AIR QUALITY EXPERT GROUP

The Potential Air Quality Impacts from Biomass Combustion

Prepared for:
Department for Environment, Food and Rural Affairs;
Scottish Government; Welsh Government; and
Department of the Environment in Northern Ireland

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This is a report from the Air Quality Expert Group to the Department for Environment, Food and Rural Affairs; Scottish Government; Welsh Government; and Department of the Environment in Northern Ireland, on the potential air quality impacts from biomass combustion. The information contained within this report represents a review of the understanding and evidence available at the time of writing.

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Freephone Air Pollution Information Service	0800 556677
Internet	http://uk-air.defra.gov.uk

PB14465

Terms of reference

The Air Quality Expert Group (AQEG) is an expert committee of the Department for Environment, Food and Rural Affairs (Defra) and considers current knowledge on air pollution and provides advice on such things as the levels, sources and characteristics of air pollutants in the UK. AQEG reports to Defra's Chief Scientific Adviser, Defra Ministers, Scottish Ministers, the Welsh Government and the Department of the Environment in Northern Ireland (the Government and devolved administrations). Members of the Group are drawn from those with a proven track record in the fields of air pollution research and practice.

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- Provide judgements on the quality and relevance of the evidence base;
- Suggest priority areas for future work, and advise on Defra's implementation of the air quality evidence plan (or equivalent);
- Give advice on current and future levels, trends, sources and characteristics of air pollutants in the UK;
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Contents

Executive Summary

The Issue

- Biomass features within several pathways to a low carbon economy and is an increasing source of secondary domestic heat.
- Measurements and inventories suggest that particulate matter (PM) from biomass burning is on the increase.
- A range of incentives that encourage the use of biomass burning for power and heat generation could have adverse air quality impacts in particular around PM and nitrogen dioxide (NO_2).

Assessment of evidence base

- National energy statistics as used within the National Atmospheric Emission Inventory (NAEI) show an increasing trend in the combustion of wood. The 2012 version of the NAEI estimates that the contribution of biomass burning to UK PM_{10} (particulate matter with a diameter less than 10 µm) emissions rose from less than 5% in 1990 to over 17% in 2012, with the largest increases occurring in the domestic sector.
- The NAEI suggests a far lower contribution of biomass burning to UK total NO_x emissions (~1%), although the increasing use of solid biomass boilers for primary heat or combined heat and power applications may influence NO_x concentrations at a local scale. The installed capacity of solid biomass boilers registered in England under the non-domestic Renewable Heat Incentive (RHI) increased by a factor of 4 between April 2014 and Feb 2016 to a total of 1651 MW. The non-domestic scheme is currently dominating the output from biomass combustion under the RHI when compared to the domestic scheme. Over half the output comes from small commercial boilers of less than 200 kWth capacity, but when summed across the whole of the UK, the total capacity of 2347 MW registered under the non-domestic scheme, exceeds the typical capacity of a large power station of around 2000 MW.
- Atmospheric measurement datasets for wood smoke are currently of insufficient duration to allow the establishment of long term trends in airborne concentrations. However, available ambient measurements from recent research campaigns show considerable (6-25%) contributions of wood burning to PM in urban areas during winter. The majority of the PM (>90%) is likely to be within the $PM_{2.5}$ (particulate matter with a diameter less than 2.5 µm) fraction.
- Much of the potential impact on air quality comes from small-scale domestic burning which is the most uncertain source sector in terms of emissions estimates.
- The implementation of the Eco-design Directive will provide lower PM emission limits than the UK Clean Air Act by approximately a factor of 2 for domestic stoves and a factor of 6 for domestic scale boilers. These new limits still suggest that domestic stoves will have higher PM emissions factors per unit energy compared to larger scale combustion devices burning biomass. In addition, the limits will apply only to

new stoves and hence their impact on UK total emissions from the domestic sector will depend on the turnover of appliance stock and what the stoves are replacing. Without incentives the replacement of existing stoves is likely to be slow since domestic stoves have a long operational life time. In addition, research studies have shown that real world emissions can be higher than legislated limits due to a number of factors including fuel quality and operator behaviour.

- Domestic emissions of wood smoke have an especially high impact on ground-level concentrations relative to those from elevated sources, and if current trends in domestic emissions continue upwards, it is inevitable that there will be an increase in airborne concentrations of wood smoke which will consequently lead to increased population exposures.
- Wood smoke is already a significant contributor to population weighted $PM_{2.5}$ exposures and is likely to make an increasing contribution into the future for the above reasons, and as emissions from some other sources decline.
- The regulatory framework for the control of biomass burning emissions is complex. This presents a significant challenge in pulling together, in a coherent and self-consistent way, the current regulatory controls and emission outcomes over the range of biomass burning emitters.

Overall recommendations

- Assessing the impacts of biomass burning on UK air quality requires action to:
    - improve long term ambient measurements of biomass combustion tracers;
    - reduce uncertainties in inventory estimates particularly around small-scale burning, including the assessment of real world emissions.
- Furthermore, there is a requirement for action on:
    - assessing the developing impact of biomass combustion on air quality and public health;
    - assessing the effectiveness of current and future emission controls and legislation;
    - further research into the development of suitable cost effective PM control methods for the domestic sector, particularly those which could be retro-fitted;
    - assessing whether providing public information and guidance on solid fuel stove operation could help to reduce real world emissions from small-scale devices.

1 Overarching issues, policy and research questions

Based on the 2013 annual submission of the GHG inventory of the European Union (EEA, 2013a), a recent European Environment Agency (EEA) report suggested that burning biomass for home heating increased by 56% within the EU between 1990 and 2011, raising serious concerns for impacts on human health due to the resulting emissions of pollutants such as particulates and nitrogen oxides (EEA, 2013b). This increase may be for several reasons. The use of biomass can potentially lower fuel costs where the wood/biomass is recovered locally, and may be increasing in response to fuel poverty issues. Domestic wood burning for the provision of secondary heating within urban areas is also increasingly becoming part of a lifestyle choice (EPUK, 2013).

The lack of particulate filters or other controls on domestic burners, coupled with increasing usage, means that households are now estimated to be the main source of fine particulate matter emissions in the EU (EEA, 2013b, 2014). In addition, in response to the Climate Change Act of 2008, the UK is committed to reducing greenhouse gas (GHG) emissions by at least 80% in 2050 from 1990 levels. Achieving such targets will require large shifts to lower carbon energy supplies across the UK, with sustainable bioenergy featuring within several of the proposed pathways to a low carbon economy set out by the Department for Energy and Climate Change (DECC, 2010). A range of government incentives to encourage the use of renewable low carbon fuels for power and heat generation are therefore also leading to increases in biomass combustion at the larger scale. Increases in wood and waste material combustion across a variety of scales from domestic to large scale power generation are therefore likely to have an impact on air quality which may grow with time and currently is uncertain. A number of questions will be addressed within this report:

- Is there is a well-developed position with respect to the current range of, and likely changes in, source distributions from biomass burning?
- Are biomass sources well represented within current UK inventories and is there any evidence for a future inventory problem?
- Will the impact of future levels of biomass burning, with proposed policy measures in place, lead to a significant change in ambient concentrations of major pollutants and further in population weighted mean exposure? The major pollutants of concern to be addressed within the report will include nitrogen oxides ($NO_x = NO + NO_2$) and particulate mass concentrations PM_{10} and $PM_{2.5}$.
- How strong is measurement evidence in relationship to pollution from biomass burning?
- Does measurement evidence currently suggest an increasing trend in the contribution of biomass burning to PM?
- Will the current set up of the monitoring networks allow us to assess future trends in the impact of biomass burning on air quality?

- Are the risks well assessed with respect to the introduction of biomass-based heat provision and combined heat and power (CHP) systems into the urban environment within UK?

2 Definition of biomass emission sources

Emissions from biomass combustion from human activities can cover a wide variety of sources. Assessing the air quality impacts will be mainly driven by the definitions of biomass sources used in the National Atmospheric Emissions Inventory (NAEI)[1]. The definitions are mainly dictated by the definitions of sectors used in the reporting of national emission inventories to the United Nations Economic Commission for Europe (UNECE) under the Convention on Long-Range Transboundary Air Pollution (CLRTAP)[2].

Emissions sources which are considered to be included within the definition of biomass burning are listed in Table 1. The NAEI provides the emissions inventory for all biomass sources necessary to fulfil the UK's obligation for reporting to the CLRTAP, and is also able to estimate emissions for some sources for information purposes, but which are not included in the total inventory. Table 1 gives an indication as to whether emission estimates for different biomass sources are made by the NAEI and whether the emissions are included in the national inventories compiled for national and international reporting.

***Table 1:** Sources of emissions from biomass combustion*

Source	**Emissions estimates available**	**Included/Excluded in national inventories**
Electricity generation Co-firing of wood, straw and poultry litter in power stations	Yes	Included
Commercial and Domestic Combustion Wood stoves, boilers, cookers and open fires	Yes	Included
Agriculture Field burning of agricultural residues[1] Muir burn[2]	Yes[1] No	Included

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[1] http://naei.defra.gov.uk/
[2] http://www.ceip.at/ms/ceip_home1/ceip_home/ceip_topnavi/home_emep/

Waste		
Burning of contaminated waste wood	Yes	Included
Municipal green waste combustion	Yes	Included
Bonfires	Yes	Included
Natural Emissions		
Forest & heathland fires etc. (accidental/arson)	Yes	Excluded

[1] The practise of agricultural field burning was banned during 1993. Emission estimates are included in the NAEI for historic years.

[2] This is a form of heathland management associated with the raising/farming of grouse.

In this report, we focus on those sources which are included in the NAEI. A further note on terminology: biomass is referred to as a 'renewable' fuel or 'renewable' energy source; that is, a source of energy that can, in principle, be renewed, for example by replanting or regrowth of the tree or other vegetation from which the biomass was obtained. It is important to note that the terminology 'renewable' fuel is distinct from 'sustainable' fuel or 'carbon-neutral' fuel; evaluation of these latter terms must also take into account the extent to which the renewable fuel source is renewed (e.g. by replanting) and the input of non-renewable energy sources in the life cycle of production and usage of the renewable energy source. This report will not focus on the wider sustainability issues related to biomass use, but rather on air quality impacts.

3 Characteristics and health relevance of emissions from wood burning.

Emissions from the combustion of wood both within power generation processes and residential heating appliances will comprise both gases and particles. These include particulate matter (PM) and gases such as carbon monoxide (CO), carbon dioxide (CO_2), oxides of nitrogen (NO_x), volatile organic compounds (VOCs), sulphur oxides (SOx) and a range of trace species including polyaromatic hydrocarbons. The sulphur content of wood is low and SOx is not likely to pose a particular problem from wood combustion. A comprehensive review of combustion processes and the main formation routes for combustion products is given in Williams et al. (2012). Particulate matter emissions are perhaps of the greatest concern since, particularly for domestic appliances, wood may have replaced gas as a heating fuel leading to a net dis-benefit to air quality in terms of PM. Of the PM produced during wood combustion in an enclosed residential wood burner, 96% was suggested to be in the PM_{10} size fraction and 93% in $PM_{2.5}$ by Houck and Tiegs (1998). This puts the majority of emissions into the respirable fraction of PM and is also of significance in terms of meeting air quality standards.

The size distribution and chemical characteristics of the emissions are likely to vary depending on the type of appliance, burn rates, fuel moisture content, and the type of wood used. Fine et al. (2002) for example, suggested that as much as 74% of the $PM_{2.5}$ discharged from domestic fireplaces burning wood was organic carbon and 1-18% was elemental carbon. Depending on the wood species burned, salts and minerals present within PM could include sodium, magnesium, potassium, calcium, zinc, ammonium, sulphate, chloride, carbonate and nitrate (Houck and Crouch, 2002). The organic compounds present in wood smoke will also vary depending on combustion conditions and wood characteristics such as lignin and resin content. McDonald et al. (2000) in their study of emissions from a range of wood burning stoves and fireplaces, suggested that most of the aliphatic and olefin hydrocarbon VOC content is made up of the C1-C6 compounds (where C1 represents organic compounds with one carbon and C6 compounds with six carbons). Dienes, especially 1,3-butadiene and 1,3-cyclopentadiene, were also found to be enriched in wood combustion emissions and were suggested to derive from the decomposition of terpenoids, although these compounds have been suggested to be unstable in motor vehicle and hydrocarbon combustion exhaust samples (Lipari, 1990; Pelz et al., 1990; Kaiser et al., 1992; Hoekman, 1993). Similar behaviour in samples from biomass emissions may impact on their quantification as well as affecting the measured VOC speciation more broadly. Significant concentrations of dicarboxylic and diterpenoid acids were found in wood smoke in a study by Rogge et al. (1998), particularly from the burning of pine woods. Furans, formed from the decomposition of wood cellulose upon heating, were found in significant quantities from softwood and hardwood emissions in the study of McDonald et al. (2000).

Of particular concern from a human health perspective is emissions of polyaromatic hydrocarbons (PAHs) which may arise via pyrosynthesis during wood burning. McDonald et al. (2000) derived emissions estimates for total PAH emission rates from fireplace/softwood and fireplace/hardwood combustion as 79.8 and 167.4 mg/kg fuel, respectively. The average emission rate for total PAH compounds from wood stoves was 74.7 mg/kg. Predominant species were found to be acenapthylene, naphthalene, anthracene, phenanthrene, benzo(a)pyrene and benzo(e)pyrene. Rogge et al. (1998) found that the smoke from the burning of synthetic logs had the highest total PAH emission rate and the most diverse range of PAHs present when compared to burning pine and oak.

