Contribution of residential combustion to ambient air pollution and greenhouse gas emissions

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Residential combustion in a village in the Spanish Pyrenees.Image: Barend van Drooge, IDAEA-CSIC, ES.

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Summary and Conclusions

This report assesses the contribution of the residential combustion sector to ambient air pollution and greenhouse gas (GHG) emissions in Europe. The use of biomass, here mainly wood, as a residential fuel shows an increasing trend in Europe. Main reasons are:

- Government incentives/subsidies promoting biomass burning;
- Rising costs of other energy sources;
- Increased public perception that biomass burning is a green option.

From a climate perspective, biomass as a residential fuel has benefits since it is a (nearly) CO_2-neutral source of energy. However, increased use of biomass in residential combustion contributes considerably to adverse effects on human health, particularly in densely populated areas. Emissions take place at low height, and often at locations where air convection is restricted (e.g. in mountain valleys). The emissions make therefore a relatively large contribution to air pollutant concentrations at ground level where people are exposed. The main reasons for high emissions of a number of potentially health hazardous air pollutants such as particulate matter (PM) from stoves installed in homes are:

- The use of old, non-regulated stoves;
- The inadequate maintenance or handling of stoves and firewood;
- The use of non-standardised fuels which hinder an efficient combustion or the use of polluted fuels.

The emission inventories from small combustion installations still suffer from large uncertainties, especially for primary $PM_{2.5}$, benzo(a)pyrene (BaP) and black carbon (BC). These uncertainties result from a broad range of available emission factors (estimates based on measurements) and the statistical information on activity data, i.e. the estimated amount of fuel that people combust in residential stoves. In addition, there are still major gaps in emission inventories with regard to the condensable fraction (i.e., secondary particle formation).

Based on data from the European air quality database 'Airbase' alone, it is not possible to clearly define areas where air pollution is mainly attributable to residential combustion. However, a correlation between BaP emissions and biomass consumption trends is detectable. Still, a direct relationship between BaP emissions and biomass burning cannot be established with the available data.

Contributions from residential sector emissions to ambient air quality have been quantified by scientific studies applying receptor modelling tools, among others. These contributions range from <5% to up to 40% of daily mean PM_{10} and $PM_{2.5}$ concentrations during the heating season.
Modern wood-burning stoves with a high efficiency and low emissions of (carbon containing) PM are becoming more and more available on the market. Such stoves are rather expensive. However, emissions can be mitigated by initiatives that use subsidies to replace old with modern stoves.
Policies at regional, national and European level, which address emissions from residential combustion, have to consider:

- Climate change issues;
- Air pollution issues, and also
- Social inequality issues.

Recent EU legislation such as the regulations under the Ecodesign Directive will play an important role for the development of better stoves and boilers over the next decade. On the fuel side, the promotion of wood pellet certification is one move forward to ensure wood pellet quality, i.e. to prevent the combustion of, for example, chemically-treated recycled wood. On the local scale, in densely populated areas or in valleys with restricted air convection, authorities might consider to ban solid fuel combustion in households.

1 Introduction

The commercial and residential combustion sector has been identified in the majority of European cities as the second-largest contributor to exceedances of PM_{10} and NO_2 limit values (EEA, 2014). The health-related impacts of residential heating, specifically for the case of wood and coal, have been addressed in detail by the World Health Organization (WHO, 2015, 2014a). The impact of the residential combustion sector on air quality is evidenced through pollution episodes such as those recorded in March 2014, when a severe air pollution episode with high PM concentrations occurred over central Europe from the south of the United Kingdom and France, to Belgium, the Netherlands, and Germany. Air quality modelling and analysis of PM samples shed light on the cause: a combination of unfavourable meteorological conditions and various emissions sources, from agricultural to traffic, in addition to residential heating (EEA, 2014). Because of episodes like these, the residential combustion sector is seen as a potential threat to air quality across Europe.

Fuels used in the residential sector are solid, liquid and gaseous. Solid fuel combustion is the oldest source of energy for residential purposes, which is used in a variety of forms and applications such as the production of heat and electricity, as well as for cooking (although less frequently in Europe). Solid fuels include a large group of materials, with coal, brown coal briquettes, smokeless coal (coal which does not produce visible smoke when burned), wood/wood waste (woodchips, pellets, etc.), and charcoal among the most frequently used ones in Europe on the residential scale. However, other types of solid fuels such as biomass waste from forest harvesting and agricultural crops, or garden pruning waste should not be discarded. Liquid fuels with presence in the residential sector are natural gas liquids (NGLs), gasoline, gas/diesel oil, kerosene, liquefied petroleum gases (LPGs), ethanol, among others. Natural gas is the most frequently used gaseous fuel, even if a small presence of biogas can also be found.

The global energy supply is currently dominated by fossil fuels. Coal, oil and natural gas together provide approximately 80% of global primary energy demand. Biomass, as a specific form of solid fuels, contributes with about 10-15% of the global energy demand (Saidur et al., 2011). In developed countries, on average, biomass contributes approximately 9-14 % of the energy supply, while in developing countries the contribution of biomass is much higher and can reach 20-30% (Demirbas et al., 2009; Hoogwijk et al., 2003; Khan A.A. et al., 2009; Saidur et al., 2011). In EU-25 in 2007, half of electricity was produced from fossil fuels, 32% of total power was generated from coal (i.e. hard coal and lignite) although in some countries, such as Poland, this contribution was much higher (>80%) (Kubica et al., 2007). With the goal of providing 20 % of primary energy from renewable sources by 2020, the EC Directive „On the promotion of the use of energy from renewable sources“ (2009/28/EC) (EC, 2009) expects residential solid biomass combustion to increase by 100 % compared to the reference year 2008. In 2007, in Austria about 40 % of solid biomass combustion was related to the residential heating sector, and this share amounted to about 80% in France, 60% in Germany, and almost 40% as the EU average (AEBIOM, 2010; EU UltraLowDust project).

Burning fuels such as wood and coal in residential stoves is an important source of directly emitted fine particulate matter ($PM_{2.5}$) and polycyclic aromatic hydrocarbons (PAH). The domestic use of solid fuel is, particularly in developing countries, an important source of indoor air pollution. Globally (including indoor cooking emissions in developing countries), household air pollution accounted for about 4.3 million premature deaths in 2012 (WHO, 2014a); for Europe this number is estimated at 99.000 (WHO, 2014b). Whereas the use of coal for residential purposes is decreasing in general over Europe (see sections below), there is currently a growing interest in the use of wood/biomass in residential energy production, as a source of renewable energy. This upward trend in biomass consumption is supported by the objective set by the European Union to achieve a minimum 20% share of its gross energy consumption from renewable sources in 2020 (EC, 2009), following climate-oriented policies. However, it has been evidenced that climate-oriented policies may not always work in line with air quality-oriented policies, and vice-versa. Interactions between energy generation and use in the residential sector, air pollution and greenhouse gas (GHG) emissions are complex. It is still a key challenge to explore the feasibility of developing an integrative framework for assessing air pollution and climate change issues linked to solid fuel combustion in the residential sector.

It is well-known that wood (and biomass, in general) is a renewable source of energy, with evident advantages with regard to climate. When combusted efficiently, it may be a (nearly) CO_2-neutral source of energy, even if it does generate emissions of other atmospheric pollutants. However, the use of wood as residential fuel under non-optimal operating conditions (e.g., with non-regulated stoves, non-commercial fuels such as recycled wood, stoves lacking proper maintenance or bad burn practises) entails negative consequences. Contributions of small combustion installations to total air pollutant emissions vary by country and depend on the pollutant type. Non-industrial energy production is applied in small-scale combustion installations (SCIs) with a thermal capacity ≤50 MW, also known as small combustion plants (SCPs). Small combustion plants are used for district heating, for commercial and institutional use, and for residential use (in addition to agriculture, forestry, fishing, and other). Low combustion efficiency (usually not exceeding 50% average per year, in particular for installation capacity below 1MW), poor fuel quality and no or little cleaning of exhaust gasses result in high emissions. Especially the combustion of coal either in residential heating appliances (i.e. stoves, furnaces) or boilers of low power is widely considered to be of the most pollutant emitting sources (Kubica et al., 2007).

Coal and biomass combustion at residential scale or in small-scale combustion plants may generate significant emissions of $PM_{2.5}$, black carbon (BC) or PAHs, thus impacting air quality (EEA, 2014, 2013a). Wood smoke particles contain several toxic organic compounds such as methoxyphenols and carcinogenic PAHs such as BaP (Sarigiannis et al., 2015b; Herich et al., 2014; Kubica et al., 2007; Saffari et al., 2013; and references therein). Epidemiological studies have shown that pollutants originating from solid fuel combustion significantly increase the risk of respiratory disease, chronic obstructive pulmonary disease and cardiovascular disease (Lighty et al., 2000; Glasius et al., 2006). In addition, air pollutants from residential solid fuel burning are emitted at heights that are conducive to high public exposure and can thus have proportionately high human health impacts in densely populated areas in Europe. Many of these areas are also hot spots concerning attainment of EU air quality standards. Impacts of residential combustion on air quality have been detected in indoor air (Canha et al., 2014; Lévesque et al., 2001; Salthammer et al., 2014), even by infiltration from nearby homes and thus affecting indoor air quality in homes without solid fuel stoves (Thatcher et al., 2014). As a result of the health impacts described above, as well as with respect to health equity and climate change, WHO has published new indoor air quality guidelines for household fuel combustion (WHO, 2014a). These guidelines provide health-based recommendations addressing the performance of fuels, and stoves as well as strategies for the effective dissemination of such home energy technologies to protect health.

The recent economic recession in Europe lead to increases in energy prices and contributed to changes in consumer habits, encouraging inter alia the use of biomass as a residential fuel. This resulted in negative impacts on air quality, which have been demonstrated for some European regions (see references in section 4). However, uniquely discriminating the contributions from residential combustion of solid fuels (biomass, but also coal) to air quality degradation is still a complex task addressed by currently ongoing studies across Europe (see sections below).

2 Objectives

This report aims to provide an up to date overview of the contribution of domestic combustion to ambient air pollution and GHG emissions and of the regulatory framework in place to mitigate air pollutant emissions from solid fuel use in domestic households. Its specific objectives are to provide:

- An overview of areas in Europe where air quality data point to air pollution attributable to solid fuel combustion.

- A literature review on contribution of domestic combustion in Europe to air pollution (source allocation) and greenhouse gas emissions.
- A review of legislation and policies in place in Europe controlling emissions from the residential combustion sector (both climate change, energy and air pollution policies, including eco-labelling etc.).

3 Combustion of biomass in domestic stoves

3.1 Physio-chemical aspects of biomass combustion

Particles resulting from combustion can be divided into two categories: primary and secondary. The first category is formed at high temperature in the fireplace and the second in the plume of smoke or in the atmosphere.

Primary particles consist in soot, organic particles and ash. Soot is mainly composed of elemental carbon (EC) and of organic matter (OM). The particles are mainly formed in the rich fuel zone of the flame via complex mechanisms. The available literature related to the formation of the soot particles coming from the combustion of biomass is limited (Obaidullah et al., 2012). During combustion, biomass breaks up into a broad variety of organic compounds with very different characteristics in terms of chemical structure and vapour pressure in particular. The quantity of volatile organic compounds (VOC) formed increases when combustion is incomplete. These compounds can be classified in three families according to their boiling point: very volatile organic compounds (VVOC), volatile organic compounds (VOC) and semi-volatile organic compounds (SVOC). According to the temperature and dilution of the combustion smoke, these last compounds can change phase. Initially present in gaseous form in the flue gas, they can be rapidly transformed into liquid aerosols by condensation or solids by adsorption on fine particles. Ashes are composed mainly of inorganic species (KCl, $K_3Na(SO_4)_2$, K_2SO_4, among others) and unburnt organic particles. The formation of the particles during wood combustion is illustrated in Figure 1.

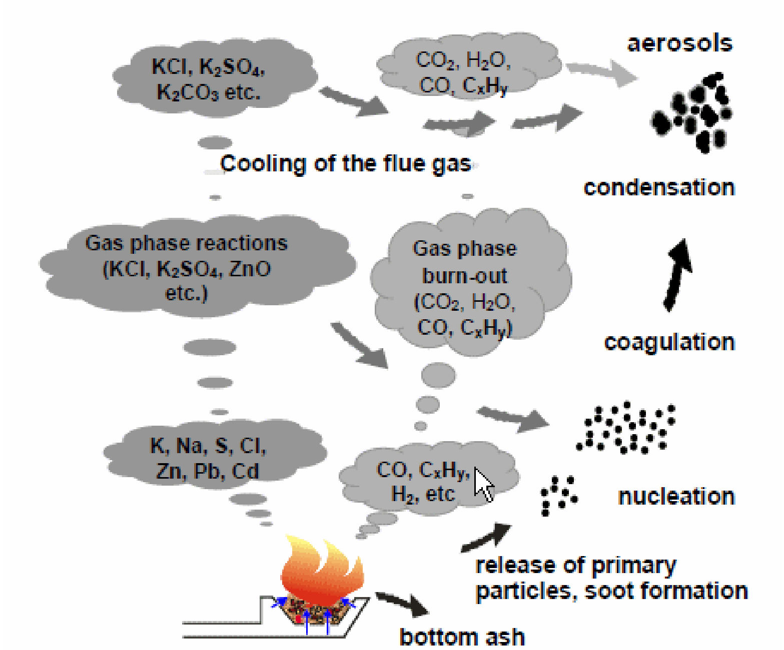

Figure 1. Aerosol formation from wood combustion (Obernberger et al., 2007).

3.2 Mechanisms of (trans)formation of the aerosols resulting from biomass combustion

Transformation mechanisms of the aerosol from biomass combustion are in particular well described by Nussbaumer et al., (2008 and 2010). Solid particles, primarily made up of elemental carbon (EC), organic matter (OM) and inorganic species, initially suspended in the flue gas, can adsorb organic molecules. The behaviour of SVOCs may vary from case to case:

- species with highest molecular weights (mostly SVOCs) may condense in particulate phase forming high condensated secondary organic aerosol (HSOA) when temperature decreases, despite the reduction of their vapour pressure due to dilution,
- species with the lowest molecular weights (mostly VVOCs) may remain in gas phase as a result of reduction of their vapour pressure related to dilution, despite the temperature drop,
- the most reactive species may be oxidised to oxygenated secondary organic aerosols (OSOA), in particular by slow photochemical reactions (photo-oxidation) in the atmosphere or by fast reactions in the plume with oxidants resulting from the combustion process, with formation of oxygenated PAHs (Nalin et al., 2014).

The mass of fine PM in ambient air arising from residential wood combustion can thus be higher than the sum of the solid particles and the condensable compounds initially present in the stack (in particular by addition of oxygen). Moreover, molecules with a higher toxicity than the initial one may be formed.

Immediately after their emission to ambient air, aerosols from residential wood combustion are deeply modified because of the above SOA formation as well as by the important dilution of the combustion smoke and the low temperatures of the ambient air. An illustration of these transformations taking place when flue gas enters in the atmosphere is presented in Figure (in Annex). This Figure describes an example of the evolution of PM_{10} concentrations in the atmosphere corresponding to 2.8 times the mass of solid particles concentrations in the stack, the difference being allotted to aerosols formed by condensation in ambient air of HSOA and to the formation of OSOA.

Box 1: Levels of biomass burning tracers in Southern European cities

Levoglucosan and K are well-known tracers of biomass burning emissions. Dedicated monitoring campaigns were carried out in 5 major Southern European cities (Barcelona, Athens, Florence, Porto and Milan) in the framework of Life project AIRUSE. Results evidenced that levoglucosan mean concentrations varied by one order of magnitude between the cities 22 (Barcelona), 53 (Athens), 287 (Florence), 303 (Milan) and 407 (Porto) ng/m^3 in $PM_{2.5}$, respectively (Figure 2) (AIRUSE, 2015a). In winter, average values of 774, 563, 548, 103 and 32 ng/m^3 were reached in Porto, Florence, Milan, Athens and Barcelona, respectively, reflecting the impact of residential wood combustion on air quality across specific regions of southern Europe. These values are probably underestimated because levoglucosan can be degraded in the atmosphere. It should be noted that in Porto and Milan some peak events were detected also in summer as a consequence of the impact of the emissions of wildfires and/or agricultural fires, although the mean winter value is higher than the summer one. The impact of biomass burning emissions on $PM_{2.5}$ levels is clearly demonstrated by the daily time series of levoglucosan levels. The lowest concentrations of levoglucosan were registered in Barcelona (Figure 2). In Barcelona natural gas is being supplied since 1870 to citizens, helping to achieve that most homes (96% according to the Barcelona City Council) are now heated by natural gas and, in contrast to the other cities, the input from biomass burning emissions is minimal.

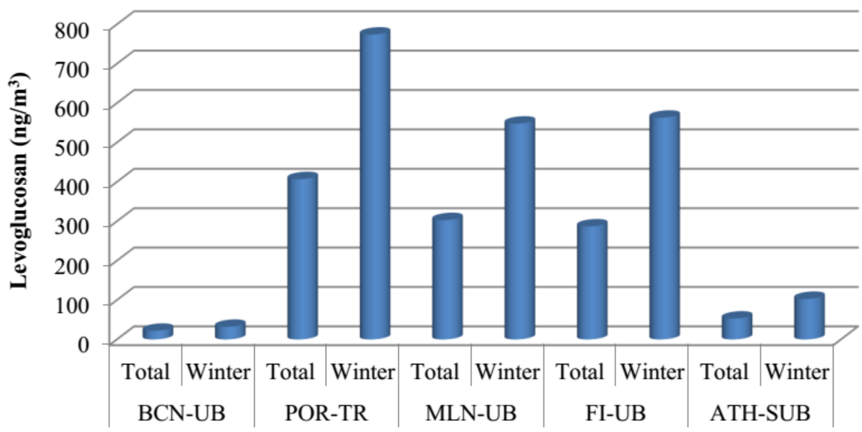

Figure 2. Average levoglucosan concentrations in PM2.5 obtained with results from all samples (1-year long sampling campaign starting in the beginning of 2013) and with winter samples (1 Jan to 21 Mar) (AIRUSE, 2015b).

3.3 Relevance of semi-volatile organic aerosol from wood combustion in emission inventories across Europe

Various studies of dilution of emissions from residential wood combustion show clearly that the semi-volatiles organic fraction is dominating (Shrivastava et al., 2006); more than half of OM emitted during residential wood combustion evaporates at a temperature of 50°C (Grieshop et al., 2009a,b). This condensable organic fraction can be largely reduced by the improvement of the combustion conditions. For larger combustion plants, higher temperature and residence time explains that this fraction has disappeared. The combustion conditions also affect the EC and OC content of PM but less its inorganic matter content. For combustion under sub-optimal conditions, we observe an increase of the more volatile fractions in $PM_{2.5}$, closely related to the increase of OM while black carbon (BC, equivalent to EC in this case) decreases (Albinet et al., 2015). Taking into account the lack of knowledge about physico-chemical processes affecting the particles from residential wood combustion and the time needed for the formation of secondary aerosols, a relation between concentrations of primary aerosol measured in the stack and concentrations in particles measured in ambient air cannot be well established. The characterization of the solid and condensable fractions is thus essential to establish emission factors and is of great interest to qualify the quality of the combustion process.

Results from the AIRUSE Life project showed that the use of secondary PM controls in domestic stoves for residential wood combustion is complex because of the condensation of SVOCs on the surfaces of the control devices, and that a condensation chamber is needed if such devices need to be installed (AIRUSE, 2015b).

A study by Denier van der Gon et al. (2014) discusses the estimates of PM emissions from wood burning across Europe. According to these authors, a new high-resolution (7× 7 km) anthropogenic carbonaceous aerosol emission inventory for Europe was recently developed. The inventory indicated that about half of the total $PM_{2.5}$ emission in Europe is carbonaceous aerosol and identified residential wood combustion as the largest organic aerosol source in Europe. Use of this organic aerosol inventory as input for chemical transport models revealed major underestimations of organic aerosol in winter time, especially for regions dominated by residential wood combustion. In their 2014 work, the authors constructed a revised bottom-up emission inventory for residential wood combustion accounting for the semivolatile components of the emissions. The new emission inventory served as input for the chemical transport models and a substantially improved agreement between measured and predicted organic aerosol was found, improving the model-calculated organic aerosol significantly. This suggests that primary organic aerosol emission inventories need to be revised to include the semivolatile organic aerosol that is formed almost instantaneously due to dilution and cooling of the flue gas or exhaust. The work concluded that the revised residential wood combustion emissions were higher than those in the previous inventory by a factor of 2-3, and additionally showed substantial inter-country variation (Figure 3). The optimised inter-country variation is especially relevant as it should allow improving modelling results in regions such as Eastern Europe with particularly challenging scenarios regarding the widespread use of solid fuels such as low-grade coal or inefficient wood burning (Kiesewetter et al., 2015).

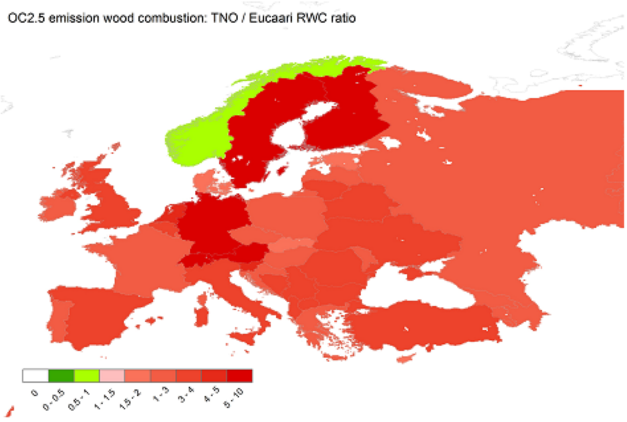

Figure 3. Ratio of the revised TNO-new-residential wood combustion inventory relative to the previous EUCAARI OC emission inventory. RWC: residential wood combustion. Source: Denier van der Gon et al., (2014).

3.4 Technology of new combustion stoves

The extent of secondary air preheating, as well as the quality of the mixing between air and volatised gases in the burnout zone, vary between appliances. Normally, the better the preheating and mixing, the better the burnout will be. In addition, both the temperature and the residence time in the burnout zone must be high enough so complete burnout can occur.

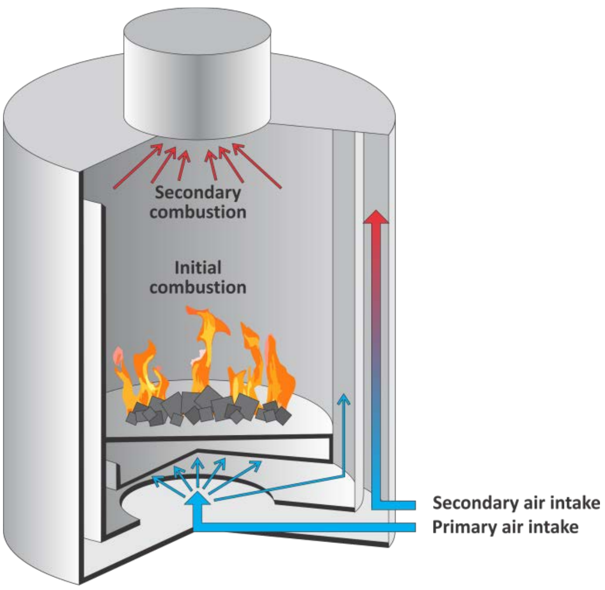

Figure 4. Burning process in a new combustion stove

Complete combustion at high temperatures is one of the key improvements in new stoves that reduces PM emissions and increases the efficiency of the stove. To achieve complete/clean combustions, several factors need to be accounted for sufficient air supply, moisture content of the fuel, and operation by user.

The combustion process needs to have sufficient combustion air and good mixing of air and pyrolysis gases. Intensive burning and feeding of secondary air can support good mixing. An important variable for good combustion is the chimney and the intake of fresh air into the house. The chimney must be able to produce a draft sufficient to supply the stove with the necessary amount of air for the current combustion conditions. On the other hand, fresh air must be allowed to enter the house. This is especially important for new, airtight, low-energy and passive houses. When the chimney or a limitation in air intake is unable to produce the necessary draft, an insufficient air supply is the result. This leads to stove operation under smouldering conditions, resulting in high emissions from wood stoves.

4 Legal framework

Main regulatory policy instruments on air pollution within the EU include the ambient air quality (AQ) directives (EU, 2004 and 2008), and the National Emission Ceilings (NEC) Directive (EU, 2001). The NEC Directive sets upper limits for each Member State for the total emissions in 2010 and each following year of the four pollutants responsible for acidification, eutrophication and ground-level ozone pollution (sulphur dioxide, nitrogen oxides, volatile organic compounds and ammonia). The directive leaves it largely to the Member States to decide which measures – on top of Community legislation for specific source categories - to take in order to comply.

The European AQ directives currently regulating ambient air concentrations of the main pollutants implement limit or target values for ambient concentrations of air pollutants, and they comprise:

- Directive 2008/50/EC on ambient air quality and cleaner air for Europe, which regulates ambient air concentrations of SO_2, NO_2 and other nitrogen oxides, PM10 and PM2.5, Pb, benzene (C_6H_6), carbon monoxide (CO), and O_3 (EU, 2008).
- Directive 2004/107/EC relating to arsenic, cadmium, mercury, nickel and polycyclic aromatic hydrocarbons in ambient air (EU, 2004).

Source-specific legislation is also available for sources such as industrial or road and off-road vehicle emissions, as well as on fuel quality standards (see also the “Policy context and targets” section of the EEA CSI040 indicator (EEA, 2016). The specific legislation for the residential sector is described below.

4.1 Legal framework at EU level

4.1.1 Medium combustion plants

At EU level, combustion installations are classified as “medium combustion plants” (MCP, rated thermal input (RTI) = 1-50 MW_{th}, thermal megawatt), or “small combustion plants” (SCP, <1MW_{th}). Above the 50 MW threshold combustion plants become "large" and subjected to Industrial Emissions Directive - 2010/75/EU (EU, 2010).

The European Commission (EC) published the Clean Air Policy Package on 18 December 2013 (EC, 2013). Within it was included a proposal for a Directive “on the limitation of emissions of certain pollutants into the air from medium combustion plants (MCP)” (COM(2013) 919 final 2013/0442 (COD)), the "MCP directive" (see Box 2). The proposed MCD Directive will impose certain administrative burdens on medium combustion plants, although less stringent than those for large combustion plants (IED directive, EU, 2010).

Box 2. Medium-sized Combustion Plant (MCP) Directive

Medium-sized combustion plants (MCPs) are used for a wide variety of uses, including domestic heating and cooling, electricity generation and providing steam for industrial processes. The approximate number of medium combustion plants in the EU is over 140 000.

Emissions from large combustions plants (>50 MW) are covered by the industrial emissions directive (IED), and there are currently discussions about setting emission standards for the smallest combustion installations (<1MW) in the Ecodesign Directive. As part of its air quality package from December 2013, the European Commission proposed a new directive to limit air pollutant emissions from combustion installations with a thermal input between 1 and 50 megawatts (MW). The new directive sets emission limit values for certain pollutants. These limits will be applied for new and existing combustion plants of medium size (between 1 and 50 MW).

The current proposal covers nearly 143,000 MCPs now in operation in the EU, which in 2010 together emitted some 554 thousand tons (kt) of nitrogen oxides (NOx), 301 kt of sulphur dioxide (SO_2) and 53 kt of particulate matter (PM). In November 2015, the Council of the European Union adopted the new directive (the European Parliament had adopted its position at first reading; EU, 2015).

