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Spatial distribution of residential wood combustion emissions in the Nordic countries: How well national inventories represent local emissions?

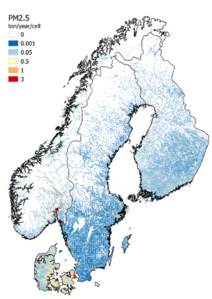
Ville-Veikko Paunu [a,*], Niko Karvosenoja [a], David Segersson [b], Susana López-Aparicio [c], Ole-Kenneth Nielsen [d], Marlene Schmidt Plejdrup [d], Throstur Thorsteinsson [e], Jarkko V. Niemi [f], Dam Thanh Vo [c], Hugo A.C. Denier van der Gon [g], Jørgen Brandt [d,h], Camilla Geels [d]

[a] Finnish Environment Institute SYKE, Latokartononkaari 11, FI-00790 Helsinki, Finland
[b] Swedish Meteorological and Hydrological Institute, SMHI, Folkborgsvägen 1, SE-601 76 Norrköping, Sweden
[c] NILU – Norwegian Institute for Air Research, Instituttveien 18, NO-2007 Kjeller, Norway
[d] Department of Environmental Science, Aarhus University, Frederiksborgvej 399, P.O.Box 358, 4000 Roskilde, Denmark
[e] Environment and Natural Resources & Institute of Earth Sciences, University of Iceland, Sturlugata 7, 101 Reykjavik, Iceland
[f] Helsinki Region Environmental Services Authority (HSY), P.O. Box 100, FI-00066 HSY, Finland
[g] Department Climate, Air and Sustainability, TNO, Utrecht, the Netherlands
[h] IClimate – Aarhus University Interdisciplinary Centre for Climate Change, Frederiksborgvej 399, P.O.Box 358, 4000 Roskilde, Denmark

HIGHLIGHTS

- New high-resolution emission inventory for Nordic residential wood combustion.
- Country level methods can produce similar spatial distributions as local level.
- Difference between urban and rural RWC is important for the spatial distribution.
- National characteristics are essential for spatial representation of RWC emissions.

GRAPHICAL ABSTRACT



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ABSTRACT

Residential wood combustion (RWC) is a major source of air pollutants in the Nordic and many other countries. The emissions of the pollutants have been estimated with inventories on several scopes, e.g. local and national. An important aspect of the inventories is the spatial distribution of the emissions, as it has an effect on health impact assessments. In this study, we present a novel residential wood combustion emission inventory for the Nordic countries based on national inventories and new gridding of the emissions. We compare the emissions of the Nordic inventory, and especially their spatial distribution, to local assessments and European level TNO-newRWC-inventory to assess the spatial proxies used. Common proxies used in the national inventories in the Nordic countries were building data on locations and primary heating methods and questionnaire-based wood

* Corresponding author.
E-mail addresses: ville-veikko.paunu@syke.fi, ville-veikko.paunu@ymparisto.fi, ville-veikko.paunu@ymparisto.fi (V.-V. Paunu), niko.karvosenoja@syke.fi (N. Karvosenoja), david.segersson@smhi.se (D. Segersson), sla@nilu.no (S. López-Aparicio), okn@envs.au.dk (O.-K. Nielsen), msp@envs.au.dk (M.S. Plejdrup), thorstur@hi.is (T. Thorsteinsson), jarkko.niemi@hsy.fi (J.V. Niemi), vtd@nilu.no (D.T. Vo), hugo.deniervandergon@tno.nl (H.A.C. Denier van der Gon), jbr@envs.au.dk (J. Brandt), cag@envs.au.dk (C. Geels).

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use estimates for appliances or primary heating methods. Chimney sweeper register data was identified as good proxy data, but such data may not be available in an applicable format. Comparisons of national inventories to local assessments showed the possibility to achieve similar spatial distributions through nation-wide methods as local ones. However, this won't guarantee that the emissions are similar. Comparison to the TNO-newRWC-inventory revealed the importance of how differences between urban and rural residential wood combustion are handled. The comparison also highlighted the importance of local characteristics of residential wood combustion in the spatial distribution of emissions.

1. Introduction

Residential wood combustion (RWC) is common in Europe (Cincinelli et al., 2019), Asia (Huy et al., 2021; Zhang et al., 2021) and elsewhere (Winijkul et al., 2016). RWC in stoves, small boilers and fireplaces is widely used for heating and for creating a cosy atmosphere in residences in the continental Nordic countries: Denmark, Finland, Norway and Sweden. Due to often incomplete combustion conditions and lack of emission control devices, RWC is also an important emission source of air pollution. Main pollutants emitted include fine particles ($PM_{2.5}$), which contain substances that are known to be linked to adverse health effects (Sigsgaard et al., 2015) and climate impacts (AMAP, 2015). Residential combustion comprised about half of the total anthropogenic $PM_{2.5}$ emissions in the Nordic countries in 2014 according to the Convention on Long-range Transboundary Air Pollution (CLRTAP) national submissions,[1] and in many cities they have a strong impact on local air quality (Kukkonen et al., 2020).

Air quality modelling is an important tool to assess the impacts of air pollution on human health. Spatially resolved emission inventories are a crucial input to these air quality models. Therefore, correct spatial representation of the emissions is vital in order to limit the uncertainty related to health impact assessments. This is especially highlighted in assessments with high spatial resolution (Segersson et al., 2017). The spatial resolution itself also affects the results of impact assessment as shown in (Karvosenoja et al., 2011; Korhonen et al., 2019; Lehtomäki et al., 2020; Zheng et al., 2017). These studies show that lower spatial resolution often leads to lower air pollution concentration and exposure estimates, since the pollutants are averaged over larger grid cells. An earlier study has shown that non-industrial combustion sources represents about 10% of the overall external costs related to health impacts caused by air pollution in Europe (Brandt et al., 2013).

RWC is an example of a diffuse source sector, which is often not practical to be described as individual point sources with exact locations for each dwelling. The number of residences with wood combustion appliances is usually large, and the locations of appliances are not necessarily known. Thus, proxy data representing the spatial distribution of emission sources is used to distribute the emissions (from, e.g., national or municipal level) into a grid (with e.g. 1 km × 1 km spatial resolution). The type of proxy data that should be used depends on the characteristics of RWC in the country in question, and availability and quality of data related to wood consumption and stove technology.

EEA guidebook 2019 (European Environment Agency, 2019) recommends proxies for residential combustion. Tier 1 (the simplest approach with highest uncertainty) recommendation is to use land cover, without specification of the land cover type to be used. Tier 2 is to use population or household density combined with land cover data. Tier 3 (the most complex with lowest uncertainty level) involves use of detailed fuel deliveries and modelled estimates using population density and/or household numbers and types. In practice, population is often used, in combination with other data, e.g. land cover, as a part of the spatial proxy for RWC (Geng et al., 2017; Li et al., 2017; Timmermans et al., 2013; Trombetti et al., 2018).

One option for proxy development is to utilize surveys. The surveys help to gather information not available in official statistics, such as the total amount of wood burned, the prevalence and types of combustion appliances, and location of the appliances (Andersen, 2015; Kaski et al., 2016; Omstedt et al., 2014). The surveys can be carried out on broad, i.e. national (Swedish Energy Agency, 2015; The Norwegian Environment Agency, 2018), or local level (Pastorello et al., 2011; Swab et al., 2019). Often surveys help to supplement other data used for assessments, and to evaluate the uncertainties in those data.

Clearly, the selection of proxies is critical for the setup of reliable emission inventories. There have been studies that have analysed the impact of proxies on the quality of spatial emission inventories. These studies have mainly compared the emissions from several national level inventories within large city areas (e.g. Timmermans et al., 2013; Trombetti et al., 2018; Vedrenne et al., 2016), compared downscale regional emissions at city level with local inventories (López-Aparicio et al., 2017a) or analysed emission distributions (Geng et al., 2017). However, to the authors' knowledge there have been only few studies that systematically analyse the quality of RWC proxies at grid level (Ferreira et al., 2013; Zheng et al., 2017).

This paper describes developed spatial proxies for RWC in the Nordic countries and presents a common gridded Nordic RWC emission inventory for the first time. The methods have been compared and harmonized between the countries. Before, the inventories were on different stages of proxy development. Finland used floor area of detached houses as the main proxy (Karvosenoja et al., 2011). Sweden used similar proxies, but with reduced wood combustion emissions in cities, taking into account the large share of households connected to district heating (Andersson et al., 2019). Denmark based gridding on a regional inventory of energy consumption for heating for oil boilers, natural gas boilers and solid fuel installations developed by the Danish Energy Agency (Plejdrup and Gyldenkærne, 2011). Norway was lacking gridded emissions altogether.

In addition to national inventories, different spatial distribution methods for RWC emissions can be found in many studies with different spatial scales. A good example of an inventory covering a larger scale is the European TNO-MACC III air pollution emission inventory (Kuenen et al., 2014). A special version of this inventory (TNO-newRWC) was made by Denier van der Gon et al. (2015) after an evaluation of official reported residential combustion PM emissions. Denier van der Gon et al. (2015) collected wood use data by country and used an appliance type split from the GAINS model (Amann et al., 2011) to recalculate the residential combustion emissions. Spatial distribution of the RWC emissions in this inventory is based on three factors: population density, urban/rural differentiation and access to wood fuel. This method creates a distribution that is significantly different from a distribution based only on the population density.

The emissions can be distributed to grid level using more than one step, with each step using its own proxies. Vedrenne et al. (2016) compared a national and a local level emission inventory in the Autonomous Community of Madrid. For non-industrial combustion, the inventories gave very different emissions (for example, local inventory had 302% higher NO_x and 83% lower $PM_{2.5}$ emissions than the national). The main reason was different fuel mix in the inventories, national having considerable share of biomass as opposed to the local

[1] https://www.eea.europa.eu/data-and-maps/data/national-emissions-reported-to-the-convention-on-long-range-transboundary-air-pollution-lrtap-convention-13, accessed 26.5.2020.

inventory, which had none. This was identified to be partly because of the spatial disaggregation proxies used for allocating national fuel consumption statistics to the province level. In contrast, local inventory used regional fuel use data. This highlights the importance of the quality of not only the grid proxies but coarser level proxies, e.g. country to municipality proxies, as well.

In a comparison between European inventories, significant differences were found for the RWC emissions (Trombetti et al., 2018). In the inventories included in the comparison, population density was a part of the proxy in every case. Other data commonly used was level of urbanization. Trombetti et al. concluded that population density alone is not a relevant proxy as national characteristics and city specific features (for example district heating zones) might show that prevalence of RWC is not directly proportional to population density. López-Aparicio et al. (2017) takes this observation further by stating that assumptions on spatial distribution of emissions between urban and rural areas based on one country does not necessarily hold in another. Example in the article is the discrepancy between France, where wood combustion occurs mainly in rural areas, and Norway, where combustion is common also in urban areas. Therefore, local knowledge is critical in spatial distribution of the emissions.

Timmermans et al. (2013) compared downscaled emissions from the TNO-MACC version I inventory and local emission inventory for the province Ile-de-France in Paris. They concluded that the downscaled emission inventory allocated significantly higher emissions to urban areas, and one main cause for this was the usage of population density as proxy. Comparison to measurements confirmed that the local bottom-up inventory had better agreement than the downscaled emissions. The study highlights the importance of the quality of spatial proxies especially in urban areas and was a motivation for an improved spatial distribution approach in TNO-MACC-III and TNO-newRWC.