Dioxins (polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs)) are also present in emissions from wood burning with a comprehensive review provided in Lavric et al. (2004). They could be a particular problem from the combustion of treated, varnished or PVC-coated wood. Nestrick and Lamparski (1983) analysed soot scrapings from the chimneys of wood-burning stoves operating in the eastern, central, and western regions of the United States. The average total PCDD/F levels in the chimney deposits were 8.3 ng/kg in the eastern region, 42.1 ng/kg in the central region, and 10 ng/kg in the west. There is obviously significant variability in the levels of dioxins found, which they attributed to the wide variability in the design of the different units, affecting sampling points and/or possible contamination of the fuel wood. The burning of waste wood found in urban areas was highlighted in the review of Lavric et al. (2004) as leading to significantly higher levels of dioxins in the fly ash from a range of combustors.

Other compounds include a variety of aldehydes, phenols, alcohols, ketones, carboxylic acids, methane, ethane, ethanol, formaldehyde, acetaldehyde, acrolein, phenol, cresol, formic acid and acetic acid (McDonald, 2000; Houck and Crouch, 2002).

Mixing between fuel and oxygen within the combustion chamber has a strong role to play in the level of emissions of carbonaceous pollutants, since the formation of soot and PAH may be enhanced in fuel rich zones where incomplete combustion occurs. The conclusion of an extensive review of Williams et al. (2012) was that increased biomass use in small units such as for space heating or cooking could lead to ineffective mixing and potential problems arising from emissions of particulates, particularly carbonaceous smoke. They suggest that biomass burned in larger combustion units leads to better mixing thus reducing carbonaceous emissions which can be additionally removed from the flue gases using a range of different pollution control options. They do however conclude that there remains a potential health problem from submicron inorganic particles. The conclusion of their report was that large combustion units may offer the best route to clean combustion of biomass due to the advantages of large scale efficient flue gas treatment plant.

4 Policy context and emissions limits

4.1 Power generation and large scale heat

A number of policies and initiatives originating both from the EU and UK government may influence the use of biomass as a power generation source within the UK in the near future. As part of a range of European 2020 targets seeking to address climate change, the UK is legally committed to meeting 15% of its energy demand from renewable sources by 2020 and to reducing its Greenhouse gas emissions by 20% compared to 1990 levels. For large scale power generation this is currently incentivised through the Renewables Obligation (RO) which is a trading scheme managed by Ofgem via Renewables Obligation Certificates (ROCs). Electricity suppliers who do not present enough ROCs to meet their obligation must pay a penalty known as the ‘buy-out price’ and hence there are financial incentives to supply electricity from renewable sources, which may include biomass derived fuels. It should be noted that this scheme is set to close to new generators in 2017 and there is currently a cap of 400 MW on the total new build dedicated biomass capacity that can expect grandfathered (i.e. free certificate allocation) support under the RO. The UK Renewable Energy Roadmap (DECC & DfT, 2013) sets out plans for accelerating the use of renewables including biomass electricity and heat. In addition, power generation facilities which use biomass exclusively as a fuel are excluded from the EU Emissions Trading System (EU ETS) and therefore do not require carbon permits, thus further incentivising the use of biomass as a fuel source.

The Fourth Carbon Budget Review (Committee on Climate Change, 2013) gives a brief evaluation of the potential impacts of replacing fossil fuels as a source of power generation and heat as part of the UK’s carbon reduction plans, and predicts an overall benefit of low carbon technologies to air quality at a national level, but does highlight the fact that biomass and biofuel burning gives rise to air pollution. The review points out that the net cost of increasing bioenergy use will depend on the technology it replaces. For example, the use of biomass for heat within buildings and in industry could cause a net dis-benefit to air quality if it were to replace gas a fuel source, whereas the displacement of coal by biomass in power stations such as Drax may have no overall dis-benefit. The review suggests that ways of limiting this potential cost could include emissions standards for boilers, and targeting rural off-grid coal and oil-burning buildings rather than on-grid, densely-populated locations. The review however, focusses on larger scale heat and power generation rather than the domestic sector.

The more detailed report available from ApSimon and Oxley (2013) provides an assessment of the air quality impacts of various carbon reduction plans provided by the Committee on Climate Change using the UK integrated assessment model, UKIAM. Particular concerns are expressed over the use of biomass within the heat sector and in particular related to particulate matter. Scenario projections show contrasting adverse impacts from the use of bio-energy, set against potential benefits from savings in fossil fuel use. The report suggests that large uncertainties exist in the prediction of both, and hence in the overall net effect, depending on assumptions made.

The Energy White paper (DECC, 2011a) sets emissions performance standards (EPSs) for power generation with a suggested initial level equivalent to 450 g CO_2/kWh (at base-load) for all new fossil fuel plant, except Carbon Capture and Storage (CCS) demonstration plants. This could provide further incentive for increased use of biomass fuel sources since biomass will be zero rated under the EPS system.

For existing plants using biomass, the control of emissions depends on the size of the installation. The regulatory regimes that apply to solid non-waste biomass appliances are:

a) For large scale installations (over 50 MWth capacity): Regulation through the Industrial Emissions Directive (IED, 2010/75/EU) administered by the Environment Agency (covering England), Natural Resources Wales, the Scottish Environment Protection Agency and the Northern Ireland Department of the Environment.

b) For large installations (of 20 to 50 MWth capacity): Individual units are regulated by the Scottish Environment Protection Agency or local authorities in England and Wales.

The EU is also proposing a new “Medium Combustion Plant Directive” or MCPD which is likely to come into force by the turn of the decade and will propose emissions limits for all combustion plants with a rated thermal input (RTI) of between 1 and 50 MW. The MCPD will not cover larger plant that is already covered by the IED or smaller devices which will be addressed via the “Eco-design” Directive to be discussed below. It should be stressed however, that the MCPD is still under negotiation.

4.2 Community and domestic scale boilers

At the community/domestic scale the Renewable Heat Incentive (RHI, DECC, 2013a), the Renewable Heat Premium Payment (RHPP) and the Green Deal system are designed to encourage the uptake of renewable heat installations including biomass boilers as part of climate change mitigation strategies. Whilst initially aimed at the commercial sectors the RHI was extended to domestic systems in April 2014. In addition, the “Merton Rule”, so called because it started in the London borough of Merton, is being more widely adopted within national planning guidance and requires 10% onsite renewable energy in large new developments (Merton Council, 2009).

The domestic RHI is a financial support scheme for renewable heat, paid at a set rate per unit of renewable heat produced (kilowatt hour or kWh), for a period of seven years to the owner of the heating system. Biomass systems are supported attracting a tariff payment which was 12.2 p/kWh of renewable heat in Nov 2014 but for new applications in 2016 will be 5.2 p/kWh. In order to attract the RHI, tariff installations must be certified under the Microgeneration Certification Scheme (MCS) and therefore biomass systems will be subject to emissions limits. Fuel also needs to be sourced from an approved supplier which should in theory provide some consistency in fuel quality. The systems currently included are

a) Biomass-only boilers (covering all solid biomass, including logs and chips)

b) Biomass pellet stoves with back boilers

Condensing biomass boilers may be included at a later date depending on the results of tests being currently undertaken on behalf of Defra. Emissions limits required for accreditation are 30 g/GJ net thermal input for PM and 150 g/GJ for NO_x (DECC, 2013b). These have been based on an impact and cost benefit study by DECC which considers the external costs of using biomass within the RHI. The study points out that where biomass displaces electricity or gas fired heat, the air quality impacts will be negative and that if the RHI tariff payments are to compensate for the costs of emissions abatement technologies, that the use of these should be enforced through emissions limits (DECC, 2013b). These emissions limits apply to new installations however, and it may be the case that current installations below 20 MWth do not meet these proposed limits. DECC has set the level of subsidy such that it is only efficient for those currently using solid and oil fuelled boilers to convert. In practice, there is some concern that users will choose to use the RHI in other circumstances. The burning of waste related fuels under the RHI still is required to meet the Environmental Permitting requirements. ApSimon and Oxley (2013) suggest that even when using conservative assumptions, based on the strict compliance of boilers with emission factors defined for the RHI, annual emissions of $PM_{2.5}$ from biomass heating in 2030 are estimated at ~7 kt. This is excluding domestic stoves and can be compared to UK total $PM_{2.5}$ emissions of 77 kt for 2012 as estimated within the NAEI (see section 7.1).

In February 2016 the number of solid biomass boilers registered with Ofgem in England under the non-domestic RHI stood at 9,110, representing a total installed capacity of 1,651 MW and a 4 fold increase since April 2014. For the whole of the UK (i.e. including Scotland and Wales) the total number of accredited new installations in February 2016 numbered 12,927 (Ofgem, 2016a) and 8,061 (Ofgem, 2016b) for the non-domestic and the domestic RHI respectively. In terms of payments to date however, the non-domestic scheme dominates, making almost 90% of payments by February 2016. The non-domestic sector is therefore currently dominating output from biomass burning for schemes registered under the RHI and just over half of this output is from small commercial boilers of less than 200 kWth capacity. Data from the Home Energy Efficiency (HEED) database from the Energy Saving Trust, suggests in excess of 18,000 homes (reported in Feb, 2014) using biomass as a main heating fuel, which suggests that many systems are not registered under the RHI. It should be pointed out that the data from HEED covers only 60% of UK domestic properties. The number of new installations from both these schemes is growing each quarter, and unfortunately, data on the fuel or heat source that the biomass system is replacing is not captured. Therefore it is difficult to assess whether the installations will lead to net positive or negative impacts on emissions of relevance to air quality.

DECC are also encouraging householders installing biomass-only boilers (but not pellet stoves with back boilers) through an optional additional payment of £200 per year to purchase an approved Metering and Monitoring Service Package from the installer. The householder and installer would then be able to view data from their system over the internet related to, for example, efficiency. This could help to improve the performance of the system, but could also be useful in terms of estimating real world emissions from these types of installations.

4.3 Domestic stoves and fireplaces

Outside of the RHI, perceived uncertainties in future gas and oil prices may encourage householders to use biomass combustion as a source of domestic heat through the use of, for example, multi-fuel stoves. Whilst emissions from large scale power generation sources and biomass boilers covered under the RHPP are (or are soon to be) controlled through emissions limits, emissions from multi-fuel stoves are currently only controlled for approved installations in smokeless zones under the Clean Air Act 1993. Fuller et. al. (2013, 2014) suggests that for many, wood burning within urban areas is a lifestyle choice; either for decoration or secondary heating. This suggestion is supported by the increased contributions of wood burning smoke to PM_{10} measured by Fuller et al. (2014) during weekends over a 6 week measurement campaign across London during the winter of 2010. Domestic stoves are also marketed as green, low carbon technologies because they can use renewable fuels. A summary of data in Fuller et al. (2013) from studies undertaken in the cities of London, Paris and Berlin suggests that at times of peak activity, up to 3 μm^{-3} of urban PM_{10} could be currently due to local domestic wood burning with suggestions of ever increasing uptake of domestic solid fuel burners. Fuller et al. (2013) conclude that there is a clear need to fully consider domestic wood burning within the remit of air quality management for major urban areas in order to ensure that increases in wood burning activity “do not off-set substantial investment in schemes to abate traffic pollution”. It is therefore of interest to compare policy, and in particular current and planned emissions limits for small-scale biomass installations, to those for vehicular sources (see section 5).

4.3.1 Implications of the Clean Air Act

Currently, under the Clean Air Act 1993, Defra provides a list of exempted appliances for use in smokeless zones within the devolved regions of the UK. Applications for exemption are handled by Ricardo-AEA. The permitted smoke emissions depend on the thermal output of the stove but vary from 6.7 g/h for a 5 kW appliance to 458 g/h for a 225kW appliance (Ricardo-AEA, 2012). For appliances between 44 kW and 240 kW, a concentration limit of 150 mg/m^3 (dry gas at 0°C, 101.3 kPa and at stack O_2) is also included since emissions in excess of this may be visible and give rise to complaints. Prescribed test protocols are used but there have been recent studies performed in New Zealand that suggest that real world emissions can vary substantially from those achieved during testing; a situation that is analogous to the problem of real world emissions from transport sources (see discussion in section 5).

4.3.2 The Eco-design Directive

The EU Eco-design Directive (2009/125/EC), will require the European Commission to set eco-design requirements for high sales volume energy-related products with significant environmental impacts, and that present significant potential for improvement in terms of their environmental impact without entailing excessive costs. Subsequently developed regulations will impose limits for a variety of local space heaters (LSH) and small boilers for both energy efficiency and emissions of pollutants PM, CO, organic gaseous carbon (OGC) and NO_x. These are likely to come into force on the 1st Jan 2018. As an example, the proposed emission limits for local space heaters (LSH) are shown in Table 2. For PM the different forms of the limit values reflect different test procedures currently in use by member states (MS) as part of national legislation. The values quoted are at rated outputs and therefore do not reflect the influence of device efficiency. A discussion of the influence of combustion device scale on emissions limits in terms of energy inputs is given in section 5.

Table 2: *Proposed Emissions Limits for PM,* NO_x, *OGC in the Eco-design Directive (2009/125/EC*[1]*)*

Appliance type	**Test methodology for PM measurement**	**Fuel type**	**Proposed Limits (PM)**	**Proposed Limits (OGC)**	**Proposed limits (NO_x)**
Closed fronted LSH	Heated filter	Any other than compressed wood pellets	40 mg/m^3 measured at 13% O_2	80 mg/m^3 measured at 13% O_2	200 mg/m^3 measured at rated output and 13% O_2
	Dilution tunnel	Any other than compressed wood pellets	5 g/kg (dry matter)		
Closed fronted LSH	Heated filter	Compressed wood pellets	20 mg/m^3 measured at 13% O_2	40 mg/m^3 measured at 13% O_2	
	Dilution tunnel	Compressed wood pellets	2.5 g/kg (dry matter)		
Open fronted LSH	Heated filter	Any	40 mg/m^3 measured at 13% O_2	80 mg/m^3 measured at 13% O_2	
	Dilution tunnel	Any	5 g/kg (dry matter)		

[1] Based on the working draft of the Eco-design Directive Draft number: WDCF14Oct2014, accessed from: http://www.eceee.org/ecodesign/products/Lot_20_local__room_heating__products/WDCF14Oct2014 No longer available, final documents published at: http://www.eceee.org/ecodesign/products/lot-20-local-room-heating-products/)

5 Summary of emissions limits and comparisons across different sizes of combustion plant

Several different sources of emissions limits for biomass combustion apply at different scales of technology. It is useful to try to compare these on an equivalent energy basis in order to assess the impact of device scale on likely emissions scenarios. The different limit values often use different units and hence certain assumptions have to be made in converting the limits to comparative units. An attempt to make this comparison in terms of g/GJ for PM emissions has been made in Table 3, and for NO_x in Table 4 on an equivalent energy input basis. An overall summary is presented in ***Figure 1***.

