are carcinogens and mutagens. Nitrogen oxide (NO_x) emissions from residential wood combustion are mainly from the nitrogen of the wood. The temperature of RWC seldom rises high enough for NO_x to form from nitrogen in the air. Sulphur emissions from RWC are low, because the sulphur content of wood is typically low (below 0.05%). Water vapour (H_2O) forms as the moisture of the wood is vaporized and also from hydrogen of the wood. Furthermore, combustion air has water vapour, and this transfers to the flue gas. (Tissari et al., 2005; Tissari, 2008)

From good combustion, particulate emissions are formed mostly from minerals in the wood. Poor combustion produces more incomplete combustion products, namely soot, tar, gaseous hydrocarbons and carbon monoxide. Fine particles ($PM_{2.5}$) that are formed in wood combustion are ash particles, such as alkali sulphates and chlorides, calcium oxides and metal oxides, organic particles or soot particles with organics condensed on the surface. Bigger particles consist of ash, charred remainders of the fuel particles, or agglomerates formed from fine particles. (Tissari et al., 2005; Tissari, 2008)

The formation process of particles is illustrated in figure 2.10. Incomplete combustion may produce liquid or tar-like parts. These are formed by gas-to-particle conversion of organic vapours in cooled flue gas. The organic compounds can be in liquid or gaseous form, depending on the environmental conditions. New particles may be formed by nucleation of heavy hydrocarbons. These hydrocarbons may also condense onto existing particles. The latter process is more common. After the flue gases enter the chimney and, further, the atmosphere, the condensation of the particles continues as the combustion aerosols cool and are diluted. (Tissari et al., 2005; Tissari, 2008)

Fine ash particles are formed by homogeneous nucleation. This happens after the flame, where the temperature and the vapour pressure of the ash species decrease. The mineral compounds are easily released during the pyrolysis of the fuel. Temperature determines how much as particles is released:

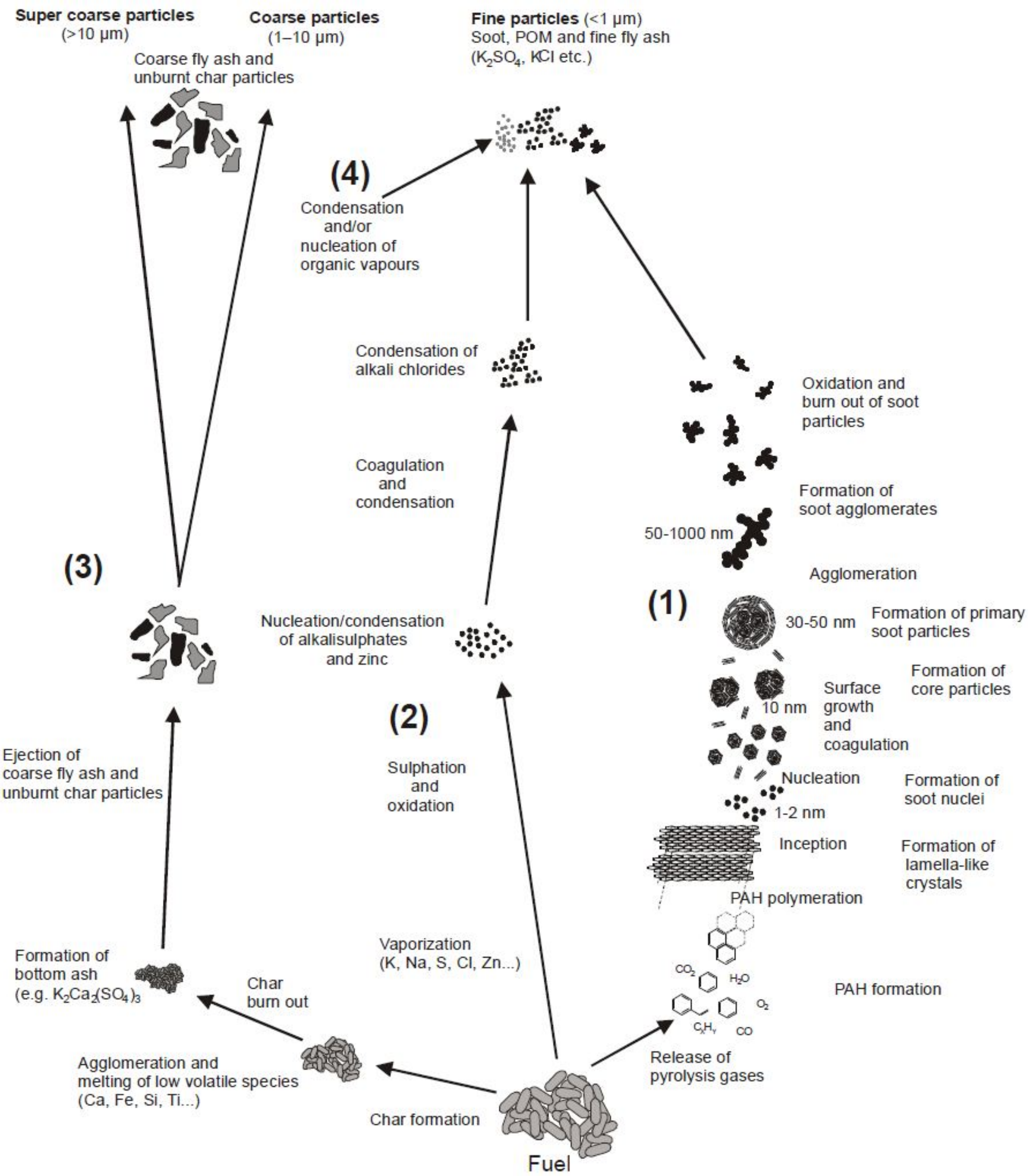


Figure 2.10: Formation of (1) soot, (2) fine ash, (3) coarse particles, and (4) particle organic matter (POM) in residential wood combustion. (Tissari, 2008)

higher temperature causes more ash particles to be released. Coarse particles ($\sim$1-10 μm) are formed from ash compounds with low volatility, and partially are unburnt char. At low temperatures the main formation process for large ash agglomerates is agglomeration. When the temperature is sufficient, the ash compounds can melt and form regular ash droplets. Residual fly ash particles lifted from the fuel bed form the super coarse particles (>10 μm). (Tissari, 2008)

Soot particles are mostly formed in the flame. Inside the diffusion flame there are fuel-rich areas, where the soot particles are formed from hydrocarbons as shown in the branch (1) of figure 2.10. These fuel-rich zones always exist in RWC, since the mixing of combustion gases and air is not sufficient. Therefore the hydrocarbons are unable to oxidise. The soot particles originate from PAH compounds. The PAH compounds polymerize, grow, and bond to the surface of core particles, finally forming primary soot particles. After the soot particles are formed, most of them are burned in the oxygen-rich area of the flame. However, small part of them are released unburned. The size and amount of soot particles is determined by the extent of soot oxidation. (Tissari, 2008)

2.3.5 Dispersion

The particle concentration and chemistry in the atmosphere caused by emissions depends on multiple factors. The topography of the environment and the weather conditions have a strong effect on local concentration levels. Other affecting factors are the source strength, distance from the source, the atmospheric processes encountered during the transport, and mixing and interaction with gases and particles from other sources during the transport (Pleijel, 2007). Lifetime of the particles in the atmosphere is quite short, from few hours up to a week (Pleijel, 2009).

The chemical composition of the particles transforms still after they are emitted to the atmosphere. When the hot exhaust gases are emitted, most likely condensable gases in the exhaust gas form new particles. Particle formation speed drops quickly after the emission due to rapid dispersion. Some of the smallest particles may evaporate. After this, the emitted organic gases can form condensable compounds through chemical reactions in the atmosphere, and these can then condense on the particles around. This is a much slower process. (Pleijel, 2009)

The temperature of the exhaust gas quickly drops after it leaves the chimney into the atmosphere. This enhances the condensation of different compounds on the surface of particles. The condensation depends heavily on temperature. It starts after the temperature drops below the boiling temperature for the compound, and accelerates as the temperature falls, until the exhaust gas is the in same temperature as the surrounding air. After this, condensation continues only with new condensable compounds produced by precursor gases by chemical reactions. This and other particle growing processes are made much slower by the strong dilution of the exhaust gases. (Pleijel, 2007)

New nanometre sized particles can also be formed by nucleation. When the temperature of the exhaust gases drop fast and by much due to cool ambient air, a certain compound can become highly supersaturated. This can also happen due to lack of surfaces for the gas to condense on. The concentration of the compound can be about 10 times at which the condensation starts. These supersaturated gaseous compounds can then from new small particles. (Pleijel, 2007)

As the emitted particles are transported further from the source, they are exposed to the particles and gases in the natural atmosphere, and the coagulation and condensation continues. The emissions are usually highly diluted now, so the processes are much slower and depend strongly on the concentration of participating components. Photochemical reactions induced by sunlight produce strong oxidizing agents, such as ozone. Chemical reactions turn non-condensable gases to condensable. Cloud droplets offer a reactive environment for the particle interactions. This can strongly affect the chemical composition of particles. (Pleijel, 2007)

2.3.6 Factors Affecting Emissions

From the emission point of view complete combustion is favourable. Three main parameters for complete combustion can be identified as follows: temperature of the combustion has to be high, air supply sufficient, and mixing of combustion air and fuel gas adequate. (Tissari et al., 2008b)

When air intake is restricted in the combustion chamber, combustion becomes smouldering. Also, if the burn rate is too high, the air supply becomes insufficient and this leads to smouldering-like combustion. Smouldering combustion is typical, for example, in old wood boilers, which don't have heat-storage tanks, and other appliances without heat storing, such as light metal stoves, since they are often used with a slow combustion rate with restricted air to keep up the heating for a long time. An insufficient control of primary air supply can lead to high particle organic matter (POM) and elemental carbon (EC) emissions.

The combustion temperature affects the emissions in a complex way. If the temperature is too low, the oxidation reactions are slower and combustion compounds don't burn out as completely as in higher temperatures. RWC appliances demand an overall excess of oxygen for local oxygen concentrations to be adequate for combustion reactions. However, the excess air lowers the temperature due to inert nitrogen being heated in the air. A low temperature and low local oxygen concentration both increase gaseous CO and volatile hydrocarbon emissions. The temperature can also be too high. In Finnish heaters it is typical that the air intakes are restricted and the combustion temperature is high. This leads to a too high gasification rate relative to the air intake size when large fuel batches are used. The supply of air is therefore insufficient and combustion becomes incomplete and emissions rise. (Tissari et al., 2008b)

The combustion temperature also affects the ash vaporization, so that in high temperatures the amount of released ash particles is higher than in lower temperatures. The chemical composition of ash is usually quite constant between different pure wood fuels, so the temperature mainly determines the fine ash emissions in RWC. (Tissari, 2008; Tissari et al., 2008b)

In batch combustion, the size of the fuel load affects the emissions. If the fuel load is too big or logs are too small compared to the heater air intake, combustion is incomplete. A large batch size increases gasification rate and results in an insufficient air supply. Tissari (2008) measured that doubling the batch size makes OGC emissions 4.0-, CO 2.2- and PM_1 1.9-fold compared to the initial batch size. Log size affected the emissions even more: small logs caused 8.7-fold OGC, 2.3-fold CO and 4.8-fold PM_1 emissions compared to larger logs with the same batch size.

The results of log size are not applicable universally, however. The observations from open fireplace emissions (Dasch, 1982; Stern et al., 1992) suggest that in these appliances bigger logs increase PM and CO factors. Furthermore, Tissari et al. (2008a) stated that there didn't seem to be a direct proportionality between the size of the logs and their gasification rate, and thus more research on the subject should be conducted.