There are generally two ways to achieve spatially distributed air pollution emissions for diffuse sources: top-down and bottom-up. Despite the regular usage of the terms there are varying definitions for them in the scientific literature. Using top-down methodology, total emissions are initially calculated for a larger area based on, for example, national statistics on fuel use. The total emissions are then distributed spatially, either directly or through an intermediate step of allocation to sub-regions, using proxies. Bottom-up can refer to a methodology where the emissions are calculated on the individual source level, e.g. per appliance (Karvosenoja, 2008), and then summed up to grid level. Sometimes the term bottom-up is used when emissions are calculated in the same manner as in top-down methodology, but using regional data for municipalities or counties (Zheng et al., 2017). Since the definitions are not always clear and are sometimes overlapping, they are not used when discussing the emission datasets used in this paper.

In this paper we present and assess new advanced gridding methods for national RWC emission inventories in four Nordic countries: Denmark, Finland, Norway and Sweden. Each country has developed their own methods and gridding proxies depending on their national characteristics in the RWC sector and the availability of relevant data. These national characteristics and how well they are taken into account in the gridding methods are discussed. The gridded $PM_{2.5}$ emissions are thereafter compared to available local level RWC emission studies in selected cities or areas to evaluate the implications of the different gridding methods. The gridded emissions are also compared to a European emission inventory to study how taking national characteristics into account affect the modelled spatial distribution of the emissions. The research questions for the paper are:

(1) How well we represent RWC emissions on a national scale in comparison to how they are represented in local assessments?
(2) How taking national characteristics into account enhance the spatial representation of emissions compared to a European emission assessment?

Finally, based on our analysis, we give general recommendations for gridding RWC emissions.

2. Methods

The national RWC emissions and their spatial distributions were created separately for the four countries, Denmark, Finland, Norway and Sweden, including country specific spatial proxies. The inventory covers years 1990–2010 on five year intervals and 2012 and 2014. The emissions were gridded in 1 km × 1 km resolution in European grid ETRS89-LAEA (EPSG: 3035). Iceland was omitted from this study as RWC is uncommon there. Only 0.2% of space heating was by other than geothermal or electricity in Iceland in 2018 (Orkustofnun, 2019). The combined Nordic inventory was compared against the European level TNO-newRWC inventory (Denier van der Gon et al., 2015). The comparison of the Nordic inventory to local RWC emission inventories consisted of four cases on four different spatial extents: Copenhagen Municipality, Oslo Metropolitan Area, Helsinki Metropolitan Area and Västerbotten County.

The four countries differ in both which kind of buildings RWC is practiced and what kind of appliances are used. Most common appliances and their emission factors are presented in Table 1. In Finland, RWC is mostly done in detached and recreational houses and masonry heaters and sauna stoves are common. One third of the wood is used in boilers, manually fed boiler with a heat accumulator tank being most common boiler type. In Denmark, RWC is mainly used for cosiness ("hygge") as supplementary heating. The appliances are dominated by woodstoves, but there are also a significant number of wood pellet boilers. The appliances are most widespread in detached houses, and case studies have shown that woodstoves in apartments have lower unit consumption than in detached houses and holiday houses (Andersen, 2015). In Norway, iron stoves and open fireplaces are the main RWC installations used for residential heating and they are used in both apartments and houses, including detached houses, town houses and duplexes. A citizen participatory approach was carried out in two areas

Table 1
Common appliances and their emission factors in the Nordic countries, and emission factors from the TNO-newRWC inventory.

	$PM_{2.5}$ emission factors [mg/MJ]	Reference
Finland		Savolahti et al. (2019)
Boiler, automatic/manual	16–135	
Masonry heater, modern/conventional	48–137	
Sauna stove	470	
Denmark		Nielsen et al. (2018)
Iron stove, pre 2005	740–930	
Advanced iron stove, 2008 forward	514	
Eco-labelled iron stove	206	
Boiler, pellet	29	
Sweden		Kindbom et al. (2017)
Boiler, pellet or chip	40–59	
Boiler, wood log	36–376	
Iron stove, modern/traditional	92–190	
Norway		Seljeskog et al. (2013)
Old iron stove, pre 1998	980	
New iron stove, 1998 forward	690	
Open fireplaces	980	
TNO-newRWC		Denier van der Gon et al. (2015)
Fireplace	900	
Traditional heating iron stove	800	
Boiler, automatic (pellet or chip)	60	
Boiler, manual	1000	

in Norway: one in a highly populated urban (Oslo) and another in an urban-rural combined area (Akershus county), to increase the understanding of RWC for heating in Norwegian urban areas (López-Aparicio et al., 2017b). This study showed that wood consumption was higher in the urban-rural combined area. However, the study also showed the importance of RWC in Norwegian apartments, as 46% of the wood in Oslo was consumed in this type of dwellings. In Sweden, solid fuel boilers are common in detached housing, especially on the countryside or smaller cities without district heating. In estimates for 2014, 64% of the firewood was used in boilers (almost equal shares from conventional boilers and modern boilers). The remaining third of the firewood was used mainly in stoves. There is an increasing trend in the number of stoves, while the number of boilers is slowly decreasing (Helbig et al., 2018).

2.1. National RWC spatial distributions

For Sweden, Norway and Denmark the emissions from the Nordic inventory were from SNAP sector 0202, i.e. residential combustion, which contains also other fuels than wood (e.g. oil or gas). However, wood is the most common fuel, and causes majority (90–95%) of the $PM_{2.5}$ emissions from the sector. Therefore, comparability of the spatial distribution of the national emissions from SNAP 0202 sector against local estimate for the RWC sector is adequate. From here onwards, the term RWC is also used to refer to SNAP sector 0202. In the following, first the methods used for setting up the national inventories will be described for each country, thereafter the local assessments are presented.

2.1.1. Finland

Finnish RWC emissions were calculated with the method applied in the Finnish Regional Emission Scenario (FRES) model (Karvosenoja, 2008). In the model, total national RWC emission per combustion appliance type, i.e. different types of stoves, boilers and fireplaces, are first calculated (presented in Savolahti et al., 2016). Then the emissions are gridded separately for boilers and stoves used in residential buildings (Paunu et al., 2013). The spatial proxies are based on several factors. Main factor is the average wood use of a house based on wood use surveys. The average wood use is determined separately for different house types that are classified based on main heating method, residential area type, and energy need for heating of the house (Table 2) (Kaski et al., 2016; Natural Resources Institute Finland). The location and main heating method of the houses were obtained from the national building and dwellings register (DVV, 2014) with exact locations of each house. The residential area type was taken from Finnish Environment Institute SYKE's YKR Spatial structure delineation dataset (SYKE, 2015) with a spatial resolution of 250 m × 250 m. Energy need for heating was described by heating degree days[2] per municipality. With these data, a relative wood use is given for each residential building, and these are aggregated into the 250 m × 250 m grid resolution. Only detached and semi-detached houses are taken into account in the proxies, as wood burning in apartment buildings in Finland is negligible (Statistics Finland, 2018).

2.1.2. Sweden

For Sweden, national total RWC emissions are calculated using annual energy balances. Biomass fuel consumption is surveyed on the three most common combustion technologies: wood boiler, wood iron stove and open fireplaces (Swedish Energy Agency, 2015). Activity data is separated into wood logs, pellets/briquettes and wood chips/saw dust. For boilers and stoves, activity data is further separated into traditional and modern technology (Swedish Environmental Protection Agency, 2019).

Table 2
Average wood use (m^3) in a house in Finland per residential area type and primary heating method (Kaski et al., 2016; Natural Resources Institute Finland).

		Residential area type			
		Helsinki metropolitan area	Urban, pop. > 20 000	Urban, pop. < 20 000	Non-urban
Primary heating	District heating	0.9	1.4	1.7	3.2
	Electricity, geothermal etc.	1.5	2.5	3.1	5.5
	Oil, gas	1.5	2.4	2.9	5.3
	Wood oven and other	2.1	4.2	5.2	9.4
	Wood central	7.9	9.3	16.1	19.1

The spatial distribution is based on statistics on the number of appliances per fire and rescue service district (which often coincide with municipalities). The appliances are divided into four types (wood boilers, wood stoves, pellets boilers and oil boilers). Due to large uncertainties for individual years and districts, the median number of appliances per district for the years 2008–2012 is used. Typical energy consumption for a Swedish detached house has been estimated for an average meteorological year using the energy model. Further assumptions regarding firing habits and fuel consumption are based on questionnaires and interview studies. Within each municipality, the emissions are distributed using living area per km^2 from one- and two-dwellings statistics. An additional weighting on municipality levels is applied to take access to district heating into account. The fraction of houses in each municipality attached to district heating is used to calculate a weighting factor between 0.2 (large number of houses attached) and 1.0 (no houses attached) (Andersson et al., 2019).

2.1.3. Denmark

Calculation of the Danish national RWC emissions is based on fuel consumption from the official energy statistics and technology specific emission factors (Plejdrup et al., 2016). The total number of RWC appliances is based on data from the Danish Chimneysweepers Association supplemented with data from the Danish Building and Dwelling Register (Nielsen and Plejdrup, 2018). The total number of RWC appliances is assumed constant throughout the time-series, i.e. from 1985 onwards. Split of the appliances into different technologies is based on replacement rates based on information about new sales from the industry, supported with information from surveys (EA Energianalyse, 2016). Gridding of the RWC emissions is based on data on primary and supplementary heating installation and fuel on building level from the Danish building and dwelling register (BBR). Buildings with RWC are categorised into three building categories (detached house, holiday house, and apartment), and different weighting factors are applied based on the building type and if RWC is used as primary or supplementary heating (Table 3). The weighting factors are based on expert judgements supported by findings by the Danish Energy Agency and information from case study surveys. A spatial distribution proxy is calculated on a 1 km × 1 km resolution, including the share of the national RWC emissions to allocate to the individual grid cells. The BBR data have large uncertainties regarding RWC heating data, as the responsibility for updating part of the information in the BBR register is placed on the building owners, which is often neglected. A higher spatial resolution is not found to be applicable, due to the relatively high uncertainty level of the spatial proxy and the emission calculation for RWC in general.

[2] Heating degree days - Finnish Meteorological Institute https://en.ilmatieteenlaitos.fi/heating-degree-days, accessed 3.2.2012.

Table 3
Weighting factors for RWC in Denmark as primary heating per technology and building group.

Technology	Building group	Primary RWC
Boiler	Detached house	1
Boiler	Holiday house	0.8
Boiler	Apartment building	1
Stove	Detached house	0.8
Stove	Holiday house	0.2
Stove	Apartment building	0.8

2.1.4. Norway

In Norway the emissions from RWC are estimated based on wood consumption per type of technology (i.e. open fireplace, closed wood stove produced before 1998, and closed wood stove produced after 1998) at county level provided by Statistics Norway on a yearly basis from 2005 to 2017 and the emission factors used in Norway for official reporting (Seljeskog et al., 2013; The Norwegian Environment Agency, 2019). Wood consumption data is collected by Statistics Norway according to two different surveys (The Norwegian Environment Agency, 2018). The emissions factors are determined based on wood stove experiments following the Norwegian Standard for testing enclosed wood heaters and smoke emissions (NS3058/NS3059), and they are considered as representative for Norwegian conditions.

The approach for the gridding of emissions consisted of downscaling emissions from county to municipality and then to the grid based on dwelling number and type (apartment, detached house, townhouse and duplex) at 1 km resolution. The number of dwellings and the type they belong to is obtained from Statistics Norway at 250 m resolution. When this work was carried out, dwelling numbers at high resolution was available for the years 2010 and 2012. For the emission inventories before 2010, the dwelling number in 2010 was used, whereas the dwelling number in 2012 was used for the emission inventories after 2010. In addition, wood consumption among the types of dwellings was distributed assuming that 70% of the wood is consumed in houses (i.e., detached, townhouses and duplex), whereas 30% is consumed in apartments. This assumption was based on previous studies on wood consumption in the area of Oslo (López-Aparicio et al., 2017b) and internal databases on wood consumption in other regions.