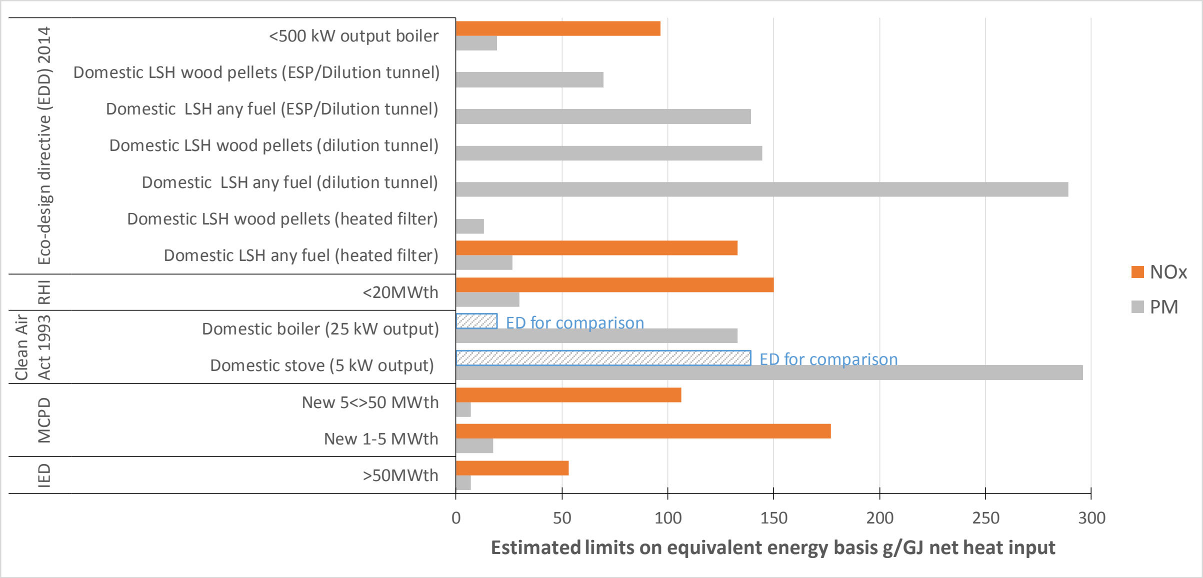


Figure 1*: Comparison of PM and* NO_x *emissions limits for different scales of combustion devices based on an equivalent energy basis (g/GJ) net heat input. The relevant test method within the EDD for the UK is the ESP/Dilution tunnel method and for ease of comparison these limits for domestic stoves and boilers are shown again directly above those based on the UK Clean Air Act. IED = Industrial Emissions Directive, MCPD = Medium Combustion Plant Directive, RHI = Renewable Heat Incentive, EDD = Eco-Design Directive.*

For PM Table 3 and ***Figure 1*** show that notwithstanding differences between test methods, emission limits are generally higher for small appliances compared with larger plant. Whilst efficiencies would also play a role in comparing emissions on an energy out basis, similar conclusions about the effects of scale could be drawn.

A straight comparison of the PM limits for local space heaters within the proposed Eco-design Directive (EDD) with those specified by the current exemption criteria for smoke controlled zones within the UK Clean Air Act is not possible since they are based on different units. However, emissions have been calculated for a 5 kW stove and 25 kW boiler based on the exemption criteria (Ricardo-AEA, 2012), and on an “energy in” basis assuming appliance efficiencies of 80 and 90% respectively (net basis). For domestic scale boilers the Eco-design Directive will lead to quite a large reduction in emissions limits of approximately a factor of 6 in comparison to the Clean Air Act, and in fact the EDD is slightly more stringent for small boilers than the limits imposed in the RHI.

For stoves, neglecting issues around real world emissions, and taking the limit based on the ESP/Dilution tunnel method as the EC interpretation of the UK methodology, Table 3 suggests that lower limits (equivalent to 139 g/GJ) will be imposed by the Eco-design Directive than currently used for appliance exemption within the UK (equivalent to 296 g/GJ). This factor of 2 is not as significant as the reduction for domestic boilers and again highlights the issue of scale of combustion device. Many existing appliances are likely not to meet the new criteria and hence the turnover of stock will be a key issue in whether overall emissions factors will reduce following the implementation of the Directive which will apply to new sales. Several studies of the impacts of stove exchange programmes have shown improvements in indoor quality following the replacement of older technologies with newer ones that meet more stringent emissions criteria (WHO, 2015). However, some caution was suggested from this WHO report in terms of the likely success of such intervention strategies due to overall costs and the long service life of appliances once they have been installed. Early implementation of the emissions limits driven by the Eco-design Directive could however, provide some benefits in terms of emissions from new stove sales.

Table 3: *Comparison of PM emissions limits for different scales of combustion devices*

Size of plant	**Source of emissions limits**	**Limit concentration**	**Units**	**Reference O_2 %**	**Estimated limits on equivalent energy basis g/GJ net heat input[1,2,6]**	**Comments**
>50MWth	Industrial Emissions Directive	20	mg/m^3	6	7.1	IED Chapter 3 (Annex V) minimum requirement for LCP. Chapter 2 BAT AEA for new LCP is 5-20 mg/Nm^3.
1-50MWth	Proposed Medium Combustion Plant Directive					Draft version (Oct 2014) circulated to member states and industry groups (not finalised).
	New 1-5 MWth	50	mg/m^3	6	17.7	
	New >5 MWth	20	mg/m^3	6	7.1	
Domestic stove (5 kW output)	Clean Air Act 1993	6.7	g/hr	-	296	Applies for appliances used in smoke control areas
Domestic boiler (25 kW output)	Clean Air Act 1993	13.3	g/hr	-	133	Applies for appliances used in smoke control areas

<20MWth	Renewable Heat Incentive	30	g/GJ (net input)	n/a	30	Incentive scheme - not a regulatory emission limit.
domestic stove (typical size range 3-12 kW)	Eco-design draft regulation (local space heaters)					Agreed 14/10/14. New products from 1/1/2022. PM limits reflect different measurement techniques and test protocols in national standards and legislation.
	Heated filter[3]	40	mg/m^3	13	26.6	Heated filter method known to understate PM emissions from batch fed, manually controlled natural draught appliances. Limit 20 mg/m^3 for pellet stoves.
	Dilution tunnel [4]	5	g/kg (dry matter)	n/a	289	2.5 g/kg for pellet stoves
	ESP/Dilution tunnel [5]	2.4	g/kg (dry matter)	n/a	139	1.2 g/kg for pellet stoves
<500 kW output boiler	Eco-design draft Regulation (boiler)	40	mg/m^3	10	19.3	Agreed 13/10/14. New products from 1/1/2018.

[1] Conversion based on assumed stoichiometric dry flue gas volume of 253 m/GJ net heat input adjusted for reference oxygen content (AEA, 2012)

[2] Conversion assumes fuel CV for dry wood of 17.3 GJ/tonne (dry basis) from Digest of UK energy statistics (DECC, 2014)

[3] German regulatory method, average of 3 tests on undiluted flue gases at rated output - underestimates PM emission compared to dilution and other techniques as does not collect condensable fraction

[4] Norwegian regulatory method, based on USEPA method, weighted average of single test results at 3 -4 burn rates.

[5] EC interpretation of UK regulatory method, assessed based on average of five results at rated output and five results at part load.

[6] Clean Air Act limit conversions assume efficiency of stove as 80% (net basis) and boiler efficiency 90% (net basis).

Table 4: *Comparison of* NO$_x$ *emissions limits for different scales of combustion devices*

Size of plant	**Source of emissions limits**	NO$_x$ **Limit concentration, mg/m3**	**Units**	**Reference O$_2$, %**	**Estimated** NO$_x$ **limits on equivalent energy basis g/GJ net heat input**[1,2]	**Comments**
>50MWth	Industrial Emissions Directive (>300 MWth)	150	mg/m^3	6	53.1	IED Chapter 3 (Annex V) minimum requirement for LCP >300 MW thermal input. Chapter 2 BAT AEA for new LCP >300 MWth is 50-150 mg/Nm^3.
1-50MWth	Proposed Medium Combustion Plant Directive					Draft version (Oct 2014) circulated to MS and industry groups (not finalised).
	New 1-5 MWth	500	mg/m^3	6	177.1	
	New >5 MWth	300	mg/m^3	6	106.3	
<20MWth	Renewable Heat Incentive	150	g/GJ (net input)	n/a	150	Incentive scheme - not a regulatory emission limit.
3-12 kW domestic stove	Eco Design draft Regulation (closed LSH - stove)	200	mg/m^3	13	132.8	Agreed 14/10/14. New products from 1/1/2022.
<500 kW output boiler	Eco Design draft Regulation, automatic boiler	200	mg/m^3	10	96.6	Agreed 13/10/14. New products from 1/1/2018.

[1] Conversion based on assumed stoichiometric dry flue gas volume of 253 m/GJ net heat input adjusted for reference oxygen content available at: http://uk-air.defra.gov.uk/assets/documents/reports/cat07/1205310837_Conversion_of_biomass_boiler_emission_data_rep_Issue1.pdf

[2] Conversion assumes fuel CV for dry wood of 17.3 GJ/tonne (dry basis) from Digest of UK energy statistics (DUKES, 2014): https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/338752/Annex_A.pdf

In a report prepared as part of the International Energy Agency's project IEA Bioenergy (Bios Bioenergiesysteme, 2011), the difficulties of controlling PM from domestic scale biomass combustors are highlighted. The report compares efficiencies of particle collection devices for residential biomass systems (<50 kW) including electrostatic precipitators (ESPs), catalytic converters, ceramic filters and condensing heat exchangers. ESPs were found to be the most promising technologies for boilers within this report, with collection efficiencies of up to 85%. However, an important conclusion of the study is the lack of available PM control systems for domestic stoves, particularly those with retro-fit options. Table 3 indicates that despite the proposed Eco-design Directive, domestic stoves will continue to make a substantial contribution to particulate emissions across Europe and that research into cost effective retro-fit PM control systems is greatly needed.

Table 4 shows the equivalent comparison across scales for NO_x emissions limits. The differences between small and large plant are much lower for NO_x than for PM, indicating that a move to decentralised provision of energy and heat through smaller combustion devices is less likely to impact on UK-wide NO_x emissions than for PM. However, localised sources of NO_x could still have implications for air quality in cases where emissions sources shift substantially from large power generation sources (usually situated away from populated areas and with tall chimneys) to smaller boilers and stoves (likely to be sited close to demand). We will return to this issue in Section 9.

In order to compare possible impacts to air quality across different pollution sources it is also of interest to compare emission limits for biomass combustion to those from typical traffic sources. In Table 5, estimated PM emissions factors based on the UK relevant limits for local space heaters within the Eco-design Directive (EDD) for domestic stoves (~3.1 g/h assumed for a 5 kW stove), and those based on the current exemption criteria for UK smokeless zones (UKSZ) (6.7 g/h) are compared with vehicle emission rates corresponding to the different Euro standards for diesel road vehicles. The vehicle emission rates for diesel cars are based on the PM emission limit values in g/km for the average speed of the NECD test cycle of 33 km/h. For heavy duty diesel vehicles, the limits are expressed in g/kWh of useful energy from the engine. Using typical fuel consumption rates of around 390 g fuel/km for a >40 tonne HGV at an urban speed of 33 km/hr and a brake-specific fuel consumption rate for a diesel engine of 250 g fuel/kWh, the PM emission limits correspond to the vehicle emission rates for HGVs in g/h shown in Table 5.

Table 5: *Comparison of PM estimated emission rates in g/h based on standards within the Eco-design Directive and UK smokeless zone exemption criteria for a 5 kW stove compared to those based on Euro standards for diesel vehicles*

Diesel Passenger Cars				**Heavy Goods Vehicles**			
	Euro 3	Euro 4	Euro 5/6		Euro III	Euro IV/V	Euro VI
	(2000)	(2005)	(2009/2015)		(2000)	('2005/2008)	(2014)
Euro standards (g/km)	0.05	0.025	0.005	Euro standards (g/kWh)	0.1	0.02	0.01

Vehicle emission rate (VER) (g/h) based on Euro standard emission limit at 33 km/h speed	1.65	0.83	0.17	VER (g/h) for >40t HGV based on Euro standard limit value and Brake-specific diesel fuel consumption at 33 km/h speed	5.2	1.0	0.5
EDD PM emission limit (g/h) – wood log stove by dilution tunnel – 2.4g/kg dry matter[1]	3.1				3.1		
UK Clean Air Act emission limit (g/h)[2]	6.7				6.7		
EDD/VER	1.9	3.7	18.2		0.6	3.1	6.2
UKCAA[2]/VER	4.1	8.1	40.6		1.3	6.5	13.0

[1] EDD - Eco-design Directive. There are two dilution tunnel limits in the Eco-design Directive, with 2.4g/kg being the UK method but at a more stringent standard than the CAA Smoke Control Area limit. 5 g/kg is the NS3058/9 Norwegian test method limit which corresponds to 6.5 g/h, [2]UKCAA - UK Smoke Control Area limits based on appliance exemption criteria in the Clean Air Act.