It needs to be underlined that the proposed MCP Directive does not prevent Member States from maintaining or introducing more stringent protective measures, for example for the purposes to comply with environmental quality standards. In particular, the proposed MCP Directive states that in zones not complying with air quality limit values, more stringent emission limit values should be applied by Member States, such as the benchmark values set out in Annex III to the proposed MCP Directive. This should also promote eco-innovation in the EU, facilitating market access of small and medium enterprises. The Clean Air Policy Package also includes the revision of the National Emission Ceilings Directive (NECD) as one major cornerstone of EU legislation on air pollutant emission control in order to update a proven and effective tool with high rates of compliance among Member States by setting stricter new emission ceilings for the 2020 to 2030 period. The proposed revision of the NECD aims at providing strong general incentives for MS to implement far reaching national emission reduction policies for the main air pollutants regulated in the NEC Directive of 2001 (EU, 2001): sulphur oxides (SO_x), nitrogen oxides (NO_x), ammonia (NH_3) and volatile organic compounds (VOCs) The scope of a revised directive, as proposed by the EC([1]), covers also primary particulate matter (PM) and methane (CH_4) emissions and provisions for black carbon. Since BC and CH_4 are radiative forcer, the latter also helps to mitigate climate change (e.g. IPCC, 2012).

4.1.2 Small combustion plants

Regarding small combustion plants (SCP), there are currently a limited number of policies and instruments implemented on EU-level, related to solid fuel combustion appliances (ULD, 2014). The Ecodesign Directive provides EU-wide rules for improving the environmental performance of energy related products (ERPs) through ecodesign. It prevents disparate national legislations on the environmental performance of these products from becoming obstacles to the intra-EU trade. This aims to benefit both businesses and consumers, by enhancing product quality and environmental

[1] The proposed Directive lays down national emission reduction commitments ("NERCs") for 2020, 2025 and 2030 for each Member State expressed as a percentage reduction of annual emissions of sulphur dioxide (SO2), nitrogen oxides (NOx), ammonia (NH_3), volatile organic compounds other than methane (NMVOC), fine particulate matter ($PM_{2,5}$) and methane (CH_4) as compared to the total of emissions of each of those pollutants discharged by each Member State in 2005.

protection and by facilitating free movement of goods across the EU. The Directive refers to energy related products, which include:

- Energy-using products (EUPs), which use, generate, transfer or measure energy (electricity, gas, fossil fuel), such as boilers, computers, televisions, transformers, industrial fans, industrial furnaces etc.
- Other energy related products (ERPs) which do not use energy but have an impact on energy and can therefore contribute to saving energy, such as windows, insulation material, shower heads, taps etc.

The Ecodesign Directive is a framework Directive: it does not set binding requirements on products by itself, but through implementing measures adopted on a case by case basis for each product group. The list of product groups to be addressed through implementing measures is established in the so-called periodic Working Plan. The Directive compiles a list of Ecodesign Measures for specific types of EUPs, which in the case of residential heating are included in the Commission Regulation (EU) No 813/2013 of 2 August 2013 (EU, 2013), implementing Directive 2009/125/EC of the European Parliament and of the Council with regard to ecodesign requirements for space heaters and combination heaters (EC, 2009)

The expected result of the introduction of new ecodesign requirements is to effectively ban all non-compliant products from being sold in the Member States. This was for example the case of incandescent lamps, for which a gradual phase-out started in the EU in 2009 under this Directive. In the case of residential combustion, the Ecodesign Directive will attempt to reduce emissions (PM, BaP, etc.) from wood stoves and combustion appliances. Recently (on 13 October 2014), proposed requirements for solid fuel boilers were approved by the Regulatory Committee (Lot 15: Solid fuel small combustion installations; Table 1). The proposed ecodesign requirements include solid fuel boilers with a rated heat output of 500 kW or less. In addition, the proposal asks the Commission to assess whether it is appropriate to include other types of equipment, including solid fuel boilers with a rated heat output of up to 1000 kW, by 26 September, 2018.

Table 1. Directives implemented at present on EU-level and generically related to solid fuel heaters (source: ULD, 2014)([2]).

Construction Product Directive (89/106/EEC)	CE-marking only if appliances comply with this Directive
Low Voltage Directive (2006/95/EC)	CE-marking only if appliances containing electrical equipment (50 - 1,000V AC / 75 - 1,500V DC) comply with EN 50165:1997
Machinery Directive (2006/42/EC)	CE-marking only if appliances containing one moving part (“machinery”) comply with this Directive
Pressure Equipment Directive (97/23/EEC)	Only applicable if pressures > 0.5 bar are used
ATEX Directive (94/9/EC)	Directive on equipment and protective systems intended for use in potentially explosive atmospheres (ATEX)

Source: Based on Ecodesign Lot15 Preparatory Study and Wuppertal Institute (2014)

In addition to the Ecodesign Directive, on 19 May 2010, the EU adopted the Directive 2010/30/EU on energy labels. Energy labels help consumers choosing products which save energy and thus money.

[2] ULD stands for the FP7 project “EU-UltraLowDust”, www.ultralowdust.eu, Project Co-Ordinator BIOS Bioenergiesysteme GmbH (Graz, AT).

They also provide incentives for the industry to develop and invest in energy efficient product design. Labels are provided for space and water heaters, and could thus be used for residential combustion appliances.
The combination of the Ecodesign and Energy Labelling Directives aims to regulate the market of new small-scale solid fuel combustion installations (SCI) and to promote Best Available Technologies (BAT). Within the framework of the EU Ecodesign and Energy Labelling process, relevant energy related products put on the market shall be regulated in order to achieve higher energy efficiency levels. For solid fuel heaters, no final regulation has been enacted yet, but it is under preparation. According to the current proposal documents, solid fuel boilers (indirect heaters) will be regulated separately from direct (room) heaters (e.g. stoves). The above mentioned directives sets emission limit values (ELVs) and minimum energy performance standards (MEPS) for different types of appliances, as shown in Tables 2 and 3.

Finally, in order to further promote the best appliances on the market, additional voluntary instruments can be used to supplement mandatory Ecodesign and Energy Labelling. For solid fuel boilers, EU Ecolabel requirements are under preparation in order to provide the basis for an EU-level endorsement label. In addition to energy efficiency or emissions, the EU Ecolabel as voluntary environment endorsement label of the European Commission may also include other environment or health related aspects.

Table 2. Proposed MEPS and ELVs for solid fuel boilers (source: ULD, 2014).

	η_s	PM	OGC	CO	NO_x
	%	mg/m^3 (@ 10 % O_2)			
Biomass solid fuel boilers					
Rated heat output < 20 kW	75	20	10	300	200
Rated heat output > 20 kW	77	20	10	300	200
Fossil solid fuel boilers					
Rated heat output < 20 kW	75	40	10	300	200
Rated heat output > 20 kW	77	40	10	300	200

Please note: ELVs for solid fuel boilers are defined in mg/m^3 @ 10 % O_2, rather than mg/MJ as otherwise throughout this document. For conversion: 1 mg/Nm^3 @ 10 % O_2 ≈ approx. 0.5 mg/MJ (for wood fuels).

Source: Ecodesign ENER Lot 15 - Solid fuel boilers, draft regulation, status of January 2014

Table 3. Proposed MEPS and ELVs for solid fuel local room heaters (source: ULD, 2014).

	η_s	PM	OGC	CO	NO_x
	%	mg/m^3 (@ 13 % O_2)			
Open fronted solid fuel local space heaters	42	40	80	1,500	200
Closed fronted solid fuel local space heaters using solid fuel other than compressed wood in the form of wood pellets	70	40	80	1,500	200
Closed fronted solid fuel local space heaters using compressed wood in the form of wood pellets	79	20	40	250	200
Cookers	70	40	80	1,500	200

Please note: ELVs for solid fuel direct heaters are defined in mg/m^3 @13 % O_2, rather than mg/MJ as otherwise throughout this document. For conversion: 1.0 mg/MJ ≈ 1.5 mg/Nm^3 @13 % O_2 (for wood fuels). If the dilution tunnel method is used for PM measurement, other ELVs have to be applied.

Source: Ecodesign ENER Lot 20 - Solid fuel local room heaters, Draft regulation, January 2014

4.2 Implementation of the Ecodesign Directive

As described in the section 4.1, the Ecodesign Directive is the main directive available at EU level addressing (air pollutants emissions from) small combustion plants. It provides consistent EU-wide rules for improving the environmental performance of energy related products through eco-design, and it prevents the application of different national legislations on the environmental performance of these products. The Directive refers to energy-using products which use, generate, transfer or measure energy (e.g., boilers, computers, televisions, transformers, industrial fans, industrial furnaces etc.), as well as to other energy related products which do not use energy but have an impact on energy and can therefore contribute to saving energy (e.g., windows, insulation material, shower heads, taps etc.).

A Mandate for programming standardisation work in the field of Ecodesign of energy-using products (EuP M/341) has already been issued to the European Standardisation Organizations. In the case of residential combustion, the Ecodesign Directive will attempt to reduce emissions (PM, BaP, etc.) from wood stoves and combustion appliances, specifically with regard to solid fuel boilers with a rated heat output of 500 kW or less, and potentially also those with a rated heat output of up to 1000 kW (the latter, by September 2018). The implementation of the Ecodesign Directive has already shown results for other types of appliances, which may serve as examples of its potential impact on the emissions from wood stoves and appliances. As an example, a study carried out by the University of Coimbra (Almeida et al., 2013) concluded that the Ecodesign requirements for power, distribution and small transformers had positive impacts on:

- cost-effective reduction of transformer electricity losses;
- cumulative reduction of the electricity consumption of about 84 TWh, corresponding to savings of 21 Mton of CO_2 by 2025 compared to the “no action” option;
- although there is an increased purchase cost, it will be largely compensated by savings during the use-phase of the product;
- reduction of the costs by economies of scale for cost-effective technologies;
- correction of market failures and proper functioning of the internal market;
- no significant administrative burdens for manufacturers or retailers;
- fair competition by creation of a level playing field.

Similar outcomes could be expected with regard to wood burning appliances.

In addition to this Directive, Directive 2010/30/EU of the European Parliament and of the Council was issued in May 2010 regarding the indication by labelling and standard product information of the consumption of energy and other resources by energy-related products. EU Ecolabel requirements are under preparation in order to provide the basis for an EU-level endorsement label for solid fuel boilers. In addition to energy efficiency or emissions, the EU Ecolabel as voluntary environment endorsement label of the European Commission may also include other environment or health related aspects. Specific labelling schemes and recommendations regarding fuels and appliances are already available in a number of countries such as Norway (www.nordic-ecolabel.org), Germany (http://www.blauer-engel.de), UK (http://smokecontrol.defra.gov.uk), France (www.flammeverte.org), and Spain (www.cambiatucaldera.com).

4.3 Ecodesign and eco-labelling norms – the national level

Even though the above policies are designed at international or EU level, minimising air pollution and its impacts also requires action at national, regional and local levels. Authorities on those levels may also adopt measures to further protect their populations and the environment.

As regards national regulations for new small combustion installations, only a small group of countries has adopted binding emission limit values (ELVs), which are generally measured according to

European Norm standards (AIRUSE, 2013; ULD, 2014). This group consists of central European countries (Austria, Germany, Belgium) and Nordic countries (Denmark, Sweden, Finland, Norway; Levander and Bodin, 2014). However, national regulations vary widely in terms of which and how strictly emissions like total suspended particles (TSP), carbon monoxide (CO), organic gaseous compounds (OGC) or nitrogen oxides (NOx) are regulated or to which product types regulations apply. Furthermore, almost all regulations include only type testing requirements for new products to be put on the market and no requirements for existing installations in the stock (ULD, 2014). At present, the most comprehensive national emission standard is German law from 2010 (Federal Law Gazette, 2010) (EEA, 2014), which addresses both, new as well as existing small combustion installations regarding their emissions.

For particulate matter (TSP, or “dust”), for direct heaters, there are TSP regulations in Austria, Belgium, Denmark, Germany, and Switzerland. Austrian ELVs are relatively high, while ELVs in Belgium start high but will be cut down to 20-26 mg/MJ fuel burnt in the third tier by 2017. German ELVs are in a similar range in their second tier as of 2015. For indirect heaters, ELVs are set in Austria, Belgium, Czech Republic, Denmark, Germany, Cyprus, and Switzerland. While limits in Belgium are comparatively high with 200 mg/MJ in the first tier today, they will be reduced to half by 2017. As of 2017, national TSP ELVs will be in the range from 66 mg/MJ in Belgium (maximum for boilers) down to 13 or 20 mg/MJ (strictest ELVs for pellet boilers in Germany and Austria) (ULD, 2014).

As mentioned above, emissions from existing combustion installations should also be considered when establishing emission limit values. So far, Germany is the only country in Europe with a regulation that controls explicitly the emissions of existing small combustion installations in the stock. The First Ordinance on the Implementation of the Federal Immission Control Act “1.BImSchV” regulates small and medium-sized firing installations. The 1.BImSchV (amended in 2010) is intended to address the four main issues of small scale solid fuel combustion, which are not sufficiently covered by the product standard relevant type testing procedures: combustion technology, maintenance of the appliances, consumer (mis-)behaviour and fuel quality.

Finally, in addition to mandatory emission limits and efficiency requirements, some countries (or regions) have set up labelling schemes in order to promote small biomass solid fuel combustion technology development through voluntary labelling, mostly with a focus on thermal efficiency (ULD, 2014). Existing national labels include the “Umweltzeichen 37” (Austria) for wood/pellet-fired heaters, the “Blue Angel” (Germany) for pellet stoves and pellet boilers, “DINplus” marking for room heaters and inserts (Germany), “Flamme Verte” (France), “P-marking” (Sweden), the “sign for ecological safety” for boilers (Poland), as well as recommendations provided by the Department for Environment, Food & Rural Affairs (DEFRA, United Kingdom). Transnational labelling schemes include the “Nordic Swan” for slow heat release, stove and insert appliances (Sweden, Denmark, Finland, Norway) or the label of the European Fireplace Association (EFA), which may be applied to solid fuel heaters all over Europe.

Box 3: The example of the Nordic Swan regulations on stoves and boilers' technology

The Nordic Swan is a voluntary eco-labelling system that evaluates the impact of a product on the environment throughout its life cycle. The Swan is the official mark that assures that climate and other environmental obligations have been taken into account. This includes limited emissions of CO_2, particles, carbon monoxide (CO), and organic gaseous carbon (OGC) in the manufacturing and in the functioning of the product. The Swan was introduced by the Nordic Council of Ministers and established in 1989, in an attempt to endorse a more sustainable environment through informed consumer purchases of high quality products and services.

In order for stoves to get the Nordic Ecolabel, they must comply with a high efficiency standard and have low emissions of PM, (CO), and OGC. Furthermore, the instruction manual must contain comprehensive information on operating and maintenance including (Nordic Ecolabelling, 2014[3]):

* information on how various fuel types (types, materials, quality, moisture content) affect output and emissions;
* instructions to the wood's moisture content should not exceed 18%, and that you can buy moisture meter to continuously monitor the proper moisture content. Firewood with a diameter of more than 10 cm and should be split;
* fuel types suitable for the stove, and that fossil fuels should not be used that Nordic Eco-labelled wood pellets should be used in pellet stoves;
* recommendations for the handling and storage of firewood, wood pellets and any other solid biofuels;
* how the stove is lit. For wood-burning stoves and inset fireplaces, a reference to "Top down" lighting method is recommended;
* instructions for filling and the volume and size of firewood on lighting/filling
* instructions for cleaning, inspection and maintenance of the stove and any particle filter; instructions describing the recommended maintenance.

The Nordic Ecolabel is well-known and highly accepted among both producers and consumers in the Nordic countries. Nordic market survey revealed that in Norway, Finland, Sweden, Iceland and Denmark 94 % of the participants recognised the Swan trademark as an Ecolabel (Bodin, S. and T. Levander, 2014). Table 4 shows are the emissions requirements from Eco-labelled stoves.

The Swan criteria for emissions from wood combustion are more stringent than the respective Norwegian, Swedish and Danish national regulations. Norwegian and Danish statutory regulations only require particle tests, and Swedish regulations only require test on hydrocarbons. There are no wood combustion emission requirements regulated by law in Finland. Test methods are also described in the document Nordic Ecolabelling, 2014.[4]

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[3] Nordic Ecolabel. (2014). Nordic Ecolabelling of Stoves. http://www.nordic-ecolabel.org/criteria/product-groups/?p=2

[4] Nordic Ecolabel. (2014). Nordic Ecolabelling of Stoves. http://www.nordic-ecolabel.org/criteria/product-groups/?p=2

Table 4: Threshold values for emissions from Nordic Ecolabelled fireplaces tested with 13% O_2. The requirement applies to a normal load, if not otherwise stated. The fireplace may not exceed the threshold values for organic gaseous carbon (OGC), carbon monoxide (CO) and PM.

Emission Limits			
	OGC	**CO**	**PM**
	mg/m^3	mg/m^3	g/kg
Valid from 1/7-2014 to 30/6-2017 Manually operated stove or insert stove for intermittent use	100	1250	3.0 (mean for up to 4 loads) 6.0 (max for each load)
Valid from 1/7-2017 to 30/6-2019 Manually operated stove or insert stove for intermittent use	100	1250	2.0 (mean for up to 4 loads) 5.0 (max for each load)
	mg/m^3	mg/m^3	mg/m^3
Manually operated heat-accumulating fireplace	100	1250	50
Manually operated sauna stove	150	1700	120
Pellet stove with automatic pellet feed	10	200	15

5 Emissions from the residential sector

5.1 Total GHG emissions from the residential sector

In order to assess the impact of residential combustion on air quality, and it's evolution in recent years, it is first necessary to evaluate the contributions from this sector in terms of emissions and their change over time. To this end, data on greenhouse gas (GHG) emissions from the residential sector (sector 1A4bi according to the Nomenclature for Reporting (NFR) format ([5])) and national totals for each EEA member country for the period 1990-2012 are discussed in this section (EEA, 2014b).

The total GHG emissions can be expressed in carbon dioxide (CO_2) equivalents. The emissions from the residential combustion sector (stationary sources) in the year 2012 are shown in Figure 5([6]). The figure shows an overview of the current status regarding total emissions volume from this sector per country, with the most recently available data. As expected, the data evidence that emissions from the residential sector are strongly linked with population, and as a result the highest emissions are reported by countries such as Germany, United Kingdom, Turkey and France.

[5] Countries must report emission data under the LRTAP Convention using an official reporting template. The latest version of the reporting template is available for download from CEIP as Annex 4 of the UNECE Emission Reporting Guidelines. EU Member States are requested to also use this template when reporting emissions and projections data under the EU's National Emission Ceilings Directive (TFEIP, 2015).

[6] Total GHG emissions in CO_2 equivalents, include here besides CO_2 the gas methane (CH_4), fluorinated gases also called F-gases (including hydrofluorocarbons, HFCs), nitrous oxide (N_2O), perfluorinated compounds (PFCs), and sulphur hexafluoride (SF_6). http://www.eea.europa.eu/publications/european-union-greenhouse-gas-inventory-2014.

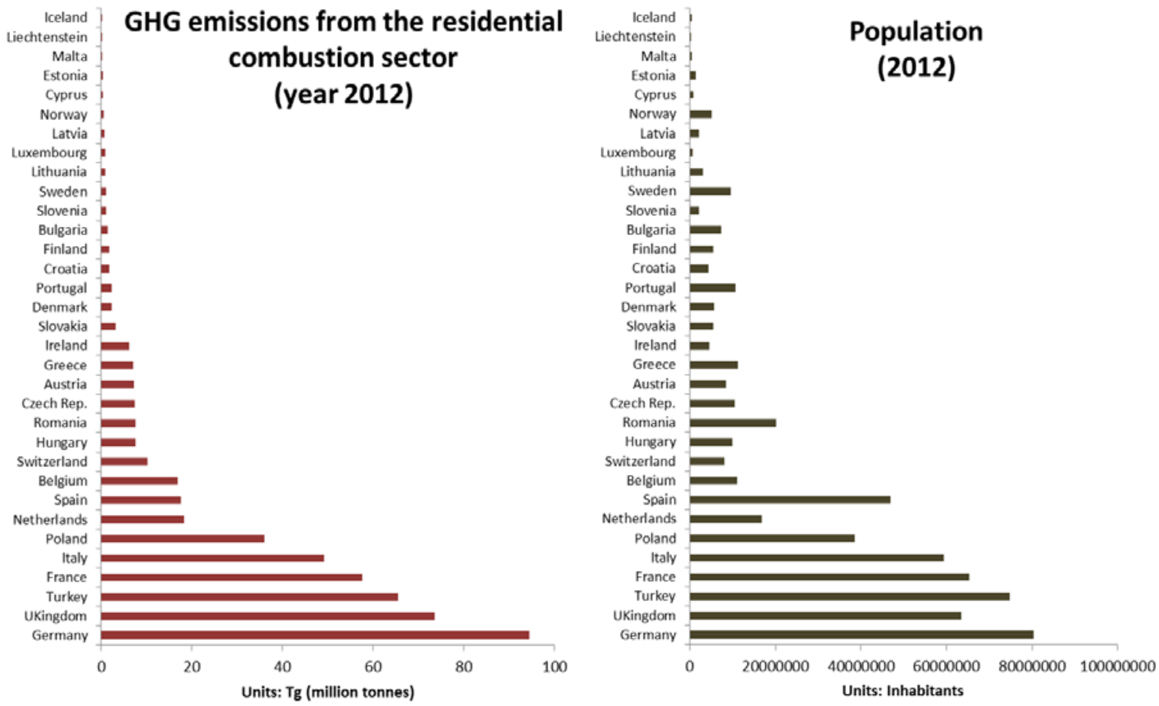

Figure 5. GHG emissions from the residential combustion sector in 2012 (source: EEA). Population in Europe in the year 2012 (source: Eurostat).

On average, for the same year 2012, the contributions from the residential sector to the national totals for EEA member countries was relatively small (Figure 6), accounting for <1% (cases of Iceland or Norway) and up to 20% (in the case of Switzerland).

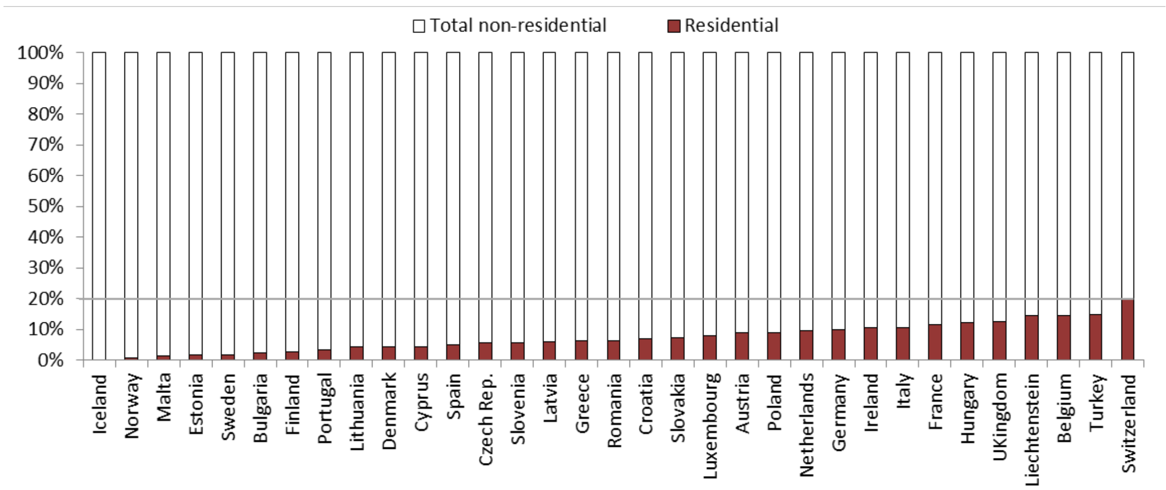

Figure 6. Contributions from the residential sector to national total GHG emission the year 2012.

However, these contributions are not constant across different years, depending on meteorological but also economic and social factors. The relative contribution of the emissions from the residential sector (1.A.4.B) over total emissions (sectors 1-7, excluding 5. LULUCF "Land use, land-use change and forestry"), per country, was calculated for every year between 1990 and 2012. The aim of this assessment was to identify statistically significant trends in the contribution from residential to total emissions, per country. The trend analysis was carried out applying the Mann-Kendall test and Sen's slope estimate (Salmi et al., 2002). Results are shown in Table 5. It should be noted that the significance of trends is probably impacted by gap-filling strategies (mainly, linear interpolation), and that this will especially affect the assessment of specific pollutants such as BaP. More details are provided in the sections below and in the section on "Uncertainties". Therefore, the uncertainty in the trends and their statistical significance for the results shown in Table 5 and thereafter should be taken into account.

Box 4: Major sources of uncertainty

- Incomplete reporting (both in time, e.g., missing years of data, and in space, e.g. missing countries): gap filling strategies are applied, however these distort trends analyses at national or EU scale.
- Emission factors: especially for components such as BaP and BC emission factors are highly uncertain. This is evidenced by the large variations in the emission ratios (e.g., BaP/total PAH or BaP/NOx of total emissions or domestic emission over the years or between countries). This variability is too large to be explained only by difference in fuel use.
- Activity data: especially in forested areas large amount of the wood combusted is not recorded in official statistics.
- Underestimation of emission due to emissions of organics and the condensable fraction (Denier van der Gon et al., 2014).
- Maintenance and usage of stoves.
- Regarding concentration maps: the largest uncertainty is linked to the scarcity of the data, and for BaP/PAH in the unknown volatile fraction (Guerreiro et al., 2014).

This analysis identified statistically significant trends for most of the countries evaluated (22 countries over 33 showing significant trends), at different levels of significance. Both positive and negative trends were detected, with increases ranging between 0.67 and 2.65%/year and decreases ranging between -0.20 and -3.82%/year for the proportion of residential/total emissions, between 1990 and 2012. These results should be interpreted as relative contributions to national totals, and therefore dependent on the remaining emission sectors. Examples of countries showing statistically significant upward and downward trends of residential sector emissions over total emissions are shown in Figure 7.

Table 5. Trend analysis for the relative contribution of residential to total emissions (% residential/ total) over time (1990-2012), for each country. The trend indicates the absolute change (as %) per year. * trend at α= 0.001 level of significance; ** trend at α= 0.01 level of significance; * trend at α= 0.05 level of significance; + trend at α= 0.1 level of significance.**

	Trend (%/year)	Signific.		Trend (%/year)	Signific.
Liechtenstein	2,7	***	**Germany**	-0,5	+
Luxembourg	1,8	**	**Estonia**	-0,5	***
Romania	1,2	***	**Cyprus**	-0,7	*
Greece	1,2	**	**Slovenia**	-0,7	
U.Kingdom	0,7	**	**Norway**	-0,8	***
France	0,3		**Poland**	-0,8	
Croatia	0,2		**Turkey**	-0,8	
Spain	0,1		**Finland**	-1,0	***
Latvia	0,1		**Denmark**	-1,1	***
Italy	-0,1		**Bulgaria**	-1,12	***
Portugal	-0,2	*	**Switzerland**	-1,6	***
Malta	-0,2	***	**Slovakia**	-1,7	***
Netherlands	-0,23	*	**Hungary**	-1,9	***
Belgium	-0,3		**Czech Rep.**	-2,0	***
Iceland	-0,3	***	**Austria**	-2,8	***
Ireland	-0,5		**Sweden**	-3,8	***
Lithuania	-0,5	+			

Note: Uncertainties due to data availability should be considered.