2.2. Local RWC assessments

2.2.1. Helsinki Metropolitan Area

The local RWC study for Helsinki Metropolitan Area (including the municipalities Helsinki, Espoo, Vantaa and Kauniainen) was done by the Helsinki Region Environmental Services Authority (HSY) (Kaski et al., 2016). The estimate for wood use for 2014 was based on a wood use survey for the Helsinki Metropolitan Area. The survey gave average wood use per appliance per main heating method of a detached house. The wood use was multiplied by the emission factor of the appliance, and appliance emissions were summed for total average emission per house. Total emissions for the area were then obtained by multiplying this average emission by the total number of houses. The main heating method and location of the houses were based on the local building and dwelling register, the same source that was used in the Nordic inventory. Also, the emission factors (presented in Savolahti et al., 2016) were the same as in the Nordic inventory for Finland described above. The study only included residential houses, i.e. recreational houses were omitted.

2.2.2. Västerbotten

For Västerbotten, Sweden, locations and type of appliances in the local inventory was retrieved from chimney sweeper register data. During 2013, a wood use survey was also carried out in four areas close to the city of Umeå (Omstedt et al., 2014). The survey comprised interviews and standardized questions regarding usage and fuel consumption to 176 households. Based on the survey, typical fuel consumption and usage was defined for different types of appliances. Emission factors used in the study are presented in Table 4. Since true locations were available for a large majority of the appliances, no proxies were required to describe the spatial distribution. The emissions were calculated for year 2011.

2.2.3. Copenhagen

The case study for RWC in Copenhagen, carried out in 2015, was based on the total number of RWC appliances from the municipality's chimney sweepers, and information of appliance age distribution and unit consumption from a survey made by the Danish Technological Institute for Copenhagen Municipality (Andersen, 2015). The survey classified the RWC appliances into four categories; house, apartment, allotment society with district heating, and allotment society without district heating. The appliance age distribution used the categories pre-1990, 1990–2005, post 2005, and eco label. Emission calculations were based on the emission factors used in the Danish national emission inventory (Table 1), which are based on national surveys and standard factors from EMEP/EEA Guidebook (European Environment Agency, 2013). The spatial distribution of RWC emissions in Copenhagen was based on the address of the RWC appliances, included in the data from the municipality's chimney sweepers. The RWC addresses were combined with the building and dwelling register (BBR) to add information on building class to allow the spatial distribution to take into account the differences in unit consumption for houses, apartments and allotments. 74% of the addresses could be linked to the BBR, and the remaining addresses were added building class according to the primary building class in the current postal number.

2.2.4. Oslo

The local RWC emissions for the Oslo Metropolitan area cover the city of Oslo and the Greater Oslo Region, with a population of around 0.63 and 1.7 million in 2018, respectively. Emissions from RWC were originally developed by Statistics Norway for the year 2002 and at district level. Even though these emissions are relatively old, they have been used for different air pollution assessments representing 2013 and 2015 (Kukkonen et al., 2020). Until now, there was no detailed information on the evolution of RWC emissions since its compilation.

The RWC emissions were estimated based on the data of a dedicated survey carried out by Statistics Norway about the use of wood combustion and the heating habits in Oslo. The results of the survey include information on the amount of wood consumed in the districts in Oslo, and information on how the wood combustion varies temporally, in terms of weeks, days and hours of the day. The amount of wood combusted was reported in the survey, in terms of the type of technology, i.e., open fireplace, wood stoves produced before 1998 and wood stoves produced after 1998. Thereafter wood consumption data was combined with the corresponding emission factors. The emissions factors were evaluated by Haakonsen and Kvingedal (2001), based on a review of the results from different tests for various fireplaces in Norway. The emission factors used were 40.0, 6.2, and 17.3 g PM/kg of dry wood for wood stoves produced before 1998, wood stoves produced after 1998 and open fireplaces, respectively. The emission results at district levels were thereafter distributed within the district to 1 km grid. There is not documentation on the proxies used to distribute emissions from district level to 1 km grid. However, the evaluation of the data seems to indicate

Table 4
Emission factors used for small scale residential heating in Västerbotten (Omstedt et al., 2014).

Appliance	$PM_{2.5}$ Emission factor [mg/MJ]
Boiler – wood logs	600
Boiler/stove – pellets	28
Boiler – oil	9
Stove – wood logs	400

that population data was used.

2.3. European RWC assessment

The developed spatial distributions were also compared with European level emission inventory TNO-newRWC (Denier van der Gon et al., 2015). The TNO-newRWC is based on TNO-MACC III inventory (Kuenen et al., 2014), but the residential combustion (SNAP 2) PM emissions were replaced with a bottom-up estimate harmonized for all counties as described in Denier van der Gon et al. (2015). In the TNO-newRWC the emissions are calculated on 0.125° × 0.0625° longitude-latitude resolution (c. 7 km × 7 km). In the inventory, the spatial distribution of RWC emissions was based on three factors: population density, access to fuel wood, and urban/rural differentiation. Population grid was weighted with surrounding forest coverage to represent local wood availability. Forest coverage was calculated for 0.25° × 0.5° (c. 20 km × 20 km) area around the cell. Furthermore, rural areas were given double weights compared to urban areas based on information from chimney sweepers in the Netherlands and Sweden (Denier van der Gon et al., 2015). From the TNO-newRWC SNAP sector 2 was used in the comparisons to our national inventories. For each country residential wood combustion comprised over 99% of the total SNAP 2 $PM_{2.5}$ emissions in the TNO-newRWC, so the sector was treated as RWC.

2.4. Comparison methods

To compare the gridded emissions in each case, an index of agreement developed by Duveiller et al. (2016) was used. The index has the benefit of being non-dimensional, bounded and symmetric. The index (λ) can be calculated as

$$\lambda = \alpha \cdot r = \frac{2}{\sigma_X / \sigma_Y + \sigma_Y / \sigma_X + \left(\overline{X} - \overline{Y}\right)^2 / \sigma_X \sigma_Y} \cdot r$$

where $\overline{X}$ and $\overline{Y}$ are the mean values of the datasets X and Y, σ_X and σ_Y are their standard deviations, and r is the Pearson product-moment correlation coefficient. α represents how much the index of agreement deviates from the Pearson correlation coefficient. The index also enables the separation of unsystematic and systematic differences, represented by unsystematic index λ_u and systematic noise f_{sys}.The unsystematic index λ_u represents the value the index λ would take if there was no systematic bias between the datasets (for example one dataset having systematically higher values than the other). The systematic noise f_{sys} represents the proportion of total deviation caused by systematic bias. The mathematics for unsystematic index and systematic noise can be found in the reference. The index of agreement was calculated so that all cells in the study area which had non-zero values in either dataset were included. This was done to reduce the number of zero-value cells, in order to better bring out the differences between the datasets.

The comparison was done on the native resolution of the original emission inventories (250 m × 250 m in the Helsinki Metropolitan Area, 1 km × 1 km in other local cases (Table 5)), and on several aggregated resolutions up to 10 km × 10 km in order to assess if spatial distributions would be more similar on coarser resolutions. To compare with the TNO-newRWC the national emissions were aggregated to 0.125° × 0.0625° (circa 7 km × 7 km). Furthermore, visual inspection was done for normalized emissions maps. Finally, the total $PM_{2.5}$ emissions were compared.

3. Results

Spatial distribution of $PM_{2.5}$ emissions in the Nordic RWC inventory is presented in Fig. 1. Total $PM_{2.5}$ emissions from RWC per country are presented in Table 6 and for the studied regions in Table 7.

3.1. Comparison to TNO-newRWC

Total national RWC $PM_{2.5}$ emissions, wood use and average emission factors from our Nordic inventory (for 2010) and the TNO-newRWC (for 2011) are presented in Table 5, and emission factors in Table 1. The emissions in the TNO-newRWC were significantly higher for Finland and Sweden, and higher for Denmark. Only in Norway the Nordic inventory had higher emissions. For wood use the TNO-newRWC also had lower value in Sweden, and the differences in all countries were smaller than the difference between the emissions. In the TNO-newRWC the emission factors were in the high end of the range or higher than the factors used in the Nordic inventory. While the years compared were not the same for both inventories, this smaller impact to the results than the other factors.

Since total national emissions had large differences between the Nordic and the TNO-newRWC inventories, the statistics were calculated for normalized emission grids, in order to be able to compare the spatial weighting only. The scatterplots and statistics are presented in Fig. 4.

Index of agreement was similar in each country (ranging from 0.76 to 0.79). However, different factors explain the differences between the two inventories in each country. A common difference between the Nordic inventory and the TNO-newRWC in each country was that the former placed more emissions on cells directly on coast. This was especially noticeable in Denmark (Fig. 3b).

In general, the use of proximity to forests as a proxy in the TNO-newRWC inventory seemed to cause more weight to be allocated into outskirts of capital areas and less into suburban areas compared to the Nordic inventory (Fig. 2). The difference between weight of urban and rural areas was also higher in the TNO-newRWC than in the Nordic inventory in Finland and Sweden. However, in Norway the Nordic inventory had larger difference between urban and rural areas, and in Denmark the Nordic inventory had much higher weighting in Copenhagen. Therefore, no single universal factor was identified that would explain the differences between the distributions in all countries. This implies that local characteristics are important when the spatial proxies are chosen.

In Finland the maximum weights were in the same areas in both inventories. However, the Nordic inventory had more even distribution, especially in the rural areas, whereas in the TNO-newRWC the emissions were concentrated into the bigger cities (Fig. 3a). In Helsinki Metropolitan Area the Nordic inventory had more weight on the residential areas with dense detached housing, and emissions in the TNO-newRWC were mainly allocated into rural and more sparse detached housing areas. The difference was due to the TNO-newRWC using proximity to forests as a proxy and different urban/rural differentiation. Since the Nordic inventory had a higher spatial resolution than the TNO-newRWC, it was possible to use more detailed land use data indicating the rural and urban areas. The TNO-newRWC grid cuts through an area

Table 5
Details of the comparisons for each case.