Measuring emissions from RWC can be tricky, as the measurement methods may significantly affect the results. Furthermore, total PM is not a suitable factor when comparing different RWC appliances, as the emissions of total PM depend on the appliance and these particles occur randomly in the flue gas, and every measurement method causes remarkable losses of coarse particles. (Tissari, 2008)

2.3.7 Emissions from Finnish Appliances

Particulate emissions from masonry heaters and sauna stoves are mainly PM_1, i.e. the aerodynamic diameter of the particles is below 1 μm (Tissari et al., 2008a). Combustion conditions affect significantly PM_1 mass emissions as well as particle number and mass size distributions. In laboratory measurements made by Tissari et al. (2008a), the main difference between the PM_1 emissions of MMH and CMH comes from the firing phase. Otherwise the PM_1 emissions in the laboratory conditions were alike. PM_1 emission factors for both appliances, were 0.7 g kg^{-1}. The particle number emissions were higher in MMH, but the particle size was smaller, so the overall PM_1 emissions were similar. The PM_1 emissions from MMH were 50% ash compounds and less than 20% organics. Tissari claimed that this indicated a possibly different chemical composition of particles, and thus variable health effects, compared to other appliances. In field measurements made by Tissari et al. (2007), the PM_1 emission factors for MMH and CMH were 0.7 g kg^{-1} and 0.6-1.6 g kg^{-1}, respectively. All in all, Tissari (2008) states that the CMH emissions vary between 0.6 and 3.3 g kg^{-1}. Tissari found NO_x emissions to be relatively low with all wood fuels due to low combustion temperature.

Emissions differ greatly between different burning phases. The differences between emission factors were 10- to 100-fold between the phases. Tissari et al. (2008a) found that the biggest proportion of PM_1 emissions come from the firing phase, and the proportion grew higher from the first to the last batch. Most OGC emissions happened also in the firing phase (18 gC kg^{-1} on average), and CO emissions were high as well (59 g kg^{-1}). This was due to a high gasification rate and resulting insufficiency of air supply and air and fuel mixing in the firing phase. After the firing phase the PM_1 emissions decreased fast. OGC emissions were still significant in the combustion phase (2 gC kg^{-1}), but low in the burnt out phase (0.39 gC kg^{-1}) and nearly non-existent from the glowing embers (0.08 gC kg^{-1}). CO emissions were lowest in the combustion phase (13 g kg^{-1}), but stayed high during the burnt out phase (21 g kg^{-1}) and from glowing embers (22 g kg^{-1}). High CO emissions were caused by the low diffusion rate of oxygen to the char and combustion chamber cooling caused by the high excess air volume. The temperature often dropped below the complete oxidation threshold for CO, i.e. 800°C.

Emissions increase from the first batch because the temperature of the firebox rises and it accelerates the gasification rate of the wood. These reasons also decrease residual oxygen concentration. Thus, the supply of air might not be sufficient, resulting in incomplete combustion and high soot and organic carbon compound numbers in the emissions. OGC, CO and average PM_1 emissions grow from batch to batch. The increasing combustion temperature causes increase in the amount of all ash species. In Tissari's measurements, the last batch had 4.5 times higher PM_1 emissions than the first batch in a CMH.

Smouldering combustion (SC), i.e. combustion where air intake is restricted on purpose to slow down the combustion rate, significantly increases the emissions from masonry heaters. Frey et al. (2009) measured six to seven times higher POM emissions factors from SC than from normal combustion (NC) in a common type of a small Finnish masonry heater. In SC, POM made up 70% of the emissions, whereas in NC POM comprised 30% of the emissions. EC comprised 32% ± 5% of total emissions in NC and 22-27% in SC. The emission factors for EC were 0.9 ± 0.2 g kg^{-1} and 2.0-2.3 g kg^{-1}, respectively. The emission factors for POM were 0.9 ± 0.3 g kg^{-1} and 5.8-6.2 g kg^{-1}, respectively.

The emissions of CO and organic gaseous compounds (OGC) show the completeness of secondary combustion. Tissari (Tissari, 2008; Tissari et al., 2008b) found that the combustion temperature was lower in SC compared to NC, which represented the best combustion practice. Smouldering combustion was achieved with small logs, big batches and closed air intakes in a CMH. Most emissions were higher in SC, like OGC, POM, CO and PM_1, but ash and particle number emissions were smaller, less than 50% compared to NC emissions. Furthermore, SC had twice the particle size of NC. In addition, the particle composition was different between NC and SC. In NC, POM composed 33% and EC 32% of the emissions, whereas in NC they made up 67-69% and 22-27% of the emissions, respectively. Fine ash emissions were 2-4 times higher in NC than SC.

There are few scientific measurements made on emissions of sauna stoves. Tissari measured PM_1 emissions from sauna stoves in laboratory (Tissari et al., 2008a) and field (Tissari et al., 2007) conditions. The emission factors were 5.0 and 2.7 g kg^{-1}, respectively. In laboratory, CO emissions were 55 g kg^{-1} and OGC 10 gC kg^{-1}. In field measurements, CO emissions were clearly higher, 120 g kg^{-1}, and OGC 13 gC kg^{-1}.

The combustion conditions have a clear effect on the composition of PM emissions. Figure 2.11 shows how the composition differs between different devices and combustion practices. Modern masonry heater represents the best combustion, sauna stove a poor combustion and conventional masonry heater something from between. Smouldering combustion is the poorest combustion quality, with the highest emissions. The most interesting point is the ratio between elemental (EC) and organic carbon (OC). Organic carbon is directly proportional to POM. When the combustion quality comes poorer, the OC/EC ratio grows.

2.3.8 International Comparison

Compared to other parts of the world, masonry heaters are more common in Finland. Pellet burners and other stoves than sauna stoves, on the other hand, are rare in Finland in comparison. Since the appliance stock is different, the emission characteristics might be as well.

Emission-wise the Finnish supplementary heating devices have low to intermediate emissions. Masonry heaters, especially modern ones, have low emissions. Sauna stoves, being much more simplistic devices, have quite

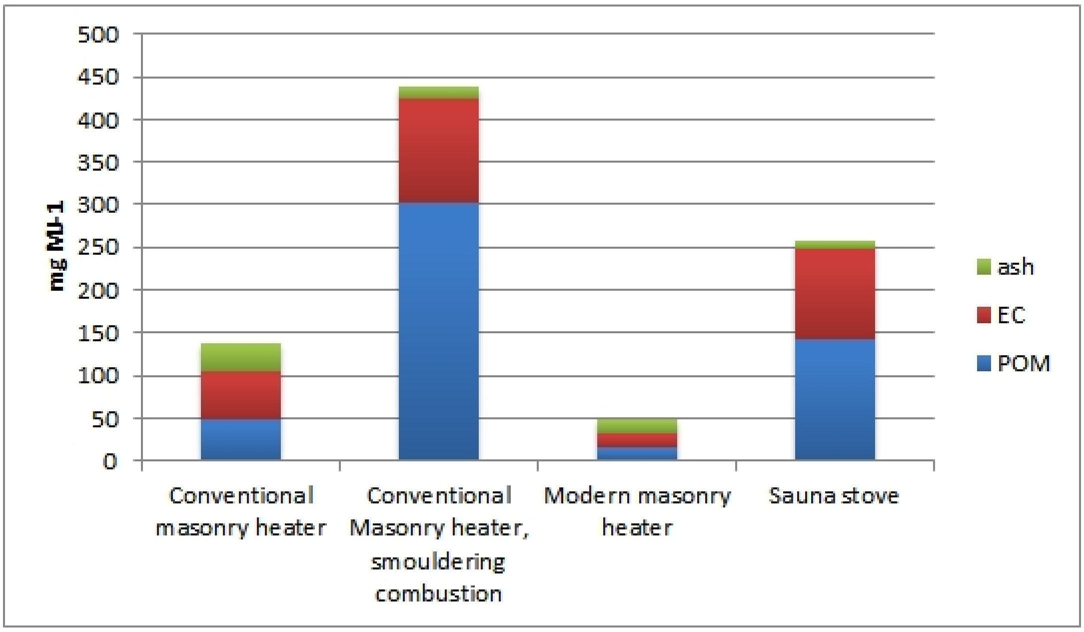


Figure 2.11: Chemical composition of PM_1 samples from masonry heaters and sauna stoves. (Tissari, 2008)

high emissions. In general, modern appliances, such as pellet burners, have the lowest emissions. Old devices, such as traditional iron stoves, old log boilers without a heat accumulator tank and open fireplaces, have high emissions. (Paunu et al., 2012)

Compared to Central Europe and Sweden, Finnish boilers are less developed and have higher emissions. Furthermore, legislation in Finland is less strict than in some other countries. For example, Swedish legislation doesn't allow updraught boilers to be used without a heat accumulation tank. (Tissari et al., 2005)

2.4 Emission Reduction Possibilities

There are numerous ways to reduce the emissions from residential wood combustion. These include the development of combustion technologies, cleaning of the flue gases, promoting better combustion practices and the use of legislation.

2.4.1 Advanced Combustion Technologies

The most straightforward way to reduce emissions from wood combustion is to change the device into a more modern one with lower emissions. Especially the replacement of old open fireplaces and log boilers with newer models may reduce the emissions significantly. The problem with this method is the price of the new devices, and the long lifetime of RWC appliances. Thus, on a larger scale, it doesn't matter how good the new appliances are, as the old ones will be replaced too slowly, if results are wanted in a few decades. By assessment made by Savolahti et al. (2012), replacing conventional masonry heaters with modern ones could reduce $PM_{2.5}$ emissions by 1% from Finland's total 2020 emission levels, with unit cost for the reductions being 253 k€/ton. Compared to other emission reduction options studied, replacement of current devices with new ones is expensive, be it log or chip boilers with boilers or masonry heaters with newer models.

Log boilers can be used with or without a heat accumulator tank. Equipping the tank allows the boiler to be used with optimal combustion rate without compromising continuous heating. Boilers without heat tank have to be operated with low combustion rate in order to produce heating at a constant rate, and this leads to higher emissions. Savolahti et al. (2012) estimated, that installing accumulator tanks to all boilers would reduce Finnish $PM_{2.5}$ emissions by 5% of the total emissions of 2020. The unit cost for the reductions was 2 k€/ton.

Harvia Oy has developed a sauna stove (Harvia GreenFlame, figure 2.12) that should have lower emissions and lower fuel use than traditional models (Harvia, 2012b). They claim that especially CO and particulate emissions are significantly lower. A patented automatic control mechanism controls the air intake. In the firing phase, the combustion air is fed from under the fire. When the combustion progresses and the stove heats up, the air is fed over the fire. In the burn out phase, as the stove cools down, the air inlets are closed. The mechanism is based on thermal expansion of metal. This device shows, that sauna stoves can be developed further without compromising the functionality of the stove.

There are some appliance types that are not so well known, but may be good choices. One of them is a rocket stove. They have an L-shaped internal design. Combustion chamber is at the end of a vertical or horizontal fuel magazine, and chimney is above the combustion chamber. The aim is a high combustion efficiency due to high combustion temperature and good air draft. These type of appliances are used for cooking in many third-world countries. A version of the device is a rocket mass heater, which combines the design with the masonry heater. A schematic is presented in figure 2.13.

Figure 2.12: Harvia GreenFlame sauna stove. The blue arrows show the air intake in the combustion phase. (Harvia, 2012b)

The emissions of rocket-type appliances should be low and efficiency high.