Table 5 compares emission limits for stoves and diesel vehicle that are expressed in different ways, but it is important to recognise that neither the regulatory limits for stoves or diesel vehicles reflect ‘real world’ operational emission rates; these are highly variable. Under real world driving conditions, emission rates for vehicles may be higher or lower than shown in Table 5 and real world emission rates for stoves will also vary with mode of operation. It is also the case that emissions only occur from stoves and vehicles when ‘under operation’ and stoves are not run all year round or over the same period that a vehicle is used. Nevertheless, it is clear that proposed emission factors for domestic local space heathers are not keeping pace with the standards introduced for diesel vehicles. Table 5 shows the ratio of g/hr emission limits for domestic stoves to limits for diesel vehicles on a comparative basis. On an equivalent hourly operational basis, a domestic stove is likely to emit a much higher mass of PM than a diesel vehicle meeting Euro 4/IV standards or above and greater than 50% of the PM emissions from a large Euro III HGV as shown in Figure 2**.** Furthermore, emissions from domestic space heaters will be seasonal. Within winter months, it might be expected that a stove will operate over several hours each day compared with much shorter commuter journeys by cars. A recent survey on domestic wood use carried out by DECC (2016) suggests a UK average of 22 hours per week usage rate for domestic wood burning appliances during the winter of 2013-14 compared to 10 hours per week in the summer.

Without further action, increased uptake of domestic stoves within cities could lead to a rising contribution of biomass burning to urban particulate concentrations within winter months, particularly during evening hours (EPUK, 2013).

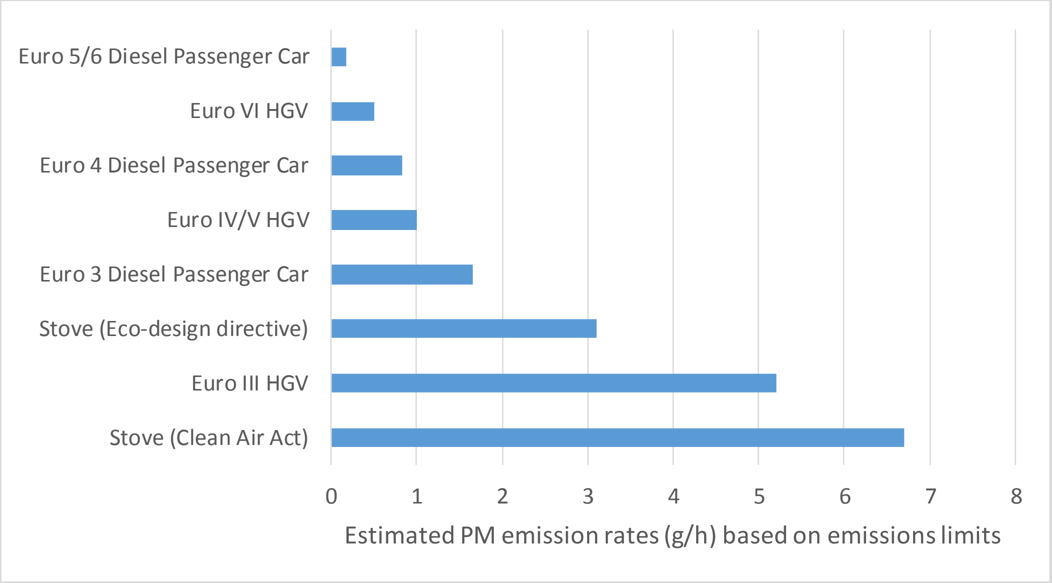


***Figure 2**: Comparison of estimated PM emission rates from a 5 kW stove (g/h) compared to typical exhaust PM emissions (g/h) from vehicles based on emissions limits as described in Table 5.*

Dispersion of course also plays a role in determining ground level concentrations that result from emissions sources and the height of the emission source will depend on the scale of the combustion device. Larger plants are likely to emit from taller chimney stacks than domestic devices. On an equivalent energy basis therefore, the impact of domestic devices on ground level pollution is likely to be larger than for larger plant.

In general, ambient concentrations of pollutants resulting from emissions from domestic wood stoves, large plant or a road vehicle will depend on the location of the sources and subsequent dispersion in addition to the corresponding rates of emission. When compared to exhaust emissions, those from domestic stoves are likely to be higher in the atmosphere, with flue design controlled by Building Regulations (2010). They are however, likely to be emitted from roof top chimney flues and therefore dispersion may be affected by downwash resulting from the presence of the building. Differences in PM emissions between different sources were discussed above. In this section, differences in ground level concentrations due to the source location and dispersion are discussed. In order to investigate the effects of dispersion features alone, we focus on an illustrative example using dispersion modelling with a constant unit (1 g/s) emission rate for all sources; more detailed calculations would be required to assess the impact of the variation in emissions during the day or with season. We consider the following scenarios:

Emissions from a wood stove. The chimney is represented as a point source at 5.6 m above ground for a single storey house of height 5 m, and at 10.6 m for a two storey house of height 10 m. Cases with and without the impact of the building on flow and dispersion are modelled in order to highlight the influence of downwash. The chimney exit is 0.6 m above the building, consistent with guidance from the Building Regulations (2010) for the required minimum chimney height. Other chimney parameters are used as follows: source diameter 20 cm, exit velocity 10.3 cm/s and flue gas temperature 100 ^{0}C. The building dimensions are assumed to be 5 m or 10 m high, 5 m wide and 10 m long (east-west direction). The chimney is located at the west end of the building.

Emissions for a road vehicle. An exhaust emission source of PM is assumed for a single vehicle with speed 20 km/hr (so that the emission is spread along the road) as a volume source with height 1 m and width 3 m to represent a single carriageway.

Emissions from a large power plant. For comparative purposes, a stack similar to that found at Drax power station has also been modelled. The stack has height 259 m, diameter 9.1 m, and effluent gases have an exit velocity of 27 m/s and flue gas temperature of 100 ^{0}C.

The dispersion model used was ADMS 5. Each scenario used one year of meteorological data from Heathrow airport for 2012 and a surface roughness length of 0.1 m. In each case the emission rate was assumed to be constant and the annual average concentration was calculated. In the case of the road source, receptor points within the road were not considered; the closest receptors to the road were at a distance of 1 m. The resolution of the modelling output points for the point sources is 5 m (with the nearest grid point 5 m from the stack), except for the power station modelling which used a 400 m resolution (with the nearest grid point 400 m from the stack). For the road source the resolution perpendicular to the road is 1 m; the concentration is constant parallel to the road. Ground level concentrations (glc) were modelled in all cases.

Table 6 shows the maximum annual mean glc for each scenario and also the ratio of that maximum to the maximum for the 10.6 m domestic stove source with building (10 m) whilst Figure 3 and Figure 4 show contours of annual mean glc. For identical emission rates, we see that the domestic stoves for the 5 m and 10 m houses result in much greater impact than a single car. However it should be noted that this effect would in reality be offset by the greater density of vehicles on roads compared with the density of houses with domestic stoves. Also of note is the impact of buildings on flow and dispersion; where buildings are present, the downwash of the plume causes much higher maximum ground level concentrations compared to the cases without buildings. For the 10 m house with the effect of building downwash included, Figure 5 shows the dependence of the maximum hourly average concentration on wind speed and atmospheric stability. The maximum glc is highly dependent on wind speed, increasing with decreasing wind speed but less dependent on stability; this is in contrast to the case without building effects where the maximum glc from an elevated source is highly dependent on stability typically occurring in unstable convective conditions.

We also see the comparatively very small impact of a power station with a tall stack on maximum glc. This suggests that the large stack height is sufficient to lead to substantial dispersion of the emitted plume before it reaches ground level when considered on an annual average basis. When considered alongside the relative emissions limits for PM for different scales of plant in Table 3, these results show that the use of biomass stoves within domestic properties is likely to have orders of magnitude larger effects on maximum local ground level PM concentrations than large scale power generation from biomass sources on an equivalent energy input basis. Considering emissions rates for a typical stack at a large power station based on data provided by regulators to be 20 g/s, and that for a 5 kW stove to be 1.39 x 10^{-3} g/s (based on 5 g/h) we can compare the relative impact on predicted maximum glc for each case. The resulting concentrations for the stove for a 10.6 m point source on top of a building, are 2.9 times that for the assumed emissions from a typical power station. Without the downwash effects this drops to 0.029 times that for the assumed emissions from a typical power station and highlights the importance of downwash effects for short roof mounted stacks that would be typical for domestic installations. The area of influence of the plume from a single domestic stove is of course much smaller than that of a major power station source and the results shown in Table 6 are based on localised maxima. However, sales figures suggest that the number of domestic stoves is growing and it would be useful to evaluate the potential cumulative impacts of a number of stoves across an urban neighbourhood.

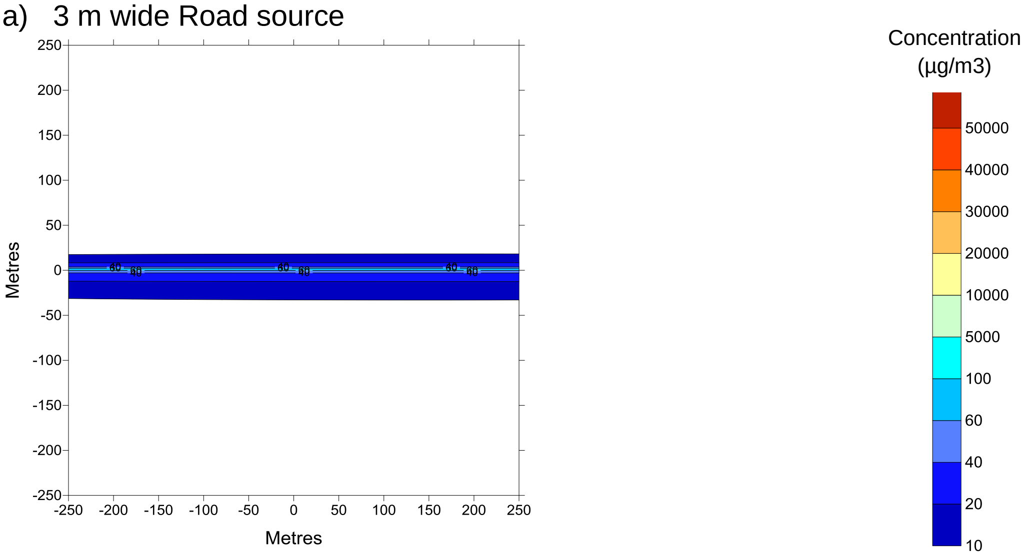
The potential for reducing the impact of downwash by increasing the chimney height for the '10 m house' is illustrated in Figure 6. This shows the variation in the maximum glc as a function of chimney height for three meteorolgical conditions. Increasing the chimney height from 0.6 m to 2 m reduces the maximum glc by 33%, 59% and 62% respectively for the convective, neutral and stable conditions modelled.

Table 6: *Maximum annual average ground level concentration for each modelling scenario*

Scenario	**Maximum ground level annual average concentration (μg/m^3) for unit release (1 g/s)**	**Ratio to results for 10.6 m point source with building**
road source	52*	0.01
5.6 m point source with building	8,282	2.32
5.6 m point source without building	154	0.04
10.6 m point source with building	3,577	1
10.6 m point source without building	28.8	0.01
Power Station	0.086	2.4E-05

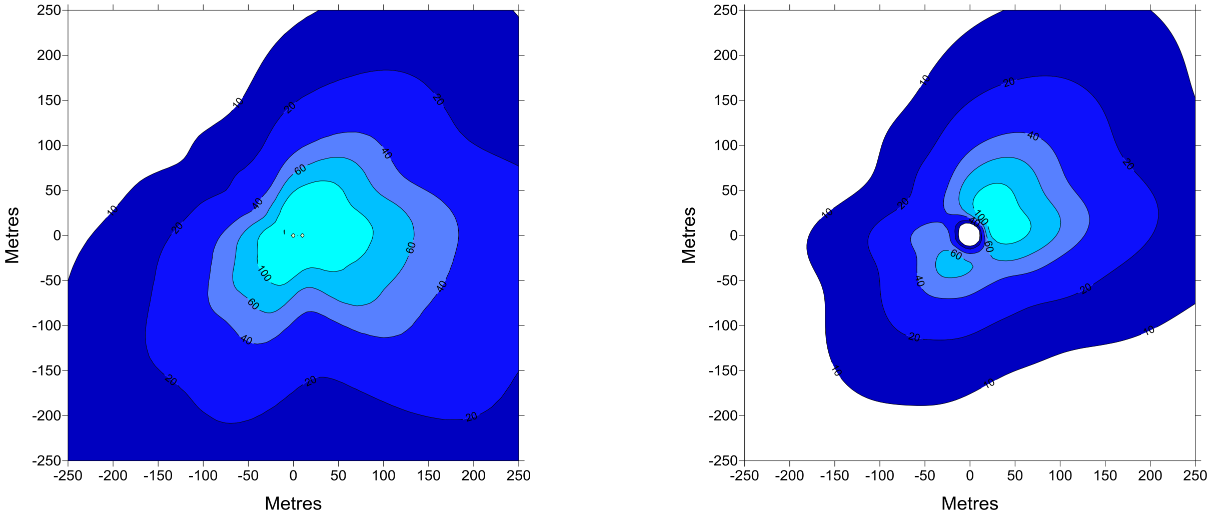
*Not including receptors within the road source

a) 3 m wide Road source



b) 5.6 m point source with building

c) 5.6 m point source without building



d) 10.6 m point source with building

e) 10.6 m point source without building

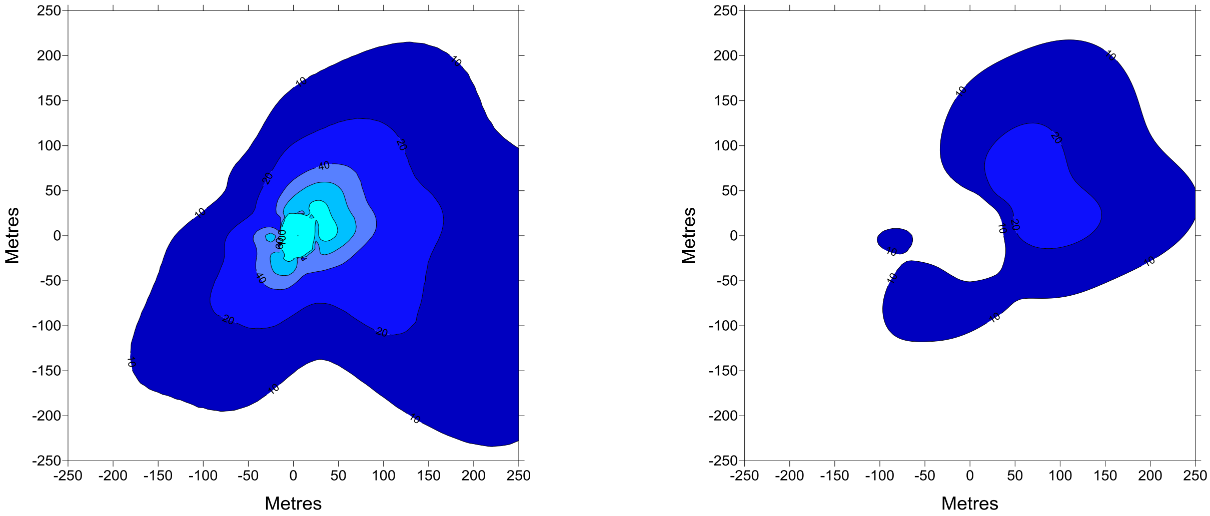


Figure 3*: Annual average ground level concentrations calculated for a release rate of 1 g/s for the different case study scenarios*