	Region	Resolution	Area (km^2)	Year, Nordic inventory	Year, compared inventory
Denmark	Copenhagen	1 km × 1 km	130	2014	2015
Finland	Helsinki metropolitan area	250 m × 250 m	790	2014	2014
Norway	Oslo	1 km × 1 km	970	2000	2002
Sweden	Västerbotten	1 km × 1 km	14 000	2010	2011
TNO-newRWC	Continental Nordic	0.125° × 0.0625°	2 300 000	2010	2011

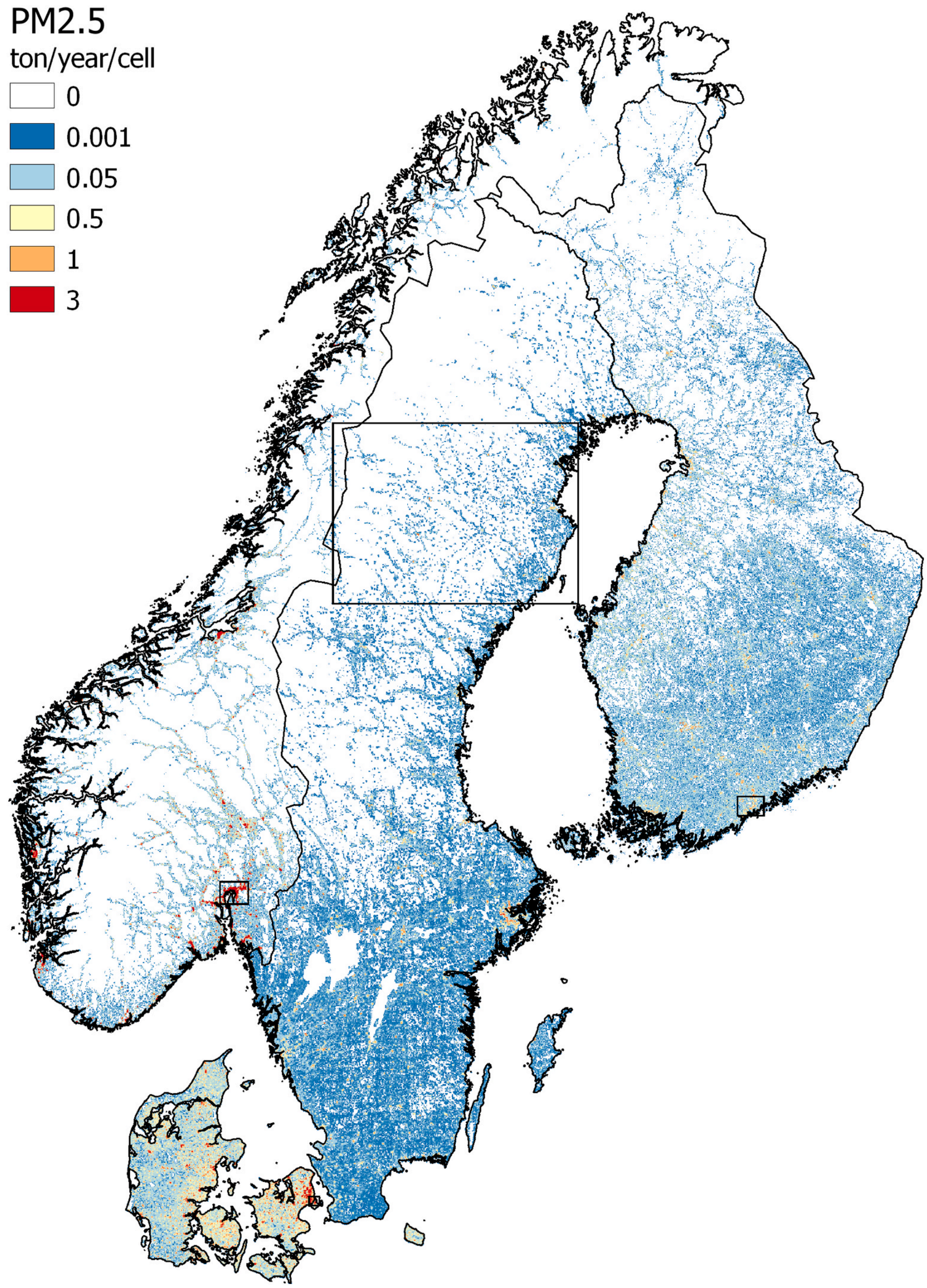

Fig. 1. $PM_{2.5}$ emissions from RWC in the Nordic countries in 2014 per 1 km × 1 km cell. Local inventory areas marked with rectangles.

Table 6
RWC $PM_{2.5}$ emissions and wood use from our Nordic (year 2010) and the TNO-newRWC (year 2011) inventories.

	$PM_{2.5}$ (kton)			Wood use (PJ)			Average EMF (mg/MJ)		
	Nordic	TNO	%-Diff	Nordic	TNO	%-Diff	Nordic	TNO	%-Diff
Denmark	17.9	27.5	54%	36.9	44.1	20%	485	624	29%
Finland	12.4	50.2	305%	58.4	70.3	20%	212	714	236%
Norway	26.8	23.6	−12%	29.7	29.6	−0.3%	902	797	−12%
Sweden	8.4	32.9	292%	47.7	43.8	−8%	176	751	327%

of high RWC emissions in Helsinki metropolitan area, enabling different land use data to affect the results significantly. Index of agreement (0.77) indicated that there were similarities in the spatial distributions, but f_{sys} was high for normalized values. This implied that the cells with high weights had higher share of the total emissions in the TNO-newRWC than the Nordic inventory. This emphasizes the observation

Table 7
$PM_{2.5}$ emission from residential wood combustion in the Nordic and local inventories in the study areas.

$PM_{2.5}$	Local [t]	Nordic [t]	Diff. (Nordic-local) %
Helsinki Metropolitan Area	175	212	21%
Västerbotten	737	493	−33%
Copenhagen	94	189	100%
Oslo	549	1672	2000%

that the Nordic inventory had more even distribution of emissions. The Nordic inventory had 15% more cells with emissions.

In Denmark, both inventories had rather even distribution of emissions. Maximum weights fell into different cells, especially around Copenhagen, where the Nordic inventory had more emissions closer to the city and the TNO-newRWC more in north closer to forests.

In Sweden, the Nordic inventory had highest weights in bigger cities, with few smaller cities receiving higher weights as well. The TNO-newRWC had more weight on the population centres (except Stockholm). The Nordic inventory had more weight in rural areas, as well as 21% more cells with emission. One interesting difference can be seen

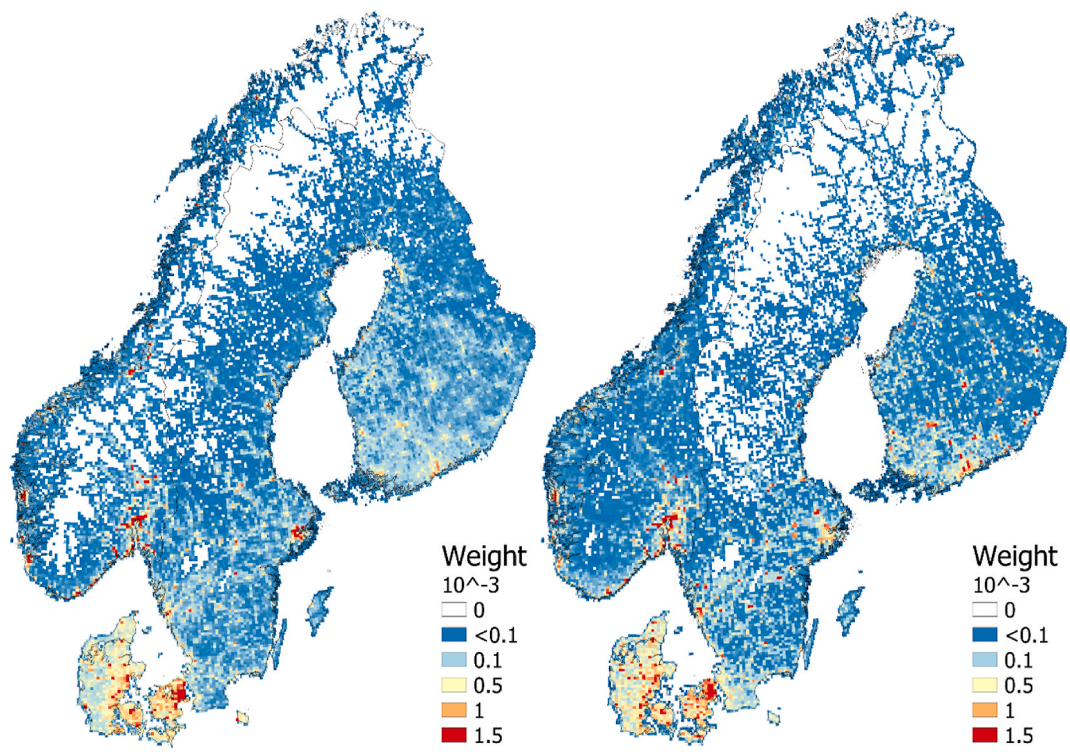

Fig. 2. Normalized RWC $PM_{2.5}$ emission distributions in 7 km × 7 km grid from the Nordic inventory on the left and the TNO-newRWC on the right. Normalization was done separately within each country. Normalization was done by dividing the emissions in the cells with the total emissions of the area.

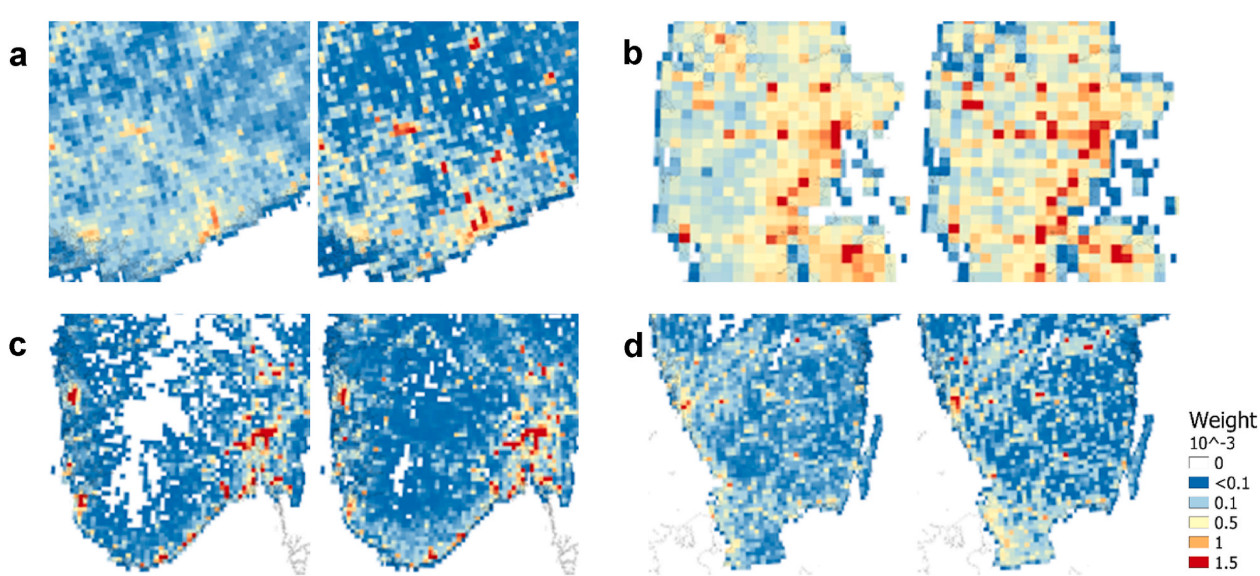

Fig. 3. Details on the spatial distribution of emissions from the Nordic inventory (on the left) and the TNO-newRWC (on the right) in (a) southern Finland, (b) continental Denmark, (c) southern Norway, and (d) southern Sweden. Note that the size of the areas shown here differs between the plots.