2.4.2 Combustion Practices and Legislation

Combustion practices have a high impact on emissions from RWC appliances. This is especially true for batch combustion, in which the user is usually solely responsible of the combustion parameters. However, it is hard to assess how much of wood is combusted with poor practices, as it is unknown how much users use their appliances in a non-optimal way.

Compared to technical measures, affecting the combustion practices could be a fast way to reduce emissions of RWC. In practice, this could be done via information campaign aimed at households that use wood as a fuel. However, the effectiveness of this kind of campaign is hard to estimate. Savolahti et al. (2012) assessed that even if the effect of such a campaign would be small, the unit cost of $PM_{2.5}$ emission reduce would be on the same level with the cheapest technical measures. This implies that even if results are uncertain prior to a campaign, it might still be worth a try.

Emissions can also be affected through legislative measures. For example, Germany and Austria have emission standards for new stoves. Limits for Finland have been under discussion, but no measures have been taken yet. However, European Union is preparing to add small-scale wood combustion

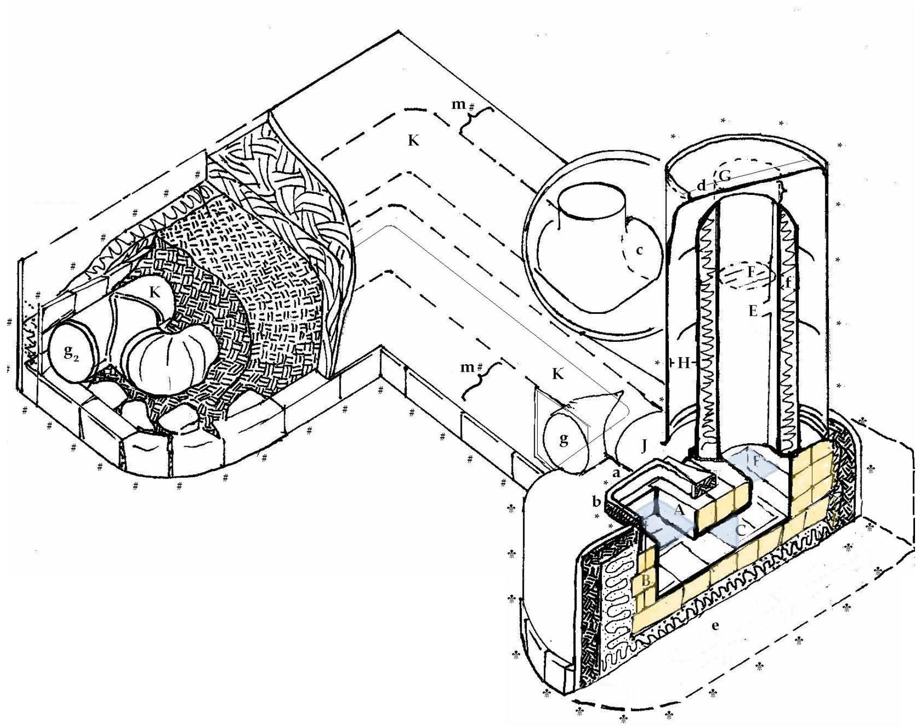


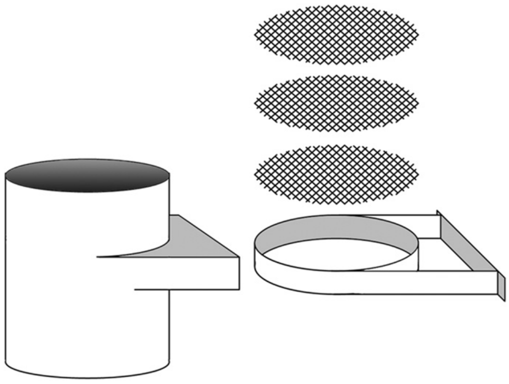
Figure 2.13: A schematic of a rocket mass heater. (ErnieAndErica, 2012)

devices into the Ecodesign Directive. As the appliances are replaced slowly, such measures would take a long time to have a significant effect. Emission limits could also be put on existing appliances, although supervising of such a law could be hard. Other measures include the banning of the use of the appliances for example during the time of general air quality problems. Legislation can also offer incentives to invest in appliances with lower emissions.

2.4.3 Flue gas cleaning

Electrostatic precipitators (ESP) are used in many power plants. In practice, small-scale ESPs are options for the future, as they are not yet commercial technology. There are other flue gas cleaning technologies in development, but ESP seems like the most effective. According to calculation of Savolahti et al. (2012), installing ESPs to all log boilers (with heat accumulator tanks) in Finland could reduce the $PM_{2.5}$ emissions by 2%, from the total emissions of Finland in 2020, with a unit costs 40 k€/ton, respectively. The unit costs for applying ESPs to chip or pellet boilers is significantly higher, making them infeasible options. It should be kept in mind that the prices of the commercial ESP units are uncertain, and it has significant impact on the unit costs of the emissions reduction. A picture of an ESP fitted to a stove is presented in figure 2.14(a).

(a)



(b)

Figure 2.14: Pictures of two main small scale wood combustion flue gas cleaning: (a) an "Airbox" ESP system installed to a stove (Hartmann et al., 2011), and (b) a schematic of a catalytic combustor (Hukkanen et al., 2012).

One technology to reduce RWC emissions is already in use: catalytic combustors have been widely used in United States. They have been found to effectively oxidise carbon monoxide and light non-methane hydrocarbons. One problem with using catalyst in RWC is the low temperature of the flue gases in some appliances. This is especially problematic in the start-up phase, when the emissions can be really high but temperature still low, preventing the catalyst to work optimally in the most critical moment. This can be solved by using heating systems, such as hot air injected through the catalyst, or placing the catalyst directly after the fire box instead of in the stack. Furthermore, the catalyst can be fouled easily, and its activity can decrease by the affected emissions. The use needs motivation from the user, as the catalyst needs to be cleaned regularly and might need surveillance. (Hukkanen et al., 2012)

Hukkanen et al. (2012) studied the reduction of sauna stove emissions with a commercial catalytic combustor (figure 2.14(b)). They measured emission drop for PM_1 from initial 391 to 252 mg MJ^{-1}, and for CO and OGC from 7900 to 6200 and 1500-1300 mg MJ^{-1}, respectively. The reduction occurred mainly during the gasification stage, except for CO. The same held for OC and EC emissions, with OC emissions reduced more than EC. EC requires a high temperature for oxidation, and was therefore affected less. However, the catalyst increased the flue gas temperature enough so that the oxidation was able to occur at all.

Chapter 3

Modelling of Population Exposure to Fine Particles from residential wood combustion

The emissions from residential wood combustion have an obvious effect on population exposure to $PM_{2.5}$. in addition to the emission strength, another important factor on the exposure is the population density; the same emission causes a bigger population exposure in a more densely populated area. Therefore, it is crucial to know where the pollutants are emitted. The aim of this chapter is to compare the $PM_{2.5}$ emissions and exposures from residential wood combustion (RWC) in different residential area types.

The spatial distribution and dispersion of $PM_{2.5}$ emissions of RWC in Finland was modelled. From the resulting concentration it was possible to estimate the population exposures, and compare these in the different residential area types with different population densities.

3.1 Methodology

3.1.1 Emission Modelling

The emission and exposure assessments were done with the Finnish Regional Emission Scenario (FRES) model (Karvosenoja, 2008). The FRES model is an integrated assessment model of air pollution emissions and impacts in Finland. The temporal and spatial resolutions for the model are one year and 1 km x 1 km over Finland, respectively. The main objective of the FRES model is to estimate the future emission scenarios, and evaluate the different pathways of emission reduction possibilities.

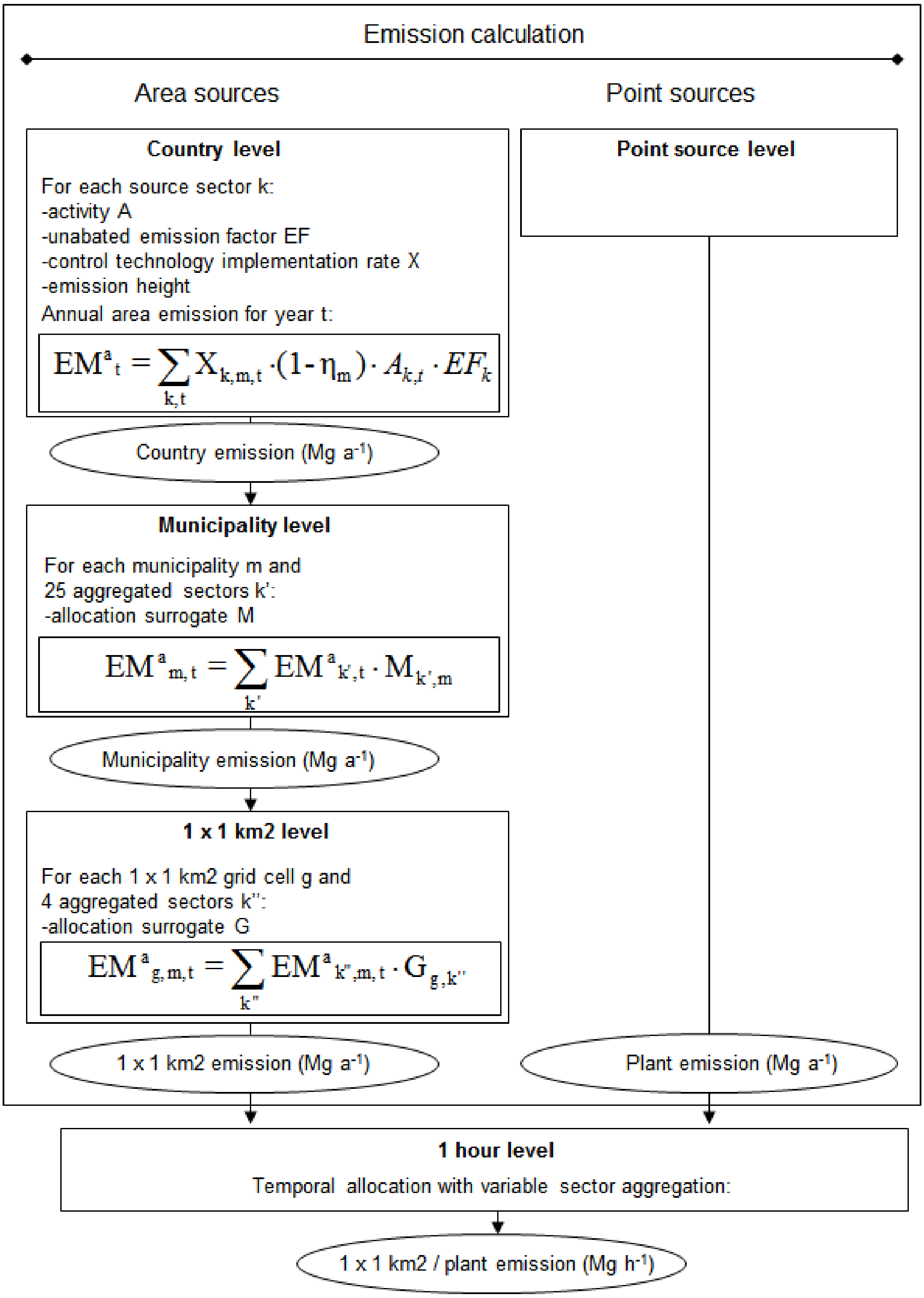


Figure 3.1: Flowchart of the emission calculation in the FRES model (Karvosenoja, 2008).