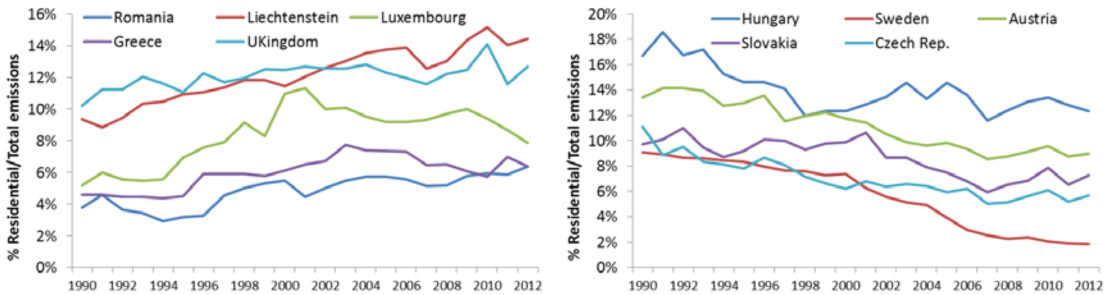

Figure 7. Examples of countries showing statistically significant upward (left) and downward (right) trends of residential sector GHG emissions over total emissions.

Note: Uncertainties due to data availability should be considered.

When analysing the data in absolute terms, both increasing and decreasing trends were also detected. Overall, absolute and relative trends were coincidental for each country, as shown for selected examples in Figure 8. It should be noted that, because the residential sector includes the sum of emissions from different types of fuels, these results may not reflect the specific trends for wood or biomass combustion. A more detailed assessment is necessary for this and is provided in the sections below.

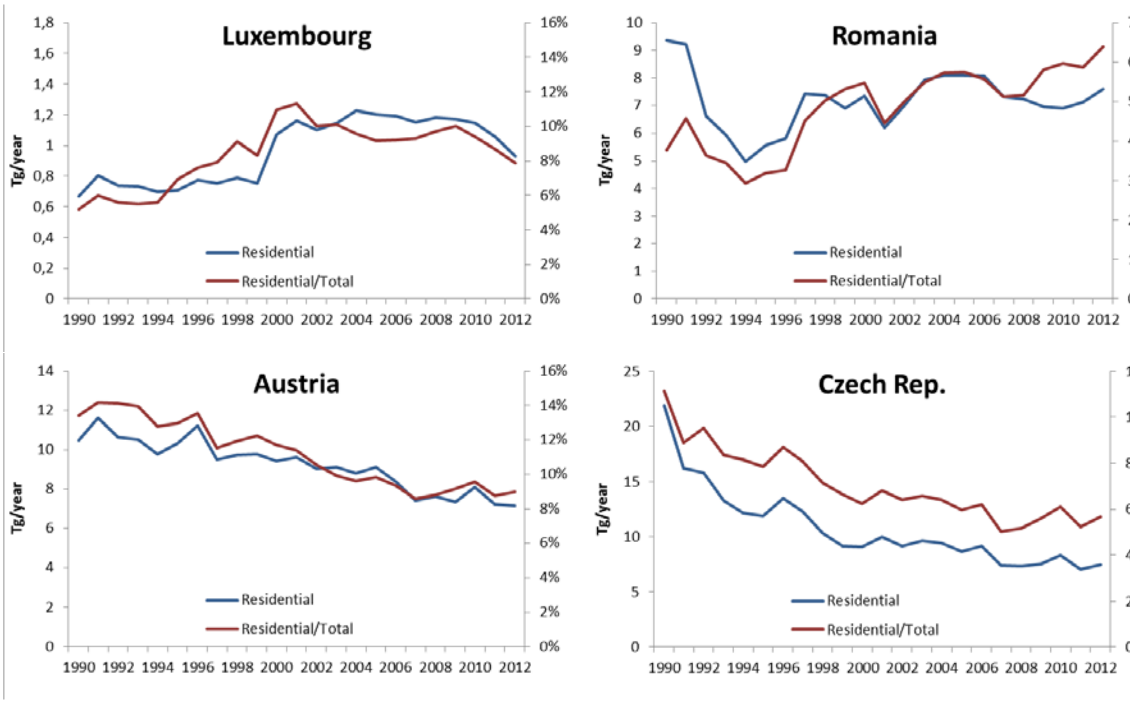

Figure 8. Examples of trends for GHG emissions from the residential sector between 1990 and 2012: results shown in absolute terms (Tg/year) and as the relative contribution to national total emissions (%).

Note: Uncertainties due to data availability should be considered.

5.2 Residential emissions of air quality pollutants

A similar assessment may also be carried out for the emissions for specific atmospheric pollutants. Emission data were collected for the sector 1A4bi, residential stationary sources (source: EEA TR 12/2014 on LRTAP emissions). With the aim to identify areas in Europe where potential changes in ambient concentrations due to changes in domestic heating might occur, the trends in the emission data for the past 10 years were calculated. The results in Table 6 show the slope of such trends together with an indication of the significance of the trend in reported data, for the different pollutants analysed.

Benzo(a)pyrene

Benzo(a)pyrene is a well-known tracer of solid fuel combustion, originating from burning coal and biomass (EEA, 2014, and references therein). Based on the emission data described above, statistically significant upward trends were identified for BaP emissions for Germany, United Kingdom, Cyprus, Lithuania, Poland, Romania and Slovakia, thus suggesting that combustion of these solid fuels in residences may have increased in these countries in the past decade. It should be noted that sufficient data were not available for Austria, Finland, Greece, Italy, Portugal, Spain and Malta. Where possible (shown in brackets in Table 6), BaP emission trends were estimated by assuming a ratio PAH/BaP of 3.3 from other EU countries (range 1.8-5.8). Similar trends were detected for $PM_{2.5}$ coincidentally with BaP in Germany, United Kingdom, Lithuania, Poland, Romania and Slovakia, even if with different degrees of significance. In the case of $PM_{2.5}$, emitted by a larger number of emission sources, solid fuel combustion should be considered as only one of the possible causes of these increased emission rates. As shown in the Table, some countries or regions show different trends for different pollutants, whereas in others data is consistent for pollutants such as BaP and $PM_{2.5}$.

Overall, Figure 9 evidences a mostly upward trend for BaP emissions across Europe, with most countries showing relative increases ranging between 1 and 7% of the national BaP emissions from the residential sector, per year and for the different countries. The highest values were obtained for Romania, where the increase in BaP emissions between 2003 and 2012 accounted for 7% of the annual BaP emissions from the residential sector, with statistical significance (α= 0.05). Statistically significant and relatively high values were also obtained for the United Kingdom and Germany, with 5.3% and 4.3% increase in BaP emissions, respectively. Countries like Poland or Slovakia showed significant increases of around 2%. Conversely, significant decreases in BaP emissions from the residential sector were also detected, as in the case of France (-4.9%). Despite the fact that these trends are statistically significant, it is essential to bear in mind the uncertainty of the input data and its impact on these trend estimates: the slopes obtained do not suggest major changes in emission trends (around 5%), which as a result may be strongly influenced (in relative terms) by the uncertainty of the input data. In addition, these results raise the question of whether it is at all possible to detect concentration trends in ambient air resulting from relatively small changes in emissions.

Table 6. Trends in emission data for 2003-2012, for pollutants from the residential sector: benzo(a)pyrene, $PM_{2.5}$, NOx, SO_2, CO. Signif.: Statistical significance.

	BaP	BaP	PM2.5	PM2.5	NOx	NOx	SO_2	SO_2	CO	CO
	Slope (Gg/yr)	**Signif.**	**Slope (Gg/yr)**	**Signif.**	**Slope (Gg/yr)**	**Signif.**	**Slope (Gg/yr)**	**Signif.**	**Slope (Gg/yr)**	**Signif.**
AT	(-0.1)		-0,1		-0,2	*	-0,6	***	-4,1	
BE	(0.4)		0,8	**	-0,5	**	-1,6	***	3,8	*
DK	0,1		0,3		0,1		-0,1	**	1,8	
FI	(0.4)		0,6	+	0,1		0,1		3,8	+
FR	-0,2	*	-5,8	**	-0,6		-2,6	**	-55,9	*
DE	1,0	+	1,1	+	-1,1		-1,2		34,1	*
GR					-0,2	**	-1,6	***	0,3	
IE	-0,1		0,1		0,1		-0,6	***	-0,3	
IT	(1.7)		2,9	***	-0,4		-1,2	***	39,3	***
LU	0,00		0,0		-0,1	*	-0,0	**	0,1	
NL	0,1		0,0		-0,8	**	0,0		0,6	*
PT	-0.1		-1,01	**	-0,1	**	-0,1	**	-5,2	**
ES	(0.5)		0,6	***	-0,1		-1,1	**	12,3	***
SE	0,1		0,01		-0,2	*	-0,1	***	0,9	
GB	0,1	**	0,6	**	-1,6	*	-0,5		3,4	
BG	0,01		0,3		0,1		-0,4		1,6	
CY	0,1	*	0,0		-0,1		-0,1	**	0,0	
CZ	-0,01		-0,1		-1,2	**	-1,2	+	-5,1	*
EE	0,1		0,3		0,1	+	-0,1	+	2,6	
HU	0,1		0,6		-0,4	+	-0,2		3,6	
LT	0,1	**	0,2	+	0,1	*	0,1		1,4	*
LV	-0,1		-0,4		-0,1		-0,1	+	-1,9	
MT					0,0	+	0,0		0,0	
PL	0,7	**	1,3	**	1,2	*	4,7	*	36,6	+
RO	1,78	*	3,9	*	0,2	+	-0,1		26,7	+
SI	0,1		0,4	***	-0,1		-0,2	***	2,9	***
SK	0,1	*	0,4	*	0,1		-0,3	**	0,2	
HR	0,1		0,1		-0,1		-0,3	**	3,0	

BaP values in brackets: BaP emission trends estimated by assuming a ratio PAH/BaP of 3.3.
Note: Uncertainties due to data availability should be considered.

Fine particulate matter

The fine particulate matter results evidenced maximum increases in $PM_{2.5}$ emissions from the residential sector in countries such as Italy (6.4% of the $PM_{2.5}$ emissions from the residential sector), Belgium (5.5%), Romania (4.6%), Germany (4.4%) or United Kingdom (4.0%). As expected, there is a lack of coincidence between the trends for BaP and $PM_{2.5}$, due to the fact that $PM_{2.5}$ is emitted by a larger number of sources than BaP. Statistically significant downward trends for $PM_{2.5}$ from the residential sector were detected for France (-6.0%) and Portugal (-4.0%). Both the positive and negative changes in $PM_{2.5}$ emissions are in the same order of magnitude as in the case of BaP, and thus the same questions regarding the interpretation of trends would be applicable here.

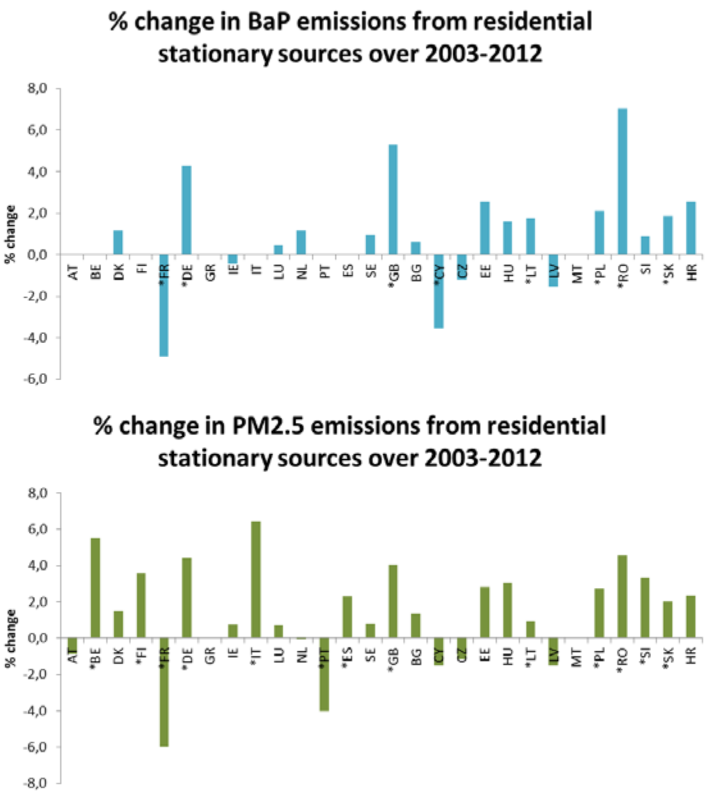

Figure 9. Relative change in BaP and PM2.5 emissions from residential stationary sources over the period 2003-2012. Countries marked with (*) indicate statistical significance of the trend.

Note: Uncertainties due to data availability should be considered.

The results of this analysis carried out for NOx, SO_2 and CO are also shown in Table 6. Contrarily to BaP and primary $PM_{2.5}$, downward trends are significant for SO_2 from the residential sector for most of the countries, and in several cases also for NOx. This is probably related to the marked decrease in coal consumption for residential use registered in recent years, as will be shown in the next section as a result of strategies for fuel conversion in domestic heating such as banning the use of coal. Mean reductions in SO_2 emissions accounted for 7% of residential SO_2 emissions in the different regions, with maxima being reported for Slovenia (-20.7%), Austria (-15.0%), Greece (-12.8%) and Italy (-12.6%). Reductions in NOx are on average lower (-2.8%) across Europe, with maximum reductions in Czech Republic (-12.1%) followed by Netherlands (-6.7%). Regarding CO, most countries show slightly upward trends (on average, 2.3%), in most cases statistically significant. Significant and relatively large decreases in CO emissions were reported by Czech Republic (-6.1%), France (-4.1%), and Portugal (-4.0%). As in the case of $PM_{2.5}$, solid fuel combustion is only one of the potential sources of residential NOx, SO_2 and CO, and therefore direct conclusions on stove emissions may not be extracted from these data.

5.3 Residential fuel consumption as a function of fuel type

Data on types of fuel for residential use in recent years was obtained from GHG inventory data from cross-reference (CRF) files for EU28 Member States as well as for EEA member countries. The detailed formal fuel definitions used internationally can be found in the guidelines of the

Intergovernmental Panel on Climate Change (Table 1.1 in IPCC, 2006). According to these guidelines, in the case of the residential sector, solid fuels include mainly coal and coal briquettes, gaseous fuels refer to natural gas, biomass includes wood/wood waste, charcoal and other primary solid biomass, and liquid fuels are mainly gasoline or diesel oil.

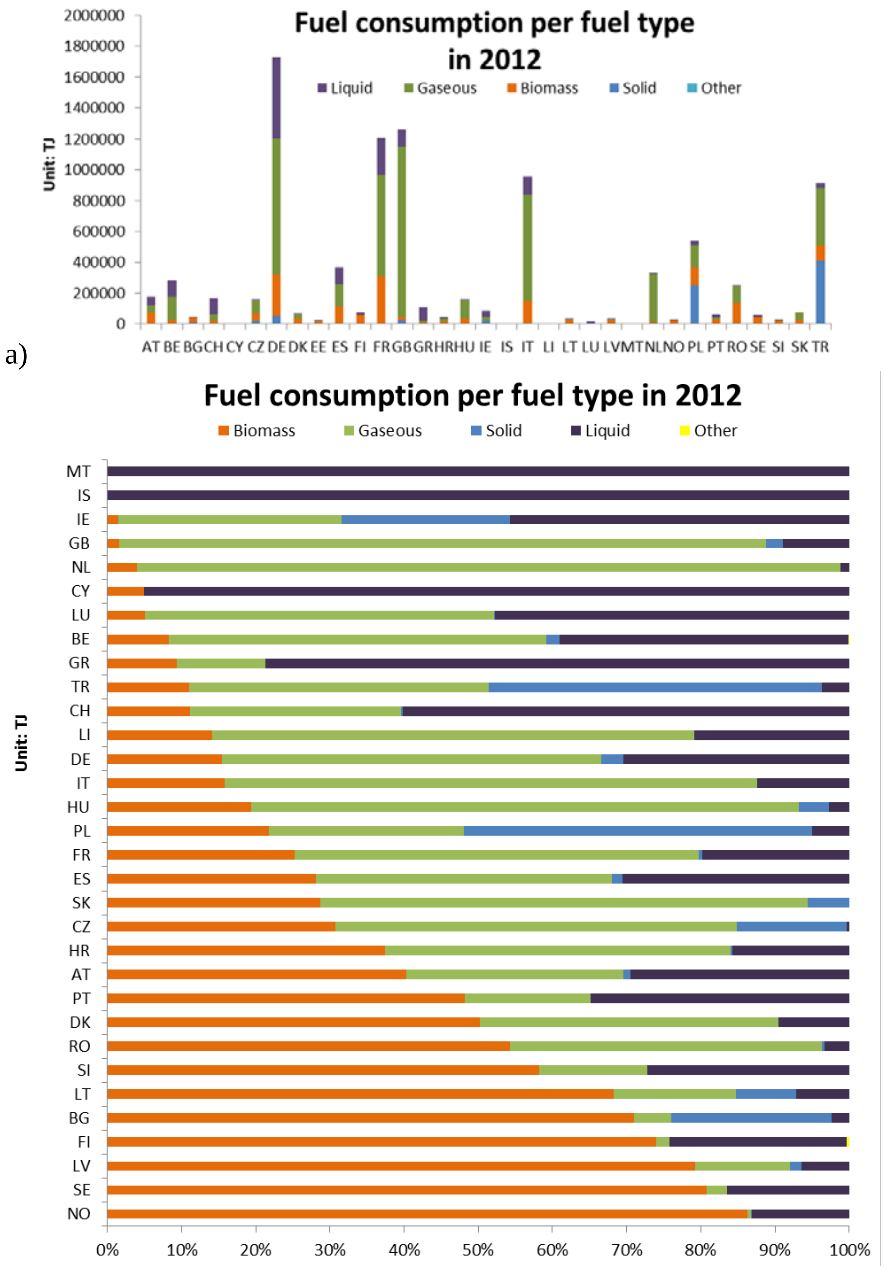

Figure 10. Residential heating. (a) Total fuel consumption (unit: TJ) per fuel type in 2012, per country. (b) Relative fuel consumption per fuel type (in %).

The residential fuel use data provide an overview of fuel consumption habits across European regions, and how these have evolved since the 1990s. In the year 2012, the fuel market in Europe seemed to be dominated by gaseous and biomass fuels, accounting for 34% and 33% respectively of the mean fuel consumed in 2012 for the 33 countries analysed (Figure 10a). These values are of course mean values and vary largely between countries (Figure 10b).

As reported for GHG emissions, fuel consumption habits in the residential sector have undergone changes over time due to economic and social aspects, as well as in response to national or EU policies such as banning or incentivising specific fuel types. In order to detect changes in these habits, trends were calculated for the proportion of each fuel type over all fuels in the residential sector (%fuel X/all fuels, on a TJ basis), for the period 1990-2012. In the case of solid fuels (mainly, coal, Figure 11), out of the 20 countries for which data were available 19 showed statistically significant downward trends, and none showed significant consistently upward trends. These results suggest an overall decrease in coal consumption for residential purposes across Europe, as shown in Figure 11, probably derived from national policies for the ban of this fuel in residences (e.g., the coal ban in Irish cities in 1990). As opposed to this, certain countries such as Turkey show an increase in the past decade and countries such as Poland show smaller decreases with the % of solid fuel over total residential fuels remaining relatively high (45-50% between 2010-2012). This could at least partly account for the high BaP emissions from the residential sector in Poland.

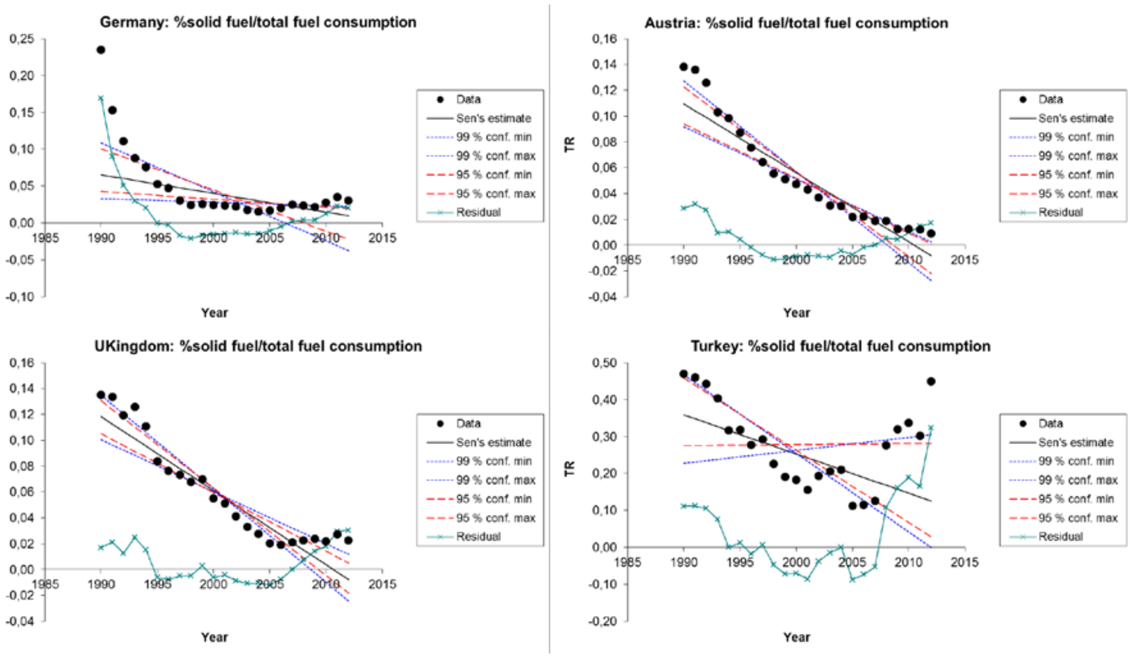

Figure 11. Selected examples of the evolution of the proportion of solid fuel consumption over all fuels in the residential sector (%solid fuel/all fuels), for the period 1990-2012.

The trends obtained for biomass consumption were not as homogeneous as in the case of solid fuels. For most of the countries (19 out of 29 with data available) statistically significant upward trends were detected, indicating a general increase in biomass consumption for residential purposes across Europe. However, downward trends were also detected in other regions, specifically for 8 countries. In certain countries (e.g., Norway, Austria, Denmark or Bulgaria,) the use of biomass at residential scale seemed to be progressively increasing, either slowly or more abruptly at a certain point in time (e.g., in Denmark around 1999; Figure 12). In other countries such as Bulgaria biomass consumption increased at a faster pace until the year 2000, when this increase became more subtle. Quantitatively, it is interesting to note differences such as those registered between Austria, where the % of biomass as residential fuel increased progressively from 30% to almost 40%, and Denmark, where this proportion more than doubled (from 15 to >50%) between 1990 and 2012.

A different scenario was observed for countries such as Spain, Greece, Hungary or Croatia, where a change in habits seems apparent (Figure 13). In these countries the contribution from biomass to all fuels decreased in general since 1990. However, a marked change in the consumption pattern was observed after the year 2005, when the contribution of biomass seemed to increase abruptly. In the case of Spain, as an example, the downward trend ceased around the year 2005, biomass consumption stabilised until approximately 2008, and increased afterwards until 2012. This change in consumption trend could be due to the economic recession (which impacted Spain after 2008), to investments in renewable sources prior to the recession (if the increases occurred prior to 2008) based on the environmentally-friendly perception of biomass, and/or the implementation of local policies such as incentivising the installation of biomass stoves in newly-built or refurbished homes. A similar pattern is observed in Croatia, only delayed by 1-2 years, as well as in Poland, even though the increased proportion of biomass as a fuel in the latter may not be statistically significant. In Hungary or Greece, on the other hand biomass consumption seemed to increase starting in 2005. In France the increase was much more subtle, and was recorded earlier around the year 2000.

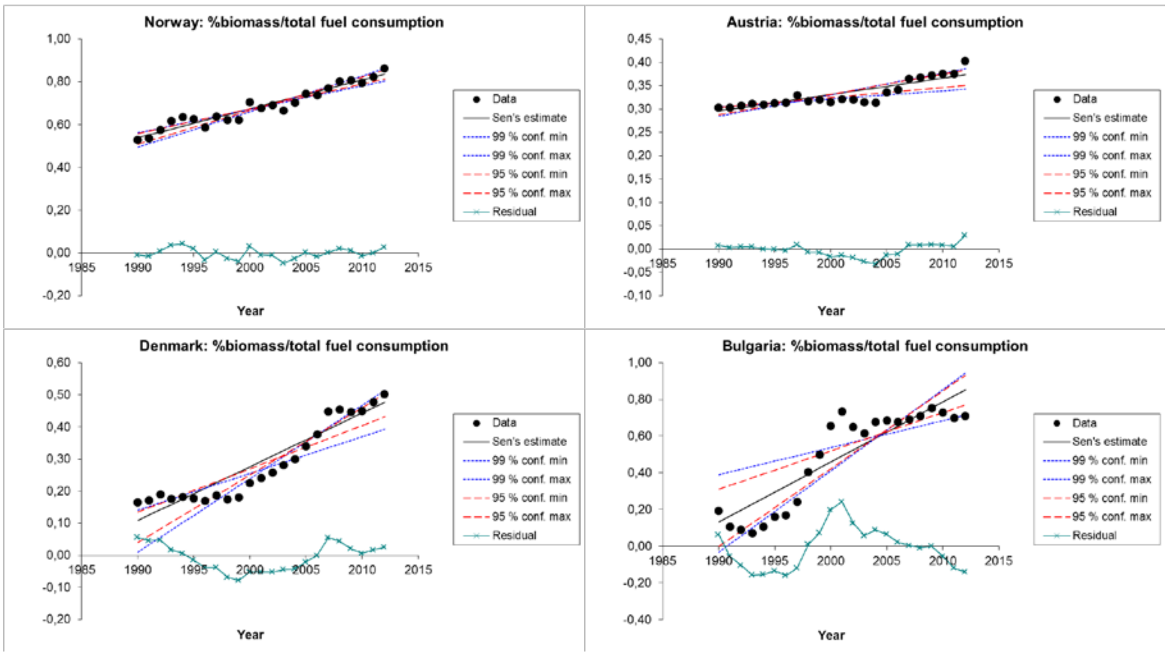

Figure 12. Selected examples of the evolution of the proportion of biomass consumption over all fuels in the residential sector (%biomass/all fuels), for the period 1990-2012, showing consistently increasing trends.

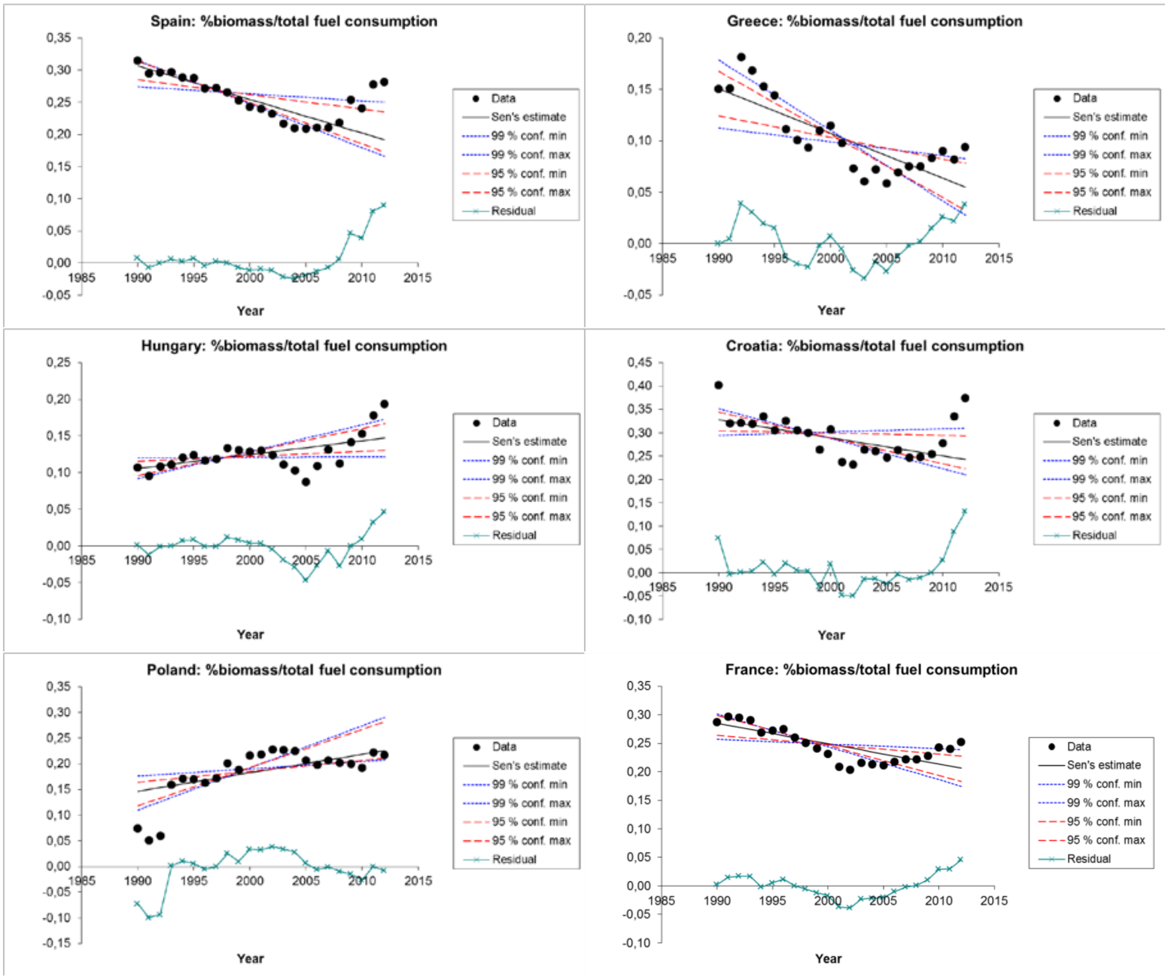

Figure 13. Selected examples of the evolution of the proportion of biomass consumption over all fuels in the residential sector (%biomass/all fuels), for the period 1990-2012, showing both decreasing and increasing trends for the same countries.