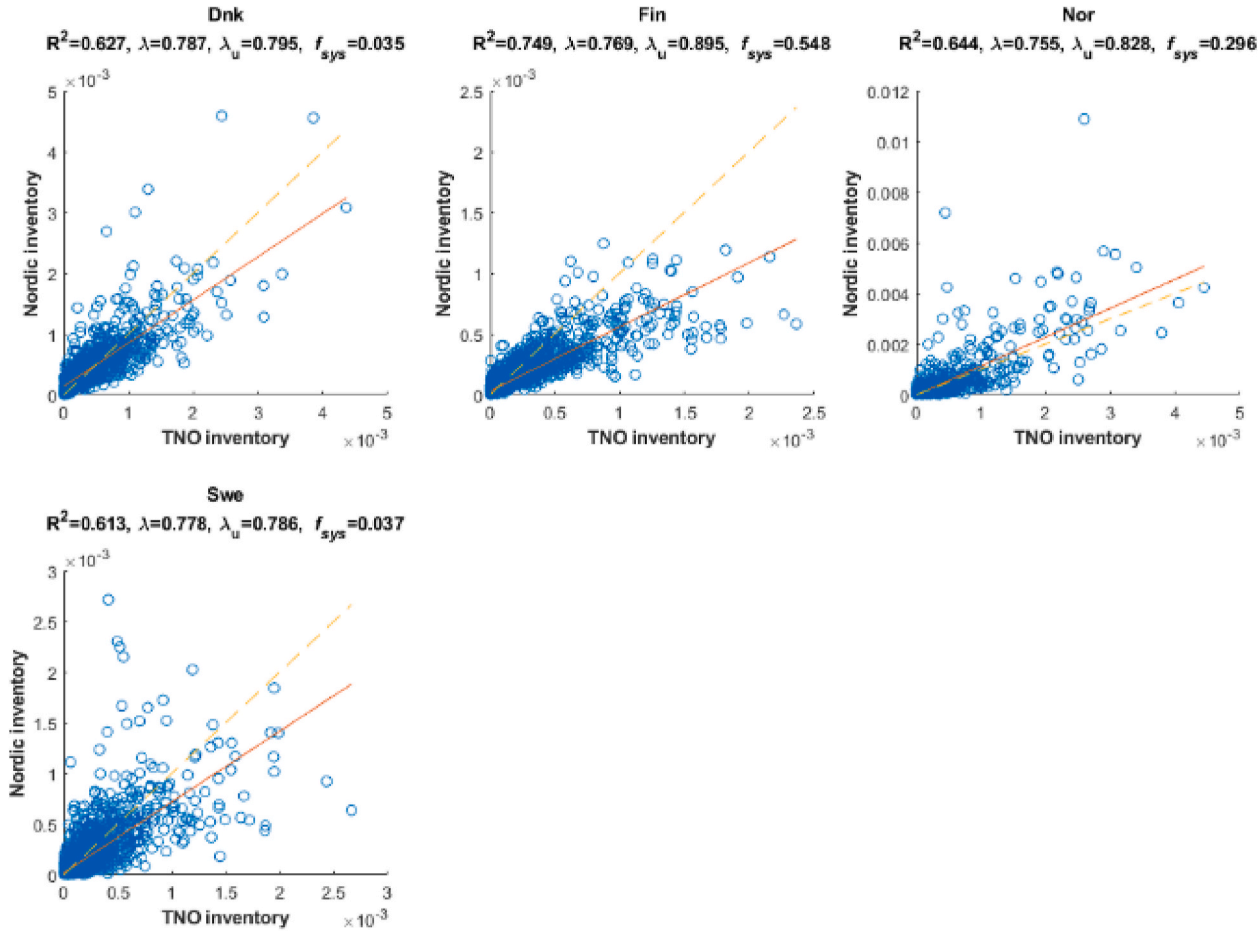

Fig. 4. Scatterplots and statistics for comparison of normalized grid values between this study and the TNO-newRWC. Orange line is the regression and dashed yellow the y = x-line. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

across the southern tip of Sweden, where the TNO-newRWC inventory had more emissions spread around the area (Fig. 3d).

In Norway, in contrast to the other Nordic countries, the TNO-newRWC had more cells (32% more) with emissions, mainly in high mountainous areas (Fig. 3c). Weights in these cells were small. In both inventories the emissions were concentrated to population centres. The Nordic inventory had higher weights in some municipalities in the west coast, e.g. Stavanger.

3.2. Comparison to local case studies

The total $PM_{2.5}$ emissions from the Nordic inventory and the local inventories in the studied areas, along with difference between the two, are presented in Table 7.

3.2.1. Helsinki Metropolitan Area

For Helsinki metropolitan area the comparison between the Nordic inventory and the local case was done for 2014. The comparison included only residential wood combustion, i.e. not recreational houses. The Nordic inventory gave 21% higher total RWC emissions than the local study. Breaking down the emissions per appliance type show similar difference (21–28%) in fireplaces and sauna stoves, but in boilers the local assessment has more than three times higher emissions. In total, boilers comprised 7% and 2% of the total RWC emissions in the local and Nordic inventories, respectively, i.e. their impact in both cases was small compared to other appliances. Wood use was 37% higher in the Nordic inventory compared to the local case, while the emission factors per appliance were identical in both inventories.

Fig. 5 shows the normalized spatial distributions of $PM_{2.5}$ emissions for the Helsinki Metropolitan Area from the local and Nordic inventories. Visual inspection suggests the distributions to be similar, which was to be expected, since in both cases the spatial locations of buildings are based on the same data source. Closer inspection reveals small differences in weights between different parts of the Helsinki Metropolitan Area. Spatial distribution of boiler emissions showed the largest difference between the inventories. The Nordic inventory allocated boiler emissions only to 3% of the cells with RWC, whereas in the local dataset almost all cells had also boiler emission. In the local study, questionnaire showed that even some houses that according to the buildings and dwellings register had other main heating method than wood boiler (e.g. electric heating) had some wood use in boilers. Therefore, all houses with that heating type were allocated with emissions from boilers. In the Nordic inventory only houses with wood boiler as main heating method were allocated with boiler emissions.

Scatterplots and calculated statistics for the Helsinki Metropolitan Area are presented in Fig. 6 for different aggregations in terms of spatial resolution. The results confirm the similarity of the spatial distributions between the datasets. On the initial resolution (250 m) the similarity is already high ($\lambda = 0.92$), approaching almost perfect resemblance on coarser resolutions ($\lambda = 0.97$). The scatterplots and indexes of agreement showed the same results as the visual inspection, with boiler emissions showing low values for index of agreement for all resolutions (from 0.11 to 0.27 from high to low resolution), and higher λ_u only on coarser resolutions (from 0.33 to 0.97 from high to low resolution).

Despite the different approaches between the inventories, both inventories showed similar emissions in the area. In the Nordic inventory

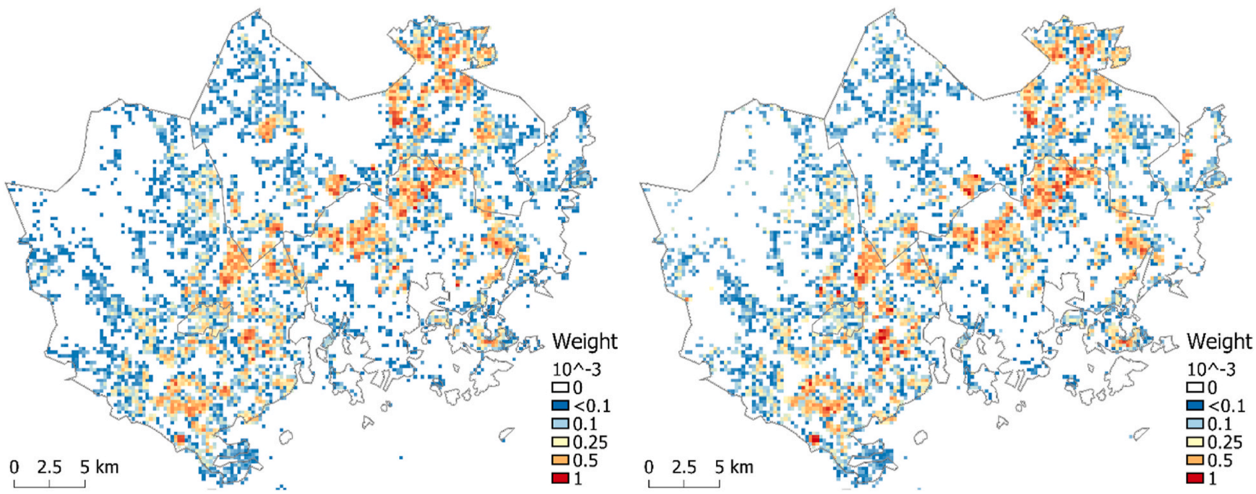

Fig. 5. Normalized $PM_{2.5}$ emission distribution in 250 m × 250 m grid in the Helsinki Metropolitan Area from HSY/local assessment on the left and from the Nordic inventory on the right. Normalization was done by dividing the emissions in the cells with the total emissions of the area.

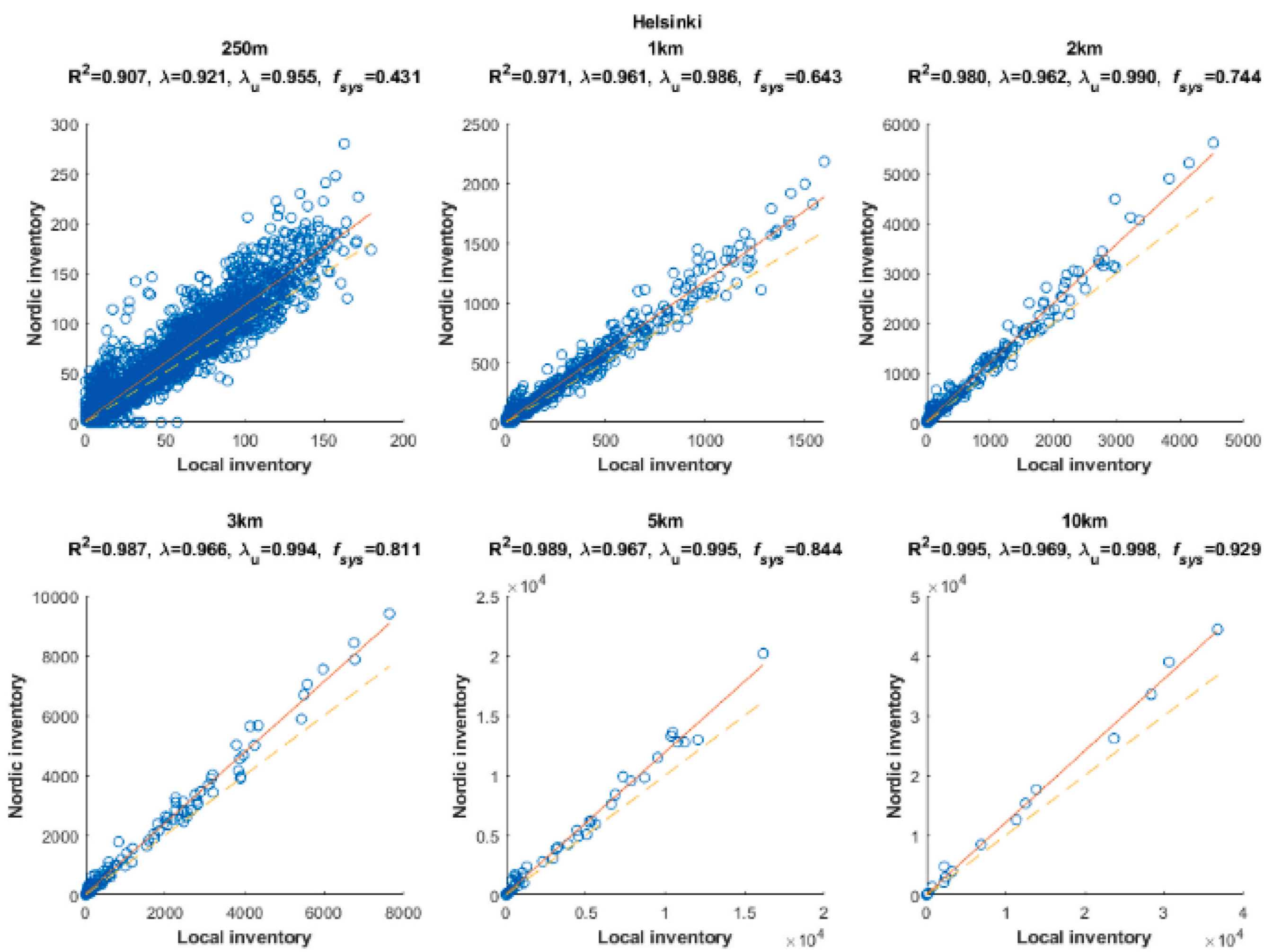

Fig. 6. Scatterplots and statistics for the Helsinki Metropolitan Area all appliances case for different aggregation levels. Unit is kg/cell. Orange line is the regression and dashed yellow y = x-line. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

the emission allocated to the Helsinki Metropolitan Area are not only affected by the weights in the area, but also weights elsewhere in Finland. Therefore, while the house data is same in both cases, and similar normalized distribution was to be expected, equal total emissions were not a given.