A flowchart of the FRES model is presented in figure 3.1. The FRES model uses both top-down and bottom-up approach for emission sources: area sources and point sources, respectively. The point sources include large plants with significant emissions on their own. Most emission sources are so small in emission quantities and sometimes also hard to pinpoint that it is not convenient to model them as point sources. Instead, these sources can be handled as area sources. For an area, there is some known or estimated number of sources, which are represented by allocation surrogates. Residential wood combustion are modelled as area sources.

In the FRES model, the emissions are calculated with activity levels, emission factors, and emission control technology removal efficiencies and utilization rates. For RWC, the activity is the annual wood use. For each source, the rate of emission per activity unit is specified with emission factors. The emission factors are determined so that the emissions represent the real life emissions of the device as accurately as possible. The most recent year in the model is 2005. This year was used in this study. The emission factors are supposed to be constant, and the changes in the factors are modelled by changing the emission control technology parameters or change in activity division.

The $PM_{2.5}$ emission factors for different appliances are presented in Table 3.1. The emissions of different RWC devices are studied in more detail in section 2. Less developed boilers, i.e. log boilers without a heat accumulator tank, have high emissions. Modern devices, especially pellet boilers, have low emission factors compared to other devices. Same holds for stoves and masonry heaters: modern masonry heaters have a low emissions factor, and fireplaces emit high emissions. Sauna stoves have quite high emissions, due to poor combustion technology. The emission factors used in the FRES model are based on literature, and are chosen to represent the real life usage. The wood uses in the FRES model in the year 2005 for primary (boilers) and supplementary (masonry heaters and stoves) wood heating are 14 and 30 PJ, respectively.

The calculation of the emissions is divided into several steps. The initial step is the fuel wood use estimation for the whole of Finland. The wood use is divided for different appliances types according to their estimated shares in the year 2005 (Torvelainen, 2009). Every appliance type has it's own emission factor (Table 3.1), which gives the $PM_{2.5}$ emissions per combusted energy unit. Therefore, the country level emissions for a device are calculated as:

$$EM_a = A_a \times EF_a \tag{3.1}$$

Table 3.1: $PM_{2.5}$ emission factors and activity levels in the FRES model for different RWC appliances (Torvelainen, 2009).

Appliance	**$PM_{2.5}$ emission factor (mg MJ^{-1})**	**Activity (PJ)**
Wood chip boiler	50	2.1
Pellet boiler	30	1.0
Log boiler with heat accumulator	80	7.3
Log boiler without heat accumulator	700	2.3
Farmhouse pellet boiler	30	0.14
Farmhouse log boiler with heat accumulator	80	1.3
Fireplace	800	2.5
Kitchen range stove	120	4.1
Conventional masonry heater	120	8.9
Masonry oven	120	7.3
Modern masonry heater	80	0.77
Sauna stove	200	6.7

where A is the activity (wood use), EF is the emission factor and a refers to the appliance type. This way the emissions of the whole of Finland are calculated for all appliance types, and then summarized to the source classes:

$$EM_s = \sum_a EM_a \tag{3.2}$$

where s refers to the source class. There are three RWC source classes: primary and supplementary wood heating, and primary wood heating in farmhouses. The classes are based on the appliance types, so the primarily wood heated houses have also supplementary wood heating. Primary wood heating class contains residential boilers, farmhouse primary wood heating farmhouse boilers, and supplementary heating all the other residential wood combustion devices (stoves, masonry heaters etc.). The farmhouse primary wood heating is included because the wood use in farmhouse boilers is bigger than in other residential house boilers (Torvelainen, 2009). The emissions of the farmhouse boilers represent this difference, not the whole emissions from the boilers, and are calculated separately. They are later included in the total primary wood heating emissions.

To achieve the 1 km × 1 km resolution the emissions of the whole of Finland are first distributed to municipalities according to source class activity, and then to the municipality's area. Every municipality and 1 km × 1 km cell has it's own weighting factor, which is comes from the allocation surrogates. The municipality emission is calculated by:

$$EM_{m,s} = \frac{S_{m,s}}{\sum_m S_{m,s}} \times EM_s \tag{3.3}$$

where m refers to the municipality, $EM_{m,s}$ is the emission of the municipality, S_m the allocation surrogate for the emission source class in the municipality, $\sum_m S_m$ the total allocation surrogate in the country and EM_s the total emission of the source class. The emission of a cell is calculated similarly:

$$EM_{c,s} = \frac{S_{c,s}}{S_{m,s}} \times EM_{m,s} \tag{3.4}$$

where c refers to the cell, $EM_{c,s}$ is the emission of the cell, $S_{c,s}$ the allocation surrogate for the emission source class in the cell, S_m the total allocation surrogate in the municipality and EM_m the total emission of the municipality.

The distribution base for RWC emissions depends on the source class. The bases are listed in Table 3.2. In practice, the emission is first distributed from the country level to municipalities, and in the next step within each municipality to 1 km × 1 km cells. All the weighting factors have first been calculated in the 250 m × 250 m resolution (the resolution for the residential area type data), and then aggregated to the 1 km × 1 km resolution.

Table 3.2: Distribution bases for different source classes.

Source class	**Allocation surrogate**	**Source**
Primary wood heating	Number of detached houses	National building and dwelling register
	Residential area type	Urban structure monitoring system
	Heating degree day	Finnish Meteorological Institute
Supplementary wood heating	Number of detached houses	National building and dwelling register
	Residential area type	Urban structure monitoring system
	Heating degree day	Finnish Meteorological Institute
	Primary heating (only cell weighting factors)	National building and dwelling register
Farmhouse primary wood heating		National building and dwelling register

For primary wood heating, the weighting factors of the municipalities and cells are determined by three factors: number of detached houses, residential area type and heating degree day. The number of the detached houses is from the Population Register Centre's National building and dwelling register (Mikkola et al., 1999) 2011 update. Residential area type is based on urban structure monitoring system (YKR) (Ristimäki, 1999), and has three options: large city urban (population of the city >20000), small city urban (population of the city <20000), and rural. The weighting factors for different area types are based on an annual wood use survey conducted by the Finnish Forest Research Institute (METLA) (Torvelainen, 2009). The annual wood uses of houses in different area types are presented in Table 3.3. The heating degree day represents the different heating need in different parts of the country. Each municipality has it's own heating degree day value in the FRES model. The values come from the Finnish Meteorological Institute (Tainio et al., 2009).

Table 3.3: The annual wood uses of houses in different residential area types. The numbers are based on an annual wood use survey conducted by the Finnish Forest Research Institute (METLA) (Torvelainen, 2009)

Residential area type	**Annual wood use (m^3/house)**
Primary wood heating	
Large city urban	9.3
Small city urban	16.1
Rural	19.0
Supplementary wood heating	
Large city urban	2.5
Small city urban	3.0
Rural	5.0

The emission distribution for the supplementary wood heating follows the same lines with primary wood heating. The basis is the detached house number from the National building and dwelling register. In the municipality level the distribution also takes into account the differences in wood use in different residential area types (Table 3.3). Municipality level also uses the heating degree day information. In the 1 km × 1 km level, the calculation also takes into account the differences of wood use in houses with different primary heating (Torvelainen, 2009). The wood uses for different primary heating types are presented in Table 3.4. The emission distribution for supplementary wood heating is structured so that the residential area type (and heating degree day) is only taken into account in the municipality level, and the primary heating type within each municipality in the 1 km × 1 km level.

Table 3.4: The annual wood uses of houses different primary heating. The numbers are based on an annual wood use survey conducted by the Finnish Forest Research Institute (METLA) (Torvelainen, 2009)

Primary heating	**Annual wood use (m^3/house)**
District heating	1.4
Oil	2.9
Electricity	3.3
Other	6.9

The emission distribution for primary wood heating in the farmhouses is altogether different. As with the two categories, the distribution weighting factors are based on the National building and dwelling register. A 250 m × 250 m cell is determined to contain a farmhouse if it contains both a detached house and a agricultural building. Cells satisfying this condition get a weight value of one. The cells are then aggregated into 1 km × 1 km resolution. These are the cell weights, and the municipality weights are the sum of the weights in the municipalities area weighted by the heating degree day factor. When the emissions have been calculated in the 1 km × 1 km resolution, the primary wood heating and primary wood heating in the farmhouses are added together to form the total primary wood heating emissions, and they are handled together from then on.

3.1.2 Dispersion Modelling and Population Exposure Assessment

As well as emission modelling, the FRES model includes $PM_{2.5}$ dispersion and population exposure modelling. The $PM_{2.5}$ dispersion modelling is based on source-receptor matrices calculated with urban dispersion modelling system (UDM-FMI) developed in Finnish Meteorological Institute (Karppinen et al., 1998). The system includes Gaussian dispersion model. The emission source size of the source-receptor matrices is 1 × 1 km^2, temporal resolution 1 hour, and the assumed emission release altitude for residential wood combustion 7.5 m, which include the initial plume rise. The emission is dispersed into a 41 × 41 km^2 grid square domain, in the centre of which the emission lies. The monthly and hourly variation is taken into account by temporal patterns shown in figure 3.2. Daily variation is assumed to be constant for residential wood combustion. (Karvosenoja et al., 2011)

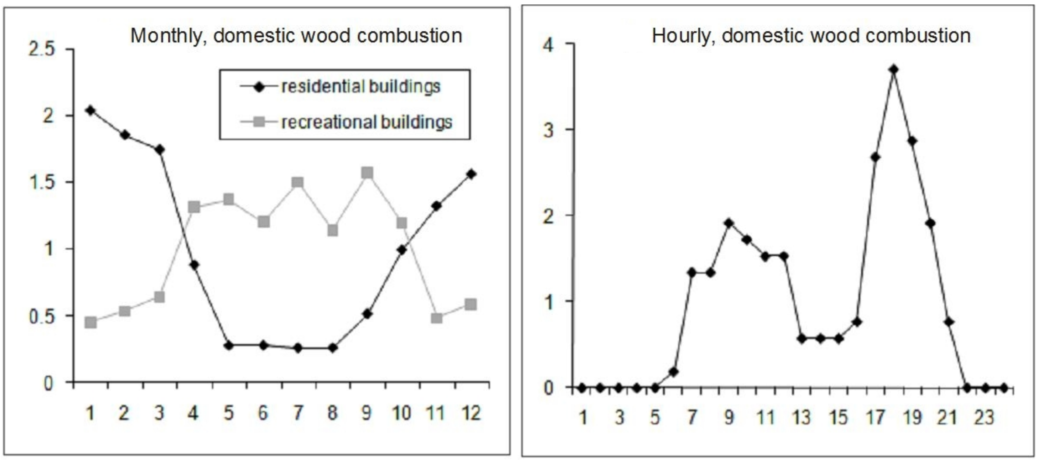


Figure 3.2: Temporal patterns of residential wood combustion emissions for monthly and hourly variation. (Karvosenoja et al., 2011)

The urban dispersion modelling system also includes MPP-FMI meteorological preprocessing model for meteorological input parameter estimation. The meteorological preprocessing model takes data from ten synoptic weather observation stations and two sounding stations as an input. The weather observation stations were chosen to represent different areas in Finland. Based on these, ten source-receptor matrices, named by the weather observation station locations, have been calculated. Figure 3.3 shows the allocation of the different matrices in Finland. The calculated concentrations represent the increase in the annual average concentration caused by an emission source. (Karvosenoja et al., 2011)

The dispersion of the $PM_{2.5}$ emissions is calculated by multiplying the emission with the source-receptor matrices. Every cell has a source-receptor matrix appointed to it, defined by the municipality the cell is in. The appropriate source-receptor matrix is placed on top of the emission cell, so that the emission cell is in the centre. The source-receptor matrix is multiplied by the emission, and the results are added to the underlying cells' concentration. In other words, the total concentration in a cell caused by a certain emission source class is the sum of the emissions multiplied by the appropriate source-receptor matrix within 41 km $\times$ 41 km square grid from the cell:

$$C_{c,s} = \sum_{d} EM_{d,s} \times SRM_{a,b} \tag{3.5}$$

where c to the cell, s refers to the source class, d to the cells within 41 km $\times$ 41 km square grid from the cell c, a to the source-receptor matrix for the municipality in question, b to the source-receptor matrix element, C is the concentration in the cell, EM is the emission of a source cell, and SRM is the appropriate source-receptor matrix.