Gaseous fuels, mainly natural gas, are dominant across Europe with an overall increase in consumption during the study period. The data reported by the majority of the countries (23 out of 28 countries) followed statistically significant upward trends, whereas only in two cases negative trends were reported. Upward trends were either progressive since 1990 (e.g., in Ireland, Spain or Czech Republic, Figure 14) or marked after a specific year (e.g., Greece, after 2005).

Finally, residential use of liquid fuels generally decreased (in 20 of the 28 countries with data available), following softly decreasing or mostly stable trends (Figure 15a). Consumption only decreased strongly or increased in certain countries (e.g., Netherlands, Figure 15b, and Ireland, Figure 15c, respectively).

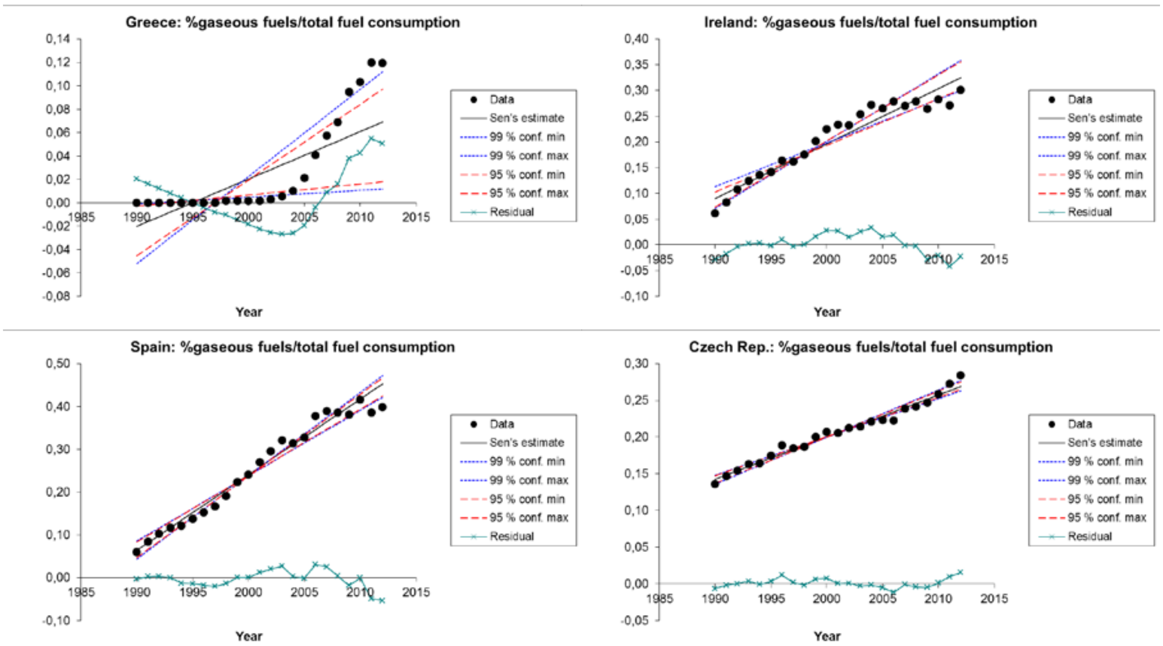

Figure 14. Selected examples of the evolution of the proportion of gaseous consumption over all fuels in the residential sector (%gaseous fuel/all fuels), for the period 1990-2012.

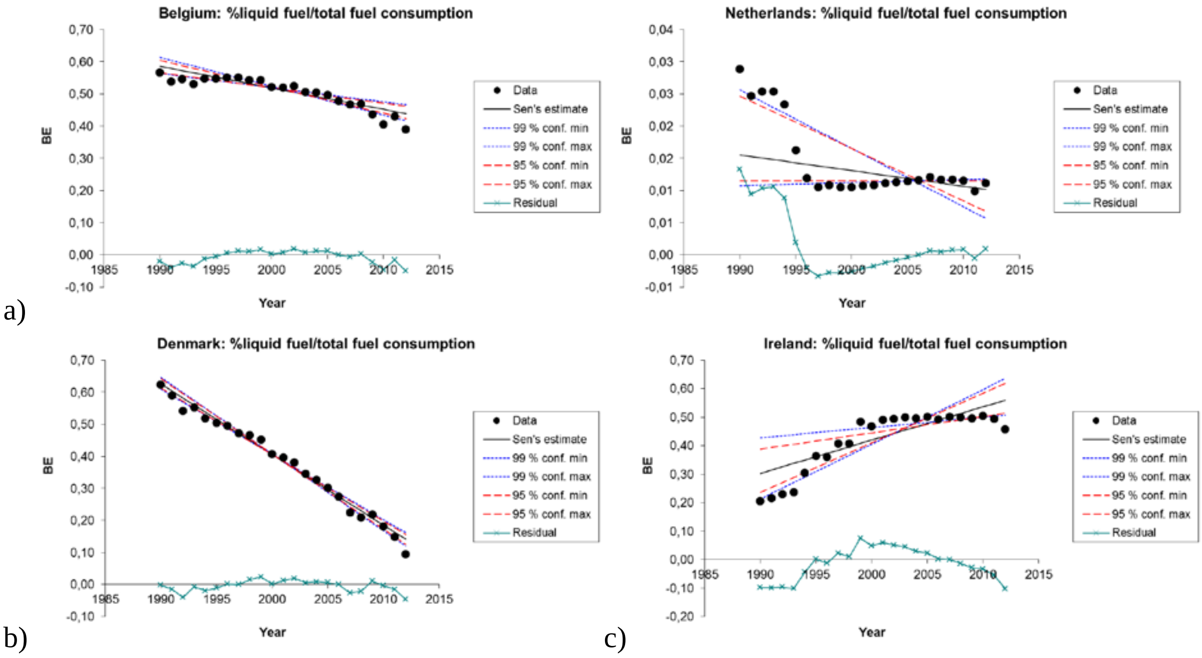

Figure 15. Selected examples of the evolution of the proportion of liquid fuel consumption over all fuels in the residential sector (%liquid fuel/all fuels), for the period 1990-2012.

In summary, with regard to fuel type consumption, the data available suggests that the use of solid (mainly, coal) and liquid fuels (gasoline/diesel oil) for residential purposes generally decreased across Europe between 1990 and 2012, whereas the use of gas and biomass mostly increased. Different patterns were observed in different countries, especially in the case of biomass. For this fuel type a change in trend (from downward to upward) was detected for several countries around the year 2005, probably related to economic, regulatory and social considerations. In addition, a potentially large uncertainty for the biomass emissions inventories should be considered. The combustion of biomass collected from nearby forests and in rural areas may not be officially registered, and therefore the emissions included in the inventories for the case of biomass should be considered the lower-end values. As an example (AIRUSE, 2013), according to a national estimation of the German Federal Environment Agency (UBA), approx. 30,000 tons of firewood were consumed in 2009 in the Berlin region, whereas in total approximately 57,000 tons of solid fuels (wood and coal) were burnt in Berlin's households in the same year.

5.4 Potential impact of fuel consumption habits on atmospheric emissions

The combination of pollutant emission and fuel type trends allows for the assessment of the potential impacts of fuel consumption habits on the kind of atmospheric emissions. This analysis was carried out for solid fuel (mainly, coal) and biomass and with special focus on BaP emissions (Figure 16). The trends obtained suggest a relationship between biomass consumption as residential fuel and BaP emissions from this sector, especially in the last decade when a decrease in solid fuel consumption was recorded. However, if residential biomass combustion were the main source of residentially-produced BaP in the atmosphere, the ratio between biomass consumption (in TJ) and BaP emissions (in Mg) should be relatively constant over time. In addition, this ratio should be mostly constant across countries. As shown in Figure 16, this is not the case. Different reasons may account for this: (a) improving combustion technologies between 1990 and 2012, which would have altered the BaP emission factors; (b) the use of different combustion technologies in different countries, which would also account for different emission factors; and (c) the fact that BaP sources from biomass but also from coal combustion, and thus coal quality and combustion efficiency should also be included in the calculations. As a result, it may be concluded that a correlation between BaP emissions and biomass consumption trends is detectable, even though a direct relationship between BaP emissions and biomass consumption cannot be extracted with the data available.

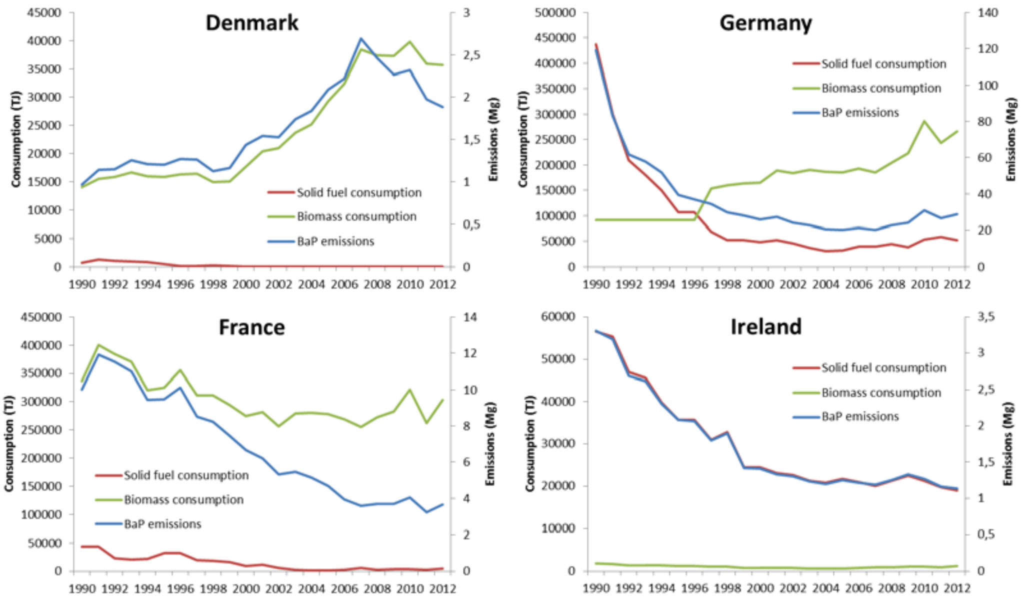

Figure 16 (continued below). Evolution of biomass and solid fuel consumption in the residential sector, as well as BaP emissions from the same sector, for the period 1990-2012.

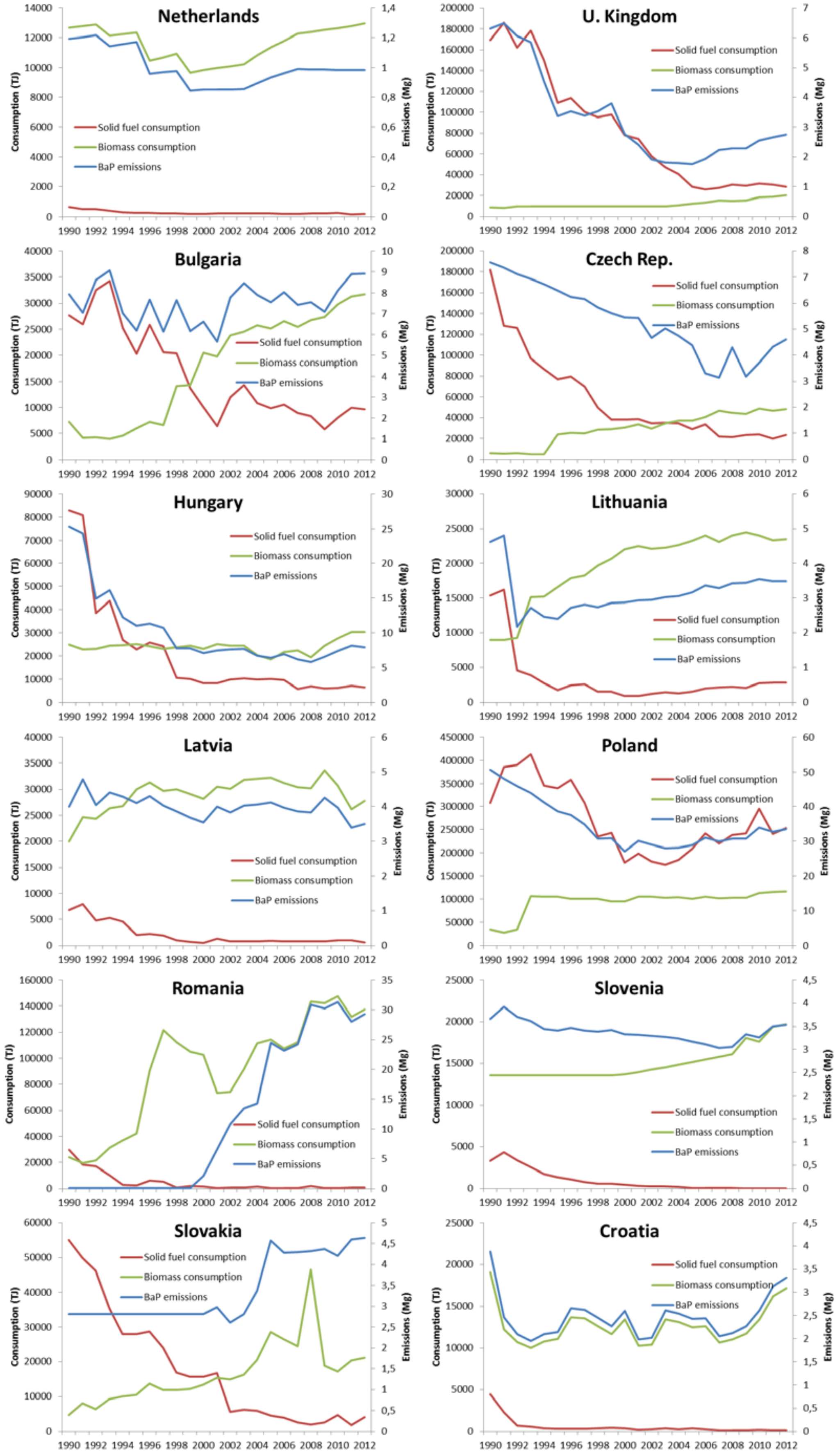

Figure 16 (continued).

6 Impact on air quality

6.1 Impact on $PM_{2.5}$ and BaP concentrations

As discussed in more detail below, wood burning gives a substantial contribution to the ambient concentrations of $PM_{2.5}$ and carbonaceous aerosol. Although source-apportionment of PAH to specific source categories is difficult because the high similarity of the PAH profile of the various categories (Dvorska et al, 2012), one may assume that wood burning contributes greatly to the BaP levels in ambient air. BaP emissions from the residential combustion sector contributes 55-95% to the total BaP emissions. BaP monitoring data suggests a strong contribution from heating emissions. Further, BaP concentrations show a strong annual profile: high concentrations during the heating season (November – February). During the summer period concentrations are very low, caused both by the low emission rates and by an enhanced photochemical removal (Figure 17).

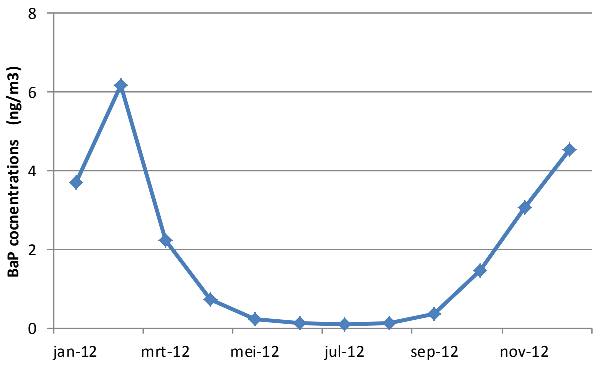

Figure 17. Monthly variations of BaP (in ng/m^3), averaged for all operational stations for which monthly data is available in AirBase, period 2013.

For the protection of human health, both for $PM_{2.5}$ as well as for BaP, reference values have been set by the EU: an annual limit value of 25 µg/m^3 to be met in 2015 for $PM_{2.5}$, and for BaP a target value of 1 ng/m^3. Both values were widely exceeded in 2012.

An interpolated map of $PM_{2.5}$ concentrations (Horalek et al, 2014) is given in Figure 18. High concentrations exceeding the limit value are mainly seen in eastern Europe and in the Po Valley. The map showing the attainment situation in the air quality management zones as reported under the Air Quality Directive gives a similar pattern (Jimmink et al, 2014); about 15% of the EU27 population lives in zones where the $PM_{2.5}$ limit value was exceeded in 2012.

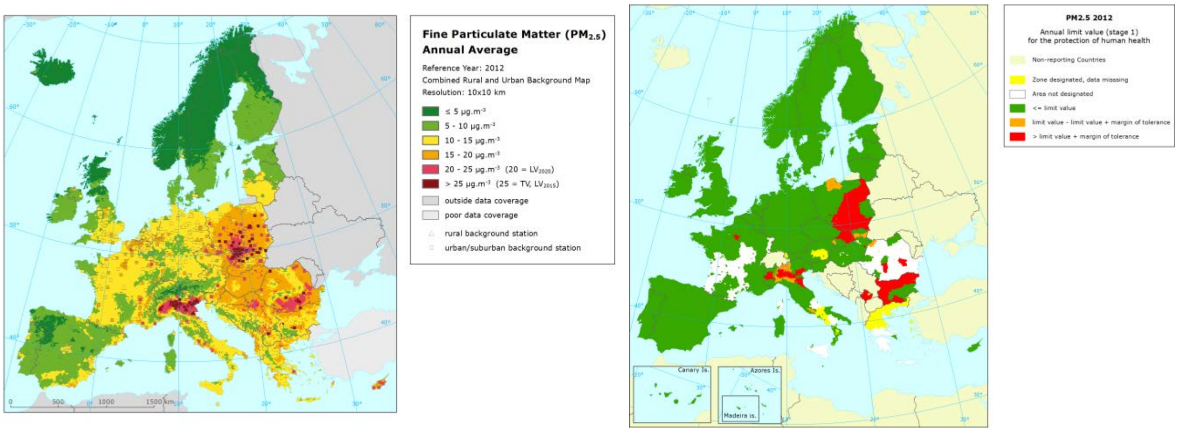

Figure 18. Annual mean $PM_{2.5}$ concentrations (Horalek et al 2014) and the $PM_{2.5}$ attainment situation in air quality management zones (Jimmink et al, 2014).

Similar maps have been prepared for BaP (Figure 19, Jimmink et al, 2014; Guerreiro et al., 2015)). Also here levels in the eastern parts of Europe are high. It should be noted that in large parts of Europe BaP monitoring data is lacking. When estimated concentrations are below 0.4 ng/m^3 EU legislation does no longer require that the assessment is based on regular monitoring data; other assessment tools like modelling or indicative measurements are optional here. The reporting under the Air Quality Directive (Figure 19) shows that more than 20% of the EU27 population lives in zones where the limit value was exceeded in 2012. In view of this and regarding the uncertainties in the model results (largely caused by uncertainties in the emission data) the uncertainty in the BaP map may be considered large. Nevertheless, at the national level the emission data and the population weighted mean concentrations show some correlation. With residential combustion being the dominant source of BaP, emissions take place at a low height and have therefore a much larger impact on the population exposure than for example industrial sources emitting in bigger heights. In Figure 20 the national BaP emission from the residential combustion sector (2012 data) have been correlated with the population weighted mean concentration using the interpolated maps (Figure 20). The correlation between the two parameters is modest (R^2 = 0.40); it should be noted that the high emissions in Germany (28.8 Mg in 2012; Figure 20) and low concentration (0.37 ng/m^3) do not fit in the overall picture. This might illustrate uncertainties in emissions, measurements and modelling results.

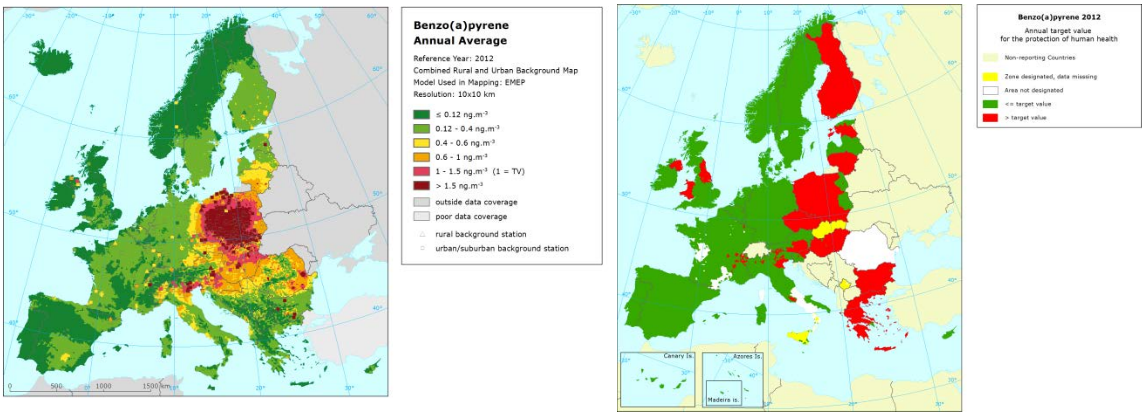

Figure 19. Annual mean concentration map of BaP (2012) interpolation based on AirBase monitoring data and EMEP model results (Guerreiro et al , 2015) Attainment situation in air quality management zones in relation to the BaP target value (Jimmink et al, 2014)

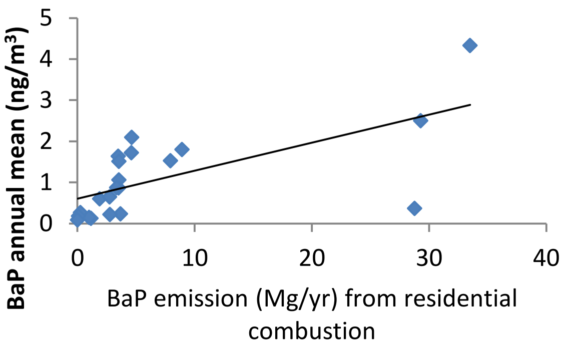

Figure 20. National BaP emissions (Mg/year; reference period 2012) from the residential combustion sector (EEA TR 12/2014), and national population weighted annual means (ng/m3) (Guerreiro et al. 2015).

Aiming at identifying areas where air pollution is mainly attributable to biomass burning, we looked at corresponding trends in emissions and pollutant concentrations in ambient air. Long-term $PM_{2.5}$ and BaP time-series covering a large part of Europe are only available since 2006 and 2007, respectively.

Focusing on urban and rural stations being the most influenced by the residential source, the trend analysis for airborne BaP resulted in a significant trend at a small number of the urban and rural stations (15 of the in total 69 stations where data were available). In 5 of 10 countries that report data, atmospheric concentrations are decreasing. At stations in Bulgaria and Poland extreme but non-significant upward trends of 0.5 (ng/m^3)/yr to more than 1 (ng/m^3)/yr are observed, although it is possible that these are caused by monitoring artefacts given the magnitude of these increments.

Refining the trend analysis by looking only at the heating season was attempted, however, as a number of Member States report BaP data only as an annual average, an analysis could only be made for a few stations in a limited number of countries. In the heating season, significant trends were observed at four stations. There might be several reasons why few concentration trends could be observed: (i) the time series are too short (6 years); (ii) high uncertainty in the observations; (iii) in most of the countries the biomass consumption and BaP emissions show in general only a very small trend. Expected small changes in the concentrations will be obscured by large variations induced by the year-to-year meteorological variability.

Long-term time series available from literature not always show a correlation between trends in national emissions and ambient concentrations. Time series as measured in the Czech Republic (Dvorská et al., 2012) do not show a significant trend during the winter seasons 1996/97 until 2008/09. In contrast, the emission inventory (Fig 12) shows a small but steady increase in biomass consumption. The BaP emissions are strongly decreasing until 2007 and fluctuate very strongly in later years.

Twenty years of PAH measurements in the UK show a significant fall in concentrations (Brown et al., 2013). However, after 2007 a slight but definite increase in concentrations has been observed, and this was most likely related to a change in PAH sampling procedures, according to the authors. This effect is particularly seen at the Northern Irish stations. The increase in BaP emissions since 2002 (Fig 12) is not detectable.

More than 25 years of measurements in the Ruhr area (Bruckmann et al., 2014) show also a significant decrease in BaP concentrations which, however, levelled off since 2004-2005. The increasing biomass consumption and the slower increasing BaP emissions in Germany as a whole (Figure 16) are not reflected in the BaP concentrations in this region.

The analysis of $PM_{2.5}$ levels in ambient air given in the EEA's Air Quality 2014 report (EEA, 2014) indicated that only for a small number of stations a significant trend – in all cases, downwards - can be observed (at 10 of the 82 urban and rural background stations included in the analysis). The set of stations that could be used for an analysis of $PM_{2.5}$ concentrations during the heating season was even smaller, and only three urban background stations (out of a total urban and rural set of 56 stations) showed a significant trend; for all three urban stations the trend is upwards. No further analysis of trends in $PM_{2.5}$ concentrations was done for this working paper.

In summary the conclusion is that - based on Airbase data alone - it is not possible to define areas where air pollution is mainly attributable to residential combustion.

6.2 Impact of wood-burning on PM_{10}, $PM_{2.5}$, BC, OC and EC ambient concentrations

Because of the growing interest in recent years in residential combustion as a source of atmospheric pollutants, numerous studies have been published in the scientific literature aiming to quantify its contribution to air quality degradation. Some of the most recent are Favez et al., 2009; Fuller et al., 2014; Jedynska et al., 2015; Schuck, 2013; Lutz, 2013; Gaeggeler et al., 2008; Caseiro et al., 2009; Sandradewi et al., 2008a, 2008b; Weimer et al., 2009; Canha et al., 2014; Gianini et al., 2013; Herich et al., 2014; Piazzalunga et al., 2013, 2011; Glasius et al., 2006; Hellén et al., 2008; Maenhaut et al., 2012; Kahnt et al., 2013; Saffari et al., 2013; Glasius et al., 2008; Minguillón et al., 2011; van Drooge et al., 2014; van Drooge & Grimalt, 2015; Denier van der Gon et al., 2014. A review work has also recently been published by Life Project AIRUSE (AIRUSE, 2015a, c), focusing on the partner countries Spain, Italy, Greece and Portugal. Several different tools are available to quantitatively assess the contribution from residential combustion to ambient concentrations of PM_{10} and $PM_{2.5}$, organic carbon (OC) and elemental carbon (EC). These tools are mainly based on:

- source apportionment by receptor modelling (e.g., Positive Matrix Factorization, PMF; Multi-linear Engine, ME),
- tracer methods (e.g., levoglucosan; levoglucosan/OC ratios, ^{14}C), and
- models based on optical parameters (aethalometer model) (Favez et al., 2009; Fuller et al., 2014; Hedberg et al., 2006; Maenhaut et al., 2012; Minguillón et al., 2011; Sandradewi et al., 2008a; Szidat et al., 2004; Yttri et al., 2005).