3.2.2. Västerbotten

The Västerbotten case study for 2011 was compared to the Nordic inventory year 2010. The local inventory had 33% higher emissions in Västerbotten than the Nordic inventory. While there are minor differences between the two years compared in relation to wood use, the main reason for the discrepancy was differences in emission factors. For the local inventory for Västerbotten emission factors were higher than those used in the national Swedish inventory (see Tables 1 and 4).

Normalized $PM_{2.5}$ emissions distributions for Västerbotten are shown in Fig. 7. The local assessment shows more evenly distributed emissions. The Nordic inventory has more emissions weighted to population centres. However, the method used in the Nordic inventory spreads the emissions into larger number of cells than the local inventory.

On the native resolution of 1 km × 1 km the index of agreement (0.63) shows that there are significant differences between the datasets (Fig. 8). As systematic noise was low, the differences are mainly due to different distributions, not the amount of emissions. However, already at the next aggregation level (2 km resolution) the index is 0.80, indicating that the emissions are in fact distributed similarly when adjacent cells are taken into account. I.e. the main differences are in the fine details of the distributions. Systematic noise stays low even in coarse resolutions. From the spatial distribution maps it is apparent that the Nordic inventory spreads the emissions to the same areas, but into more cells within those areas, compared to the local inventory. This somewhat smooths the distribution and explains also why the agreement quickly gets better as the emissions are aggregated into lower resolution. When compiling the Nordic inventory emissions for Sweden were transformed from original SWEREF99 TM projection to ETRS89/LAEA Europe. This caused some smoothing in the distribution, enhancing the difference between the Nordic and local inventory.

3.2.3. Copenhagen

The Copenhagen case study for 2015 was compared to the Nordic inventory year 2014. Spatial distributions of normalized $PM_{2.5}$ emissions for Copenhagen are presented in Fig. 9. The study area was the smallest of all cases, with the least cells in the study area. The small number of cells (i.e. small sample size) could've affected the index of agreement results. Especially on the coarser resolutions the sample size was too small to give meaningful results.

In Copenhagen the Nordic inventory had double the emissions of the local assessment. This was caused by the use of lower unit wood consumptions for appliances in the local assessment, based on results from a local survey on RWC in Copenhagen, compared to the national average unit consumption applied in the national model used in the Nordic inventory. The possible difference between the different years had small impact by comparison. The index of agreement (presented in Fig. 10) showed low values in the original resolution (0.56). However, systematic noise was high, and λ_u (0.83) indicated that without the large difference in the total emissions the agreement would have been good already in high resolution. As the emission distributions presented in Fig. 9 confirm, the distribution patterns had much in common. The most noticeable difference was that the local inventory had higher maximum values. In both cases, the distribution was based on chimney sweeper data, and therefore the location and type of appliances were the same. However, in the local inventory there was more detailed knowledge of fuel consumption for different appliance types (stove and boiler) in different building types (single-family house, holiday house and apartment building). Same type of information was not available on a national level to split the fuel consumption by technology or by building type, and the emission calculation was based on assumptions on fuel consumptions for stoves, open fireplaces and boilers. This caused some losses in the details of the distribution and explains the difference between the distributions.

3.2.4. Oslo

The local Oslo inventory for 2002 was compared to the Nordic inventory year 2000. The Nordic inventory had three times higher $PM_{2.5}$ emissions than the local assessment. There were several reasons for the difference. Wood use estimate in the local inventory was low and inconsistent with latest Statistics Norway wood use estimates. This seems to be caused by different methods used (personal communication with Statistics Norway). Furthermore, the inventories used different emission factors. For pre 1998 stoves the local inventory had 2.4 times higher emission factor than the Nordic inventory, but for newer stoves (1998 onwards) the Nordic inventory had almost two times higher emission factor. This implies that the difference in emissions could have been even larger if similar emission factors had been used. Both the emission factors and wood use estimates of the local inventory are now considered outdated, showing that local level assessment does not automatically give better results as inventory with broader scope.

Normalized $PM_{2.5}$ emission distributions for Oslo are shown in Fig. 11. The local data was in a different projection than the Nordic inventory, so the local data was resampled using ArcGIS with nearest neighbour resampling method. This method is usually only used for categorical data but was chosen as bilinear resampling would have averaged the values in the cells, and thus changed the spatial distribution. Nearest neighbour preserved the distribution and total $PM_{2.5}$ emissions as closely as possible. There were small number of cells with changed values, but this did not have noticeable effect on the results.

The Nordic inventory distributed the emissions more evenly than the local assessment, which had higher emission shares standing out in few

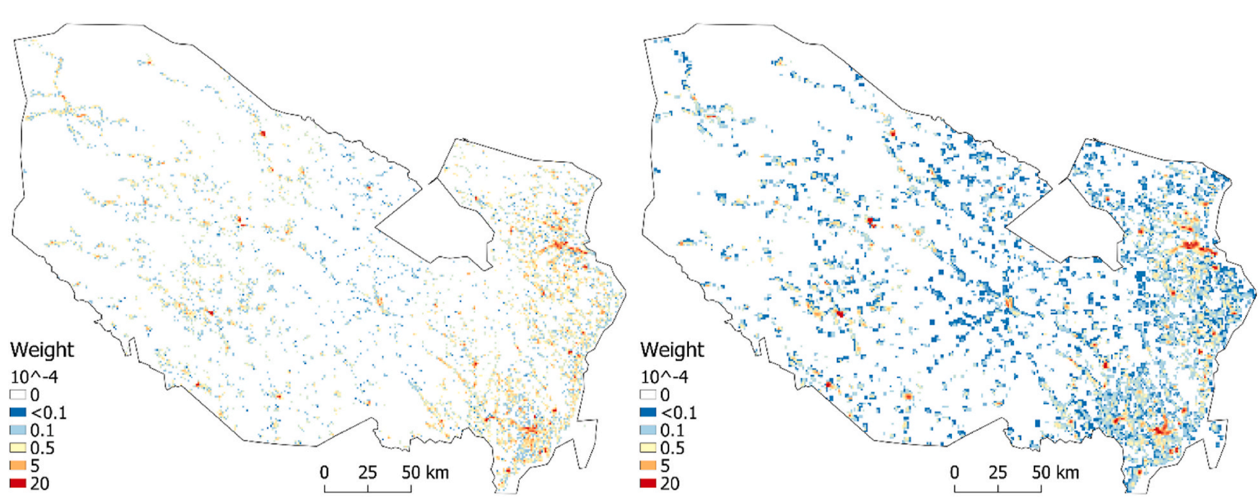

Fig. 7. Normalized $PM_{2.5}$ emission distribution in Västerbotten from local assessment on the left and the Nordic inventory on the right. Normalization was done by dividing the emissions in the cells with the total emissions of the area.

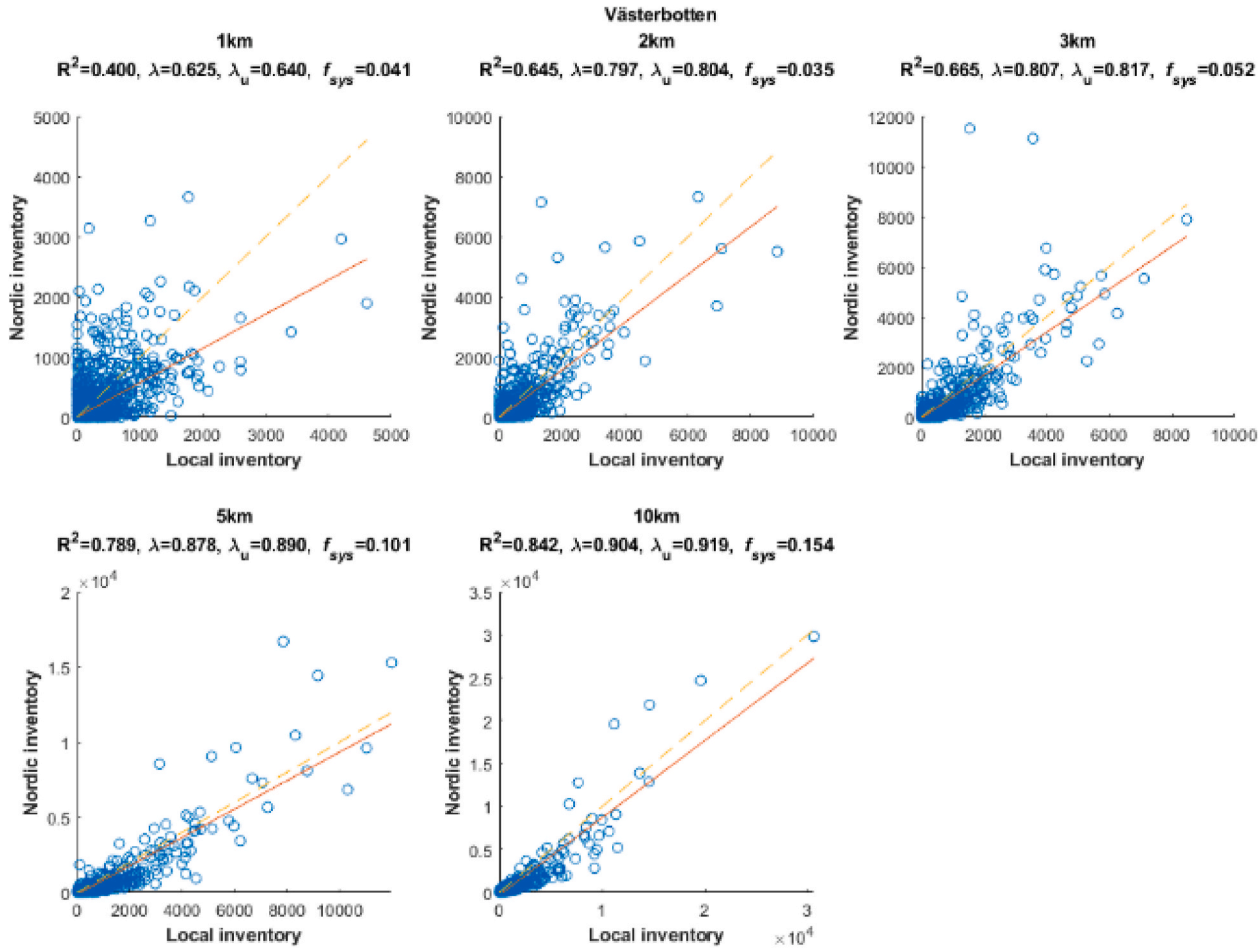

Fig. 8. Scatterplots and statics for the Västerbotten case for different aggregation levels. Unit is kg/cell. Orange line is the regression and dashed yellow y = x-line. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

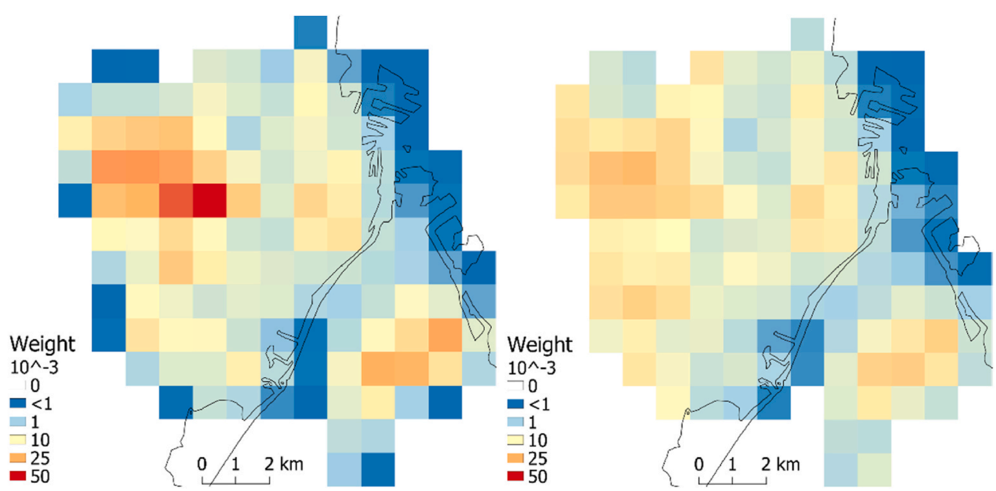

Fig. 9. Normalized $PM_{2.5}$ emission distribution in Copenhagen from local assessment on the left and the Nordic inventory on the right. Normalization was done by dividing the emissions in the cells with the total emissions of the area.

cells characterized by high population areas. Overall, the emissions were distributed into the same areas with differences in the details. Water and forest areas were left with zero emissions in both inventories.