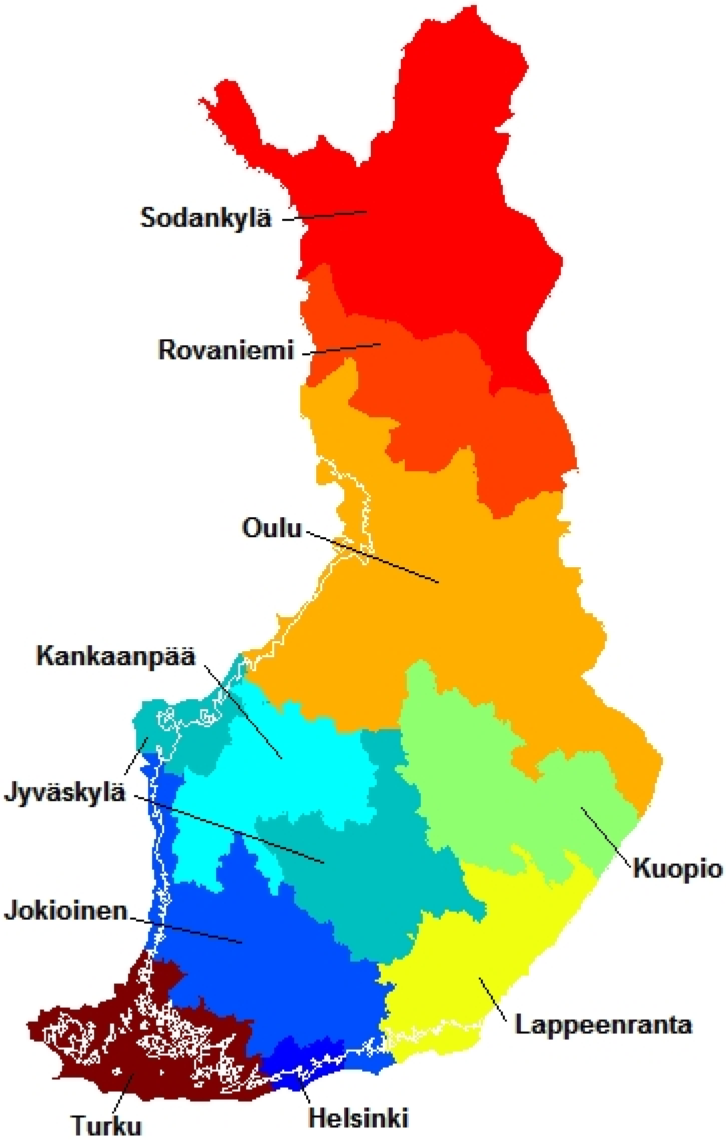


Figure 3.3: Allocation of dispersion matrices in Finland.

Population exposure to $PM_{2.5}$ is assessed with population weighted concentration (PWC). PWC is calculated by weighting the concentrations by population densities:

$$PWC = \frac{1}{Pop} \sum Pop_c C_c \tag{3.6}$$

where Pop_c is the population of the cell, C_c is the concentration in the cell and Pop is the total population of the area studied (whole Finland in this case). Population data was based on the locations of permanent residents from the national building and dwelling register (Mikkola et al., 1999). The population map for Finland is presented in figure 3.4.

In order to assess how important emissions from different residential areas were, an exposure/emission ratio was calculated. Exposure/emission ratio represents how big of an exposure a unit emission causes. A big ratio indicates, that the emissions cause big population exposure. The ratio is calculated by:

$$EER = \frac{PWC}{EM} \tag{3.7}$$

where PWC is the total population exposure from the residential area class and EM is the total emission from the class.

3.1.3 Residential Areas

Finland was divided into seven residential area classes: *block building, detached house, scattered detached house, other urban residential, villages, small villages* and *rural residential area*. Areas that were outside these classes were defined as *non-residential areas*. A map of the classes is presented in figure 3.5.

The residential area classes were based on urban structure monitoring system (YKR) (Ristimäki, 1999) developed in the Finnish Environment Institute. The criteria for the classes are presented in Table 3.5. The residential land use in a cell is represented by efficiency. The efficiency is calculated by dividing the total residential floor space with the land area. The criteria are applied for each cell, the size of which is 250 m x 250 m. The data was then aggregated into 1 km x 1 km -resolution. The highest (most densely populated class) determined the class of the aggregated cell. It is notable that the buildings in the FRES model can be situated in *non-residential area*.

The $PM_{2.5}$ emissions were calculated for each residential area class for primary and supplementary wood heating. The dispersion of the emissions was modelled as well. From the resulting concentrations, the PWCs caused by emissions from each class was calculated. In other words, the PWC of the

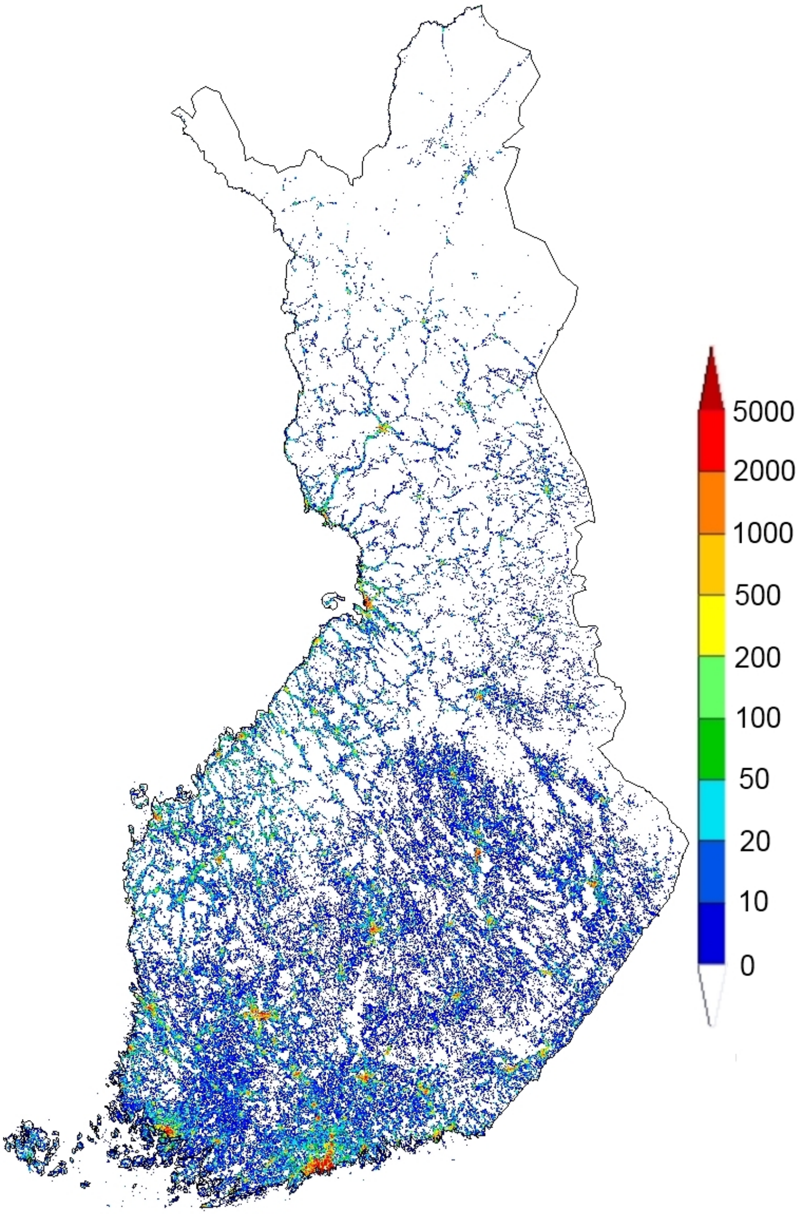


Figure 3.4: Population density (population/km^2) in Finland.

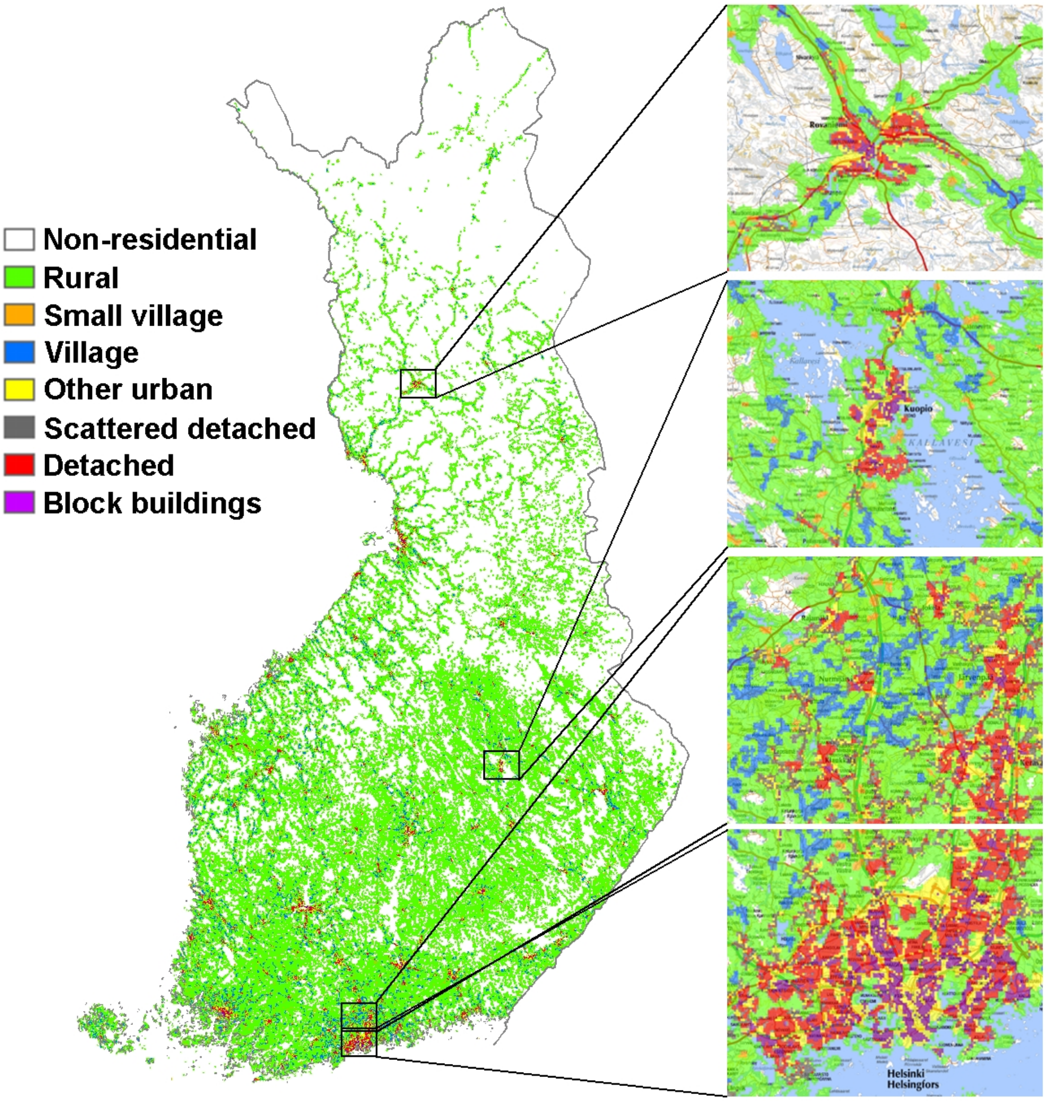


Figure 3.5: Residential area classes in Finland. Zooms are, from top to bottom, Rovaniemi, Kuopio, Nurmijärvi and Helsinki Metropolitan Area.

class represents the total exposure the emissions from that class cause in the whole Finland, not only within the class.