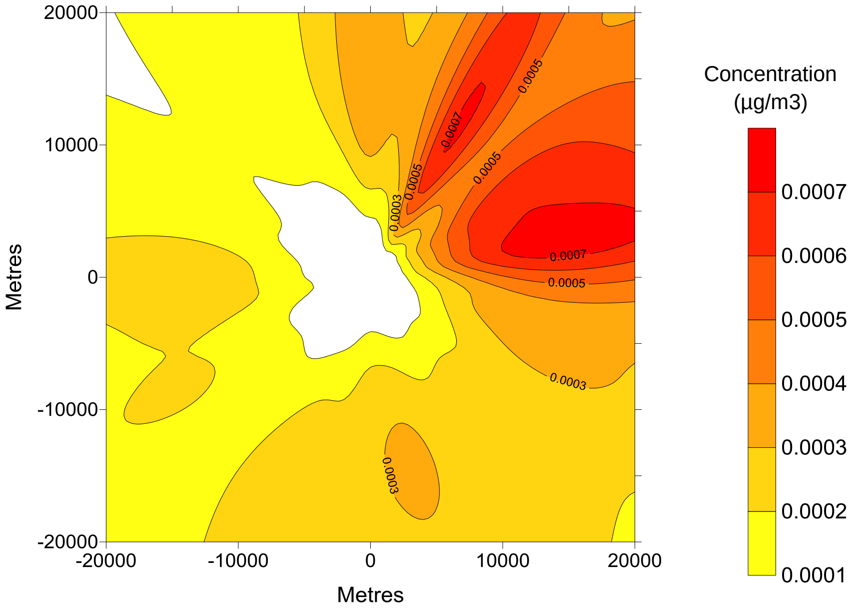


***Figure 4**: Annual average concentrations calculated for a release rate of 1g/s from a large power station*

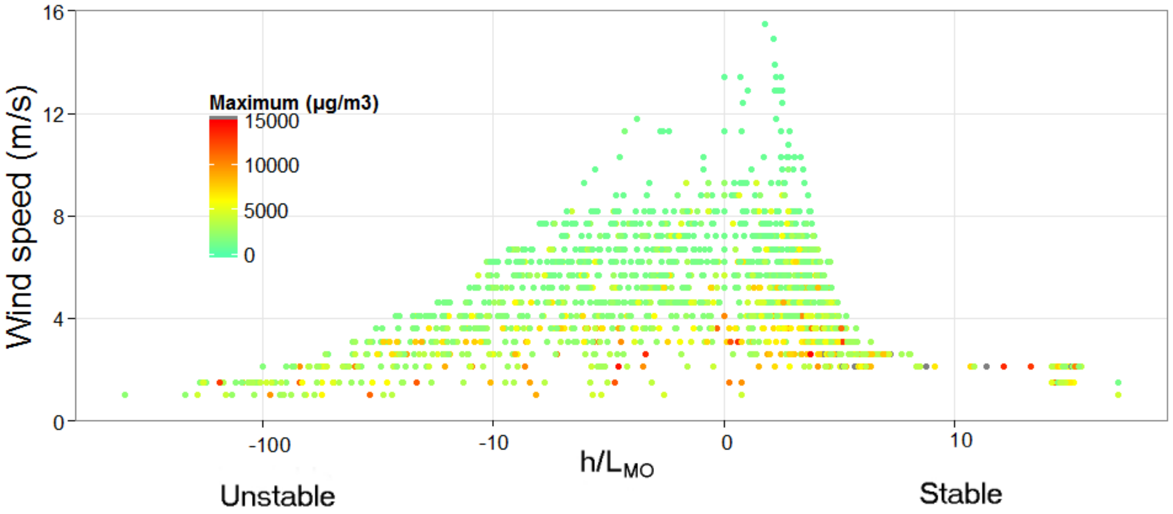


***Figure 5:** Dependence of maximum ground level hourly average concentrations on stability and wind speed for a 0.6 m chimney above a 10 m building for 2012 meteorology (January, April, July and October). The stability is represented by h/L_{MO} where h is the boundary layer height and L_{MO} the Monin-Obukhov length. Stability ranges from very convective (large negative h/L_{MO}) to neutral (h/L_{MO} =0), through to stable (positive h/L_{MO}).*

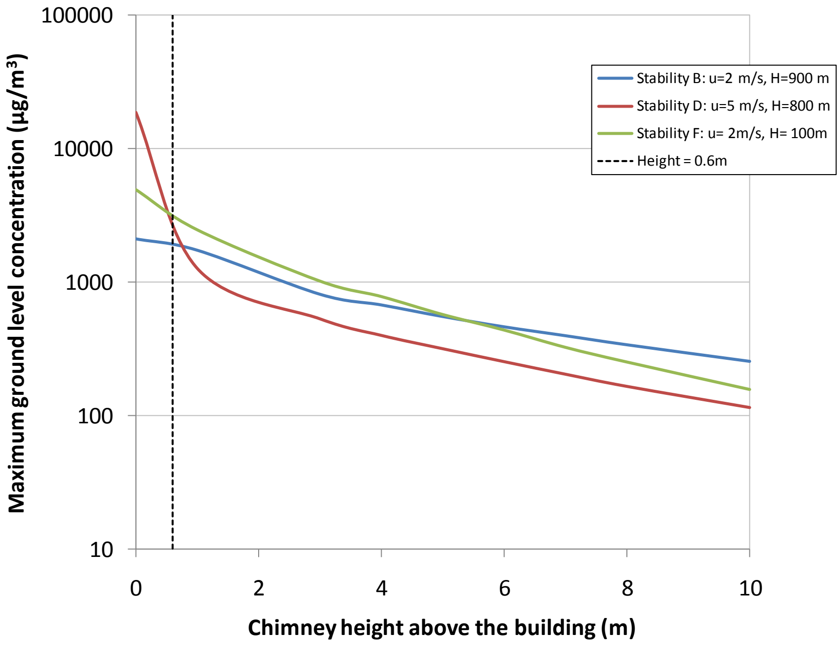


Figure 6: *Maximum ground level concentrations as a function of chimney height for three meteorological conditions for the 10 m building modelled using ADMS 5. Stability categories B, D and F represent unstable, neutral and stable conditions respectively. 0.6 m above the building is the minimum height from the building regulations.*

6 Estimating Emissions from Biomass

6.1 Introduction

Emission estimates for biomass combustion are the most uncertain of any major fuel type covered in the UK inventory[3]. This is because the amount of different types of fuel used in different sectors, especially the residential sector, is not well-known and is challenging to estimate, and also because emissions of most pollutants are highly dependent on combustion conditions, meaning that emission factors can vary greatly, even for similar sources. It is also difficult to estimate the extent to which biomass is used in older equipment rather than in modern equipment that is considerably more efficient and less polluting. The recent survey by DECC (2016) suggest that >50% of closed domestic stoves are between 1-5 years old, which contrasts with the age of open fires which tend to be much older and linked to the age of the property.

This section discusses the challenges in estimating emissions from biomass combustion for large point sources down to domestic and small-scale combustion sources, the basis for the NAEI estimates and their uncertainties.

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[3] http://naei.defra.gov.uk/

6.2 Biomass fuels

In recent years there has been a substantial increase in the use of biomass in larger plant for electricity generation and in domestic and small-scale combustion appliances.

The term biomass is used to include a range of different fuels and forms. Whilst biomass is typically the same as wood (in a range of different forms), there are other non-wood biomass fuels. The following lists a range of fuels included in the NAEI under the heading of biomass:

- Wood logs
- Wood chips
- Wood pellets
- Sawdust/fines
- Waste woods
- Non-woody biomass (e.g. miscanthus, straw)
- Food wastes
- Bioliquids
- Biogases

The EN 14961 suite of standards provide specifications for non-industrial fuel (eg EN14961-2 A1 or A2 wood pellets) but there are also internationally-traded specifications for industrial wood pellets (as used in power stations for example). EN 14961-1 lists the following different types of solid biofuel:

- Woody biomass
- Herbaceous biomass
- Fruit biomass and blends and mixtures

Fuels can also be transformed (e.g. gasification) or treated (physical treatment, liquefaction, torrefaction). It is also possible that fuels require further sub-division when estimating emissions. For example, waste wood needs to be divided into ‘clean’ and chemically-treated wood when estimating emissions of some metals and Persistent Organic Pollutants (POPs).

Estimating emissions requires estimates of amounts of these fuels used by different sources (activity data) and emission factors, taking account of the types of combustion technologies and emission controls.

The primary source of activity data used in the UK inventory is the “*Digest of UK Energy Statistics*” (DUKES) published annually by DECC (2014). For example, this source provides a time-series in total UK domestic and industrial wood combustion, energy from waste combustion and animal- and plant-based biomass combustion for generating heat or electricity each year.

6.3 Large and Medium Point Sources

Information on biomass consumption in the electricity generating sector and other large point sources are sourced from EU-ETS data.

Larger combustion plant are required to comply with both national and international legislation that imposes limits on emissions. As a result, such plant are expected to be well-managed, often utilising various emissions control technologies. Similarly a number of smaller combustion plant are captured under UK ‘Part B’ regulation, and therefore are required to comply with emission standards. The proposed EU Medium Combustion Plant Directive is expected to result in future controls for combustion plant of 1-50 MWth (for PM, NO_x and SO_2).

There are requirements for large point sources to report emissions estimates to the regulators. The NAEI compiles estimates of emissions from power stations burning wood, poultry litter, straw and waste using operators’ data reported to the Environment Agency’s Pollution Inventory (PI), the Scottish Pollutant Release Inventory (SPRI) or Northern Ireland’s Pollution Inventory (NIPI). These may be used in combination with literature emission factors. Comprehensive data on the amounts of biomass co-fired in power stations are readily available and emission factors are considered to be relatively more accurate than many other biomass sources because emissions are typically directly measured.

The procedures for reporting emissions data for large point sources to regulators are well established, with strong QA/QC systems in place, and both the reported biomass consumption and emissions data are therefore considered to be robust and of a good quality. Due to the lack of detailed source data available prior to the mid-1990s, the emission factors and UK emission estimates for that early period in the inventory time-series are associated with higher uncertainty than the factors and estimates for later years.

In future, the availability of RHI data for boilers burning wood may help to reduce uncertainties in emissions estimates for sub-ETS installations. In addition, operator and national reporting required by the proposed EU Medium Combustion Plant Directive is expected to provide improved emissions estimates for 1-50 MW thermal input plant (EU 2015). Article 11 of the Medium Combustion Plant Directive (MCPD) will require operators to monitor emissions, except as allowed for in MCPD exemptions – principally about plant with low operating hours. Furthermore, by 1 October 2026 Member States will be required to provide “*an estimate of the total annual emissions of* SO_2*,* NO_x *and dust from medium combustion plants, grouped by plant type, fuel type and capacity class*.”

6.4 Domestic and Small-scale Combustion

6.4.1 Estimating Fuel Consumption

The use of wood in the domestic sector has traditionally been a challenge for emission inventories in general, because there is no simple method for collecting data that allows the determination of the annual consumption of wood among different types of combustion devices. Whilst UK census data provides information on a household’s main central heating system it does not currently collect information on sources of fuel for secondary heating. In addition, solid fuel is treated as a single category and there is currently no differentiation between coal and wood. Indirect indicators of activity trends in the UK (for example Clean Air

Act exemption applications and stove industry sales figures) suggest that growth in the use of residential wood burning stoves is large and ongoing[4].

Other European countries undertake extensive surveys to estimate the annual quantities of wood burned in the residential sector. Studies in Scandinavian countries have shown that more extensive surveys do provide better results (TFEIP, 2010). New Zealand gathers census data on fuel type used for heating in all houses (Scott, 2005). An older UK survey study by the Forestry Commission (1997) showed that 36% of homes with wood heating used it regularly but only 8% used wood as the main household fuel. Of the responders 50% gathered wood themselves and hence this usage would not be not well captured by national sales statistics. Wood consumption data for the UK domestic sector that is robust and more up to date could be obtained by investing in data collection/surveying. This investment is needed if the current levels of uncertainty are to be addressed. In designing such a survey, it would be helpful to understand the relationship between the level of investment required in terms of data collection compared to the level of detail and quality of the data returned. Such a survey on domestic wood use was recently carried out by DECC (2016) for around 1000 households in the UK. It provides useful data on temporal and geographical usage patterns, age of devices, and to a certain extent fuel type. The survey was a one off and would need to be repeated regularly to capture information on trends[5]. As an alternative, census data, for example, might in principle be used in the future to gather information on specific fuels used for both primary and secondary heating but is unlikely to be able to provide information on fuel usage rates.