The differences in total emissions and similarities in the spatial distribution between the assessments can also be seen in the index of agreement results and scatterplots presented in Fig. 12. Systematic noise was high on all resolutions. Index of agreement itself was low (0.34), but λ_u (0.96) shows that without the systematic difference λ would have values showing good agreement in the spatial pattern. Systematic noise accounted for 94% of the differences between the datasets. The large difference between the emissions kept λ low even in coarse resolutions.

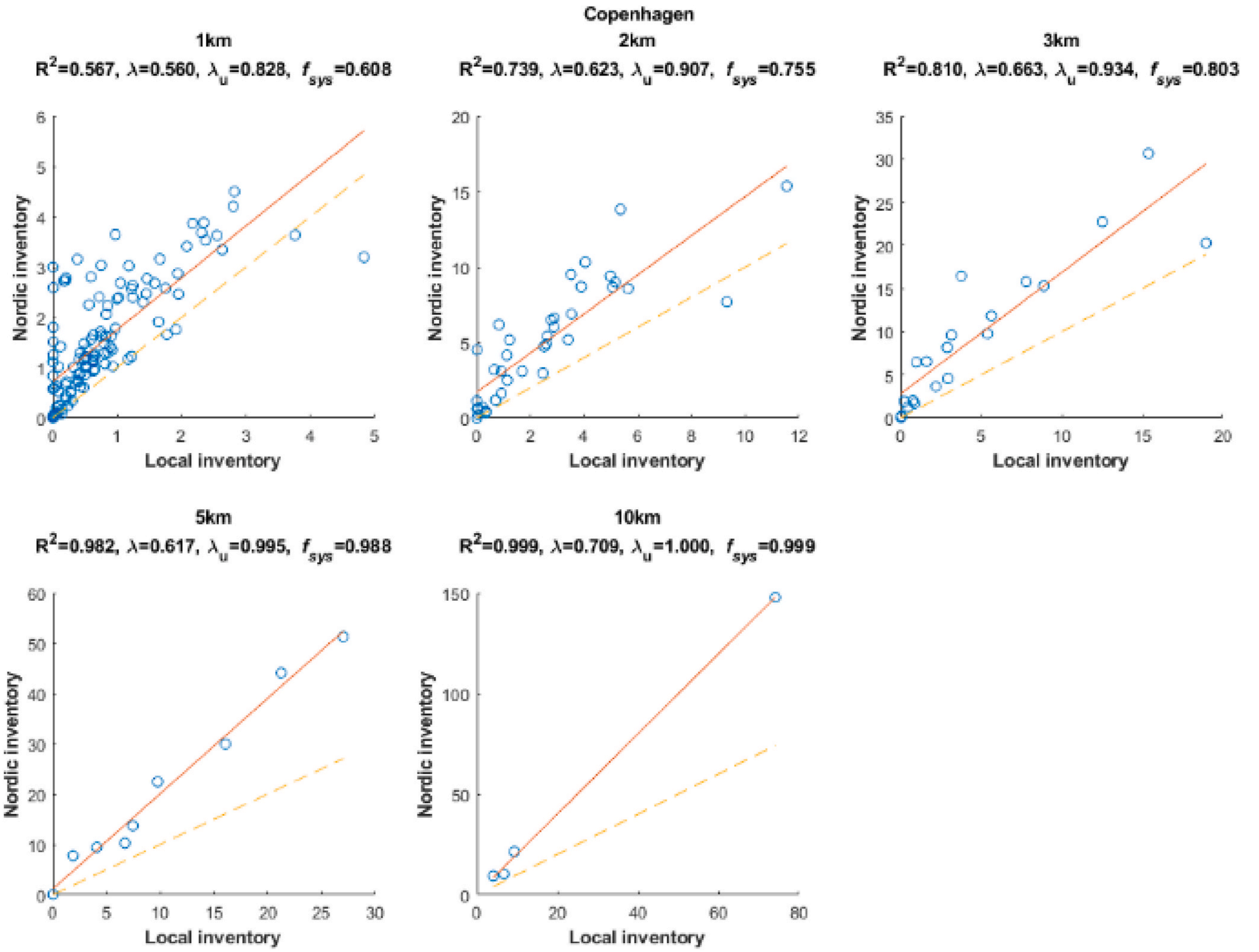

Fig. 10. Scatterplots and statics for the Copenhagen case for different aggregation levels. Unit is kg/cell. Orange line is the regression and dashed yellow y = x-line. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

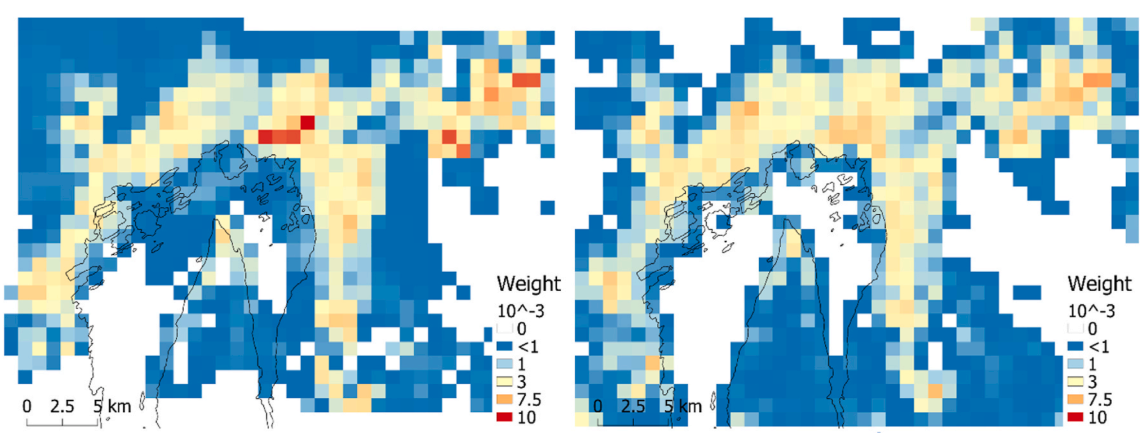

Fig. 11. Normalized $PM_{2.5}$ emission distribution in Oslo from local assessment on the left and the Nordic inventory on the right. Normalization was done by dividing the emissions in the cells with the total emissions of the area.

4. Discussion

There are several factors that influence the spatial distribution of emissions from RWC which would be beneficial to take into account in emission inventories, and which are not adequately represented by using population density as a proxy for spatial allocation of emissions. First, typically RWC takes place predominately in detached and other small houses. This was shown to be the case also in the studied Nordic countries. Norway was to some degree an exception as considerable amount of wood is combusted also in apartment buildings. For the spatial distinction between the different housing types, the data are typically available in building and housing registers.

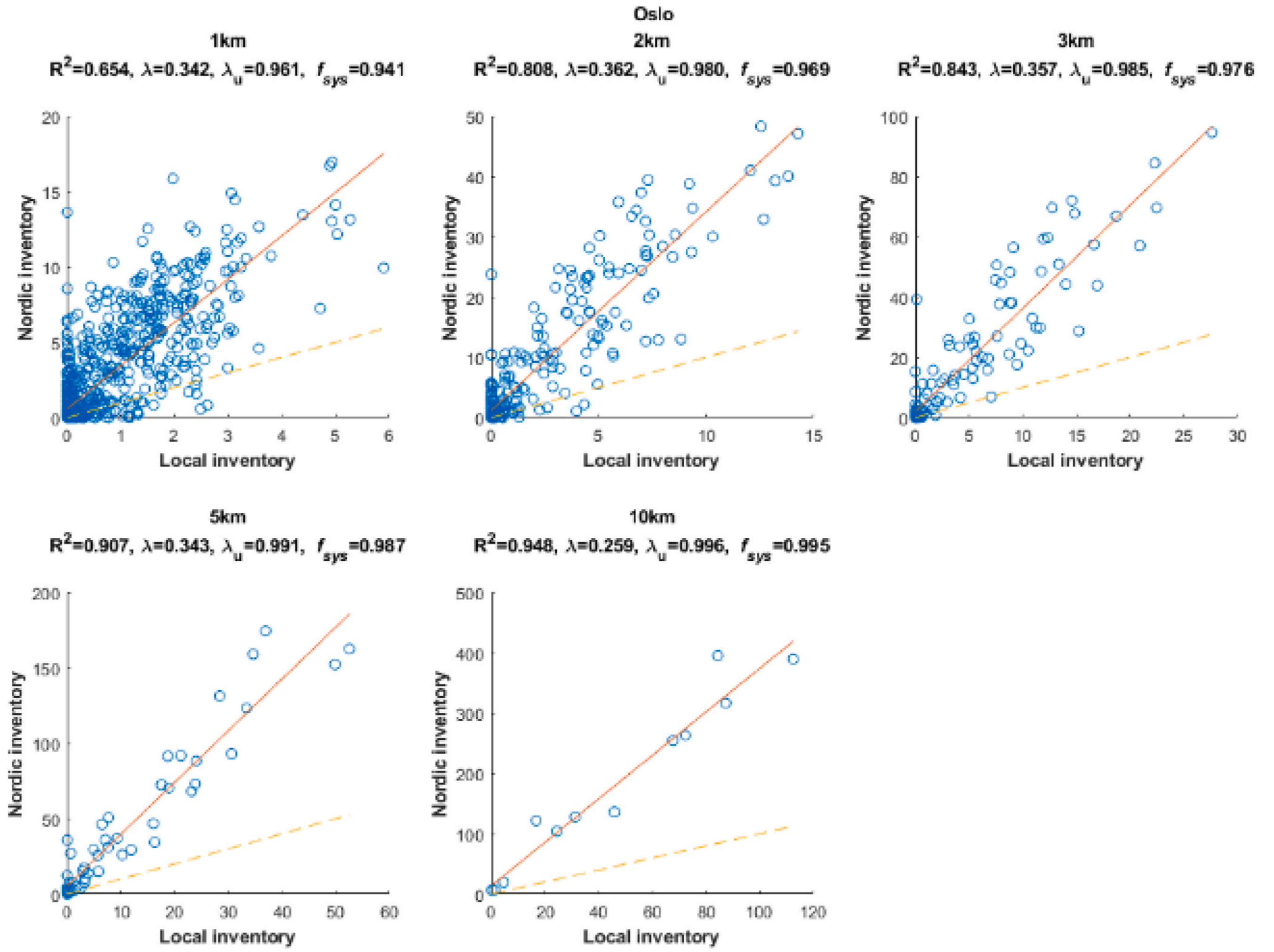

Fig. 12. Scatterplots and statics for the Oslo case for different aggregation levels. Unit is kg/cell. Orange line is the regression and dashed yellow y = x-line. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Second, the prevalence and spatial locations of different combustion appliance types would be beneficial to know. Both emission factors and average amounts of wood used in different appliances differ strongly between appliance types. This has an implication for both total emissions in certain areas and the spatial distribution of emissions in cases where the appliance types are not uniformly distributed over the area. Especially strong differences occur typically between the use of wood boilers that are used for primary heating purposes and stoves for supplementary or occasional use. The data on the average prevalence of combustion appliances in a certain area can typically be obtained from questionnaire surveys, if available. However, these often do not offer information about the spatial distribution of the appliances. Building and housing registers might include information about primary heating methods and could therefore be used for the identification of wood boilers. However, detailed information about the number and types of less frequently used stoves are typically not available. One potential source for such detailed information is chimney sweepers' registers in some countries or regions. However, such register data or the format might often not meet the needs for systematic use in national or other large-scale emission inventories.