Table 3.5: The criteria of the residential area classes per cell based on YKR.

	Criteria
Block buildings	Residential floor space at least 40% of total floor space and at least 400 km^2 OR floor space at least 20% and at least 1000 km^2; Efficiency at least 0.02; block building floor space at least 60% of total residential floor space.
Detached	Same as above, except detached house flood space at least 40% of total residential floor space.
Scattered detached	At least one residential building and efficiency smaller than 0.02.
Other urban	Not any of the classes above; area with at least 200 residents.
Village	At least six residential buildings in total in the cell and adjacent cells; non-urban; at least 40 residents.
Small village	Same as above, except at least 20 residents.
Rural	Non-urban, non-village, non-water, at least one residential building within one kilometre.
Non-residential	Non of the above.

Chapter 4

Results

The $PM_{2.5}$ emissions and population exposures were calculated for the eight residential area classes. The total $PM_{2.5}$ emissions from RWC was 8230 Mg a^{-1}, which corresponds 26% of the total $PM_{2.5}$ emissions of Finland in 2005 (Hildén et al., 2008). The total PWC was 815 ng m^{-3}. The emissions and PWCs for primary and supplementary wood heating were 2440 and 5790 Mg a^{-1}, and 93 and 722 ng m^{-3}, respectively.

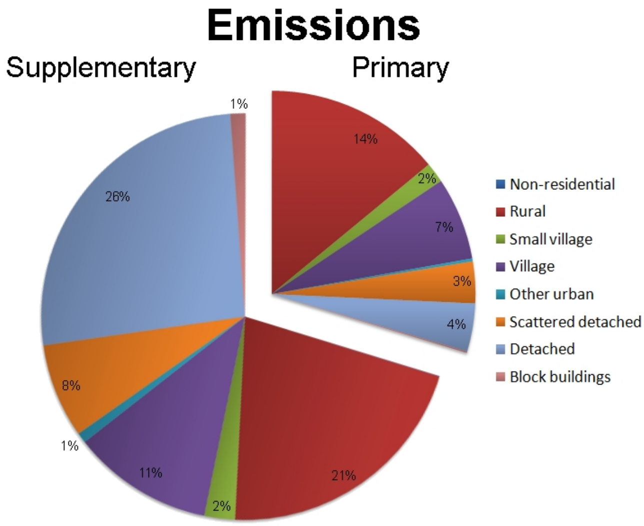
The emissions from supplementary heating were 70% of the total emissions from RWC, but they comprised 89% of the total PWC. The share in PWC was bigger, because the emissions of supplementary heating occurred closer to high population density areas in contrast to primary heating.

Primary wood heating caused 30% of the total RWC emissions, but only about 9% of the detached houses in Finland are primarily heated with wood. These houses have, therefore, bigger share of the emissions than the building stock. This reflects the fact that primary wood heating devices, i.e. boilers, use more wood than supplementary heating devices, i.e. masonry heaters and stoves, and the average emission factors are relatively high.

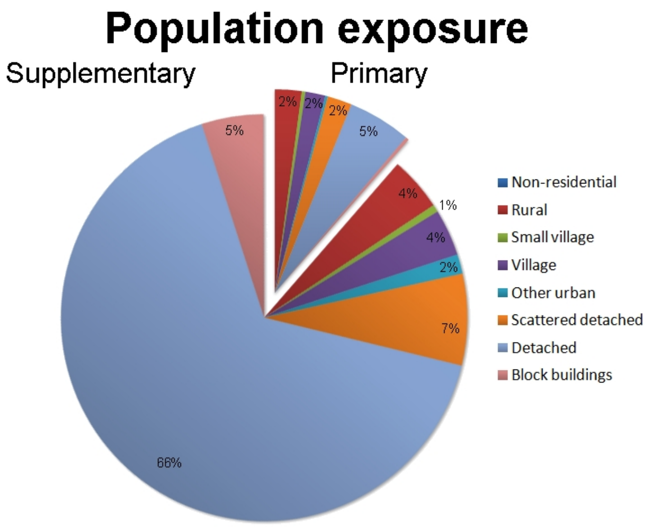
The $PM_{2.5}$ emissions and concentration maps are presented in figures 4.3 and 4.4. The maps show that the spatial distribution of the emissions is quite different for the primary and supplementary wood heating. The $PM_{2.5}$ primary heating emissions are distributed somewhat evenly. The emissions occurred mainly in rural areas, with highest emissions in Western Finland.

For supplementary wood heating, the cities clearly stand out from the maps. The Eastern and Southern Finland have higher shares than in primary heating case. As the highest emissions are in cities, they are close to areas with high population density. Therefore, the emissions have higher impact on the population exposure. In general the emissions from supplementary heating are higher than form primary heating in urban areas.

The $PM_{2.5}$ emissions, PWC caused by them and exposure/emission ratios are presented for different residential area classes in Table 4.1. The shares of emissions and PWCs from different residential area classes for primary and supplementary wood heating are presented in figures 4.1(a) and 4.1(b).



(a)



(b)

Figure 4.1: Share of $PM_{2.5}$ (a) emissions and (b) PWC from residential area classes for primary and supplementary wood heating.

Table 4.1: $PM_{2.5}$ emissions, PWC and exposure/emission ratios from primary and supplementary wood heating from different residential area classes.

Source	Emission (Mg a^{-1})	% of total emissions	PWC (ng m^{-3})	% of total PWC	Exposure/emission ratio (ng m^{-3} (Mg $a^{-1})^{-1}$ $\times$ 1000)
Primary wood heating					
Non-residential	1	<0.1	<0.1	<0.1	8.3
Rural	1150	47	17	19	15
Small villages	130	5	2.3	3	17
Villages	540	22	13	14	24
Other urban	20	1	1.6	2	86
Scattered detached	273	11	16	17	57
Detached	310	13	41	44	130
Block buildings	9	0.4	2.1	2	220
Supplementary wood heating					
Non-residential	3	<0.1	<0.1	<0.1	16
Rural	1730	30	35	5	20
Small villages	200	4	4.8	0.7	24
Villages	920	16	30	4	33
Other urban	70	1	13	2	190
Scattered detached	620	11	59	8	96
Detached	2150	37	540	75	250
Block buildings	100	2	40	6	400

The $PM_{2.5}$ emissions from primary wood heating were highest in *rural* and *villages* residential area classes, in general more scarcely populated areas, and slightly lower in more densely populated areas. The emissions from *rural* areas were 1150 Mg a^{-1} which comprises 47% of the total emissions from primary wood heating. For *villages* and *small villages* the emissions and shares were 540 and 130 Mg a^{-1}, and 22 and 5%, respectively. *Scattered detached* and *detached* area classes had emissions of 273 and 310 Mg a^{-1}, with shares of 11 and 13%, respectively. Each of the *other urban, block building* and *non-residential* areas had less than 1% of the total emissions.

The distribution of the $PM_{2.5}$ PWCs caused by primary wood heating differ from the emissions. The areas with higher population densities increase their significance. The PWCs for *rural* areas, which had the highest emissions caused by primary wood heating, was 17 ng m^{-3}, which was 19% of the total PWC. *Villages* and *small villages* had smaller shares of the total PWC than the emissions, with PWCs of 13 and 2.3 ng m^{-3} and shares of 14 and 3%, respectively. *Scattered detached* and *detached* areas increased their shares to 17 and 44% with PWCs of 16 and 41 ng m^{-3}, respectively. *Other urban* and *block building* areas had PWCs and shares of 1.6 and 2.1 ng m^{-3}, and 2 and 2 %, respectively. The PWC of *non-residential* areas was negligible.

For supplementary wood heating, *rural* and *detached residential* classes dominate the $PM_{2.5}$ emissions. *Detached areas* had emissions of 2150 Mg a^{-1}, which was 37% of the total emissions from supplementary wood heating. *Rural* areas were the second biggest contributor, with emissions of 1730 Mg a^{-1} and a share of 30%. *Villages* and *small villages* had emissions and shares of 920 and 200 Mg a^{-1}, and 16 and 4%, respectively. For *scattered detached* areas, the emissions were 620 Mg a^{-1} and the cut from the total 11%. *Other urban* and *block building* areas had emissions that were less than 100 Mg a^{-1} and they comprised less than 2% of the total emissions. *Non-residential* areas had negligible emissions.

The exposures were concentrated into areas with dense population. Namely *detached* residential areas, for which the PWC was 540 ng m^{-3}. That was 75% of the total PWC from supplementary heating. All the other classes caused less than ten percent each of the total exposure. *Block buildings* and *scattered detached* had PWCs and shares of 40 and 59 ng m^{-3}, and 6 and 8%, respectively. *Other urban* areas had 2% of the total PWC with 13 ng m^{-3}. *Villages* and *small villages* comprised 4 and 0.7% of the total PWC, with 30 and 4.8 ng m^{-3}. *Rural* areas caused a PWC of 35 ng m^{-3}, which meant a share of 5%. Same as with emissions, the PWC of the *non-residential* areas was negligible.

The *rural, detached* and *villages* area classes were the biggest emission sources for both primary and supplementary wood heating. In both cases they made up over 80% of the emissions. Only *rural* areas for primary heating had emissions higher than the four biggest contributor classes in supplementary heating. Each area class had higher emissions from supplementary heating, the values being from 1.5- to 11-fold.

For PWC, *detached* areas were the biggest contributors for both heating types. It was also the second most densely populated residential area class, which partly explained the highest PWC shares. *Rural* and *villages* (i.e. scarcely populated) classes had higher shares in primary heating than supplementary. Even though *detached* areas were the biggest contributor to the PWC from primary heating, the actual PWC was smaller than the PWC from supplementary heating *scattered detached* areas, emphasising the higher exposures caused by supplementary heating. As with emissions, each area class had higher PWC from supplementary than primary wood heating. *Detached* areas seemed to be the key source for $PM_{2.5}$ from population exposure point of view. Furthermore, supplementary wood heating in these areas had the biggest impact on population exposure. These factors made supplementary wood heating in *detached* areas important target for emissions abatement measures when the reduction of the health impacts of fine particles are considered.

The exposure/emission ratio tells how big of an exposure a unit emission causes. The exposure/emission ratios for different residential area classes for primary and supplementary wood heating are presented in figure 4.2. In general, the ratios were bigger in areas with higher population densities. The ratios for non-urban areas (*non-residential* - *villages*) were close to each other, and were much smaller than ratios for urban (*other urban* - *block buildings*) areas. The emissions in non-urban areas exposed only few people due to scattered population. The highest ratios were (in order from the smallest to the biggest) for *other urban*, *detached* and *block buildings.* The emissions from *block buildings* areas were relatively low, but the exposure/emission ratio was high due to high population density in those areas. The *detached* area had both high emissions and high exposure/emission ratio, highlighting it's importance from population exposure point of view. This further indicated that emission control would be more important in higher population density areas.