Methods for estimating the national domestic consumption of wood for heating also include direct energy modelling or a gap analysis. In the latter, the total domestic energy requirement is estimated based on temperature data. The shortfall between the modelled total domestic energy requirement and the known supplies of fossil fuels and electricity to the domestic sector provide an estimate of the energy provided by combustion of wood and other biomass.

Separate to government subsidies through the RHI, it is understood that sales of wood burning stoves have been running at a high level in recent years. The Stove Industry Alliance estimated that around 150,000 units were installed in the UK in 2009, with single room direct heating units accounting for 90% of 2009-10 sales of wood stoves (HHWT, 2010). The study from the Heating and Hot Water Taskforce (HHWT, 2010) suggests a strong market for wood stoves that that grew between 2007 and 2010 by around 40% per annum. However, as discussed above, the 1997 Forestry Commission survey indicates that it is likely that these together with the existing fleet of stoves and open fires are widely used sporadically for secondary rather than primary heating. This can lead to characteristic air pollution signals in housing estates on cold Friday and Sunday nights as well as on special occasions (see Section 8.2). Such intermittent use is also a source of uncertainty in emissions estimates. Whether these units replace gas or higher pollution fuels such as oil or coal is also unclear. HHWT (2010) report anecdotal evidence that large numbers of consumers displace some high carbon fuel use by wood burning in homes with gas and oil

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[4] Information gathered from https://www.nomisweb.co.uk/census/2011/qs415ew

[5] Note that the one off survey carried out by DECC (2016) has led to a significant increase in the domestic wood consumption figures used in the latest version of the NAEI published in 2016. See section 7.1

fired central heating, possibly shortening their gas/oil heating season by displacing use over the winter. Further data on fuel displacement/replacement would be helpful in assessing current and future emissions trends.

6.5 Combustion Technologies

There is a wide range of technologies from domestic room heaters (which covers basic fire baskets, open fires to fully-automatic pellet stoves), small boilers, medium and large boilers (incorporating all sizes of automatic boiler technologies and sizes), as well as gasification units which can theoretically be coupled to engine technology. Liquid and gaseous biofuels can also be burned in engines or boilers.

The range of appliance types in the domestic sector and small commercial sectors is extensive:

- Traditional open fires are no longer an option for new buildings as they are inconsistent with energy efficiency targets, but remain an option for the majority of the existing building stock.
- Modern ‘open’ fireplace replacements are covered by a harmonised EN Standard (hEN) but are not realistic in a new build for the reasons given above.
- Closed inserts, stoves and a range of other appliance types which also have hEN Standards.
- Pellet stoves offer a modern, high combustion efficiency room heater but at higher prices, and require an electricity supply and a manufactured fuel.

Obtaining quantitative information on the range of domestic and small-scale biomass boilers and stoves is a major challenge to reducing the uncertainty in emission inventories for these sources. The issues, approaches and uncertainties specific to the UK inventory are discussed in Section 6.6.2.

For the domestic sector, the EC has proposed Eco-design regulations for solid fuel boilers < 1MWth and solid fuel room heaters as discussed above. However there have been problems with the Eco-design process, arising from the fact that different emission measurement methodologies give very different emission results.

The RHI has set emission requirements (for PM and NO_x) for biomass boilers. The product standard EN 303-5 sets other non-mandatory emission classes for solid fuel boilers <500 kW output at rated output.

6.6 Emission measurement issues

One key issue for domestic scale appliances is the variability in emissions recorded by different measurement techniques. Across Europe there are a range of test methods (for PM emissions) which have been established for room heaters and in particular batch-fired, manually-controlled, natural draught appliances. There are significant differences between the German ‘DIN’, Norwegian ‘dilution tunnel’ and UK ‘Clean Air Act’ test regimes (as well as differences with test regimes in Australia and the USA). Issues are also apparent for pellet stoves and small (EN303-5) automatic boilers, although considered to be less important.

Furthermore, emission 'type-tests' at rated output for all of these equipment types are unlikely to reflect real-life emissions.

The Scandinavian states have made strong suggestions that proposed test procedures within the Eco-design Directive be as realistic as possible since real world emissions will depend on a large number of factors such as stove type, fuel type and quality, dilution conditions, operator know-how, ambient weather conditions etc. A report from the US EPA (http://www.epa.gov/ttnchie1/conference/ei10/pm/houck.pdf) suggests emissions factors for open fires burning wood to be on average 11.6 g/dry kg i.e. more than twice the final proposed Eco-design limit. A study in New Zealand (Scott, 2005) compared emissions from the same appliances under "real world" conditions as simulated within a laboratory and under real world conditions where the wood burners were operated normally by the householder, using their own firewood supply. The results suggested that real world emissions may be up to 4-5 times those achieved within legislated test cycles and laboratory "real world" studies. Even burners with authorisation particulate test emissions below 1.5 g/kg, could achieve real world emissions factors of up to 13.0 g/kg. Coulson et al. (2013) suggest that in situ studies which take account of the "operator" variable are essential for understanding real world emissions. They present a review of such in situ studies for New Zealand which highlights the variability. Overall only 22% of stoves met their legislated limits in the real world. Wood moisture, flue temperature and oxygen supply were the major factors affecting emissions rate variability. In the UK context, a report from EPUK (2013) based on work undertaken by Brighton and Hove City Council, highlights an increasing number of gross pollution and nuisance cases based on solid fuel appliances being dealt with by Local Authorities. These are commonly linked to appliances which are poorly installed, being misused, or burning inappropriate fuels. The report also suggests evidence of poor public knowledge of Smoke Control Area compliance and safety awareness of solid fuel combustion leading to DIY installations. The need for public information and guidance on solid fuel stove operation is highlighted in the EPUK (2013) report.

6.6.1 Emission factors for inventories

Recent years have seen a substantial increase in the measurements conducted on a range of different domestic wood burning appliances to determine emission factors for different pollutants. As a result, many of the old emission factors published in emission inventory guidebooks[6] have been superseded.

However, emission test methods used in the literature often tend to be non-standard test methods and with poor or unknown uncertainty. Test protocols on domestic appliances which seek to reflect real-life activity may be extremely variable and will be difficult to compare with EN or other standard protocols which may only test a limited range of operation. Furthermore, emission factors that are now available often relate to equipment found in Scandinavian countries and the alpine regions of Europe. These are not necessarily applicable to equipment used in the UK.

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[6] EMEP/EEA Emissions Inventory Guidebook

It is challenging to obtain reliable information on the age profile of the UK stock of domestic wood burners in the UK. As a result it is difficult to estimate the percentage of modern equipment in the UK stock that is considerably more energy efficient and less polluting.

In practise, there are many factors which will impact on the emissions from each individual unit. These include: installation, user behaviour, the way in which fuel is introduced/stacked, whether the unit is started/stopped frequently, the choice of fuel and the quality of the fuel.

6.6.2 Sources of information used in the NAEI and their uncertainties

For smaller scale wood combustion, in the commercial and domestic sectors, the availability of wood consumption data is mixed and higher in uncertainty than consumption data for power stations. Emissions from boilers in the commercial, domestic and public sectors are estimated by the NAEI using emission factors mostly taken from literature sources such as the US EPA Compilation of Air Emission Factors (US EPA, 2009), the EMEP/EEA Emission Inventory Guidebook (EMEP/EEA, 2013) and UK emission factor surveys (Walker *et al*, 1985) or derived from data for certain plant in the Pollution Inventory.

The NAEI does not currently use information on combustion technologies when estimating emissions from biomass consumption in the domestic sector. The NAEI methodology is simplistic and ideally would reflect the range of technology (including open fires) in use for all domestic solid fuels, but the approach is limited by a lack of detailed activity data on use of wood and (to a lesser extent) a lack of emission factors. UK energy statistics provide the total consumption of wood, but robust data on how the fuel is used or the appliance stock is extremely limited. There is also an issue regarding the impact of the type and quality of wood burned on emissions, for example the extent to which the wood has been dried or seasoned, whether it is hardwood or softwood, or whether it has been collected locally etc.

Estimating the uncertainties in emissions from domestic and small-scale biomass combustion is itself highly uncertain and difficult to quantify. However, inventory uncertainties are higher for these sources than for biomass emissions from power stations and other large point sources. Moreover, the uncertainties in inventory estimates for 20-50 MW boilers are likely to be better than for the residential, the commercial/public and small industrial sectors. This is due to improving activity data from EU-ETS (Emission Trading Scheme). Uncertainties in estimates for <20 MW boiler plant using biomass fuels will also improve in time with data from the RHI.

For domestic biomass combustion, the NAEI considers (based only on expert judgement) that the uncertainty range in emission estimates could be an order of magnitude, with substantial uncertainties in both the emission factors for appliances and the activity data (number of appliances, usage and fuel used) contributing to this uncertainty range estimate. However, the NAEI generally makes conservative assumptions so that, if anything, the actual emissions are more likely to be lower than the inventory estimates. This is because the NAEI is largely adopting emission factors for uncontrolled, older appliances for all domestic sources. The NAEI is aware of the current issues and is working with other inventory experts in Europe to reduce the levels of uncertainty, for example by including estimates of the mix of appliances including more modern stoves.

Estimates are made of emissions from stationary combustion of straw in agriculture using DECC energy statistics and default emission factors from the USEPA and EMEP/EEA Emissions Inventory Guidebook.

Emission estimates in the UK inventory for small-scale waste burning comprise emissions from combustion of agricultural and domestic waste, and also from burning of treated wood (i.e. treated with fungicides and used in construction). For all sources, the activity data are not routinely collected as annual statistics across the time series, and the NAEI generates time series estimates of activity based on available survey data and published statistics, together with proxy data to extrapolate across years where data are missing. Activity estimates were recently refined in the light of a national waste burning habits survey of a thousand UK households completed on behalf of Defra in 2010 (Whiting et al 2011). This survey provided the frequency of domestic waste burning in households with and without allotments. From this, an overall figure for the number of waste burning fires per household per year was derived. This figure was then combined with statistics on the number of households in the UK in each year to derive a consistent emission trend from this source over the inventory time-series.

Emissions of NO_x, $PM_{10,}$ VOCs, dioxins and furans from all the small-scale burning sources are based on composite emission factors derived from estimates of the individual waste types burnt and factors for specific waste types from UK and US research (US EPA, 2009, Coleman et al., 2001 and Perry, 2002). The uncertainty in emission estimates for waste burning sources is particularly high because the type and amount of wood being disposed of by combustion is not well characterised.

The celebration of bonfire night in the UK (5th November) is treated as a separate source from other domestic burning events due to the large scale organised nature of the event and potential air quality impact over a short period of time. Emission estimates for bonfire night are based on estimates of the quantity of material burnt in bonfires. Emission factors for domestic wood fires (in the case of PM_{10} and PAH) and disposal of wood waste through open burning (in the case of PCDD/PCDF) are used to generate emission estimates.

Field burning of agricultural residues as a general management practice was banned in 1993. However estimates for earlier years are based on UK farm surveys and fertiliser sales data. Muir burn is practised in Scotland, but emissions estimates are not currently made.

Emissions from wildfires are included in the inventory (whether these are identified as arson or otherwise). Statistics are available from the Fire Services on the number of fires attended each year, and an indication on the size of the fire is also provided. This allows an estimate to be made of the amount of biomass that is burned, and hence the emissions determined.

6.6.3 European studies of potential inventory uncertainties

Denier van der Gon et al. (2015) report a new high resolution anthropogenic carbonaceous aerosol emission inventory for Europe which indicates that about half of the total $PM_{2.5}$ emission in Europe is carbonaceous aerosol. Residential wood combustion was identified as the largest source of organic aerosol in Europe. The inventory was used as input for two chemical transport models (CTMs) which revealed major under-estimations of organic aerosol in winter time especially for regions dominated by residential wood combustion when compared with ambient measurement data. The discrepancies were not universal but appeared to differ by country. Further development of a revised bottom-up inventory for residential wood combustion accounting for the semi-volatile components of the emissions led to emissions estimates higher by a factor of 2-3 than in the earlier inventory. When this was used as input for the CTMs, a substantially improved agreement between measured and predicted organic aerosol was found. These findings suggest that many national inventories substantially under-estimate emissions of primary organic aerosol from residential wood combustion because they fail to include the semi-volatile organic aerosol that forms during the dilution and cooling of the flue gases. Unfortunately, the UK was not one of the countries studied in this research but the paper highlights the need for national studies to test inventories through comparison of modelled predictions with measurements.

7 Trends in Emissions from Biomass Combustion According to the NAEI

7.1 Trends in Historic Emissions

The NAEI has currently estimated the trend in UK emissions from biomass and other sources up to 2012 (Passant et al, 2014).

Figure 7-Figure ***11*** show trends in emissions of NO_x, PM_{10}, $PM_{2.5}$, benzo(a)pyrene (BaP) and dioxins from 1990-2012 for biomass sources in the UK. Whilst the NAEI estimates emissions from 16 different PAHs, BaP is suggested as a suitable indicator for the carcinogenic fraction of total PAH in ambient air by the World Health Organisation (WHO, 2000). BaP also has a target value set as a National Air Quality Objective (Defra, 2007) and hence we focus on it here. Biomass emissions of NO_x, PM_{10} and $PM_{2.5}$ show an increasing trend over this time period as levels of activities have increased against a backdrop of decreasing emissions in the total UK inventory due to reductions from other sources. As a consequence, the share of biomass emissions has been increasing significantly.

Emissions of PM_{10} and $PM_{2.5}$ from these sources increased by over 100% over the period from 2002-2012, mainly due to increases in domestic wood combustion and stationary combustion of straw in agriculture. This reflects the increase in activities for these sources over this period as shown in Figure 12 for domestic combustion of wood. Figure 9 indicates that the share in total $PM_{2.5}$ emissions from these biomass sources increased from 9% in 2002 to almost 25% in 2012. To put this in context, road transport (including exhaust and non-exhaust sources) contributed 26% of total UK $PM_{2.5}$ emissions in 2002 and 21% of total UK $PM_{2.5}$ emissions in 2012. For PM_{10} emissions, the share of biomass sources increased from 7% in 2002 to 17% in 2012 compared with the road transport share of 23% in 2002 and 20% in 2012. Emissions of NO_x from biomass combustion are also rising, mainly due to the increase in wood combustion in power stations, but overall, biomass still makes a much smaller contribution to UK emissions of NO_x (<1%) than PM.