Dispersion models and emissions inventories have also been used, although less frequently, for this purpose (Laupsa et al., 2009; Lutz and Rauterberg-Wulff, 2013).

A non-exhaustive literature review was carried out to identify those studies quantifying the impact of residential combustion (mainly wood, also referred to as biomass in some of the works) on air quality across Europe. The results of the literature review are summarised in Table 7 and Figure 21. For additional studies from Southern Europe please consult AIRUSE (2013, 2015b). As shown in Table 7, the reviewed results refer to a wide range of study areas (from urban to suburban, regional background and rural sites), to different seasons (annual, winter or summer means) and different study years, thus limiting the direct comparability between results. However, this assessment aims to provide an overview of contributions from residential combustion to PM and carbonaceous aerosols, and the results shown are considered sufficient for this purpose. According to analysed studies, residential combustion (mainly wood) accounts for 3-20% of ambient annual mean PM_{10} levels in different European regions, with maximum contributions of up to almost 40% of PM_{10} during the winter period and in specific regions (in this case, in rural areas in the vicinity of Stuttgart, Germany). The highest winter contributions are reported in the Alpine Valleys, the Po Valley, Oslo, Zurich, and rural areas in Austria and Germany. As expected, the lowest contributions are reported for Southern European regions (e.g., Barcelona for the annual mean, 3% of PM_{10}). The case of Madrid in winter is obviously an exception, certainly due to the fact that the study was carried out prior to local measures to reduce coal consumption (Salvador et al., 2012).

In the case of $PM_{2.5}$, residential combustion contributions account for 2% to almost 30% on an annual mean, ranging between 20-30% during the winter heating season. The areas where highest winter contributions were reported are Berlin and Paris suburbs, as well as Oslo. It is important to remember that the studies consulted here refer mainly to wood combustion, and therefore the contributions from combustion of other fuels (e.g., coal, liquid or gaseous fuels) are not included in this summary. In both cases (PM_{10} and $PM_{2.5}$), contributions are in general higher in rural and regional background areas than in urban or suburban ones. This implies that regional-scale wood burning contributions to atmospheric aerosols constitutes an additional source of PM in urban areas, where the local contributions from the residential sector to the PM load is super-imposed over the regionally-transported aerosols from the same source. Overall, several of the studies carried out in urban areas (e.g., Vienna, Berlin, Zurich) report that the PM_{10} or $PM_{2.5}$ from residential combustion sources originates mainly from regional-scale transport, and that only a minor proportion is emitted locally. In the case of Vienna, where wood smoke accounted for 10-15% of the measured PM_{10} levels on winter days with PM_{10} higher than $50 \mu g/m^3$, but with only around 1-1.5% coming from sources within the city.

The data available for organic carbon (OC) and elemental carbon (EC); or black carbon, BC) are scarcer than for PM, but still representative for their respective study areas (Figure 21). In Lombardy (Italy), wood burning contributed to ambient EC concentrations with a maximum of 27% in the years

2004-2005, which increased by 10% in 2006-2007. In Lombardy, this increasing trend was also observed for PM, suggesting a change in fuel consumption habits in this region which was probably related to the promotion of wood as a renewable fuel, and not to the economic recession (given that it was registered earlier). In Zurich, wood burning also accounted for a similar percentage of BC (25%). As for OC, this source accounted for 14-36% of OC on an annual basis in the different regions studied, reaching maximum values of 50% of OC during the winter heating period and especially in central Europe (Austria, Po Valley).

Table 7 (continued below). Non-exhaustive literature review of studies quantifying the impact of residential combustion (mainly wood, also referred to as biomass) on (a) PM_{10}, (b) $PM_{2.5}$, and (c) BC, EC and OC, across Europe.

a) PM_{10}

Study area	Season	Contribution (%)	Reference
Zurich	Winter	21	Giannini et al. 2012
Zurich	Annual	14	Giannini et al. 2012
Milano-Alpine valleys	Winter	30	Larsen B. et al. 2008
Milano-Po Valley (rural)	Winter	25	Larsen B. et al. 2008
Milano	Winter	10	Larsen B. et al. 2008
Madrid	Winter	9	Viana et al. 2010; Moreno et al., 2013
Oslo	Summer	7	Yttri et al. 2009
Oslo	Winter	24	Yttri et al. 2009
Barcelona	Annual	4	Minguillón et al., 2011; Reche et al., 2012; Viana et al., 2013b
London	Winter	9	Fuller et al. 2014
Vienna larger area	Winter	11	Caseiro et al. 2009
Austria rural (Salzburg, Styria)	Winter	21	Caseiro et al. 2009
Belgium (7 sites: UB, suburban, rural)	Annual	13	Maenhaut et al. 2012
Lombardy (6 sites: UB, rural; 2004-2005)	Winter	16	Piazzalunga et al. 2011
Lombardy (6 sites: UB, rural; 2006-2007)	Winter	24	Piazzalunga et al. 2011
Berlin/Brandenburg (urban and background)	Winter	16	Schuck et al. 2013 (refs therein)
Lower Saxony (rural)	Annual	21	Schuck et al. 2013 (refs therein)
Saxony (rural)	Winter	21	Schuck et al. 2013 (refs therein)
Stuttgart (rural; daily max)	Winter	38	Schuck et al. 2013 (refs therein)
NRhein Westphalia	Winter	12	Schuck et al. 2013 (refs therein)
Essen	Annual	8	Schuck et al. 2013 (refs therein)
Switzerland (urban)	Winter	21	Schuck et al. 2013 (refs therein)
Flanders (local hotspots)	Winter	11	Schuck et al. 2013 (refs therein)

Table 7 (continued).

b) $PM_{2.5}$

Study area	Season	Contribution (%)	Reference
Oslo	Annual	28	Laupsa et al. 2009
Berlin suburbs	Winter	30	Lutz M. 2013
Berlín	Annual	4	Lutz M. 2013
Barcelona	Annual	4	Minguillón et al., 2011; Reche et al., 2012; Viana et al., 2013b
Londres (suburbano)	Annual	2	Harrison R.M. (pers. Comm., AIRUSE, 2013)
Oslo	Annual	12	Jedynska et al. 2015
Netherlands	Annual	5	Jedynska et al. 2015
Munich/Augsburg	Annual	9	Jedynska et al. 2015
Catalonia (Spain)	Annual	5	Jedynska et al. 2015
Paris (suburban)	Annual	15	Favez et al. 2009
Paris (suburban)	Winter	30	Favez et al. 2009
Denmark (suburban/rural)	Winter	20	Glasius et al. 2008
Denmark (suburban/rural)	Winter	20	Olesen et al., 2010; 2012

c) EC, BC, OC

Study area	Parameter	Season	Contribution (%)	Reference
Zurich	BC	Winter	25	Herich et al. 2011
Lombardy (6 sites: UB, rural; 2004-2005)	EC	Winter	27	Piazzalunga et al. 2011
Lombardy (6 sites: UB, rural; 2006-2007)	EC	Winter	38	Piazzalunga et al. 2011
Barcelona	OC in PM10, PM2.5 y PM1	Annual	21	Minguillón et al., 2011; Reche et al., 2012
Oslo	OC in PM2.5	Annual	29	Jedynska et al. 2015
Netherlands	OC in PM2.5	Annual	23	Jedynska et al. 2015
Munich/Augsburg	OC in PM2.5	Annual	22	Jedynska et al. 2015
Catalonia (Spain)	OC in PM2.5	Annual	14	Jedynska et al. 2015
Austria regional background	OC in PM10	Winter	50	Caseiro et al. 2009
Belgium (7 sites: UB, suburban, rural)	OC in PM10	Annual	36	Maenhaut et al. 2012
Lombardy (6 sites: UB, rural; 2004-2005)	OC in PM10	Winter	39	Piazzalunga et al. 2011
Lombardy (6 sites: UB, rural; 2006-2007)	OC in PM10	Winter	50	Piazzalunga et al. 2011
Finland (suburban residential)	VOC in PM2.5	Winter	48	Héllen et al. 2008
Spain (rural site in the Pyrenees)	Organic aerosol in $PM_{0.5}$	Warm/cold	1%/84%	Van Drooge & Grimalt, 2015
Spain (rural site in the Pyrenees)	Organic aerosol in $PM_{0.5\text{-}7.2}$	Warm/cold	2%/27%	Van Drooge & Grimalt, 2015

It should be stressed that the list of studies presented here is not exhaustive, and that other studies are available (e.g., Denby et al., 2010; Glasius et al., 2006; Herich et al., 2014; Saarikoski et al., 2007; Saffari et al., 2013; Sillanpää et al., 2005b) where quantitative estimates of residential combustion emissions are provided. However, because they are not presented as a percentage to PM_{10}, $PM_{2.5}$ or OC mass, they were not included among the above for comparison.

Finally, it is well-known that wood is a renewable fuel with clear climatic benefits given that it is considered to be quasi neutral with regard to GHG emissions. However, the literature review presented above evidences that residential combustion of wood has an impact on local and regional-scale air quality (quantified here for PM_{10}, $PM_{2.5}$, BC, EC and OC). There are several reasons for the relatively high emissions of toxic pollutants from residential wood burning. Open fires for burning agricultural or garden waste forms an important source, although in most areas waste burning is no longer allowed during most periods of the year (EEA, 2013b). Also, other reasons are the use of non-regulated stoves, the inadequate maintenance of stoves installed in homes, and/or the use of non-standardised fuels (like treated, painted or not sufficiently dried wood) which hinder an efficient combustion (Kubica et al., 2007; AIRUSE, 2013; see also conclusions from the JOAQUIN Workshop Wood burning, Elemental/Black/Brown Carbon, 2013). However, modern woodstoves with a high efficiency (>75%) and low emissions are becoming more and more available on the market. These modern stoves meet criteria set by eco-labelling initiatives (see e.g. Nordic Ecolable, 2014).

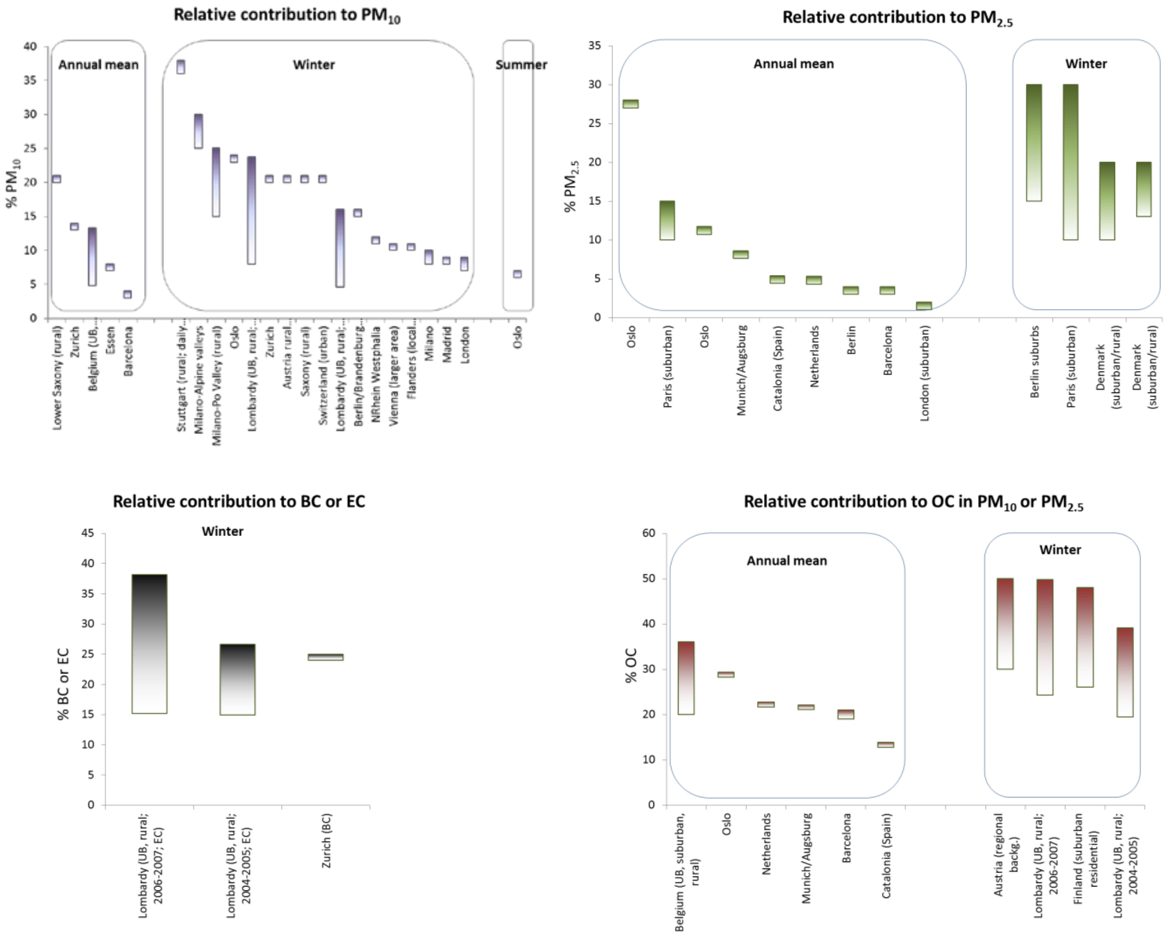

Figure 21. Non-exhaustive literature review of studies quantifying the impact of residential combustion on PM_{10}, $PM_{2.5}$, BC, EC and OC across Europe.

7 Mitigation of emissions from residential wood combustion

The following list on new technologies and upgrade possibilities for reducing black carbon and PM emissions from residential wood combustion was adopted by the latest Arctic Contaminants Action Program (ACAP, 2014).

1. Optimizing combustion process:

Introducing fans for more controlled combustion air supply.

A gasification concept using a large fuel charge, which gasifies in a controlled way producing constant heat output over longer time intervals.

Heat storage solutions. The higher the mass, the lower and more constant the heat output to the room. Storing the heat means higher efficiencies and the ability to burn at nominal effect without overheating the room. Downscaled light steel plate and cast iron stoves, with minimal heat storage and lower nominal effect, can be designed to fit newer building standards for low-energy and passive houses.

Box 5: Overall recommendations by the World Health Organisation (WHO)

WHO proposed in their recent report (WHO, 2015) a set of interventions (technical and non-techical) that decrease emissions, improve outdoor and indoor air quality and improve human health. These covered:

Fuel switching: with examples of the coal ban in Dublin and replacement of wood stoves with electricity in Launceston, Tasmania.

Heater and wood stove exchanges: with examples of successful community wood stove exchange programmes such as one in Libby, Montana, where 95% (n = 1100) of older (not certified by the United States Environmental Protection Agency, EPA) wood stoves were replaced with EPA-certified appliances or other heating sources over the course of four years.

HEPA filtration: discussing that in-home HEPA filtration might reduce health impacts from wood smoke.

Educational campaigns: educational campaigns run at the city, county and national levels can also encourage switching to alternative energy sources and avoiding unnecessary recreational combustion.

The summary of this report states that encouraging fuel switching (away from coal and other solid fuels) and use of more efficient heating technologies (such as certified fireplaces or pellet stoves) can reduce the emissions from residential wood and coal heating devices. Filters may reduce health effects from indoor air pollution. Educational campaigns may also be useful tools to reduce emissions from residential solid fuel heaters (WHO, 2015).

2. More robust wood stove design that prevents end-user interference in its operation.

3. More advanced wood stove solutions:

Wood stoves with enhanced pre-heating and mixing of secondary air with the volatised gases for improved burnout.

Wood stoves with automatic air regulation, using either a fully automatic system with a lambda probe or a mechanical solution regulated using a bi-metal spring.

Wood stoves operating in a quasi-gasification mode, achieved by more or less separating the primary and the secondary combustion zones.

Box 6: Recommendations by the Arctic Contaminants Action Program

The latest Arctic Contaminants Action Program report (ACAP, 2014) discusses estimates on emission reduction based on current knowledge and previous laboratory experiments on the particle emission characteristics of wood stoves. The estimates are derived from dilution tunnel sampling according to the Norwegian standard (NS3058/NS3059). Any efficiency mentioned is calculated using the effective heating value of the fuel (EHV) according to the standard EN 13240.02 (ACAP, 2014). The key findings are listed below:

* Increased knowledge regarding the operation of wood stoves, combined with correct operating techniques, and might reduce PM emissions significantly. The potential is a reduction of between 93 percent and 88 percent (12 to 20 g/kg weighted emissions down to approval emission level, which for the best current stoves is 2–3 g/kg weighted). Lighting the fire from the top using kindling wood in combination with fuel tablets can reduce emissions under cold stove lighting conditions by as much as 30–50 percent.
* An improved efficiency, from 50–65 percent for old stoves and up to 75–85 percent for modern stoves, will reduce PM emissions as less fuel is used to produce the same amount of heat.
* Replacing old stove with modern ones will reduce PM emissions by an estimated 90 percent (from between 20 to 30 g/kg, measured according to the Norwegian Standard, down to between 2 to 3 g/kg today), and most probably by as much as 94 per cent (down to 1–2 g/kg) within the next 5 years. This assumes correct use, preferably top-down lighting and a sufficiently low wood-moisture content. A conservative estimate, assuming stove usage "as before", indicates a reduction in emissions of 70 percent (from 20–30 g/kg down to 6–9 g/kg) and most probably of as much as 82 percent (down to 3–6 g/kg, three times higher than the most optimistic estimate above).

7.1 Primary measures to abate emissions from small combustion installations

In relation to small combustion installations, primary measures of emission reduction include technological activities for reducing primary emissions from incomplete combustion, such as TSP, PM, CO, NMVOC, PAH, PCDD/F as well as heavy metals and SO_2, and NO_x. These actions, preventing or reducing emissions, include several possibilities such as (Kubica et al., 2007):

- pre-cleaning, pre-treatment of raw coals and improvement of their quality to reduce the fine sub-fraction that achieves the reduction of ash content and sulphur content as well as chlorine and mercury;
- modification of the fuels granulation by means of compactification processes, e.g. briquetting, pelletizing, selection of grain size in relation to heating appliances requirements (stove, boilers) and supervision of its distribution;
- replacing of coal by upgraded solid derived fuel, biomass, oil, gas. For example, thermal upgrading of raw coal will reduce the fuel's volatile content (typically from around 35% to around 9%) to produce "smokeless" fuels as briquettes or as coke (the volatile contents is ranged about 2%);
- replacing of high sulphur content oil by high quality liquid fuels;
- Regulating coal quality.
- avoiding the combustion of recycled wood (frequently treated with chemicals), in order to reduce emissions of SO_2 and metals, among others;
- partial replacement of coal with biomass (implementation of co-combustion technologies) so that SO_2, and NOx as well as CO, TSP, VOCs, PAHs emissions are reduced;
- application of combustion modifiers, e.g. catalytic and S-sorbent additives ;
- homogenization and stabilization of the moisture contents in the fuel, especially in the case of solid biofuels;

- replacement of low effective heating appliances with newly designed appliances, and supervision of their distribution by obligatory certification systems;
- implementation of advanced technologies in fire place, stove and boiler constructions (implementation of BAT for combustion techniques and good combustion practice);
- applications of catalytic converters, in particular for biomass combustion appliances;
- combustion process control optimization, mainly in small combustion installations with capacity above $1MW_{th}$.

Increasing of appliance efficiency (for all fuels) leads to decreasing emissions of CO_2 due to improved energy efficiency, i.e. less fuel is needed per energy unit.

7.2 Secondary measures to abate emissions

For small combustion installations secondary measures can be applied to remove emissions, in particular of PM. Simultaneously, emissions of pollutants attached to the PM, such as heavy metals, PAHs and PCDD/F are also reduced. For particulate matter the following options can be considered (Kubica et al., 2007):

- settling chambers; gravity separation (although with low fine dust collection efficiency of about 35%);
- cyclone separators: to achieve high effectiveness of 94-99%, units with multiple cyclones (cyclone batteries) are applied, and multi-cyclones for increased gas flow rates;
- electrostatic precipitators with an efficiency between 99,5% to 99,9%) or fabric filters (with efficiency about 99,9%).

Wood combustion appliances, stoves in particular, can be equipped with a catalyst. When flue gas passes through the catalytic combustor, smoke that otherwise would leave the chimney as dirty, wasted fuel is recirculated and burnt. The catalyst decreases emissions caused by incomplete combustion by reducing the temperature that smoke catches fire at so that it can safely burn while still inside the stove. The catalytic converter is a cellular or honeycomb heat ceramic monolith covered with a very small amount of platinum, rhodium, or combination of these. It is usually placed inside the flue gas channel beyond the main combustion chamber. How efficient a catalyst is in reducing emissions depends on the catalyst material and on its construction (active surface, the conditions of flue gases flow inside converter, temperature, flow pattern, residence time, type of pollutants, etc.). The operation of a catalytic stove is a little more complicated compared to a non-catalytic one because they have a lever-operated catalyst bypass damper which is opened for starting and loading and the stove has to be burned hot before the catalyst is engaged. This means that these stoves are most efficient when you load them full, unlike to the non-catalytic stove. Further, the catalytic element degrades over time and must be replaced; its durability is largely in the hands of the stove user. However, due to the greater fuel capacity and efficiency, the fuel load lasts longer.

As will be discussed in section 3.2, a key issue with regard to emissions and secondary measures are the emissions of condensable compounds, which form particulate pollutants shortly after emission. Condensable compounds are complex to address from an emission reduction point of view. They should be included in emissions inventories in order to more accurately represent the particle emissions from stoves.

Secondary measures with reference to NOx and SO_2 are not applied for small combustion installations due to technical and economical restrictions (Kubica et al., 2007). Because of the significant share of PM and the linked substances, technical methods for their reduction are under development, particularly for small sources with a capacity below $1MW_{th}$.

7.3 Quality of combustion in different appliances

The better the quality of combustion, the lower the emissions. Pellet appliances are in general less emissive than log appliances because of fuel size, air distribution and automatic mode of operation. Emissions of pellet appliances are thus also more stable than log appliances which are characterised by cyclic emissions (ignition, rapid combustion, low combustion and end). Wood-log appliances are more difficult to classify according to their emissions. Emissions can be very variable from one apparatus to another. The European Eranet project Biohealth proposed a classification of combustion quality of appliances according to their levels of pollutants emission (Table). These emission factors take into account solid particles and condensables. Appliances are classified based on particle and gaseous emissions given in Table 8.

Table 8. Classification of different types of combustion.

	Typical emission factors				
Combustion class	PM_1 (mg/MJ)	CO (mg/MJ)	TVOC (mg/MJ)	TC (mg/MJ)	PAHs[7] (ng/mg particles)
Efficient	0-20	0-200	0-5	0-2	0-300
Intermediate	20-50	200-1000	5-50	2-10	300-2000
Incomplete	50-100	1000-2500	50-250	10-30	2000-20000
Smouldering	>100	>2500	>250	>30	>20000

An efficient combustion is currently accessible especially by pellet appliances. Log appliances of last generation are able to reach only an intermediate combustion because they are dependent on the fuel heterogeneity and on their various conditions of use. Some technologies minimise the formation of particles. The improvement of log appliances performances can be obtained by:

- an increase in temperatures in the combustion chamber (introduction of preheated air, insulation of the combustion chamber, limitation of parasite air),
- a better temperature homogeneity and turbulence of gases in the combustion chamber (geometry of the combustion chamber allowing to avoid cold zones or accumulation of the fume, deflector location, air intake location, distribution between primary and secondary air),
- an increase in the residence time of combustion gases at high temperature in the combustion chamber (number and appropriate location of the secondary air intake, deflectors location),
- adequate excess air by adjusting the air flow according to the evolution of combustion. The excess air must be neither too weak (combustion degradation) nor too strong (generation of unburnt compounds).

The amount of secondary air must be sufficient to burn combustion gases. Among all these improvements, the management of primary and secondary air is essential in order to obtain an optimised combustion with low particulate emissions (in particular condensable fraction) and best performances.

Pellet appliances are among the most efficient of the market from an energy as well as an environmental perspective. These appliances make it possible to obtain high heat efficiencies (approximately 90%) due to a good mixture and a good ratio fuel/air, the low size of the pellets facilitating air access, and a fuel automatic loading. The performances of these appliances can be appreciably increased by the use of a good regulation of air. At reduced output, because of air adjustments, the performances of these appliances remain also much better than those of log appliances. However, the performances of pellet appliances are in general very disparate from one apparatus to another according to their design.

[7] 7 PAHs recommended by Directive/2004/107/CE

Furthermore, EU Life project AIRUSE evidenced a very wide range of emission factors for pellet combustion, with the lowest emissions being recorded for EN Plus certified pellets which were in addition much lower than those from other commercial non-certified pellets (Vicente et al., 2015). For some of the latter, high metal emission factors were obtained due to the fact that probably recycled industrial wood (chemically treated) was used for the manufacture of the pellets.

High pollutant emission reduction could however be reached only if wood-burning appliances are used under good conditions. A log appliance of last generation used inappropriately can lead to more important emissions than an old appliance. For that reason, one of the tracks of improvement of log appliances consists in limiting the intervention of the user by adjusting automatically the air flow according to the evolution of combustion. However, it is essential to remember that even under optimal combustion conditions the use of poor quality logs or pellets will still result in high pollutant emissions.

In addition to the performances of the appliance and their mode of use (ignition of the logs from the bottom, operation at reduced output, adapted load), fuel quality (moisture, wood treated) and maintenance (cleaning of the chimney) also play an important role in pollutant emissions.

8 Case studies

8.1 Southern Europe – AIRUSE project

Because of the specificities of the Mediterranean region with respect to climate and combustion fuel types and appliances, a dedicated Life project (AIRUSE) focused on residential combustion practices in Southern Europe (Madrid, Barcelona, Athens, Florence, and Porto). The main results obtained from this project with regard to emission factors from fuels and stoves are described below.

PM emissions from fireplace, traditional cast iron stove, eco-labelled stove and pellet stove were, respectively, in the following ranges: 312-1135, 149-703, 61-156 and 25-156 mg MJ-1 biofuel burned (dry basis). PM emissions from fireplaces were about 3, 12 and 15-fold higher than those from traditional woodstoves, eco-labelled appliances and pellet stoves, respectively. Emissions from the traditional woodstove exceeded 5 to 6 times those of the two more modern combustion devices. However, even the pellet stove, for most of the biofuels, does not meet the emission limits stipulated in countries where certification of combustion appliances is required (e.g. 50 mg PM_{10} MJ-1 in Denmark and Switzerland; 35 mg MJ-1 for wood fuels and 25 mg MJ-1 for pellets in Austria; 27 mg MJ-1 in Germany).

Among all the biofuels tested, only one type of pellets, with EN plus quality certification, complies with the limits. Thus, it is suggested to adopt as much as possible these certification processes, either of pellets or of combustion appliances.