For primary wood heating, the exposure/emission ratios were lower in all of the area classes compared to supplementary heating. In urban areas, the ratios for supplementary heating were almost double compared to primary heating. Therefore, a unit emission caused almost double the population exposure from supplementary heating. This indicated that most emission from primary heating came from low population density areas, and even the emissions of primary wood heating that occurred in urban environment seemed to occur in areas with lower population density compared to supplementary heating. This supports the view that the emissions from supplementary wood heating seemed to be more important from exposure point of view.

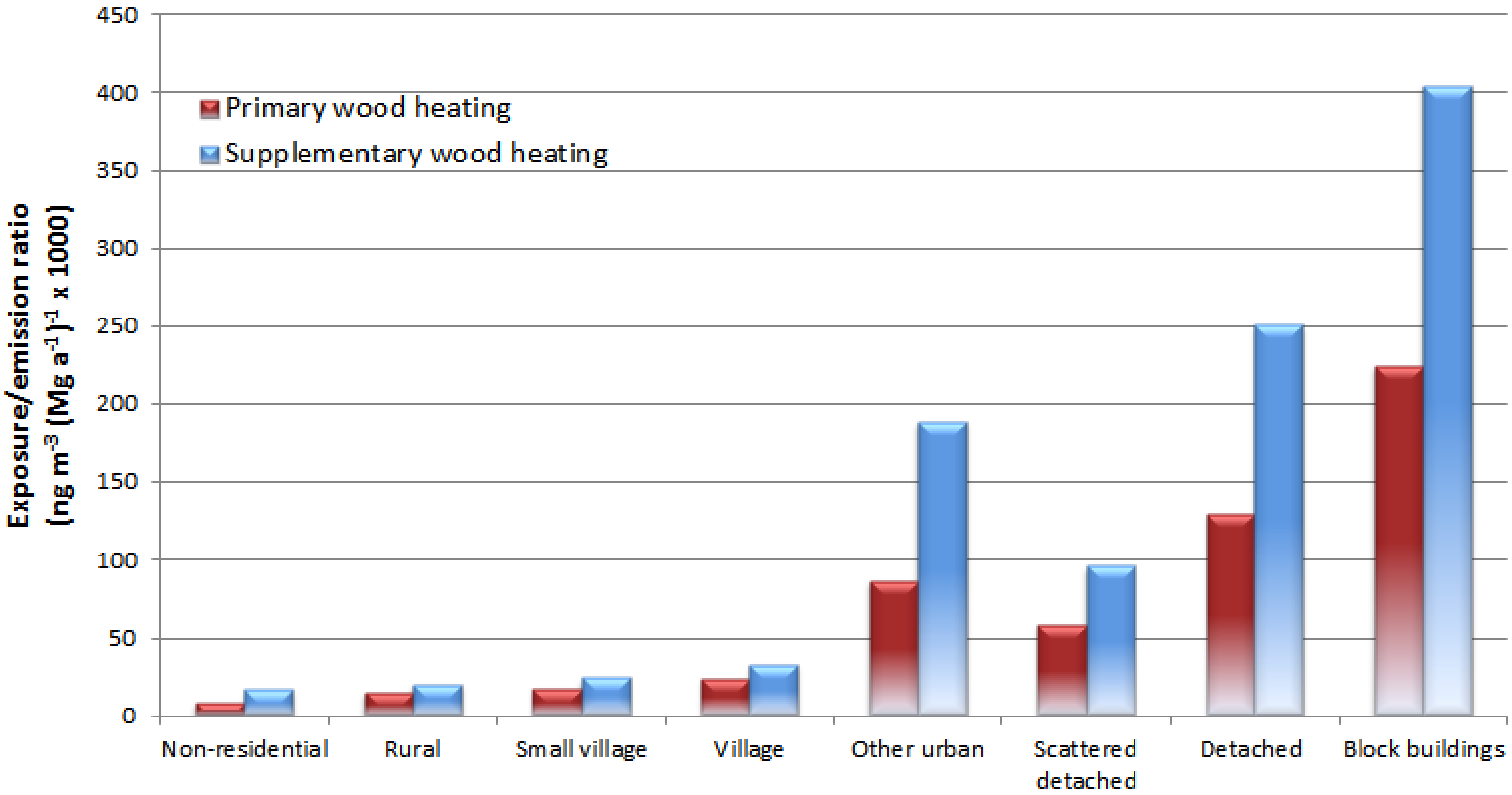


Figure 4.2: $PM_{2.5}$ exposure/emission ratios for primary and supplementary wood heating in the different residential classes.

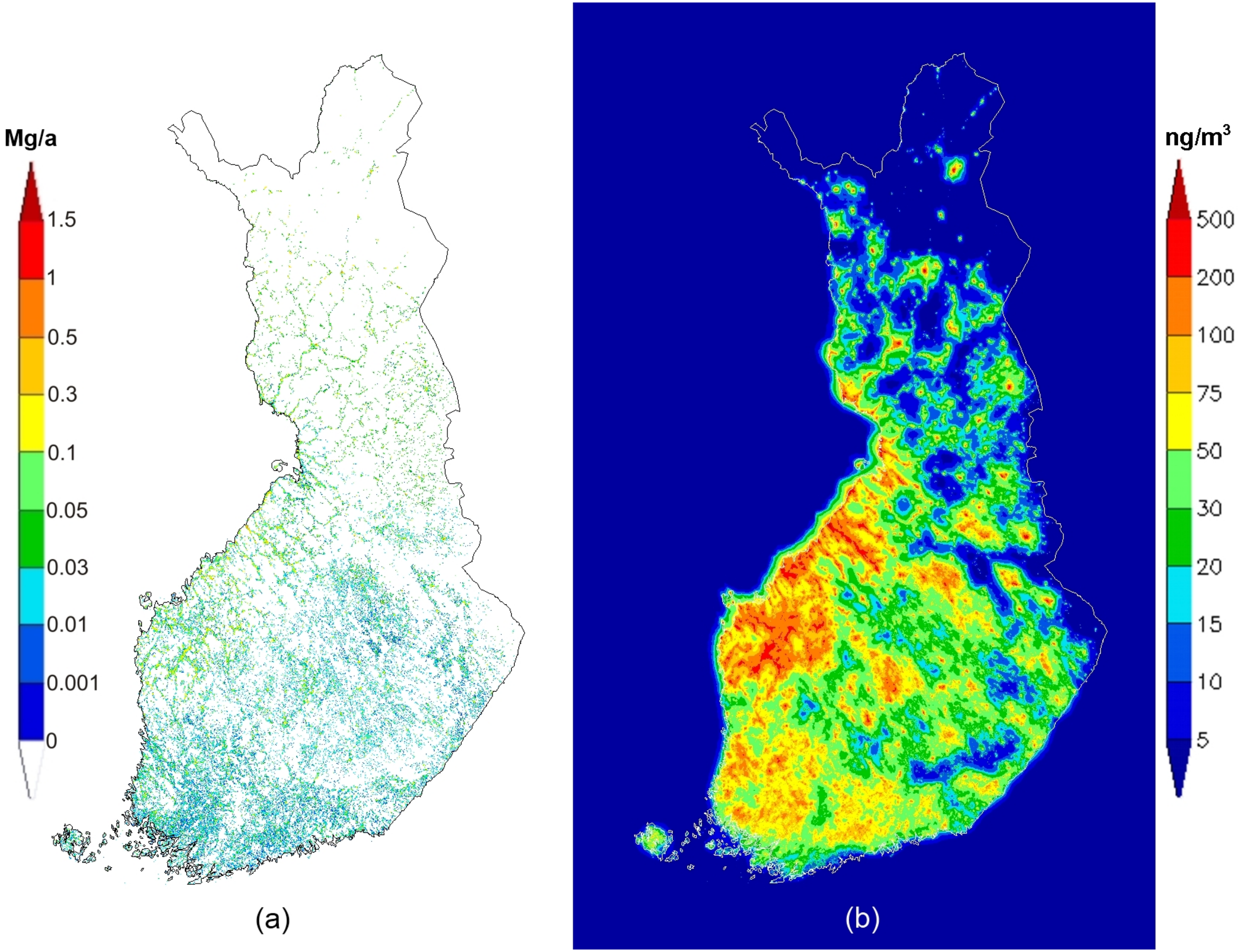


Figure 4.3: $PM_{2.5}$ (a) emissions and (b) concentration caused by primary wood heating.

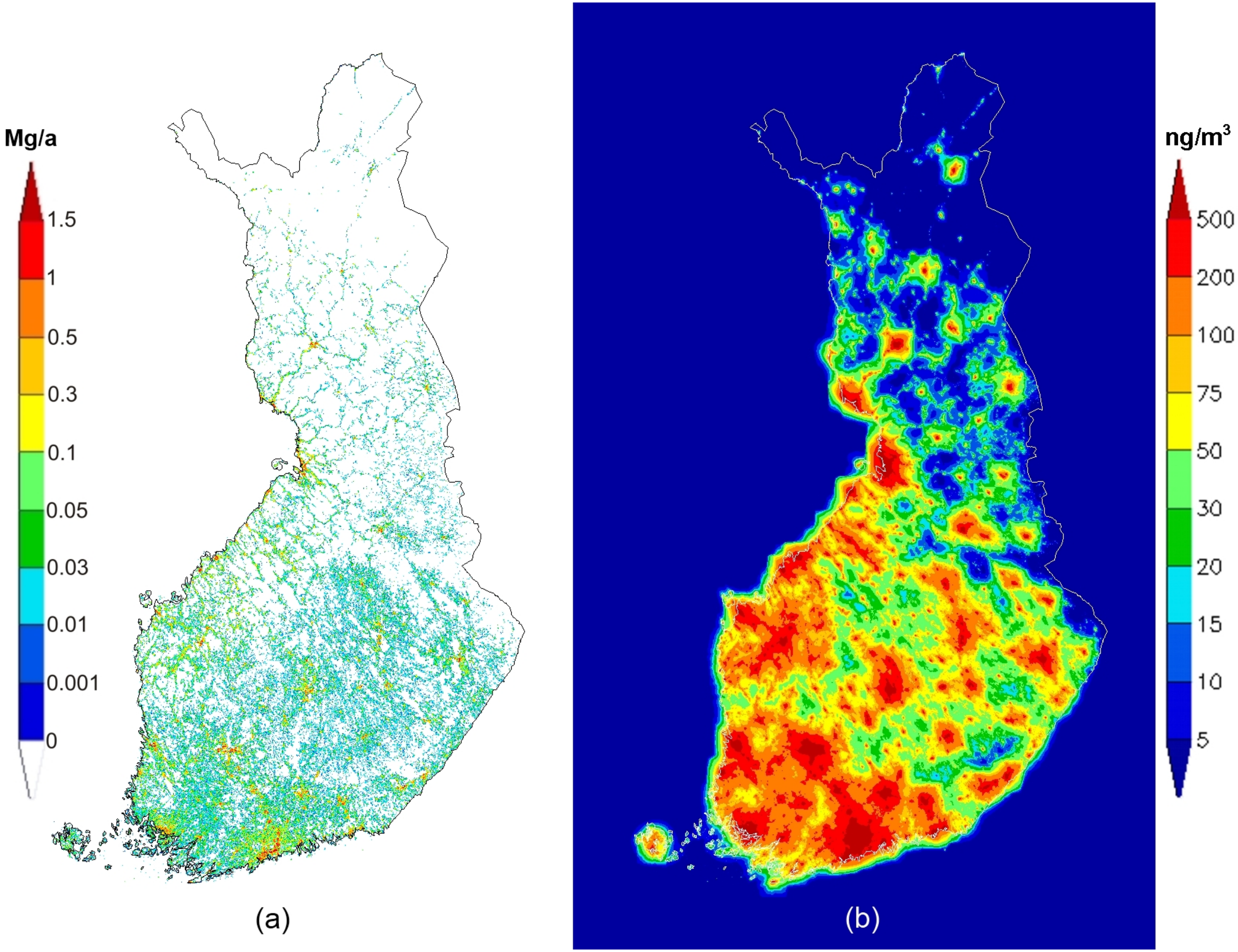


Figure 4.4: $PM_{2.5}$ (a) emissions and (b) concentration caused by supplementary wood heating.