Note that post drafting of the main body of this report, DECC has significantly revised its statistics on UK domestic wood fuel consumption for recent years (DECC, 2015). The figures have been increased substantially and this in turn has led to a significant increase in the NAEI estimates of emissions from this source in the latest version of the inventory covering years up to 2014, and published in early 2016, compared with the version referred to elsewhere in this report (covering years up to 2012). The increase in DECC's estimates of domestic wood fuel consumption in DUKES followed a one-off large-scale survey commissioned in 2014, the purpose of which was to provide a new baseline for domestic wood fuel use in the UK (DECC, 2016). The new baseline puts the quantity of wood used for domestic fuel consumption in 2012 at 3.9 million tonnes, nearly a factor of 3 increase on the previous figure in DUKES for 2012. As a consequence, the latest NAEI estimate of $PM_{2.5}$ emissions from domestic wood consumption in 2012 is 31.0 ktonnes compared with the previous figure of 10.6 ktonnes from the 2012 version of the NAEI which forms the basis of all the figures discussed elsewhere in this report.

This change was the main cause of the significant revision to the overall $PM_{2.5}$ inventory in the latest version of the NAEI and means that domestic wood consumption is now estimated to be responsible for 30% of all UK $PM_{2.5}$ emissions in 2012 compared with our previous estimate indicated by figures elsewhere in this report of 14% (see Figure 9).

To further put these revised consumption figures from DECC into context, about 4.4 million tonnes of wood was used in 2014 for domestic combustion compared with 22 million tonnes of diesel fuel consumed by road transport.

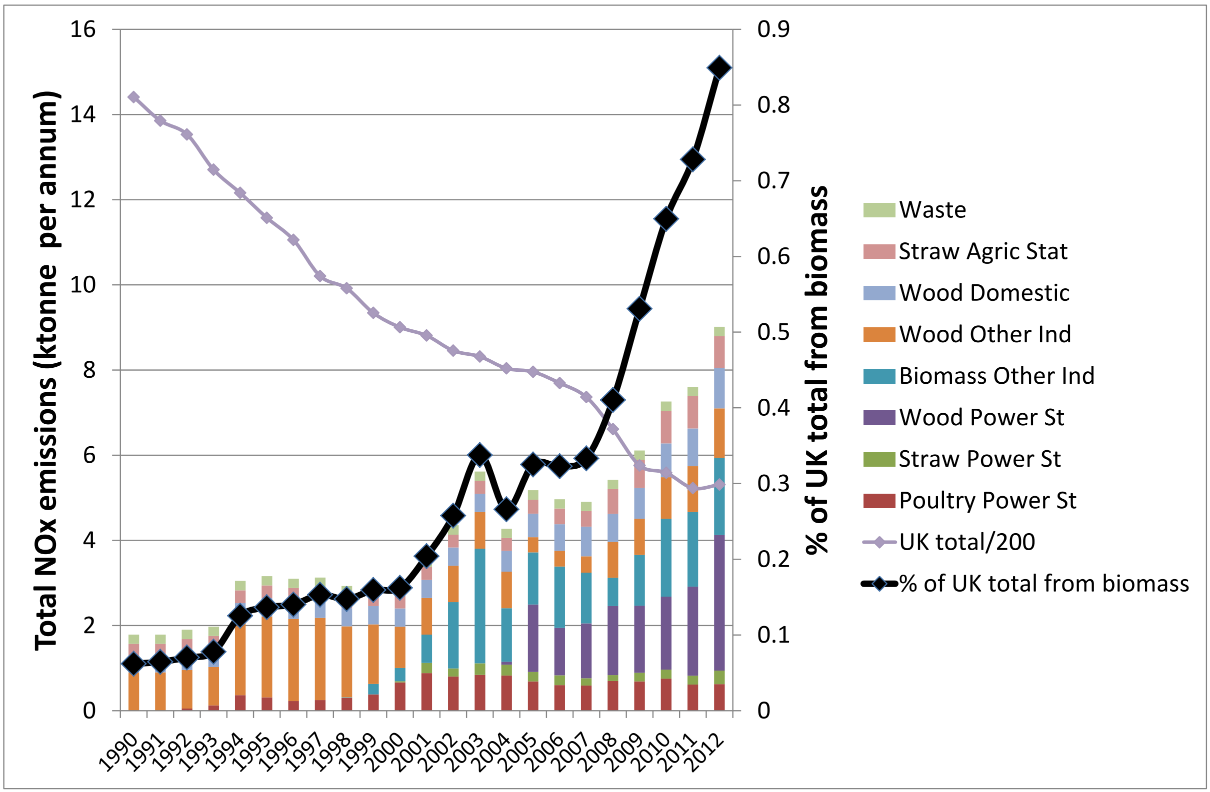

Figure 7: *Trends in annual UK* NO_x *emissions from biomass sources according to the NAEI (Passant et al, 2014)*

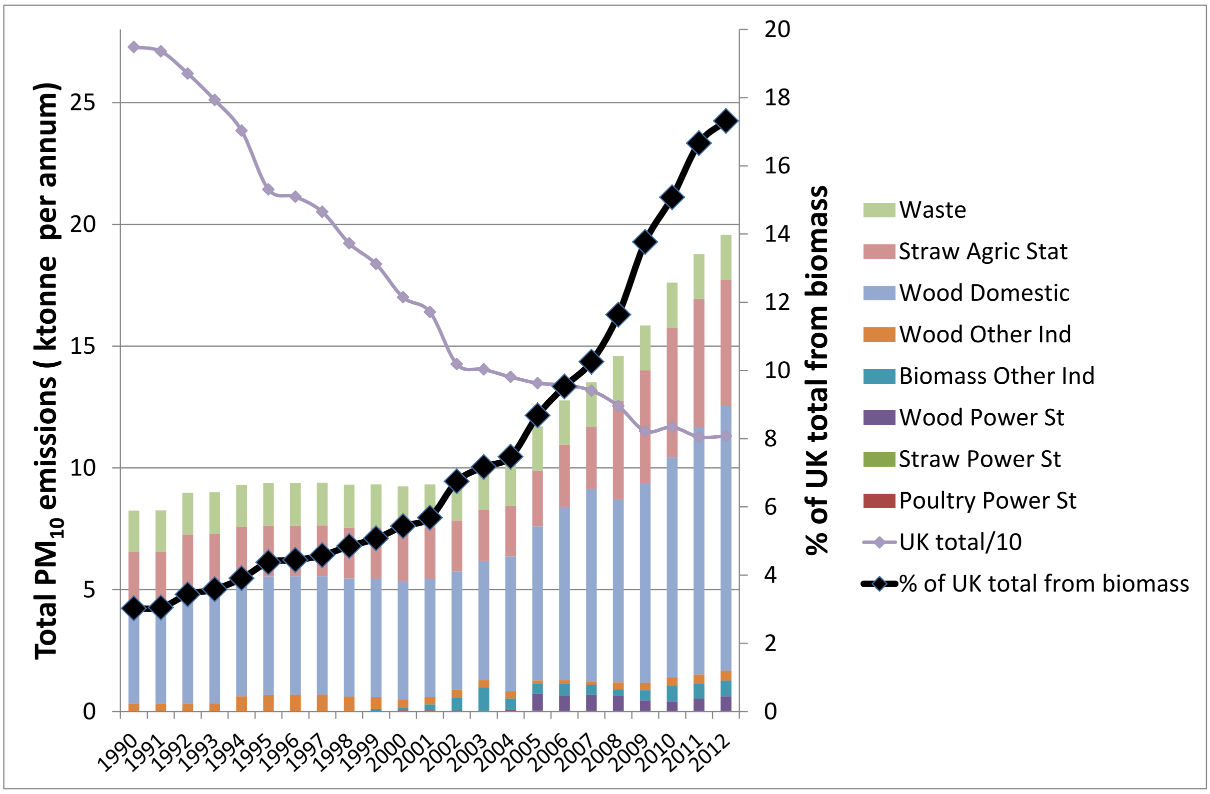

Figure 8: *Trends in annual UK PM$_{10}$ emissions from biomass sources according to the NAEI (Passant et al, 2014)*

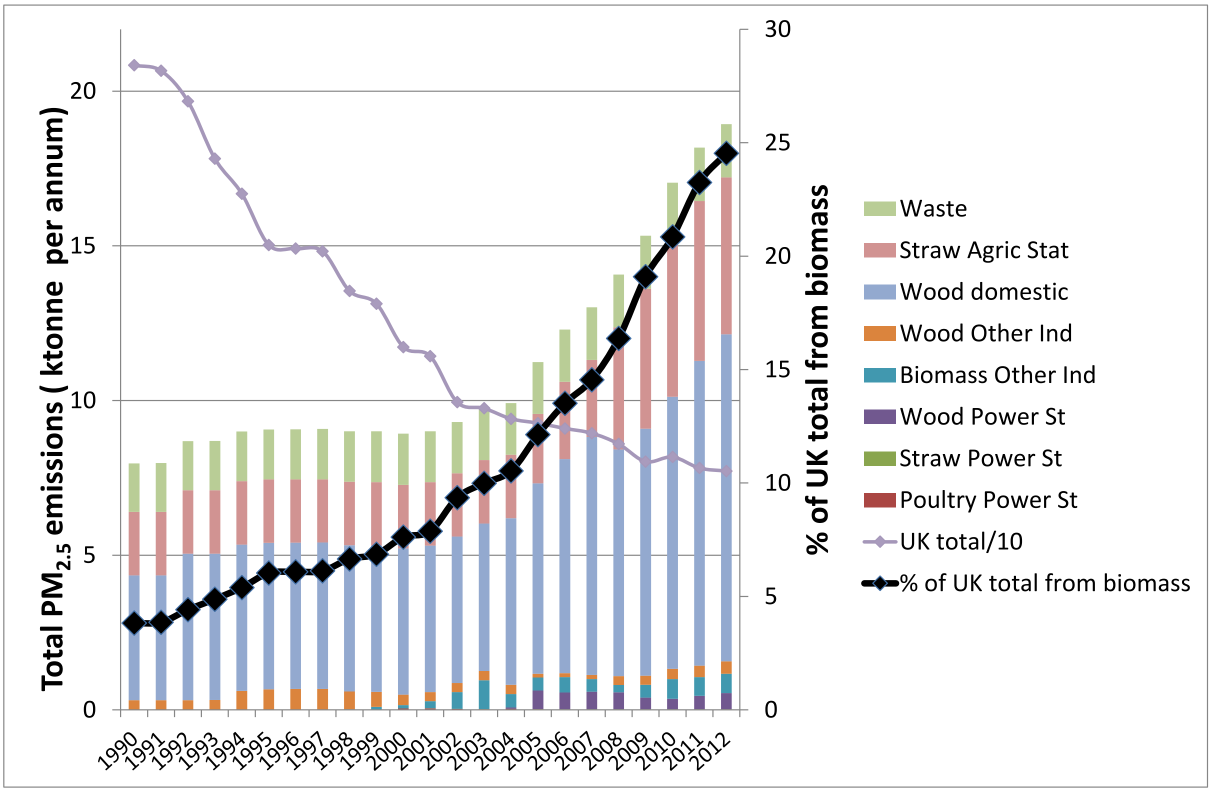

Figure 9: *Trends in annual UK PM$_{2.5}$ emissions from biomass sources according to the NAEI (Passant et al, 2014)*

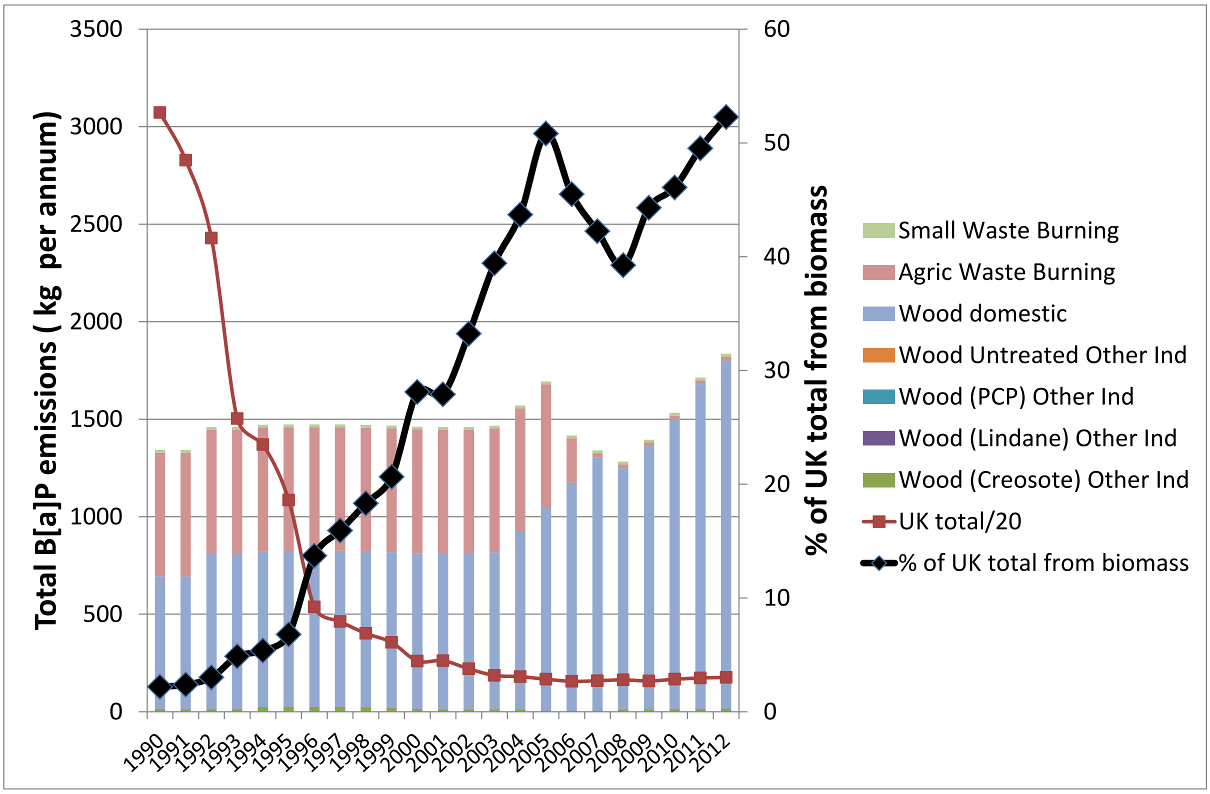

Figure 10: *Trends in annual UK B[a]P emissions from biomass sources according to the NAEI (Passant et al, 2014)*