Table 9 summarises the results of AIRUSE experimental averaged results on $PM_{2.5}$ or PM_{10} and BaP emission factors (dry basis) for different combustion devices and biofuels. As for PM, the lowest BaP emission factors were registered for the pellet stove. The lowest emitting wood species generated more than 8 times higher BaP concentrations than pellets. This study shows that flue gas from modern small scale heating systems, such as the eco-labelled stove, could produce elevated BaP emissions, especially during the combustion of conifer logs. These resinous woods are characterised by higher burning rates, which result in very hot flame and short, local drop of oxygen concentration during the combustion. Thus, although a "new" combustion technology contributes to the reduction of overall PM emissions compared with "old" burning appliances, higher combustion temperatures in the modern logwood stoves may lead to higher PAH emissions.

Table 9. Results of AIRUSE experimental averaged results on $PM_{2.5}$ or PM_{10} and benzo[a]pyrene emission factors (mg PM emitted per Megajoule of fuel burnt) for different combustion devices and biomass. nd: not detected.

FIREPLACE							
	Softwood			Hardwood		Briquettes	
mg $PM_{2.5}MJ^{-1}$ biofuel	390			939		767	
µg BaP MJ^{-1} biofuel	14			26		1.7	
TRADITIONAL WOODSTOVE							
	Softwood			Hardwood		Briquettes	
mg $PM_{2.5}$ MJ^{-1} biofuel	202			750		501	
µg BaP MJ^{-1} biofuel	2.6			18		4.7	
ECO-LABELLED STOVE							
	Softwood			Hardwood		Briquettes	
mg PM_{10} MJ^{-1} biofuel	62			114		---	
µg BaP MJ^{-1} biofuel	86			8.1		---	
PELLET STOVE							
	Pellets I	Pellets II	Pellets III	Pellets IV	Olive pit	Shell of pine nuts	Almond shell
mg PM_{10} MJ^{-1} biofuel	27	86	102	76	168	117	112
µg BaP MJ^{-1} biofuel	0.24	nd	nd	0.26	nd	0.92	0.50

Results from several measurement studies, together with disaggregation of emissions factors by technology and fuel type, lead to quite large differences, especially between old-type residential appliances, which dominate in Southern European countries, versus modern woodstoves and boilers with higher combustion efficiency. With regard to human health, emission requirements for the eco-labelling of small-scale combustion appliances for wood logs and pellets must be mandatory in all countries. The requirement for selling only certified pellets should also be widespread. The product testing should be performed by qualified and recognised laboratories. Similarly to what has already been implemented in a few countries, other regulations should be imposed so that aspects such as storage of wood logs or pellets and transportation are taken into account.

It is important to highlight that some woods, wood products and wastes arising from specific industrial sectors may be subjected to phyto-treatments with metals and if these products are directly used as biofuels or as raw material for manufacturing pellets, upon burning, the emission into the atmosphere of heavy metals, including Cu, Cr and As, as well as preservatives (e.g. pentachorophenol) will take place. For this reason, it is very important to regulate and exclude them from the biomass labelling in terms of combustion. It should be noted that this issue is not specific of Sothern Europe. The new EC directive on mid-size combustion plants is a good opportunity for this classification and labelling. In AIRUSE some heavy metals (such as Zn, Pb, Fe and As, Figure 22) were found to be higher in PM10 emissions from some non EN certified pellet types (II, III and IV; Figure 22). These pellets were made of recycled wood products, wood waste and wood residues, especially from the furniture manufacturing industry. The inclusion of wastes from preservative-treated wood could not be discarded.

Given the interest in increasing the use of pellets as a renewable fuel, standards need to be established in the European Union for elemental composition of commercial wood pellets and chips to avoid the inclusion of extraneous materials. Only Germany has standards containing extensive trace element limits. Such standards would prevent the environmental impact of toxic species that would be emitted when the wood is burned. The establishment of enforceable European standards for elemental composition of commercial wood pellets and chips would help exclude inappropriate materials and promote cleaner combustion. Presently, in most countries, it is almost impossible to track beyond the pellet production process to the source of processing materials and to determine if elevated elements result from harvesting practices, use of waste materials, processing impurities or inappropriate handling during production and distribution. Only through the use of programs or regulations that look beyond the current pellet standards will the European Union be able to assure the use of clean wood pellets. Inappropriate storing conditions of the raw material, poor handling concepts, use of waste woods and lack of standards for raw materials as well as binders are probable contributors to product quality issues. Certain elements may still remain in elevated concentrations in the combustion equipment bottom ashes. Thus, further analysis of clean pellets should be accomplished to support the development of policies to guarantee appropriate use of wood ash for soil supplementation or solid waste disposal.

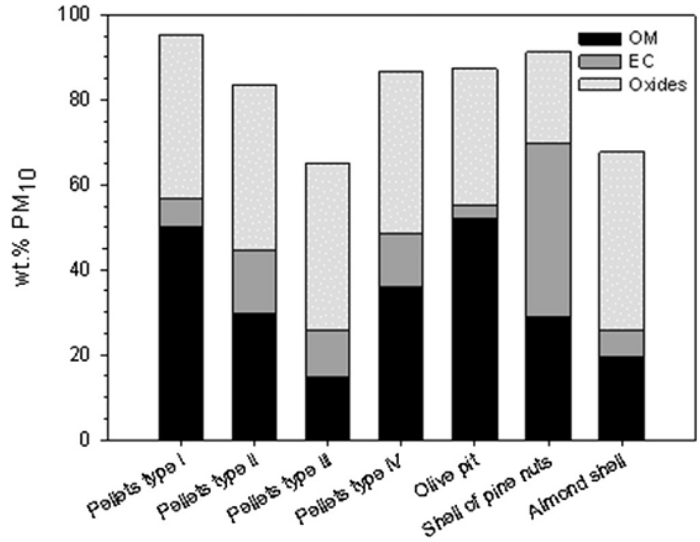

Figure 22. Chemical composition of PM10 (sum of chemical constituents = OC x 1.8 + EC + Oxides), for the combustion of the distinct biofuels (Vicente et al., 2015).

8.2 Norway

In Norway, wood fuel is the second most important source of energy for household heating, after electricity (which in Norway is primarily hydro). Its consumption has shown a stable trend until 2012. In the last two years, wood fuel consumption has dropped, which might be due to relatively mild winters It can also be seen in Figure 23 that within the last decade, fuel wood consumption used in ovens produced after 1998 increased from 34 to 55 percent of the total wood consumption, whereas, there was a decline in wood consumption used in ovens produced before 1998 from 62 to 40 percent of the total from 2005 to 2014.

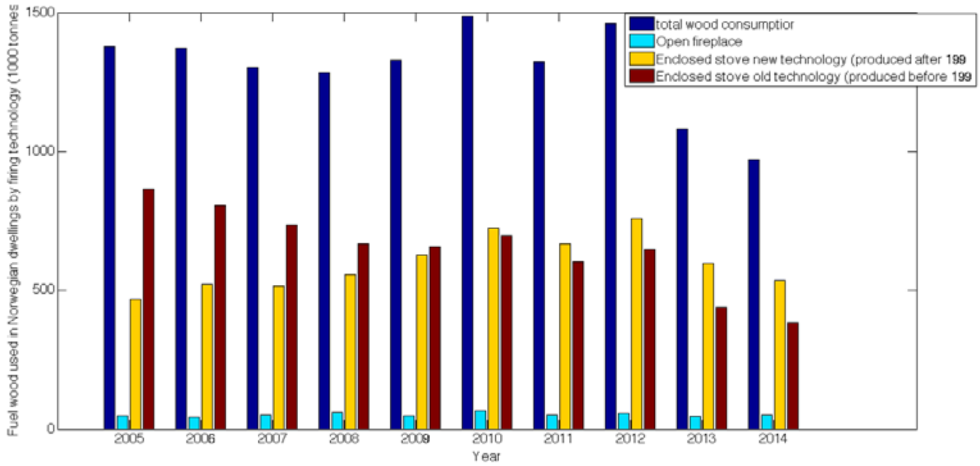

Figure 23. Yearly evolution of fuel wood used in Norwegian dwellings from 2005 – 2014 consumed in stoves with different combustion technology[8].

This strong reduction in fuel wood use in older stoves is believed to be related to the fact that Norway has engaged in more extensive efforts to encourage replacement of old stoves through regulatory and economic instruments, scrapping payment and subsidy programs, as explained in the next sections.

In Norway, the development of technical solutions to reduce emissions from wood stoves is driven by the need to cut PM emissions. To support that process, Norway introduced an emission limit for new stoves in 1998. Since then, enclosed wood heaters must be approved for sale and use in Norway according to Norwegian standard NS 3058([9]). The Norwegian standard NS 3058 defines the test facility, fuel and heating patterns and provides norms for the determination of particles sampled in a dilution tunnel. The norm consists of four tests at different burn rates, which shall reflect real-world firing habits in Norway and take into account that stoves can handle a range of burn rates without compromising particle emission. The stoves and fireplaces have to meet the emissions requirements described in NS 3059([9]) (for more detail see section 2.3). This standard requires an average emission of less than 10 g of PM per kg dry wood in four test runs and has an upper threshold of 20 g/kg for an individual test run. However, those current technical regulations and emission limit values in Norway are out-dated and do not provide motivation for manufacturers to further reduce emissions. Considering the fact that producers of wood stoves act on the global market, they often design their products to meet the most stringed standards. It is possible to develop new technologies with both higher efficiency and reduced emissions as seen in Nordic Ecolabel Standard (see Box 6).

8.2.1 Norwegian regulations on wood stoves technology

Iron stoves/metal stoves are made of sheet metal or cast iron with a closed fire chamber, a refractory brick base and adjustable air control. In Norway, a distinction is made between old stoves, produced before 1998, and new stoves, produced after 1998. This distinction is due to the regulation on emission limits for new stoves, which was introduced in 1998. The main difference between the old and new stoves is that all new appliances for wood log combustion are based on preheating and optimised

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[8]http://statbank.ssb.no/statistikkbanken/Default_FR.asp?PXSid=0&nvl=true&PLanguage=1&tilside=selectvarval/define.asp&Tabellid=09703

[9] NORSK STANDARD, NS-3058 (1994), Enclosed wood heaters, Smoke emission

feeding of secondary air, as well as insulation of the primary combustion chamber and sometimes partly the secondary burnout zone (see Figure 23).

Box 6. Instruments to reduce emissions from residential combustion in Norway

Regulatory instruments: In 1998 Norway introduced a requirement that all stoves to be sold in the Norwegian market should emit no more than 10 grams of particles per kilogram of wood burned. The producers of wood stoves are obliged to pay for testing and approval of their products, in accordance to the Norwegian standard NS3058/ NS3059.

Economic instruments: Enova is a public enterprise owned by the Norwegian Ministry of Petroleum and Energy with the objective of driving the transition to more environmentally friendly energy consumption and production in Norway. Financial support is given by Enova for the installation of more environmentally friendly heating solutions and better insulation in private households (e.g. balanced ventilation, heat control, heat recovery, heat pumps and replacement of old stoves). In addition support is given for Industry (e.g. for energy efficient buildings, environmentally friendly restructuring of energy use and production, renewable heating for a more climate-friendly use of energy, etc.).

The financing of the business is through funds allocated from the Energy Fund, which in turn is financed through a small charge levy on electricity bills. In addition, the Energy Fund administers the return of the "Fund for climate change, renewable energy and energy conversion". This fund is per January 2014 of NOK 40 billion (3.5 billion euro) and will be strengthened by 5 billion (400 million euro) in 2015 and 5 billion (400 million euro) in 2016.

Key parameters of intensive burning are the moisture content of the logs and the size of the logs. The lower the moisture and the smaller the logs, the faster drying during combustion occur and the hotter they burn. In Norway log size and moisture content of the log is regulated in the Norwegian quality standard for fuel wood NS4414. The allowed moisture content varies between 20-25% depending on the wood type.

Considering the fact that most of the residential wood combustion devices are manually operated, emissions of PM and BC are greatly affected by end-user operation. Key factors relating to operation and PM emissions are start-up (ignition) procedure, fuel load, burn rate and air supply.

To avoid the end-user operation of manual stoves, the most advanced models of wood stoves are equipped with automatic fuel feeding systems and controlled air supply. At the moment, only a small number of automatic wood stove solutions exist on the market. They are produced by a Danish company (mechanical stove, electronically controlled stove). The mechanical stove, with its heat sensitive spring, ensures the correct air supply for the different combustion phases enabling an optimal combustion. The electronically controlled stove is a brand new solution for automatic combustion control. The user lights the fire as usual and uses the included remote control to adjust the room temperature as desired. The remote control gives a sound when it is time to reload the stove. The device operates by measuring both the temperature and the oxygen content in the flue gas that regulates an integrated air box, providing an optimum predefined amount of air at all times (ACAP, 2014). The PM emissions from the mechanical and the electronically controlled stove should be on average less than 4g/kg dry wood, as the products conform to the Nordic Ecolabel standard (The Nordic Swan; see Box 6).

Reduction of PM emissions through new technology and improved operation

The estimates on the reduction of PM emissions through new technology and improved operation shows a strong reduction. However, as a recent study from Seljeskog et al., 2013 indicates, the estimates from the Arctic Contaminants Action Program (see Box 6) might be optimistic. Seljeskog et al., 2013 performed wood stove experiments using representative Norwegian stoves with both old and new combustion technology in a way that more closely represents actual use. The experiment used the Norwegian standard NS 3058. This norm consists of four tests at different burn rates, which shall reflect real-world firing habits in Norway and take into account that stoves can handle a range of burn rates without compromising particle emission.

The aforementioned experiment resulted in emission factors for new wood-burning stoves (produced after 1998) for total suspended particulate matter (TSP) of 12.2 and 13.4 (g/kg) for medium and normal firing, respectively. Using 1.6 kg wood per hour is considered as medium firing pattern, and 1.25 kg/h reflect normal firing pattern. Emission factors results for stoves with old technology for TSP are 17.4 and 22.7 (g/kg) for medium and normal firing, respectively. From this estimates, the reductions would be in the range of 30 percent and 40 percent for medium and normal firing respectively. In addition, the authors indicate, based on previous results, that stoves which have been in use for several years emit more PM than brand new ones.

Reduction of pollutant emissions with exhaust gases can be obtained by either avoiding formation of such substances (primary measures) or by removal of pollutants from exhaust gases (secondary measures) (Kubica et al., 2007).

8.2.2 Examples of scrapping payments and subsidy programs in selected Norwegian municipalities

Scrapping payments and subsidised programmes were introduced in several Norwegian cities to accelerate the transition from old to new stoves in conformance with the Norwegian standard (Box 6). In the following paragraph we describe the examples of three Norwegian cities.

Oslo

The city of Oslo introduced a scrapping payment plan to promote the replacement of old stoves. To increase the proportion of clean burning stoves, Oslo residents can apply for grants from the city's climate and energy fund to replace old stoves, a measure introduced in 1998 that is still on-going today. The health impact of particle emissions is an important motivation for the support scheme, and the size of the grant is differentiated. The grant is NOK 3,000 (350 euros) in central areas of the city, where air pollution is worst and the need to replace old stoves is greatest. In other areas, the grant is NOK 1,500 (175 euros). The annual operating cost for the City of Oslo of this local support scheme varies from year to year depending on the number of grant applications and payments made. In the period 1998–2015, a total of NOK 23 million was granted for the replacement of 8677 stoves.

Bergen

To target air quality problems in Bergen, the city of Bergen introduced a grant program to accelerate the replacement of old stoves. The grant amounts to NOK 5000 (550 euros). Between 1999 and 2011, approximately 7000 new stoves were installed as a result of this municipal effort, representing a total reduction of approximately 85 tons of particulate matter. In 2011, 65% of the boiler stock met the Norwegian emission standards established in 1998. (Bodin, S. and T. Levander, 2014).

Trondheim

In Trondheim, there was an official effort to improve the environmental performance of older stoves by providing installation of an afterburner. The afterburner consists of a sheet of metal, which is mounted in the oven where it provides a secondary post-combustion chamber. The afterburner was installed in 100 homes, and proved to perform better in smaller stoves. Laboratory experiments have shown that the installation of afterburners could reduce particulate emissions by up to 75% (Bodin, S. and T. Levander (2014)).

8.2.3 Climate change mitigation for residential heating and implications on air quality

As a climate change mitigation measure, the Oslo municipality has regulated the phasing out the use of fuel oil for residential combustion by 2020. Greenhouse gas emissions from fossil fuel heating in Norwegian buildings are equivalent to 1.6 million tonnes of CO_2 (Klif, 2010). This is 3% of all greenhouse gas emissions in Norway. In order to phase out the use of fossil fuel by 2020, the Oslo municipality will introduce and reinforce various incentives using the Climate and Energy Fund and information campaigns. Oslo will work to ensure that the state imposes increased fuel surcharge and a ban on oil combustion as quickly as possible and at the latest by 2020. Other municipalities in Norway, e.g. Skedsmo, are following Oslo's example and implementing similar measures to phase out oil fuel for heating.

Phasing out the use of oil fuel for heating is a good climate measure, may have significant consequences for local air pollution if not properly considered and planned. One recent report by NILU-Norwegian Institute for Air Research and the Norwegian Asthma and Allergy Association (Tønnesen and Høiskar, 2013) investigated the consequences of the abolition of oil heating on air quality in Oslo. The report focuses on ambient air concentration changes of PM_{10} and NO_2 for the following three different scenarios of oil heating replacement:

- S1. All heating based on oil is replaced with electricity or other energy source without emission to air;
- S2. Half of the oil-fired plants are replaced with pellet stoves and half with electricity or other energy without emission air;
- S3. All oil heating systems were replaced with pellet stoves.

The annual emissions of NO_2 and PM_{10} in Oslo for 1) the current situation (all oil-fired plants remain); scenario 2 (half of the oil-fired plants are replaced with pellet stoves); and scenario 3 (all the oil-fired plants are replaced with pellet stoves) are shown in Table 10.

Table 10. Estimated annual emissions corresponding to fuel use and consumption.

Scenario	Annual consumption	Emissions NO_X	Emissions PM_{10}
100% fossil fuel use	50 kt	125,2 t	13,4 t
50 % pellets	56 kt	53,8 t	72,8 t
100 % pellets	112 kt	108,6 t	145,6 t

The results of this report indicate that the replacement of all oil heating systems by pellet stoves (scenario 3) will have a minor positive effect on the level of concentration of NO_2 in Oslo. On the other hand, it will lead to a considerable increase in PM levels in the city. The PM_{10} daily mean concentrations, which are under the EU limit value by a good margin for the current situation simulation, will considerably increase under scenario 3, leading to the exceedance of the EU daily limit value over ca 3 km^2 area in the city centre. Scenario 3 will also lead to the exceedance of the Norwegian guideline of 35µg/m^3 over large areas of Oslo.

8.3 Spain

In relation to other countries, the overall consumption of biomass and other renewable sources in Spain was below the European average (<4 MToe([10]), tonnes of oil equivalent) in the year 2003, while

[10] Toe = tonne of oil equivalent.

countries such as France, Finland or Sweden reported consumptions >6 MToe (Hernández Gonzálvez, 2005). In 2013, the total residential energy use in Spain was 15015 ktoe, a 3.3% decrease with regard to the previous year (IDAE, 2014).

Figure 24 shows the spatially distributed biomass consumption statistics for the year 2004. However, in 2007 it was estimated that the potential biomass resources in Spain could account for 19000 kTep([11]), out of which 13000 kTep correspond strictly to residual biomass (IDAE, 2007).

Despite this potential, the use of renewable energies in the residential sector is relatively minor when compared to other types of fuels. As shown in 25, in 1990 biomass accounted for slightly over 30% of the fuels used in the residential sector in Spain, a percentage which decreased steadily until 21% in 2007, only to increase again reaching 28% in 2012. While the share of gaseous fuels (mainly, natural gas) showed a marked increase from 6% to 40% during the same period and dominates the market currently, the residential use of liquid (oil) and solid (mainly, coal) fuels decreased mostly after 1996. The case of coal is especially interesting here, given that until the early 1990s a major proportion of buildings in the city of Madrid used coal-fuelled central heating boilers, especially in the densely populated city centre. In a large metropolis such as Madrid, residential coal combustion was expected to be negligible, especially in central heating systems. However, in the year 2007 residential coal combustion accounted for 8-9% of the mean annual PM_{10} mass concentrations in ambient air at a central location in the city (Moreno et al., 2013; Viana et al., 2010). The number of individual coal fed heating devices in Madrid (residential and commercial sectors) decreased from 2350 units in 1999 to 1107 in 2008 (Ayuntamiento de Madrid, 2010). As a consequence, the consumption of coal decreased by 55% (from 89,578 tonnes in 1999 to 40,593 tonnes in 2008) in this period (Madrid, 2010; Salvador et al., 2012). This decrease was achieved by active discouragement of coal combustion within the city by the local government, promoting and financially supporting the replacement of old existing heating boilers. In the year 2012 coal combustion had further decreased to 27,771 tonnes (Madrid, 2012). Currently, the use of solid fuels has decreased significantly with the result that the use of coal is today negligible (0%) in Madrid (Figure 26; López Jimeno, 2013). The Madrid Town Hall authorities are currently working on an updated census of coal, biomass and gasoil (Type C) boiler rooms in the city.

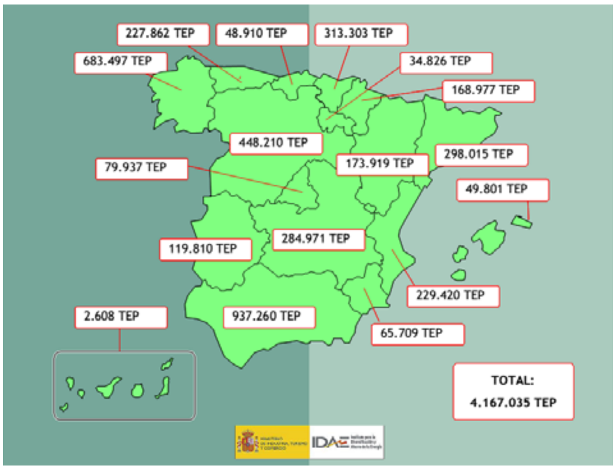

Figure 24. Total biomass consumption in Spain in 2004. TEP: Tonnes of oil equivalent (TOE). Source: (Hernández Gonzálvez, 2005).

[11] Tep = Ton equivalent petrol.

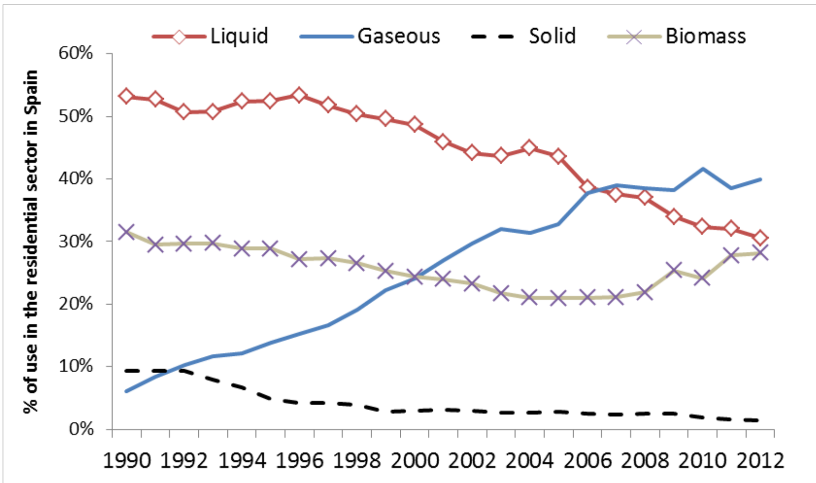

Figure 25. Relative use of solid, gaseous, liquid and biomass fuels in the residential energy sector in Spain between 1990 and 2012. Source: EU28 MS CRF inventory data.

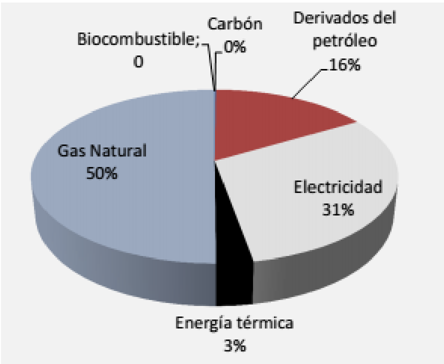

Figure 26. Residential fuel consumption in Madrid in 2012, as a function of the type of fuel. Gas Natural: natural gas. Biocombustible: biofuel. Carbón: coal. Derivados del petróleo: oil and its derivatives. Electricidad: electricity. Energía térmica: thermal energy. Source: López Jimeno (2013).

Regulatory instruments

The legal framework in Spain is based on national law 54/1997 for the electrical sector, which already back in 1997 identified the need to favour the use of renewable energy sources. As a result of this, the national-scale *Plan for Renewable Energies in Spain (2005-2010)* (currently ongoing for the period 2011-2020) was made publicly available in August 2005. The plan focuses mainly on biomass, biogas and biofuels. At municipal level, regulatory instruments in the city of Madrid are the Local Strategy for Air Quality in Madrid 2006-2010 (*Estrategia Local de Calidad del Aire de la Ciudad de Madrid 2006-2010*) and the Air Quality Plan for Madrid 2011-2015 (*Plan de Calidad del aire de la Ciudad de Madrid 2011-2015*), both of which include measures referring to the substitution of coal-fires stoves and boilers.

Economic instruments

Two main instruments have been implemented in the city of Madrid to increase energy efficiency in the residential sector: a plan for the renewal of residential boilers and stoves, and an energy efficiency certification for buildings.

8.3.1 Plan for the renewal of residential boilers and stoves

Following the recommendations in the national plan, the city of Madrid implemented in 2007 a regional plan for renewal of residential boilers and stoves (*Plan Renove de Calderas Individuales de la Comunidad de Madrid*), which aims to incentivise the exchange of residential stoves or boilers using coal, LPG, gasoil, electricity or natural gas, by condensation boilers running on LPG or natural gas. Therefore, this scheme does not directly address biomass as a residential fuel. The ultimate goal is to increase both safety and energy efficiency. This scheme is still ongoing (until September 2015, or until the funds approved by the regional government are covered). The *Plan Renove* is aimed at individual residential boilers/stoves, or those in residential buildings, with thermal power lower or equal than 70 kW. Specific requirements for application are:

- Installation within the Madrid region.
- Installation of new appliances, which should be registered in the database of efficient boilers www.cambiatucaldera.com.
- Minimum reduction of 20% in consumption with regard to the period before the installation.
- Installation by an official operator/company, which should be registered in the Plan Renove database.
- The stove/boiler removed must be disposed of and inutilised.

The scheme offers to cover for a maximum of 30% of the new installation. Only energy efficient boilers and stoves, listed in the website www.cambiatucaldera.com, are incentivised. The ultimate goal was to reduce the number of coal-fired boilers in Madrid to <500.

As a result of the implementation of the *Plan Renove de Calderas,* in the year 2013 more than 80.000 individual domestic boilers/stoves had been renewed since 2010 (López Jimeno, 2013), as well as 1.180 boiler rooms in residential buildings (including 439 coal-fuelled boilers). From the 7224 applications received between 1999 and 2010 in the framework of this Plan, 90% of the boilers were exchanged for natural gas installations. Initial estimations suggest that the *Plan Renove* had in 2013 had positive impacts regarding (Table 11):

- Primary energy savings: 103.600 MWh/year
- Emissions reduction: 21.096 tonnes/year CO_2
- Emissions reduction: 67.3 tonnes/year NOx

Table 11. Environmental impacts of the Plan Renove for residential boilers and stoves in the Madrid region in 2013. Source: López Jimeno (2013)

ENVIRONMENTAL IMPACTS			
PLAN RENOVE	Energy savings (MWh/year)	Emissions reduction: CO_2 (T/year)	Emissions reduction: NOx (T/year)
Individual boilers	80.000	16.282	52.0
Boiler rooms	23.600	4.814	15.3
TOTAL	103.600	21.096	67.3

Finally, this scheme also includes a component for renewal of boilers in industrial settings, which is out of the scope of this work.