Third, there are considerable differences in average wood use amounts per household or appliance type between different types of neighbourhoods. Especially, as the Finnish wood use surveys show, differences in wood use per household between urban and less urban areas can be many-fold (Torvelainen, 2009). To obtain such differences, questionnaire surveys could be implemented on a regular basis, or estimates from other areas or countries could be used. These could then be applied in the inventories, e.g., by combining housing register and land use data.

The above-mentioned factors are taken into account in varying degrees in the studied inventories. All countries used building and dwelling registers, with Finland and Denmark also utilising heating method data from the register. Denmark used chimney sweeper data for RWC appliance numbers. Surveys were used by all countries for fuel wood estimates per appliance and household type. All local case studies based their estimates on local surveys. Västerbotten and Copenhagen cases also utilised chimney sweeper data for appliance numbers and locations. Helsinki Metropolitan Area case relied on building and dwelling register data for the location of houses (and, as a proxy, appliances).

Based on the proxies used in the Nordic inventory it seems that data from chimney sweepers is the best possible data for RWC spatial assessment, at least in these countries. However, this kind of data was not available in all countries. For example, in Finland the data are scattered across each sweeper's own database, which can include non-digitized material. Other data identified as good or even essential were building and dwelling data containing house locations and primary heating methods. These help to locate the possible wood combustion appliances and their estimated wood uses. Furthermore, residential area type, whether this means e.g. detached housing versus apartment buildings or population density of the population centre, can also be used as indication of the prevalence of RWC. To fully implement this type of data it needs to be accompanied with questionnaire or other data on how much wood is used in different settings. Even if all described data would be available, it does not guarantee that the resulting spatial distribution represents the activity very well. It is important to identify the country characteristics in order to identify the best sources of proxy data.

An example of a sophisticated spatial distribution is presented in Grythe et al. (2019). They describe the MetVed model for emissions from RWC at high spatio-temporal resolution. The MetVed model is based on the combination of downscaling with bottom-up principles to estimate a wood burning potential at 250 m resolution. In order to define the wood burning potential, the MetVed model combines several databases, including statistics for energy consumption in households at the municipality level and per dwelling type, placement of fireplaces as points from the fire and rescue agencies registry, and the geographical position and type of dwellings and the available technologies for heating (e.g., district heating). An interesting innovation is the use of web-crawled data from real estate advertisement portal for dwelling details, such as heating types and wood combustion appliances (Lopez-Aparicio et al., 2018). The method could be applied in other countries as well and could provide the otherwise missing information on installed appliances on dwelling level. This method was not implemented in the Nordic emission inventory for RWC as it was not available at the time.

The current comparisons were done in aggregated resolutions in order to study if the spatial distributions would be more similar on a larger scale. The aggregation method affects the results of this assessment. For example, this type of simple aggregation can be done by summing together cells, for example, to the east, south and southeast of a cell, or west, north and northwest of it. If there are large weights placed into adjacent cells in the inventories, different aggregations might increase the index of agreement between them or not have an impact. The aggregation can also be done in a more sophisticated way, e.g. taking account the emissions in all adjacent cells, or doing the aggregation by neighbourhoods or larger housing areas.

While the local inventories only concentrate on the area in question, in national inventories emission distribution weights in other parts of the country have an impact on the local emissions when the emissions are distributed from a national or county level. Furthermore, there might be more than one step from national emissions to grid level, e.g. first distributing the emissions to municipalities. Thus, the total emissions in an area might be different between two inventories, even with similar spatial distributions.

The spatial distributions are sensitive to the assumptions made on the proxies that could correlate with the source in question. Seemingly similar and reasonable but fundamentally different proxies might lead to quite different spatial distributions. An example in our study could be seen for Stockholm (Fig. 13), where the Nordic inventory had more weight concentrated into smaller area in the east side of the city centre, compared to the TNO-newRWC inventory, which had more weight to north and south in the outskirts of the city. Resolution is coarse for city-level health impact assessment, but this comparison highlights how proxies can create notably different emission distributions on densely populated areas. This can have significant impacts on health impact

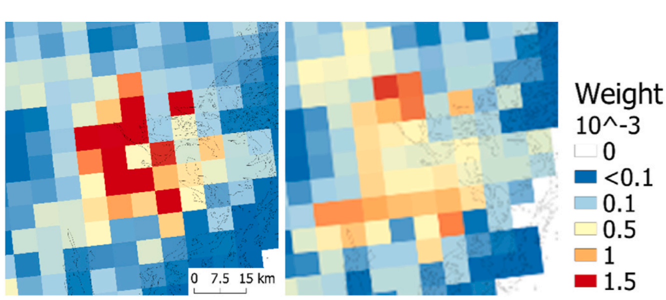

Fig. 13. Normalized $PM_{2.5}$ emission distribution in Stockholm from the Nordic inventory on the left and the TNO-newRWC on the right.

assessments based on the emissions.

Timmermans et al. (2013) suggest that downscaled emission inventories should be used with caution, as they may overestimate urban emissions. However, we argue that the challenge is in the data availability and spatial proxies, and with correct proxies top-down inventories may represent local emissions as well as bottom-up inventories. We agree with the authors that the downscaled inventories need to be evaluated.

As seen from the inventories in our study, there is a wide range of emission factors for different RWC appliances. Emission factors depend on number of factors, i.e. the appliance type and structure, quality of fuel and operation of the appliance. It is a challenge to take all these into account when calculating the emissions. Furthermore, appliances differ between the countries, and, therefore, some appliances have a small number of emission measurements on which the emission factors are based on. This of course increases the uncertainties related to the factors. On the other hand, if generalized emission factors are used for several countries, the risk is that national characteristics are not fully represented. In general, in order to have comparable results, factors based on measurements from diluted flue gas should be used.

An important point when comparing the Nordic inventory to the European TNO-newRWC inventory is the modifiable areal unit problem (MAUP). As the resolution of the European grid is roughly 7 km $\times$ 7 km, the way the grid is setup across cities might have an impact on the spatial distribution. If, for example, the grid cell splits a residential area with high emissions in two, the emissions might be spread to twice the area than it would if the whole area would be in one cell. The grid cell size is also larger than many residential areas in the Nordic countries, meaning that the emissions are averaged into larger areas than where they actually happen. This is true with higher resolution as well, but as cells get smaller more details can be described (assuming the data allows higher resolutions).

All proxies presented here are static in time, i.e. they do not take into account how the emission distribution has changed between the years. This is a common feature in many emission inventories. For the past years it would be possible to change the proxies with the correct data, if such data can be found. The development of future scenarios is more challenging, as it is not necessarily known where new housing areas are built and how the prevalence of different appliances develops. Urbanization may result in higher population densities and change the spatial distribution of the emissions from residential heating, and it would be important to catch the changes in emission spatial patterns to assess the health impacts of e.g. different policy options.

5. Conclusions

In this paper we have studied the spatial proxies used for spatial distribution of residential wood combustion emission in the continental Nordic countries in a novel Nordic emission inventory. Common data used for the proxies were: building data on locations and primary heating methods, and questionnaire-based wood use estimates for appliances or different primary heating methods. Other data used were residential area type and heating degree day (Finland), supplementary heating installations (Denmark), and living area (Sweden).

The $PM_{2.5}$ emissions were compared to selected local case studies. The comparisons, especially in the Helsinki Metropolitan Area, showed that it is possible to reach similar spatial distributions through nation-wide methods as with more local studies based on local specific data. However, this didn't guarantee that the total emissions were similar.

The national emissions were also compared to the European-wide inventory TNO-newRWC. The comparison revealed the importance of how differences between urban and rural residential wood combustion are handled. In several Nordic countries, while wood combustion in general is more common in rural area houses, high level of activity is also reported in urban areas. Urban areas can also have much more dense housing, concentrating emissions into these areas. Therefore, while appliance types differ between urban and rural areas (for example, in Finland boilers are more common in rural areas), and more wood is combusted per appliance in rural areas, the number of appliances is the main factor affecting the differences in the calculated emissions between the area types. These issues need to be considered with care, as their impact can be large on the emission distribution especially on the border between urban and rural areas.

The comparison also showed the importance of local characteristics in spatial distribution of the emissions. The method used by the TNO-newRWC created distributions that had similar values for the index of agreement *λ*, representing correlation between the datasets, in each Nordic country. However, the differences in the distributions were not due to the same factors. For example, in Finland and Sweden the Nordic inventory weighted more emissions to the rural areas, whereas in Norway the TNO-newRWC did. The Nordic countries are often seen as countries with much resemblance. However, even between these countries there are different appliances and different building types where wood is burned. Therefore, a common method applied to all countries cannot cover the spatial distribution in as good detail as one applying local information. This is a major challenge for European wide studies and products as they need to cover at least 30 countries (EU-27, UK, Norway and Switzerland).

While local characteristic should guide the choice of proxies, it is ultimately dependent on available data. First crucial step is to identify the areas where residential wood combustion happens, and then what available data would best represent those areas and the activity within them. Based on our study the best proxies would be based on data from chimney sweeper registers, since those can have detailed, up to date appliance and usage information. Building and dwelling register and/or residential area type data offer potential surrogates for appliance location data, assuming general appliance prevalence is known. As total wood used in residential combustion is often unknown, wood use surveys, conducted in different area types, help to assess the use and also the differences in usage in, for example, urban and rural areas. Finally, it is important to avoid including data just for inclusions sake. Any new data should have a notable and justified positive effect on the quality of the result.

CRediT authorship contribution statement

Ville-Veikko Paunu: Conceptualization, Methodology, Formal analysis, Writing – original draft, Visualization. **Niko Karvosenoja:** Conceptualization, Methodology, Writing – original draft, Supervision, Project administration, Funding acquisition. **David Segersson:** Methodology, Writing – review & editing. **Susana López-Aparicio:** Methodology, Writing – review & editing. **Ole-Kenneth Nielsen:** Methodology, Writing – review & editing. **Marlene Schmidt Plejdrup:** Methodology, Writing – review & editing. **Throstur Thorsteinsson:** Methodology, Writing – review & editing. **Jarkko V. Niemi:** Methodology, Writing – review & editing. **Dam Thanh Vo:** Methodology, Writing – review & editing. **Hugo A.C. Denier van der Gon:** Methodology, Writing – review & editing. **Jørgen Brandt:** Project administration, Funding acquisition, Writing – review & editing. **Camilla Geels:** Project administration, Funding acquisition, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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