Chapter 5

Discussion

5.1 Emissions and Spatial Distribution

The FRES model emission factors for residential wood combustion are chosen to represent real life emissions. According to Tissari (2008), the normal combustion PM_1 emission factors for modern and conventional masonry heaters are 42 and 38-198 mg MJ^{-1}, respectively. The FRES model emission factors are 80 and 120, respectively. The FRES emission factors are supposed to take into account poor combustion, which causes higher emissions. Therefore, the factors are higher than the smallest measurement values. It is hard to assess how much of the combustion is poor. Furthermore, only one emission factors is used per appliance. Therefore, the factor has to represent the average emissions as accurately as possible. The emission factors of the FRES model seem to fall between the extremes of the measurement results. In order to further developed the values used in the model, for example combustion practice surveys should be implemented. Since PM_1 comprises 90% of the PM emissions from residential wood heating, the PM_1 and $PM_{2.5}$ emission factors can be compared to each other. More inaccuracy comes from the assessment of combustion practices and their effect on the factors than from the emissions of particles that fall in between 1 and 2.5 μm.

The total $PM_{2.5}$ emissions of Finland in 2005 were 32 000 Mg a^{-1} (Hildén et al., 2008). Thus, the emissions from residential wood combustion, 8230 Mg a^{-1}, were 26% of the total emissions. It is notable, that not only does RWC have a big share of the total emissions, but the emissions also occur within residential areas close to population, which makes them important to population exposure. Furthermore, the pollutants are emitted from low altitude, so the particles are less diluted than from high stacks, i.e. from power plants.

The share of $PM_{2.5}$ emissions from primary wood heating is higher than the share of houses that are primarily heated with wood. There are approximately 90 000 houses heated primary with wood combustion, and little under a million detached houses altogether. Where primary wood heating is used in about 9% of all the detached houses, the emissions were about 30% of the total emissions. This reflects the fact that the primary heating devices use annually more wood per device. The emission distribution between primary and supplementary heating followed the distribution of wood use, as 32% of the wood fuel is used in primary heating.

The spatial distributions of the emissions from the two cases had a clear difference. Primary wood heating was rare in big cities (especially in Helsinki Metropolitan Area), and more common outside urban areas. West Finland stood out with the highest emissions. Supplementary heating, in contrast, had the biggest concentrations in the most populated cities. Therefore, the lower urban wood use per appliance is not enough to compensate the higher device numbers compared to non-urban areas. In other words, the prevalence of supplementary wood heating appliances in urban areas cause higher $PM_{2.5}$ concentrations compared to non-urban areas, even though their wood use per device is smaller.

A high proportion, 74%, of the primary wood heating emissions occurred in non-urban areas (*non-residential - villages*). In contrast, 51% of the emissions from supplementary heating were emitted in urban areas (*other urban - block buildings*). Non-urban areas use more wood per residence, but the sheer number of the houses is much bigger in urban areas. Primary wood heating is much more common in non-urban areas than in urban (Torvelainen, 2009). From population exposure point of view, this can be thought to be good. In these areas, less people are exposed to the emissions, which, especially in the cold months, are continuous.

5.2 Population Exposure

The $PM_{2.5}$ population exposures differed significantly between primary and supplementary wood heating, in contrast to the emissions. The PWC for supplementary heating was 7-fold compared to primary heating. The main reason for this was that more of the emissions of supplementary heating occurred in more densely populated urban areas. Thus, more people were exposed to the pollution. This highlights the importance of spatial modelling of the emissions.

The effect of $PM_{2.5}$ exposure to mortality can be estimated with the PWC, exposure-response functions (ERF) and background incidences. Karvosenoja et al. (2012) assumed the mean ERF for mortality to be 0.62% change in non-accidental mortality per 1 μg m^{-3} change in $PM_{2.5}$ concentration. Background non-accidental mortality for the year 2005 was obtained from Statistics Finland (Official Statistics of Finland (OSF), 2010). Calculating with these assumptions, the total PWC from RWC (815 ng m^{-3}) caused 200 premature deaths in 2005.

The CAFE estimate for premature deaths caused by $PM_{2.5}$ in Finland in the year 2000 is 1300. Therefore, the RWC would comprise the order of magnitude 15% of this estimate. The CAFE estimation is mainly from exposure to long-range transported pollution, as it takes into account the local sources poorly. Long-range transport pollution cannot be controlled by local measures. Considering other significant particle emission sources, the PILTTI project estimated, that traffic caused about 800 premature deaths in Finland in 2000. Traffic emissions are already controlled by legislation, and fine particle emissions from traffic exhaust are decreasing. Large plants in Finland have small impact on the population exposure (Karvosenoja et al., 2010), because the particles are emitted in high altitude (because of high stacks), and the emissions are relatively low due to flue gas cleaning.

In total, it can be estimated that RWC causes a significant part of the total premature deaths from fine particles, with a share of at least 10%. RWC emissions are not yet controlled by legislation in Finland, and it's activity is increasing in the future. Therefore, it's share and significance is increasing. In contrast to traffic and long-range transported particles, RWC emissions have clear potential for abatement. There are undeveloped devices, and incomplete combustion causes significant emissions.

Urban areas (*other urban - block buildings*) withheld 25% of the $PM_{2.5}$ emissions of the primary wood heating, but 65% of the exposure. For supplementary wood heating the figures were 51% and 91%, respectively. From the total emissions and exposure from RWC, the supplementary heating in urban areas comprised 36% and 80%, respectively. All these indicate the fact that supplementary stove heating in densely populated areas should be in the focus when emission impacts and reductions are considered.

The exposure/emission ratios reflect the fact that the emissions in areas with higher population density cause bigger population exposures. Furthermore, they indicate that supplementary wood heating is more significant than primary heating for the population exposure. An emission of the same amount causes higher population exposure from supplementary heating than primary. This highlights the importance to concentrate the emission abatement measures to supplementary heating, i.e. masonry heaters and stoves.

5.3 Emission Reduction

Since the population exposure to $PM_{2.5}$ from supplementary wood heating is 7-fold compared to primary heating, it would be more efficient to target the reduction measures to supplementary heating. Since the renewing rate of masonry heaters is so slow, most feasible option for the reduction would be the enhancement of the operation practices and flue gas cleaning. As flue gas cleaning technologies are not yet widely commercially available, information campaigns and legislation measures are the the most promising ways to affect the emissions in the short term.

Since urban areas cause most of the population exposure to $PM_{2.5}$, the information campaigns and legislation measures would be most efficient when targeted there. The measures could take into account the population densities of the residential areas. Areas with higher density could have stricter limits or have more emphasis in education.

Masonry heaters are the most common supplementary wood heating devices alongside with sauna stoves. Masonry heaters are typically made out of masonry products, so once they are build, they tend to stay put. Therefore, new appliances with lower emissions would affect the country-wide emissions slowly. Improvement of operational practices would be the most viable reduction method. Especially fuel quality should receive attention. Burning wet wood or waste in the device produces high emissions. Sauna stoves are possibly renewed quicker, as they are easier to replace. For them, new appliances with set emission limits would be efficient way to cut down emissions. Operational practices are important for sauna stoves as well.

Boilers are typically used with continuous combustion. This means that the user affects the combustion conditions less than in batch combustion. However, especially fuel quality (for example moisture content) has significant effect on the emissions, and poor quality can cause multifold emissions. Therefore, education for boiler users could be important emission reduction method. Technology wise, an important issue is the use of heat accumulator tanks. Boilers without heat accumulator might be used with smouldering combustion, which causes high emissions. Equipping all boilers with heat accumulators would be a cost-effective emission reduction method. Flue gas cleaning technologies, which are an option in the future, work better with continuous combustion devices, and offer notable emission reduction potential.

Boilers or stoves in Finland don't yet have binding emissions limits. However, the European Union's Ecodesign Directive would affect Finland also. The Directive is going to set emission limits to small scale wood combustion devices. The effect of the Directive to the emissions depends on the renewal rate of the devices.

Residential wood combustion has much emission reduction potential compared to other important particle emission sources. Traffic emissions are already controlled by legislation measures, and the traffic exhaust emissions are decreasing. The emissions of big plants in Finland are controlled by legislation, and they have small impact on the population exposure to particles. Long-range transported particles cause significant population exposure to particles, but it cannot be controlled by local or national measures. Therefore, residential wood combustion should receive the attention of decision-makers in order to decrease the adverse health impacts of fine particles.

Chapter 6

Conclusions

The goal of this thesis was to identify the characteristics of Finnish residential wood combustion (RWC), study the $PM_{2.5}$ emissions and the population exposure they cause from different residential area types, and assess the emission reduction options for the future. RWC was found to be an important source for $PM_{2.5}$ emissions, and it caused significant population exposure to $PM_{2.5}$.

The total $PM_{2.5}$ emissions from residential wood heating was 8230 Mg a^{-1}, which amounted to 26% of the total emissions in Finland in 2005. Supplementary wood heating, i.e. stoves and masonry heaters, caused 70% of these.

Emissions were studied in different residential area classes in order to identify the most important source areas. Non-urban areas, with a share of 74%, were more important emission source than urban for primary wood heating, i.e. boilers. For supplementary wood heating, the urban areas made up 51% of the emissions.

The picture of population exposure to $PM_{2.5}$ was quite different from the emissions. Supplementary heating caused 89% of the total $PM_{2.5}$ exposure from RWC, with 80% of the total exposure coming from urban areas. In total, RWC was estimated to have caused around 200 premature deaths in 2005. From population exposure point of view, supplementary wood heating with stoves and masonry heaters is much more significant than primary heating with boilers.

Since the RWC is increasing in the future, the reduction of the emissions is important for the public health. The reduction measures should be targeted at supplementary wood heating, as it comprises the majority of the population exposure. Supplementary wood heating has two main combustion appliance types: masonry heaters and sauna stoves. For sauna stoves, the renewal of the devices may offer emission reductions in the near future.With sauna stoves, consumers should be directed to invest in new, low emission technology. New stoves should also have legally binding emission limits. The renewal of masonry heaters offers emission reduction potential as well. However, their renewal rate is slow, so the short term reductions might be small.

A viable way to reduce the emissions in the sort term could be to affect the operational practices. The emissions reduction potential seems to be remarkable, as smouldering combustion causes significant increase in the emissions. A potential way to affect the practices is through an information campaign. Although the influence of such a campaign is hard to assess, it is a cost-effective option even with small effects compared to other emission reduction measures.

In the future, more studies are needed about how users operate their RWC appliances. It would be crucial for the emission and health assessment, as the emission factors and emission reduction potential could be estimated with higher accuracy. Furthermore, residential wood combustion should be studied more with combined health and climate effects assessment. The increase of wood combustion is promoted as a measure to battle the climate change. However, this is likely to increase the adverse health effects. A cost-benefit analysis between different scenario options would shed light to the real benefits of wood combustion promotion.

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