8.3.2 Energy efficiency certification for buildings

According to the EU (https://ec.europa.eu/energy/en/topics/energy-efficiency/buildings), buildings are responsible for 40% of energy consumption and 36% of CO_2 emissions in the EU. While new buildings generally need less than three to five litres of heating oil per square meter per year, older buildings consume about 25 litres on average. Some buildings even require up to 60 litres. Currently, about 35% of the EU's buildings are over 50 years old. By improving the energy efficiency of buildings, we could reduce total EU energy consumption by 5% to 6% and lower CO_2 emissions by about 5%.

Following the recommendations of the EU 2010 Energy Performance of Buildings Directive and the 2012 Energy Efficiency Directive, in April 2013 the Spanish national decree RD 235/2013 was published, establishing the technical requirements for energy labelling of residential buildings. According to this RD, all residences being sold or rented after this date must provide an up-to-date energy certification, provided by the national or regional authorities or by authorised companies. This certification evaluates the building's energy efficiency (insulation, windows, etc.) as well as energy consumption and CO_2 emissions. Based on this evaluation the building receives a label (from "A" to "G", "A" referring to the highest efficiency and "G" to the lowest) regarding primary energy consumption, and another for CO_2 emissions (Figure 27).

Figure 27. Energy efficiency grading scheme for buildings. Source: https://ec.europa.eu/energy/en/topics/energy-efficiency/buildings

In Spain, the certification grades are obtained based on an algorithm and a series of coefficients which may be found in the technical Annex "*Condiciones de aceptación de procedimientos alternativos*" of RD 235/2013, and which are cumulative (i.e., the applicable coefficients for a given building are summed up). The lower the result of the algorithm, the higher the energy efficiency. As an example, label "A" is only awarded to buildings with <0.15 points. In the case of biomass boilers and stoves, the coefficient determining the conversion of primary energy to CO_2 is equal to zero, given that it is considered that biomass is CO_2 neutral on the global cycle). As a result, this scheme incentivises the installation of biomass boilers in residences given that it results in lower grades by the algorithm and therefore higher rankings in the energy certification. It must be stated that the installation of biomass boilers is not obligatory for the obtention of an "A" certification, even if it in facts does help to obtain it.

As a result, this energy efficiency mechanism has the indirect effect of incentivising the installation of biomass boilers in buildings in Spain. This would be beneficial for air quality and climate in rural or sparsely populated areas, but detrimental for air quality and population health in densely populated urban areas such as Madrid or Barcelona, as discussed in previous sections.

8.3.3 Scientific research

Because of the interest in favouring the increase of the share of renewable energy sources in the residential sector, a number of research projects have been funded in Spain by the European Commission to focus on biomass as residential fuel (e.g., BioMaxEff project, *Cost efficient biomass boiler system with maximum annual efficiency and lowest emissions*, www.biomaxeff.eu; *Life AIRUSE, Testing and Development of air quality mitigation measures in Southern Europe*, www.airuse.eu). From a national perspective, the Spanish Ministry of Economy actively supports scientific research on this topic through dedicated institutions such as the Institute for Diversification and Energy Saving

(IDAE), the Centre of Energy, Environmental and Technological Research (CIEMAT), and the National Research Council (CSIC).

Case study: the combined effect of emission sources and meteorological factors

The case of rural areas in North-Eastern Spain is presented here as an example of increasing ambient BaP concentrations in rural areas in recent years, expected to arise mainly from the combination of meteorological factors and residential and local combustion emissions, but for which no clear conclusion may be drawn regarding the main pollution source(s).
The towns of Bellver de Cerdanya (2075 inhabitants in 2014, www.ine.es) and Manlleu (20279 inhabitants in 2014, www.ine.es) are located South of the Pyrenees region in Northern Spain (Figure 28). Whereas Bellver is mostly rural, Manlleu may be considered an urban nucleus. In recent years, mean annual ambient BaP concentrations showed an increase in the rural areas which did not correlate with BaP in nearby urban areas such as Barcelona (Figure 29). BaP concentrations in Manlleu and Bellver followed similar trends between 2008 and 2012 (annual BaP means increased from 0.2 to 0.7-0.8 ng/m^3). In 2013, the Manlleu station registered unusually high BaP concentrations, exceeding the 1 ng/m^3 target value, which were not detected in Bellver. It should be noted that there was a change in the monitoring station location (from *IES Antoni Pous* to *Hospital comarcal*). In 2014, the target value was not exceeded but mean concentrations reached 0.8 ng/m^3. As evidenced by Figure 29, the BaP trends observed in the rural areas had no apparent relationship with the concentrations registered in the Barcelona urban area, and thus suggest different emission sources.

Figure 28. Location of Manlleu, Bellver de Cerdanya and Barcelona (Spain).

In order to interpret the origin of these BaP trends, the same analysis was carried out for PM_{10} concentrations. As shown in Figure 29, PM_{10} concentrations followed similar trends at the three locations, with decreasing levels across the period 2007-2014 which were especially evident for the more densely populated areas (Barcelona and Manlleu). These results suggest that the main driver of PM_{10} concentrations during this period was regional-scale meteorology, which favoured dilution or concentration of the locally emitted pollutants. The similarities between the Barcelona and Manlleu trends suggest that the main source of locally emitted PM_{10} was vehicular traffic. However, the differences between the PM_{10} and BaP trends point towards different sources of both pollutants.

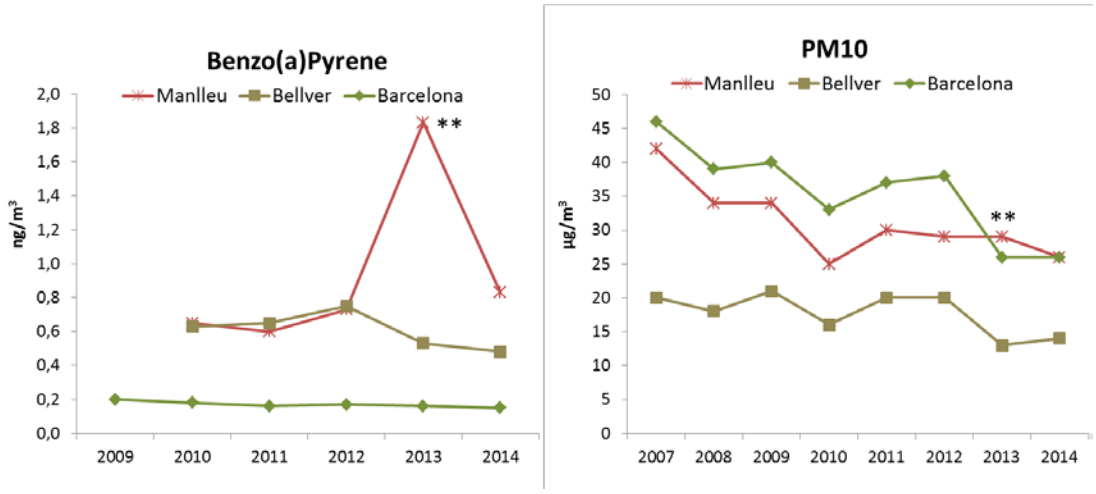

Figure 29. Mean annual benzo(a)pyrene (BaP) and PM_{10} concentrations in Barcelona (station Gràcia-St. Gervasi), Manlleu (stations IES Antoni Pous and Hospital comarcal), and Bellver de Cerdanya (station CEIP Mare de Déu del Talló), for the years 2007 to 2014 where available. **: the data after 2012 correspond to a different monitoring station in Manlleu (*IES Antoni Pous* <2012; *Hospital comarcal >2012)*.

The seasonal variation of both pollutants (BaP and PM_{10}) may also be assessed in Barcelona and Manlleu. As shown in Figure 30, winter BaP concentrations were one order of magnitude higher in Manlleu than in Barcelona during this period, whereas in summer similar results were obtained in both areas. This might hint at the residential origin of the BaP concentrations. However, because the Manlleu area is frequently affected by thermal inversions, winter PM_{10} concentrations were of a similar order of magnitude and even higher during certain years in the mountain area than in Barcelona, despite the larger size of the urban agglomeration (1.6 million inhabitants in Barcelona vs. 20279 in Manlleu in 2014). This could suggest that the high winter BaP concentrations registered could be linked to meteorological factors, instead or in addition to emission sources. However, in 2008-2009 thermal inversions were not the cause of high BaP concentrations and there is a clear gradual increase in BaP from 2008-2013. Finally, it is interesting to note how winter BaP concentrations seem to follow an increasing trend in Manlleu, whereas this is not the case in Barcelona (Figure 30).

Based on these data, the interpretation of the origin of the increased BaP concentrations in the Pyrenees region is not straightforward. The use of solid fuels in local activities (residences, small-scale industries, etc.) could be a source of the elevated BaP concentrations. However, the regional-scale meteorology, characterised by frequent thermal inversions, could also play a relevant role in the accumulation of atmospheric pollutants.

Other studies carried out in this region also evidenced the presence of biomass burning emissions. Van Drooge and Grimalt (2015) collected samples of atmospheric PM at urban and rural sites during warm and cold seasons in Catalonia (Spain). Organic tracer compounds, such as levoglucosan, isoprene, pinene oxidation products, PAHs and quinones, were analysed, concluding that the strong predominance of biomass burning residues at the rural site during the cold period involved atmospheric concentrations of PAHs that were 3 times higher than at the urban sites. In addition, BaP concentrations above legal recommendations, indicating that rural communities undergo higher exposures to this carcinogenic compound as a consequence of biomass burning.

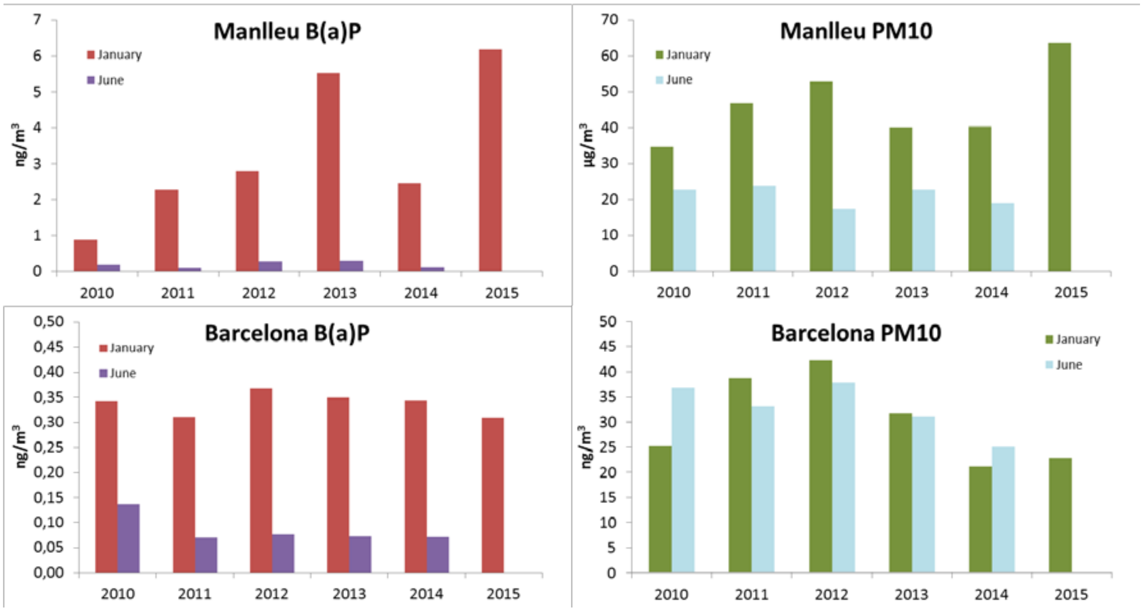

Figure 30. Mean monthly PM_{10} and BaP concentrations in Barcelona (station Gràcia-St. Gervasi) and Manlleu (stations IES Antoni Pous until 2013, and Hospital comarcal after that date) during the winter and summer months. Source: Generalitat de Catalunya.

Similarly, other studies focused on major cities such as Barcelona and Madrid (van Drooge et al., 2014). In these urban areas biomass burning was not expected to be an important local emission source. However, regional emissions from wildfires, residential heating or biomass removal (e.g., agricultural) may influence the air quality in the cities (e.g., (Minguillón et al., 2011; Reche et al., 2012; Viana et al., 2013). In their 2014 study, van Drooge et al. (2014) observed that the ambient air concentrations of the organic tracers for biomass burning increased by an order of magnitude at both sites during winter compared to summer. Overall, there was little variation between the street and roof sites in both cities, suggesting that regional biomass burning sources influence the urban areas after atmospheric transport. Despite the different atmospheric characteristics in terms of air relative humidity, Madrid and Barcelona exhibit very similar composition and concentrations of biomass burning organic tracers.

In order to reduce air quality impacts from the residential sector and to increase energy efficiency, a number of economic instruments are in place in Catalonia. Some of these instruments have been discussed in detail in previous sections, including the energy efficiency certification for buildings and the plan for the renewal of residential boilers and stoves. These instruments were mainly in place during the years 2011 and 2012, and they were implemented by the regional administration (Generalitat de Catalunya, and the Institut Català d'Energia). At present, other schemes are in place such as the Catalan Strategy for the Renewal of Buildings, which aims to decrease the energy consumptions from buildings by 14% in 2020, targeting 60% of the buildings in the region.

8.4 France

Thanks to its renewable character, relative low cost, and source of local jobs, using wood as primary source of residential heating is strongly supported in France in the context of climate mitigation to reach the national target of 23% renewable energy by 2020. The number of individual stoves in France is therefore expected to increase from 5.7 millions in 2006 to 9 millions in 2020.

In France, primary $PM_{2.5}$ emission attributed to indoor residential solid biomass burning reach 27% on average. Emphasising that it is mainly when operated in inappropriate conditions that woodburning is detrimental to air quality, the French Agency for Environment and Energy Efficiency (ADEME) is actively supporting the development of best practices. In addition, ADEME is promoting installation of stoves complying with the eco-label “Flamme Verte 5*” that corresponds to particulate matter emissions of less than 125mg/Nm^3, or 3kg/yr when used as primary source of heating, with an energetic yield of at least 80% (ADEME, 2013).

The average age of stoves in France is relatively high (15 years) with 18% of open fire places, and the turnover is only 4%, i.e., 25 years for a full replacement of the park. Supporting the implementation of newer stoves is therefore expected to substantially reduce emissions. It should also be noted that retrofitting existing stoves is not common in France. Several studies (Fraboulet et al, 2012; Le Dreff et al, 2012) supported by ADEME have been performed to evaluate such techniques. The main conclusions were that although some of them allow to reduce particulate matter emissions, they remain difficult to implement partly due to their costs and to the French legislation which is not adapted (CSTB, 2013). In 2015, ADEME (ADEME ERFI, 2015) has launched a benchmark study in order to look at the suitability of implementation of retrofitting techniques for old existing stoves and open fires in France as an alternative to replacement.

In order to investigate the efficiency of subsidising the replacement of existing stoves at the National scale, ADEME has set up an innovative programme aiming to replace 3,200 existing devices with efficient stoves emitting less than 50mg/Nm^3 PM in the Arve Valley (Alps) with the ambition to reduce by 25% total emission of $PM_{2.5}$ attributed to residential woodburning by 2016 in the area. The Arve Valley, in the vicinity of Chamonix and the Mont Blanc, has been selected for this experiment because of the high number of PM_{10} exceedances that legitimated the deployment of an air quality management plan agreed in 2012 and an emergency air quality management plan in 2013. The total number of residential woodburning devices in operation in the Arve Valley is about 20,000, of which about 13% are open fireplaces, and 40% of the stoves installed before 2000 (Fond Air Bois, 2013).

The corresponding subsidising fund has been set up in June 2013 as a joint initiative of the State (through its local representation in Haute Savoie), ADEME, and the local management venture (Syndicat Mixte d’Aménagement de l’Arve et ses Abords, SM3A). It is funded by ADEME and local authorities and managed locally by SM3A. The value of the fund is 800k€ per year for four years. It aims at subsidising private individuals up to 1000€ for the installation of an efficient stove as long as it does not exceed 50% of the total budget of the new installation. The subsidy applies for the replacement of open fire place or stoves installed before 2002 in accommodations used as primary residence over any of the 41 commune concerned with the air quality management plan. The newly installed stove must comply to the Flamme Verte 5* eco label and guarantee an emission of less than 50mg/Nm^3 of PM. Last, the installation must be performed by a qualified practitioner (Fond Air Bois, 2013).

After 1.5 years of operation, the fund reached 1000 of replaced woodburning devices (Fond Air Bois, 2015) , 44% of them being insets, 35% stoves and 17% open fire places (which is encouraging given that the average proportion of open fire places in only 13% in this area). The average age of the appliances replaced was 25 years.

Given the relatively recent deployment of the subsidising programme, it remains irrelevant to attempt a quantification of the efficiency in terms of improvement in air quality. It should be noted however that the programme is accompanied by a dedicated atmospheric chemistry research project that builds upon PM source apportionment monitoring to track any change in the composition of ambient PM that could be attributed to the subsidising programme (Research Project Decombio, funded by the Primequal Programme, results expected in 2017). It was also supplemented by an additional field study to reduce uncertainties in quantification of emission from fireplace and woodstoves in the Arve valley.

8.5 Greece

This case study summarises the effects of the economic recession on air pollution caused by residential heating in case of Thessaloniki (Greece). The integration of renewable energy sources into existing energy systems has emerged as pivotal in the context of sustainable energy planning (Toka et al., 2014), both in Greece and in other European countries and especially taking into account the EU 2020 renewable energy target. However, the implementation rate of eco-innovative technologies in the energy sector varies considerably among countries and regions, mainly due to the lack of attractive policy schema to incentivise their use. Among other renewable energy applications, the regulated deployment of biomass heating systems by residential users in Greece is considered to be hindered both by economic and non-economic influence factors and it remains below expectations Figure 31, despite the readily available supply of feedstock and the positive impact on the environment, if used appropriately. A recent analysis (Toka et al., 2014) assessed the impact of various interventionary policies on promoting the deployment of biomass heating systems in residences, aimed at the scientific but also the country's energy regulatory authorities. It concluded that, without any intervention, 85% of the total projected adoptions of biomass-based systems for residential heating is expected to take place until 2030 in Greece, with only 12% being attained by 2021.

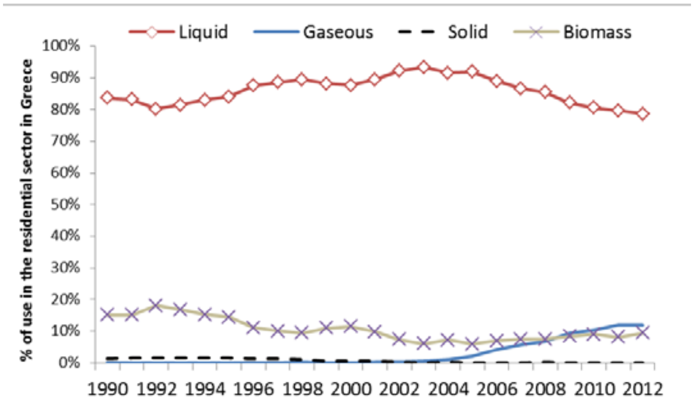

Figure 31. Relative use of solid, gaseous, liquid and biomass fuels in the residential energy sector in Greece between 1990 and 2012. Source: EU28 MS CRF inventory data.

Other studies focus on the analysis of household fuelwood consumption and the determinants of the choice to use the specific energy for heating and cooking in Greece (Arabatzis and Malesios, 2011). The results show that household sociological and economical characteristics as well as more general environmental issues are suitable to explain differences towards fuelwood consumption for space heating and cooking. Examples of these characteristics are the number of members in the household (fuelwood consumption is positively linked with this parameter) or the type of household (detached houses seem to consume more fuelwood than apartments, perhaps because there is ample storage space available, but also because the owners either live in rural areas or have higher incomes). The study concludes that advisable and rational policies in the fuelwood market should be based on studies of the profile and patterns of fuelwood consumers.

As a result, it seems evident that there is a growing interest in Greece in implementing biomass-based heating solutions at residential scale, even if the necessary economic instruments (incentives) are not yet in place and/or the necessary socio-economic mechanisms are not yet fully understood.

In this framework, the recent economic crisis prompted a rapid increase in the un-regulated combustion of solid fuels in Greece, which led to air quality degradation. One example of a serious wintertime air pollution episode in Thessaloniki was assessed by Saffari et al., (2013). According to these authors, air quality deterioration was mostly due to the increased price of fuel oil, conventionally used as a source of energy for domestic heating (Figure 32), which encouraged the residents to burn the less expensive wood/biomass during the cold season. A wintertime sampling campaign for fine particles ($PM_{2.5}$) was conducted in Thessaloniki during the winters of 2012 and 2013 in an effort to quantify the extent to which the ambient air was impacted by the increased wood smoke emissions. The results indicated a 30% increase in the $PM_{2.5}$ mass concentration as well as a 2–5-fold increase in the concentration of wood smoke tracers, including potassium, levoglucosan, mannosan, and galactosan Figure 32. The concentrations of fuel oil tracers (e.g., Ni and V), on the other hand, declined by 20–30% during 2013 compared with 2012. Moreover, a distinct diurnal variation was observed for wood smoke tracers, with significantly higher concentrations in the evening period compared with the morning. The combination of these tracers clearly pointed towards combustion of solid fuels as the source of these emissions. In addition, correlation analyses indicated a strong association between reactive oxygen species (ROS) activity and the concentrations of levoglucosan, galactosan, and potassium, underscoring the potential impact of wood smoke on PM-induced toxicity during the winter months in Thessaloniki. Because the residential use of solid fuels was mostly un-regulated, it is not detected in the country's emissions inventories (Figure 31; data available only until 2012, at national scale).

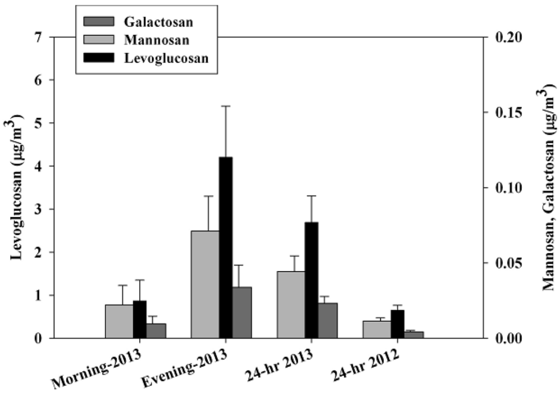

Figure 32. Average concentrations (µg/m^3) of organic wood smoke tracers (levoglucosan, mannosan, and galactosan) in $PM_{2.5}$ samples in Thessaloniki. Error bars represent one standard error. Source: Saffari et al. (2013).

The authors concluded that active involvement of public authorities and local air quality control agencies was urgently required to implement effective air pollution control strategies in the study area. One of the practical long-term solutions to the general air pollution issue in Thessaloniki could be natural gas distribution in residential areas. Developing catalytic domestic wood burners and increasing the energy efficiency of existing buildings were also discussed as other possible solutions prior to the economic crisis (Saffari et al., 2013).

The presence of biomass smoke had previously been detected in Greece, although the main source on previous occasions was long-range transport of wildfire emissions (e.g., from Russia; Diapouli et al., 2014). The relative increases in organic (OC) and elemental (EC) carbon were higher due to solid fuel combustion (Saffari et al., 2013) than to wildfire emissions (Diapouli et al., 2014), thus highlighting the relevance of the impacts on air quality and health of local un-regulated solid fuel combustion at residential scale. Other studies (Paraskevopoulou et al., 2015) also detected the impact of wood combustion (dominant fuel for domestic heating) to air quality in Athens, which massively started in winter 2013, by means of the almost 30% increase in the contribution of particulate organic matter to the urban aerosol mass in the city. With regard to the impact of the economic recession on air quality, these authors observed that the inter-annual trend of PM masses reveals a decrease of ambient aerosol masses through the studied 5-year period (2008-2013), which is attributed to measures adopted during the last decades but also due to further reduction of anthropogenic emissions since the economic recession originating in Greece.

Subsequently, the health impact and monetary cost of exposure to particulate matter emitted from biomass burning was also assessed for Thessaloniki (Sarigiannis et al., 2015a). The years under study were 2011-2012 and 2012-2013. Health impact was assessed based on estimated exposure levels and the use of established WHO concentration-response functions (CRFs) for all-cause mortality, infant mortality, new chronic bronchitis cases, respiratory and cardiac hospital admissions. Monetary cost was based on the valuation of the willingness-to-pay/accept, to avoid or compensate for the loss of welfare associated with illness. Results showed that long term mortality during the 2012-2013 winter increased by 200 excess deaths in a city of almost 900,000 inhabitants, or 3540 years of life lost, corresponding to an economic cost of almost 200-250m€. Estimated health and monetary impacts are more severe during the cold season, despite its smaller duration (4 months). Considering that the increased ambient air concentrations (and the integral of outdoor/indoor exposure) are explained by shifting from oil to biomass for domestic heating purposes, several alternative scenarios were evaluated. Policy scenario analysis revealed that significant public health and monetary benefits (up to 2b€ in avoided mortality and 130m€ in avoided illness) might be obtained by limiting the biomass share in the domestic heat energy mix. As a result, the authors concluded that fiscal policy affecting fuels/technologies used for domestic heating should be reconsidered, since the net tax loss from avoided oil taxation due to reduced consumption was further compounded by the public health cost of increased mid-term morbidity and mortality (Sarigiannis et al., 2015a). If biomass continues to be used, then following recommendations were suggested to reduce population exposure to fine and ultrafine particles: (1) closed fireplaces are more efficient in terms of heating performance as well PM emissions; and (2) emission factors for fine particles are highly dependent on fuel characteristics and burn conditions (smoldering vs. flaming), and emission factors for specific organic chemicals are influenced by fuel moisture content and burning conditions, thus frequent ventilation of indoor spaces, good combustion conditions and high quality, low moisture wood would ensure relatively controlled direct exposure to the urban population.

8.6 Denmark

This case study summarises the impact of wood combustion on particle number concentrations: case study in a Danish residential village. Impacts of wood combustion on air quality are generally monitored using PM, OC, K+ and/or organic compounds (e.g., levoglucosan, galactosan, mannosan) as tracers. However, few studies have so far focused on the potential of size resolved particle number concentrations for this purpose. This issue was addressed in a study carried out in a rural area in Denmark (Wåhlin et al., 2010), where fine particles ($PM_{2.5}$), particle number size distribution, NOx, CO, soot, PAHs and monosaccharide anhydrides were measured at two sites, one exposed by smoke from woodstoves in a small village (Slagslunde), and the other acting as background reference outside of the village. The particle size distributions were grouped into four log-normal modes at 13 nm, 28 nm, 65 nm, and 167 nm. By studying the time series and the wind direction dependence of the individual modes, some episodes and wind directions were identified with contributions from other local sources than wood stoves. Most important were contributions to the 28 nm mode from local traffic in the wind sector 90°-130°, and contributions to the 13 nm mode from the district heating and power station. Based on wind direction analyses a cleaned data set was created in which all time periods with presumed influence from other sources than wood combustion was removed.

Curves for the average diurnal variation of the increments measured with high time resolution and based on the cleaned data set have almost the same shape for $PM_{2.5}$, for particle volume and for soot, thus providing strong evidence for the existence of only one major source, i.e. wood burning. Furthermore, the shape of the curves is typical for normal firing habits with low values during most of the day until the middle of the afternoon when residents return from work. At this point the values start to increase to a higher level and remain high in the evening until normal bedtime before or around midnight. The diurnal variation of the total particle number concentration was slightly different, showing some increase already when the morning traffic starts. By an examination of the different modes it was evidenced that this behaviour was mostly due to the smallest particles, represented by the 13 nm mode and the 28 nm mode. The behaviour of the 65 nm and 167 nm modes were very similar to the behaviour of $PM_{2.5}$, particle volume and soot. The authors concluded that the increments of $PM_{2.5}$, soot, particle volume and particle number in the larger size fractions were almost entirely due to wood combustion, while some part of the particle number in the finer size range (28 nm mode) was due to traffic.

As a core result of the study the authors defined a wood smoke source profile, i.e. a vector which links the three parameters measured ($PM_{2.5}$, V and soot) to each other, so that from the knowledge of the increment of one of these, the value of the other two may be calculated provided that the source responsible for the increment is wood smoke. Particle number N did not have a close link to these three parameters. In conclusion, particle number concentrations are unable to yield a fair impression of pollution from wood combustion, and are therefore not an optimal tracer for these emissions. Even though a certain degree of correlation may be found between particle number and other tracers of wood smoke pollution, particle number concentrations is not suitable as predictor for these other measures (Wåhlin et al., 2010).

8.7 Czech Republic

This case study summarises the impacts of wood and coal combustion in a complex urban aerosol mix in Mlada Boleslav (Czech Republic). As described in the previous sections, contributions from residential fuels to air quality may range from <3% to >30% of fine particle mass, depending on the area in Europe and also on the season of the year. Numerous works have described the need for specific tracers or combinations of tracer elements to identify such contributions, given that one of its main markers (organic carbon, OC) is a frequent tracer of other anthropogenic (e.g., traffic, certain industrial activities, secondary organic aerosol, etc.) and natural (biogenic emissions) sources. As a result, the clear discrimination of a residential combustion emission source is usually complex within the urban aerosol mix, especially in the presende of other dominant sources.

An example of this scenario was published recently by Hovorka et al. (2015). In this work, the long-time centre of the automobile industry in the Czech Republic Mlada Boleslav, typical example of urban development, was the subject of study. The automobile factory was founded and has grown continuously within the urbanised area of the city, currently occupying about one third of the urban area. The aim of this work was to identify the major PM emission sources in the city, given that industries in close proximity to residential districts are a historical phenomenon of urban development frequently causing air quality problem in European cities.

Receptor modelling, time series of organic tracers for wood and coal burning, small-scale multiple-site PM_{10} monitoring and mobile PM_{10} measurements were combined to identify sources of fine ($PM_{0.15-1.15}$) and coarse ($PM_{1.15-10}$) particles in a residential district in winter 2013. The receptor model was applied to hourly concentrations of organic and elemental carbon in fine PM and 27 elements in fine and coarse aerosol particles. Results showed that wood burning, coal combustion, traffic, and industry were identified as the fine particle sources (contributing with 49%, 34%, 16% and 1% of the fine PM mass, respectively). Combined wood burning and coal dust, road dust with salt, and abrasion of car brakes were identified the coarse particle sources (contributing with 80%, 14%, and 6% of the coarse PM mass, respectively). The large contributions of wood and coal combustion were surprising for this residential district that is a block of flats with district heating. High correlations were observed between the wood combustion contributions and the levoglucosan and mannosan concentrations. Lignite combustion was specifically identified. Peak concentrations in excess of 500 mgm^3 of PM_{10} recorded during mobile measurements along with visible plumes from home heating, confirmed the interpretation of the results. These results evidence that the application of adequate tracers and with the necessary time resolution can support the discrimination of residential combustion sources, even within a complex urban aerosol mix.

9 Co-benefits for health and climate of reducing residential heating emissions

Co-benefits are win–win outcomes for sectors other than the one from which a policy originates. They include health benefits that arise from actions that are primarily motivated by an interest in mitigating climate change, and climate mitigation benefits produced by actions that are primarily motivated by an interest in improving public health (WHO, 2015). Reducing emissions of health-relevant air pollutants – especially those that are also climate active pollutants (such as CH_4 and BC) – can have short- and medium-term co-benefits for health, as it can immediately reduce exposure to associated particulate air pollution. Accounting for these health co-benefits can produce a more complete economic picture of the costs and benefits associated with efforts to reduce heating-related emissions, such as woodstove change-out programmes. Increasing efficiency and tightening restrictions on emissions from wood and coal heating throughout the world would both slow down the current rapid speed of global warming (relating to BC in fine particles and VOCs and CH_4 that promote ozone formation) and reduce the burden of disease caused by combustion-derived particles (especially organic compounds carried by BC and contaminants in coal) (WHO, 2015).

Wood and other forms of biomass are often considered renewable and climate-friendly fuels because trees take up CO_2 as they grow and store it in the form of carbon (it is in this sense that biomass is considered a "CO_2 neutral" source of energy). However, as wood is burned this carbon is released back to the atmosphere, not only as CO_2 but in most household combustion also in the form of short-lived climate pollutants such as BC, CO and VOCs including CH_4. Thus, to be perfectly "carbon neutral", wood fuel has to be not only harvested renewably but also combusted completely to CO_2. As has been discussed in detail in the previous sections, this is rarely the case. In addition, forestry activities, wood storage and drying, and transport and distribution of wood to households also contribute to GHG emissions.

A World Bank study found that replacing current wood stoves and residential boilers used for heating with pellet stoves and boilers and replacing coal fuel with coal briquettes (mostly in eastern Europe and China) could provide significant climate benefits. It would also save about 230.000 lives annually, with the majority of these health benefits occurring in Organisation for Economic Co-operation and Development nations (Pearson et al., 2013). Another study coordinated by the United Nations Environment Programme and the World Meteorological Organization found that widespread dissemination of pellet stoves (in industrialised countries) and coal briquettes (in China) for BC mitigation could improve health, since these interventions lead to reductions in $PM_{2.5}$. Major reductions in annual premature deaths expected as a result of these interventions include about 22.000 fewer deaths in North America and Europe, 86.000 fewer deaths in east and southeast Asia and the Pacific, and 22.000 fewer deaths in south, west and central Asia (UNEP & WMO, 2011).

If Arctic climate change becomes a focus of targeted mitigation action (because of threats from rising sea levels, for example), widespread dissemination of pellet stoves and coal briquettes may warrant deeper consideration because of their disproportional benefit to mitigating warming from BC deposition in the Arctic (UNEP & WMO, 2011). The World Bank found that replacement of wood logs with pellets in European stoves could lead to a 15% greater cooling in the Arctic (about 0.1 °C). For Arctic nations the modelling strongly indicates that the most effective BC reduction measures would target regional heating stoves for both climate and health benefits (Pearson et al., 2013)

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Annex

Analysis of emission factors

Residential wood combustion is an important source of fine particles. For this sector and the associated pollutants, the use of emission factors is indispensable to develop reliable national emission inventories.

Particle emission factors determination requires common methods and practices at the European level, but these conditions are not yet met at the moment. Emission factors are supposed to be representative of emissions of existing wood-burning appliances as well as of the fuels used. They have to involve lifespan of this equipment, estimated between 20 and 40 years and have to include old technology as well as the current state-of-the-art appliances. Inventories have also to reflect the reduction of emissions driven by the wood-burning appliance replacement rate. Furthermore, they should be representative of particulate and gaseous emissions, and should include the condensable fraction.

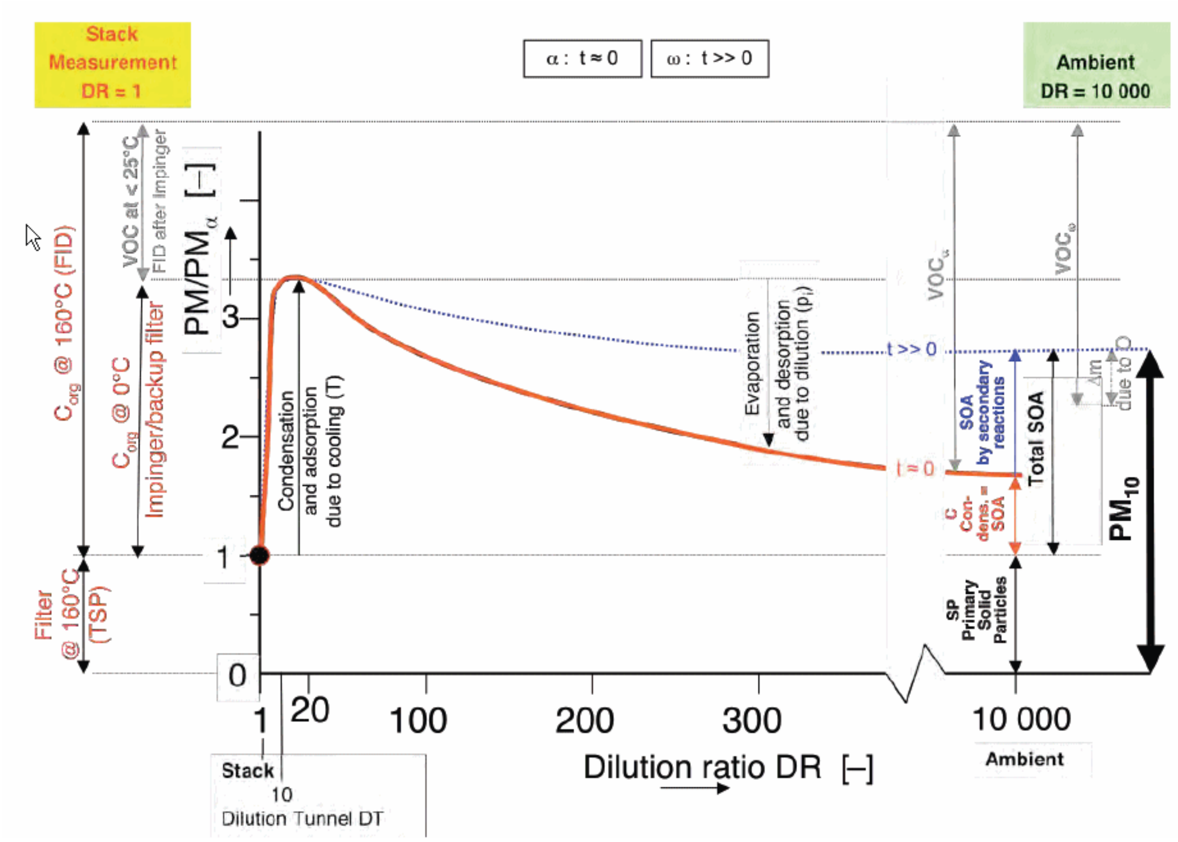

Figure A1. Conversion of stack emissions of PM and organics to PM_{10} in the ambient air (Nussbaumer et al., 2010).

Sampling procedures and their influence on the determination of aerosol emissions

Different particle sampling procedures can be used to characterise particulate matter mass emission from residential wood combustion. Many of them were already described by Nussbaumer et al., (2008). The four main approaches are described hereafter:

- Method SP (solid particles): the gas sampled from the flue gas is filtered directly or after a cutoff inlet for PM_{10} or $PM_{2.5}$ if a reference particle size is requested. The mass of particles collected on the filter (TSP, PM_{10} or $PM_{2.5}$) is determined by gravimetry. This approach is used at the European level for stationary emission sources (EN

13284-1 - temperature of filtration: 160°C, method DIN+ - temperature of filtration: 70°C). The higher the temperature of filtration, the more the condensable organic fraction collected on the filter will be volatilised,

- Method SPC (solid particles and condensable organic fraction): after hot filtration (cf. method SP), the sample passes through a series of impingers maintained to less than 20°C, containing water (method EPA 5H - temperature of filtration: 120°C) or a solvent to trap condensable VOCs (method derived from EN TS 15439: 2006 used for biomass gasification). After evaporation of water or solvent, the mass of particles collected is added to the mass of particles measured on the filter,
- Method DT (dilution tunnel): the flue gas from the appliance is collected and diluted with ambient air via an exhausting hood (dilution tunnel). A sampling on a filter is then carried out (English standard BS 3841-1, Norwegian NS 3058-2 and EPA method 5G). Because of dilution, the temperature decreases and the condensable species are transformed into aerosols. A sample is taken in the dilution tunnel and particles collected on a filter, again with or without a PM_{10} or $PM_{2.5}$ cut-off inlet.
- Method SPFID (solid particles and flame ionization detector): solid particles are measured according to method SP. The condensable fraction is estimated from a measurement of the total volatile organic compounds (TVOCs) made with a flame ionization detector. There have been intents to establish correlations between the TVOCs measurements and the condensable fractions, but at the moment no reliable correlation exists.

Method SP (solid particles), which is simple to implement, does not take into account or largely under-estimates the emitted condensable VOCs, and therefore highly underestimates the quantity of emitted particles and the quantity of particles in ambient air resulting from combustion. It should thus not be used to determine emission factors of particles. It is not either sufficiently discriminant to identify the most efficient wood-burning appliances. Let us recall that the improvement of combustion conditions leads to a significant reduction of condensable organic compounds, which are not measured by this method. In addition, depending on the equipment used, the actual temperature of filtration can strongly fluctuate and lead to a greater variability of the results obtained (the temperature of filtration cannot be maintained constant during the whole of the test if the setpoint temperature is too low). To overcome this shortcoming, future European standard envisages raising the temperature of filtration to 180°C. Method SPC (solid particles and condensable) has the advantage of quantifying solid particles and condensable organic fraction as they are emitted. Let us recall that some condensable compounds are present in gas phase exposed to high temperature in the stack. This method allows to determine these compounds owing to the condensation stage which reaches a temperature lower than 20°C. Although more complex than the method SP, this method can still be implemented relatively easily (Fraboulet et al., 2011). In addition, it makes it possible to determine the ratio method SPC/method SP which seems interesting for better evaluating the combustion quality of a wood-burning appliance (Nussbaumer et al., 2008).

Method DT (dilution tunnel) takes into account the particulate and condensable fractions. The condensable inorganic fraction being negligible, the results obtained by means of this method should be similar to those obtained by means of method SPC. Actually, this does not seem to be completely the case because dilution that induces a lower vapor pressure of VOCs, some organic compounds present in solid or liquid aerosols in the gas effluent are partially or completely volatilised after dilution at ambient temperature. Taking into account the low dilution ratios inherent at this method (about 10 to 20) and the still high temperatures in the dilution tunnel (30 to 50°C approximately), the behaviour of aerosols is not identical to the actual behaviour of the aerosols emitted in ambient conditions by a domestic wood-burning appliance (where the dilution ratio considered is quite higher and the temperature is lower). The chemical reactions that occur outdoor do not have enough time to take place in the dilution tunnel.

Method SPFID (solid particles and flame ionization detector) aims to give results in real-time but requires a robust correlation between TVOC and the condensable fraction, which is not the case at the present time (see below). In addition, FID measurements are sensitive to implement on this type of effluent loaded with condensable organic compounds. These species may easily condense in case of a cold spot along the sampling line or in the analyzer with the risk of obtaining more scattered results. Correlations between these methods have been looked for in United States and in Europe (CEN work), but at the moment no reliable correlation exists.

Indeed, in Europe, tests to prepare a European standard were carried out in 2011 in order to compare methods DT, SPC and SPFID. The good correlation observed, in particular, between the condensable fraction made up of organic compounds of high molecular weights and the volatile organic compounds presents signs of weakness:

- very low number of tests carried out and operating conditions tested,
- different correlations were observed in other studies (e.g. EN PME TEST project: "Common European method for the determination of particulate matter emissions of solid fuel burning appliances and boilers"),
- the wide confidence intervals linked to these measurements,
- the measuring principle of the flame ionization detector. The existence of a very good correlation would suppose that the proportion between volatile organic species and condensable species remains constant and that the response factors from one analyzer to the other are identical. This is generally not the case.

Consequently, the method SPFID is not appropriate to establish particle emission factors. The tests carried out in the framework of the EN PME TEST (Fraboulet et al., 2015) project confirm that even if the TSP evolution and combustion gases are in good agreement, it is not possible to establish reliable correlations between these parameters. It confirms the interest of characterizing TVOC in addition to TSP to determine appliance performance, without using it to make an evaluation of the condensable fraction as it was initially proposed by the CEN/TC295.

In the United States - Canada, the correlation established between methods SPC and DT, with a significant number of appliances and operating conditions, in order to compare the results achieved by these two methods is relatively poor (r^2 = 0,52). It could be related to the sampling method where gas inorganic species sometimes interfere to form solid compounds in water. This bias could be avoided using an organic solvent which does not interfere with inorganic gas species to trap the condensable organic fraction.

<u>Which method can be recommended in order to establish emission factors?</u>

Nussbaumer et al., (2010) mentions ratios DT/SP ranging between 2.5 and 10 and ratios SPC/SP between 3 and 6. Consequently, studies of PM emission factors must be linked to the method which was used (see Annex). In the objective to establish emission factors of pollutants, methods SPC or DT, which reflect better the PM emissions by taking into account the condensable organic fraction, should be recommended. Method SPC makes it possible to compare results with those of method SP, largely used up to now.

<u>Towards a European common protocol taking into account secondary organic aerosols?</u>

The determination of wood combustion pollutant emission factors is usually performed by measurements directly carried out at the emission source. In these conditions, the results obtained are not representative of the PM chemical composition and concentrations observed in ambient air due to aerosol ageing between stack and ambient air.

In EN PME TEST project, a method aiming at evaluating the potential of formation of secondary organic aerosols using a smog chamber has been evaluated (Keller et al.). In spite of its interest, this method is not reliable and mature enough yet to be proposed as candidate method at a European level right now but it would be advisable to carry on in this way.

Comparison between field *vs*. laboratory conditions

In addition to the method used, the establishment of representative emission factors also needs to adapt protocols of emission measurement. Indeed, the normative tests as well as the laboratory tests do not fully represent the real conditions of use of domestic wood-burning appliances. These conditions must be taken into account to establish emission factors of pollutants but also to better determine the performances of these appliances. Energy and environmental performance evaluation of wood-burning appliances is carried out under optimum conditions according to European standards (NF EN 13229, 13240 and 14785). The standards tests should be carried out:

- only at nominal output,
- with a chimney constant draught (12 Pascal),
- with a hot combustion chamber, a preliminary wood load (pre-load) beeing burned before the realisation of the test,
- with an optimal wood load which corresponds to the nominal output of the appliance,
- with wood species that minimise emission; beech, charm or birch are in general used,
- with dry wood (humidity : 16% ± 4% of moisture) and small log size (low diameters are used),
- during a rapid combustion (the starting of the test takes place 3 min after the loading), not taking into account the whole combustion steps (starting, rapid combustion, slow combustion and end) but only the rapid combustion at the origin of the weakest emissions. The highest particle emissions take place during the fire ignition and during the end of combustion (too important excess air).
- on new appliances, in private households, over the years, taking into account that joint wear and deformations can take place and generate parasite air intake.

These tests are carried out by a notified body that controls their implementation and this enables to minimise the formation of pollutants (optimal wood load, logs location in order to favour air flow distribution). Too small or too strong wood load induce a high increase of particulate emissions (excess air or incomplete combustion). A too frequent loading of the wood-burning appliance tends to cool down the combustion chamber (too important air intake). These conditions are relatively far away from real conditions of use where users will often operate:

- at reduced output:
  - because their heat requirement does not correspond to the nominal output delivered by the wood-burning appliance,
  - if the appliance has been dimensioned to provide heat during great cold periods,
  - to compensate for a too high chimney draught (see below),
  - if users are not physically present to reload the combustion chamber (before night for example),
- with a variable load and a variable wood quality (size, diameter, logs moisture and wood species). All these parameters play an important role on combustion quality and thus on particulate emissions:
  - quality of combustion is degraded, in the two following configurations: a too important load cools the combustion chamber; air supply will be insufficient,
  - a too weak load cannot maintain the combustion chamber at a high temperature, air supply will be too important,

    - a too important wood moisture also cools the combustion chamber, but a user does not have always the possibility to store wood during 1 to 2 years before using it. Particulate emissions notably increase when wood moisture is beyond 25 to 30%
    - some species, in particular coniferous trees, are at the origin of stronger emissions (coniferous tree > oak > beech),
    - log size influences air paths in the combustion zone and thus on the homogeneity of the combustion chamber temperatures. The larger the log size, the poorer the combustion quality.
- wood-burning appliances are designed to run with a chimney draught lower than 20 Pa. The more important the height of chimney and the lower the outside temperatures, the more important the chimney draught. In some cases, the 20 Pa will be exceeded for chimney higher than 4 meters. An excessive chimney draught (beyond 20 Pa) may create a too important excess air, cause a too rapid combustion and lead to an excessive wood consumption as well as excessive heat losses (very high smoke temperature). To avoid overheating the room, the user has the possibility:
    - either, to reduce the quantity of wood introduced into the combustion chamber, but this may lead to high excess air, to a reduced temperature in the furnace and consequently a degradation of combustion,
    - or to close the air intake, fire is then fed with parasite air what might also lead to a strong deterioration of combustion. To avoid this, it is possible to install on the chimney a regulator but relatively few residences are equipped with it in some European countries. Conversely, a high chimney increases heat transfer and limits particulate emissions by deposit along the stack. The main differences between real conditions and laboratory conditions are summarised in Table A1.

To illustrate the influence of these parameters on pollutant emissions, we can mention Albinet et al., (2015), who used dedicated on-line instrumentation (TEOM, Aethalometer, ACSM and PILS-LC/PAD) to study the dynamic emission of PM and biomass burning tracers (Black Carbon and levoglucosan) in controlled «real» conditions. Results obtained showed that the pollutant emission dynamic and concentrations were closely linked to the combustion conditions (wood load, firing rate) and phases (ignition, pre-load). As an example, BC accounted for the main part of the solid fraction of $PM_{2.5}$ during the all combustion process at nominal firing rate conditions while, at the beginning and at the end of the combustion cycle, the emission of other species (i.e. OC) was important in reduced output conditions (Figure A2). It has also been shown that the emission of the widely used residential wood combustion molecular tracer levoglucosan especially seemed to be emitted under specific combustion conditions (cold start, pre-load, reduced output, large wood load inducing a decrease of the temperature – (Figure A3). Levoglucosan is a very hydrophilic molecule belonging to the family of the SVOCs.

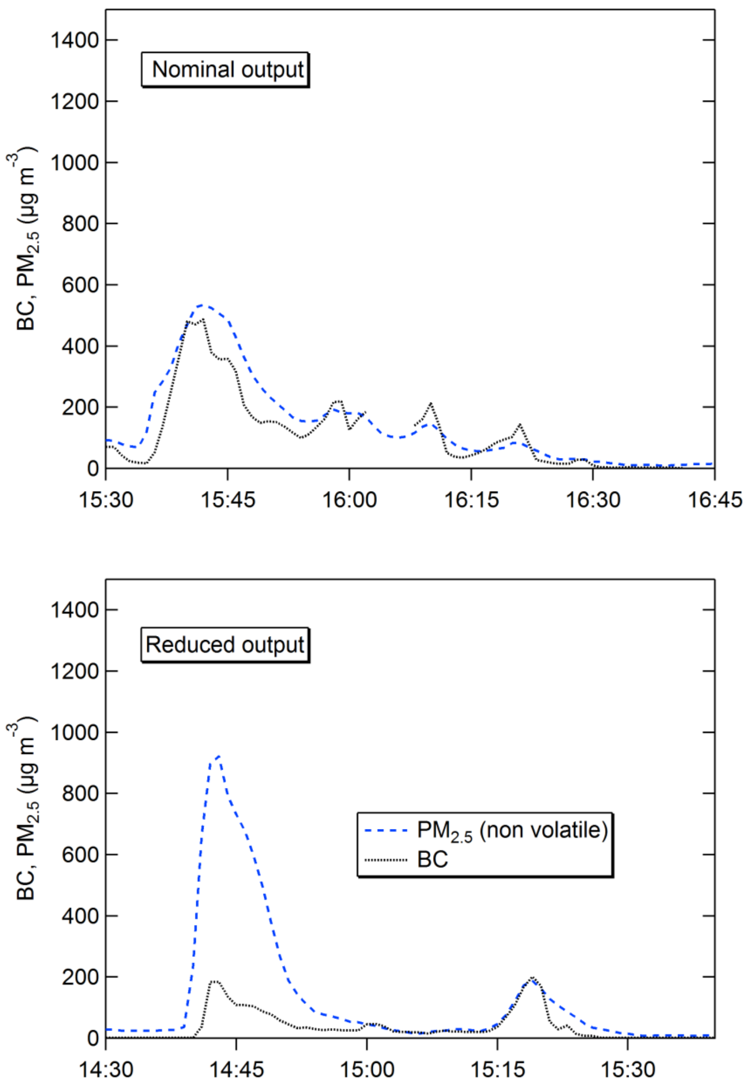

Figure A2. Time evolution of BC and $PM_{2.5}$ (non-volatile) concentrations using a conventional wood-burning appliance at nominal and reduced firing rates (Albinet et al., 2015).

Table A1. Comparison between real and laboratory conditions, with regard to impact on PM emissions.

Real condition (user)	Laboratory conditions	Impact on pollutant emissions
Use at reduced firing rate, under the following conditions : - **oversizing of the combustion chamber,** - **to compensate a too important chimney draught,** - **operation of the appliance without user**	Tests carried out at nominal output	Strong Increase in emissions, low reduction in heat efficiency
Strong chimney draught for stack higher than 4 m (> 20 Pa)	Chimney draught fixed by standard : 12 Pa	Operation beyond the nominal output which led to an overconsumption of wood and a loss in heat efficiency
Chimney draught variable according to the external conditions	Controlled and constant : 12 Pa	Low impact
Chimney height variable	Constant height	Increase in chimney draught, increase in particle deposit in stack and in heat efficiency (slightly) depending on the stack height
Cold start	Hot start	Strong Increase in emissions
Variable wood species	Selected species	Increase in emission with resinous wood
Variable wood moisture	Wood humidity : 16 ± 4%	Increase in emissions with the moisture content of wood, heat efficiency and output decrease
Variable wood load	Optimum load, according to the appliance power	A too weak load or a too strong load lead to a deterioration of combustion quality (cooling of the combustion chamber or lack of oxygen)
Variable log size	Selected in diameter	Impact on temperature and air distribution
Combustion cycle (ignition, rapid combustion, soft combustion, end)	Rapid combustion (European standard)	Highest emissions are noted during ignition and end (too important excess air)
Non-Professional user	Professional user	
Ageing appliance	Only new appliance	Performance reduction over the years (parasite air)

Note: INERIS rédigée pour le groupe de travail « Foyers domestiques » en vue de définir la position française pour la Directive Eco-design, INERIS-DRC-12-126318-14205A, 2012

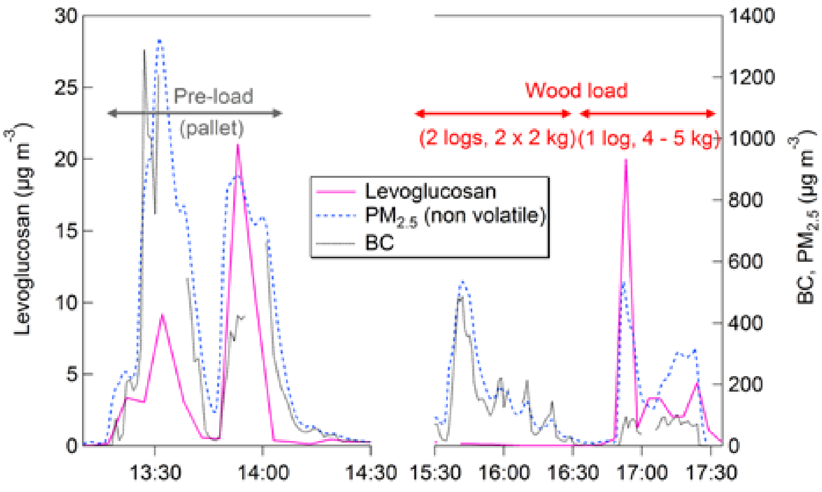

Figure A3. Time evolution of levoglucosan, BC and $PM_{2.5}$ (non volatile) concentrations for different combustion conditions and wood loads (nominal firing rate, wood load: beech, 12% moisture) (Albinet et al., 2015).