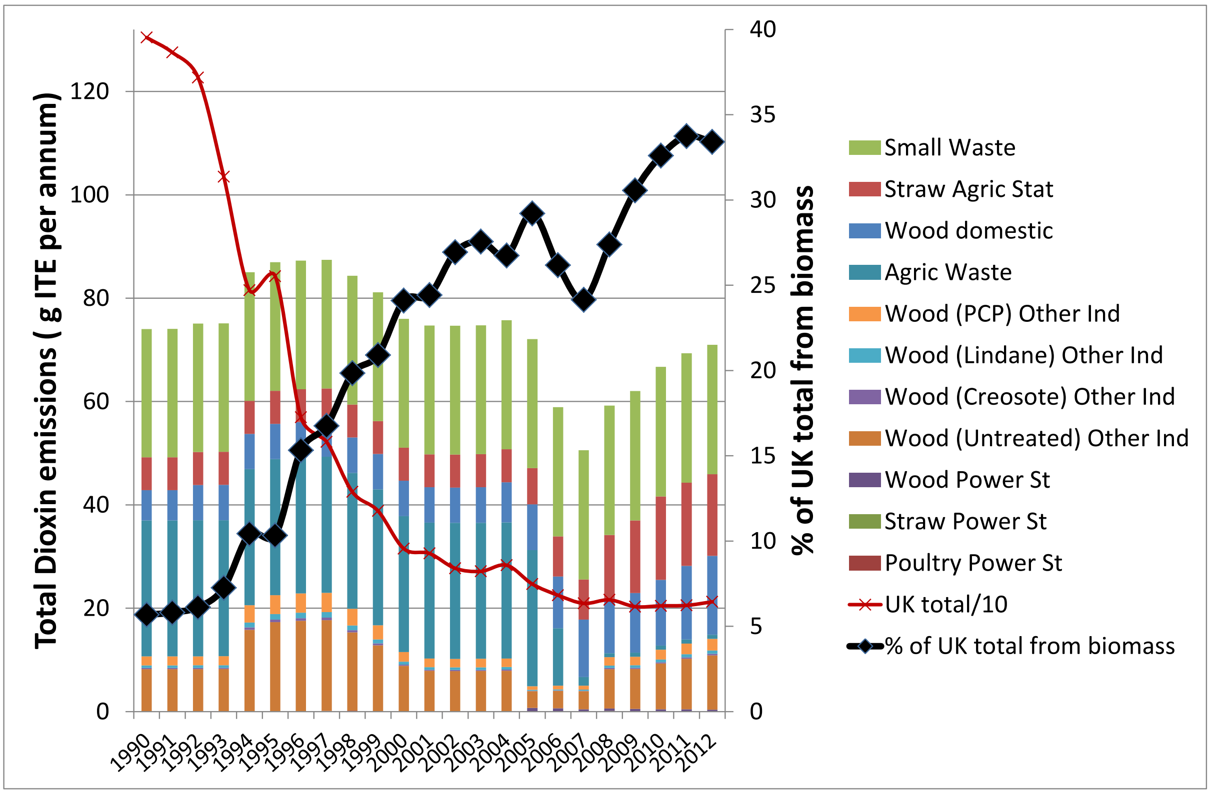

Figure 11: *Trends in annual UK Dioxin emissions from biomass sources according to the NAEI (Passant et al, 2014)*

Domestic wood combustion is the dominant source of BaP emissions from all biomass sources. Although overall biomass emissions of BaP have not increased so significantly since the 1990s as the increase in domestic wood combustion emissions has been partly offset by a reduction in emissions from agricultural waste burning. However, the percentage of total UK BaP emissions has grown from 2% in 1990 to over 50% in 2012 due to the decline in emissions from other sources.

Emission factors for BaP are around a factor of two higher for domestic wood burning relative to coal on a comparative energy-basis and since there are virtually no BaP emissions from natural gas combustion, switching from gas to wood burning for domestic heating will lead to a large increase in BaP emissions. Emissions of metals and other persistent organic pollutants are also likely to increase.

Biomass combustion is a major source of dioxin emissions according to the NAEI. Overall biomass emissions of dioxins decreased until 2007 due to reductions in emissions from agricultural waste burning, but have since increased with increase in emissions from stationary combustion of straw and domestic combustion of wood. The percentage of total UK dioxin emissions has grown from 6% in 1990 to 33% in 2012 due to the decline in emissions from other sources.

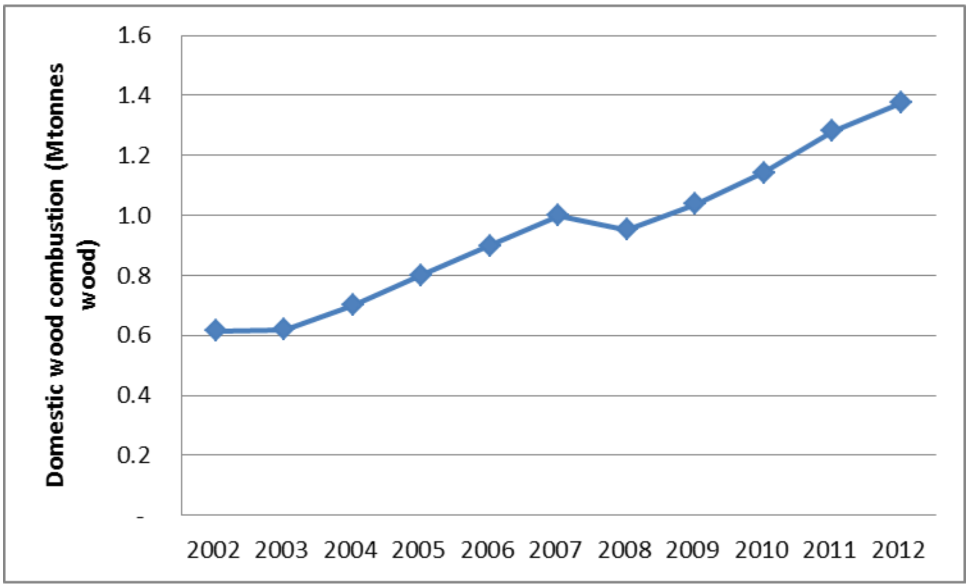

Figure 12: *UK consumption of wood for domestic combustion: 2002-2012. Source: Digest of UK Energy Statistics (DECC, 2014)*

7.2 The Potential Influence of Fuel Switching

The biomass emissions are not entirely additional emissions as they will have offset the emissions that would have occurred from other fuels which biomass has displaced. However, since domestic combustion of wood is for a variety of purposes, some aesthetic as well as functional (to produce heat), it is not possible to estimate the quantity of other domestic fuel (e.g. gas) this may have displaced.

To gain some insight on how emissions can change with fuel switching to biomass, data from the NAEI have been used to estimate comparative emission factors on an energy content basis for domestic consumption of wood, coal, oil and gas. These are shown in Table 7 for NO_x and $PM_{2.5}$ in units of grammes of emission per gigajoule energy content of fuel.

Table 7: *Comparative emission factors for domestic consumption of fuels implied by energy content of fuel*

	NO_x **(g/GJ)**	$PM_{2.5}$ **(g/GJ)**
Wood	56.5	625
Coal	81.5	387
Burning oil	73.7	3.2
Gas	21.6	0.5

These factors are derived by dividing total emissions from domestic consumption of each fuel by the total quantity of fuel used in the inventory calculations converted to energy units using net calorific values for each fuel reported in DECC's Digest of UK Energy Statistics (DECC, 2014). For domestic wood, according to DECC, the net calorific value is based on seasoned logs at 25% moisture content used as received. The calorific value for coal refers to domestic house coal, oil refers to burning oil and for natural gas DECC provides a weighted average calorific value for gas entering the national transmission system.

Some caution should be applied in drawing quantitative conclusions from these factors. As stated earlier, the NAEI estimate for emissions from domestic biomass consumption is based on total emissions from all wood burned and using conservative emission factors for mainly old uncontrolled appliances. Lower factors may be expected for more modern, efficient appliances currently in use or to be installed in the future. The trends in Figure 7-Figure ***11*** therefore reflect these conservative emissions factors and it is likely that emissions per device are likely to reduce in the future, particularly in the light of the Eco-design directory. Nevertheless, the much higher comparative PM emission factors for wood combustion compared with natural gas does illustrate the potential negative impact (an increase) on PM emissions that would result if there was a switch from domestic use of natural gas to biomass. It also reinforces the need for effective regulation on emissions from modern domestic biomass appliances.

The EMEP/EEA Emissions Inventory Guidebook (2013 edition[7]) is a further source of information on typical emission rates from all sources of emissions to air drawing on information from the literature. It is considered to be the central reference manual for the compilation of national emissions inventories of air quality pollutants.

The "Guidebook" includes detailed information on the emission rates from different types of residential appliances, including biomass boilers and stoves utilising different technologies. Data from the Guidebook are presented below, and whilst these are not UK specific emissions data, they do provide a good overview of the impacts that arise from fuel switching.

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[7] http://www.eea.europa.eu//publications/emep-eea-guidebook-2013

Figure 13 below illustrates the emissions from open fireplaces. Emissions of NO_x, PM_{10} and $PM_{2.5}$ per unit of energy generated are presented for the use of different fuel types in open fireplaces. Emissions of NO_x are broadly comparable across the different fuel types, with wood giving lower emissions of NO_x than both coal and gas. However, wood burning in open fireplaces produces more PM_{10} and $PM_{2.5}$ than any other fuel type by a considerable margin (emissions from gas oil and gas are too small to be easily seen on the graph). The figure clearly illustrates that switching from coal to wood in open fireplaces gives rise to more than twice the emission of PM_{10} and $PM_{2.5}$.

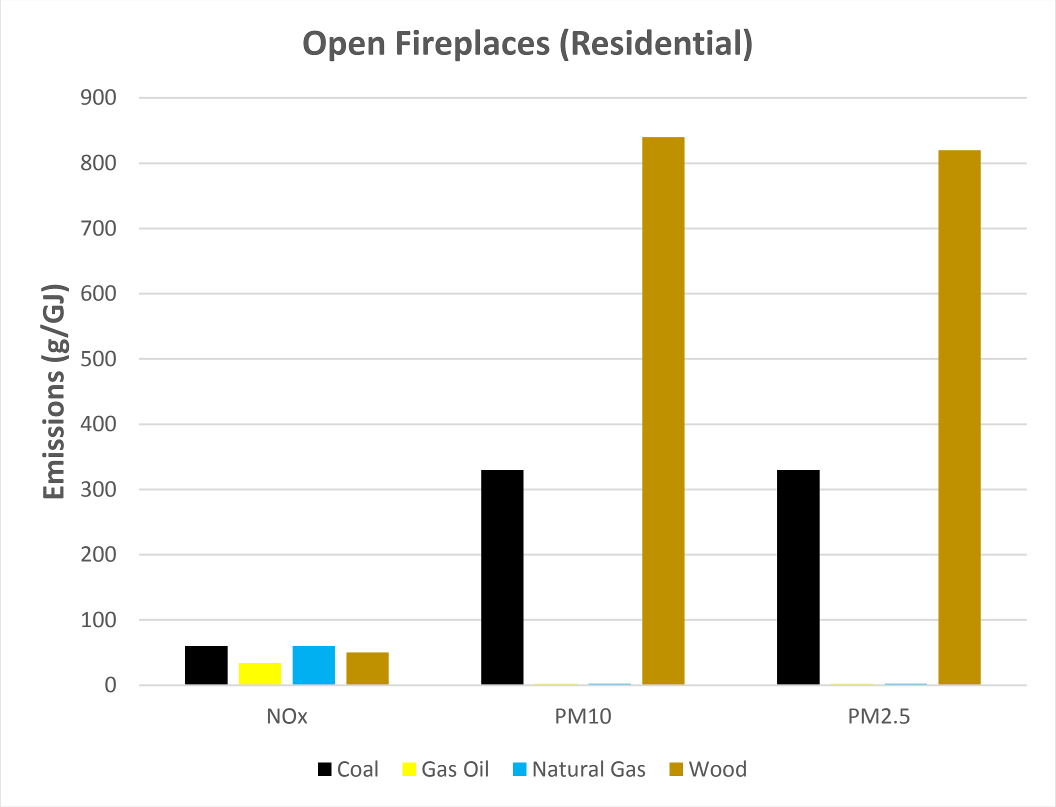


Figure 13: *Emissions of* NO_x*,* PM_{10} *and* $PM_{2.5}$ *from open fire places utilising different fuel types*

A similar result is observed when residential boilers/stoves are considered. Figure 14 shows the emissions of NO_x, PM_{10} and $PM_{2.5}$ from a range of different residential boilers and stoves, utilising different types of fuel. The figure includes different types of boilers and stoves using wood as the fuel.

Boilers and stoves using wood emit less NO_x than coal fired boilers/stoves on an equivalent energy output basis. Gas appliances are lower emitters by more than an order of magnitude.

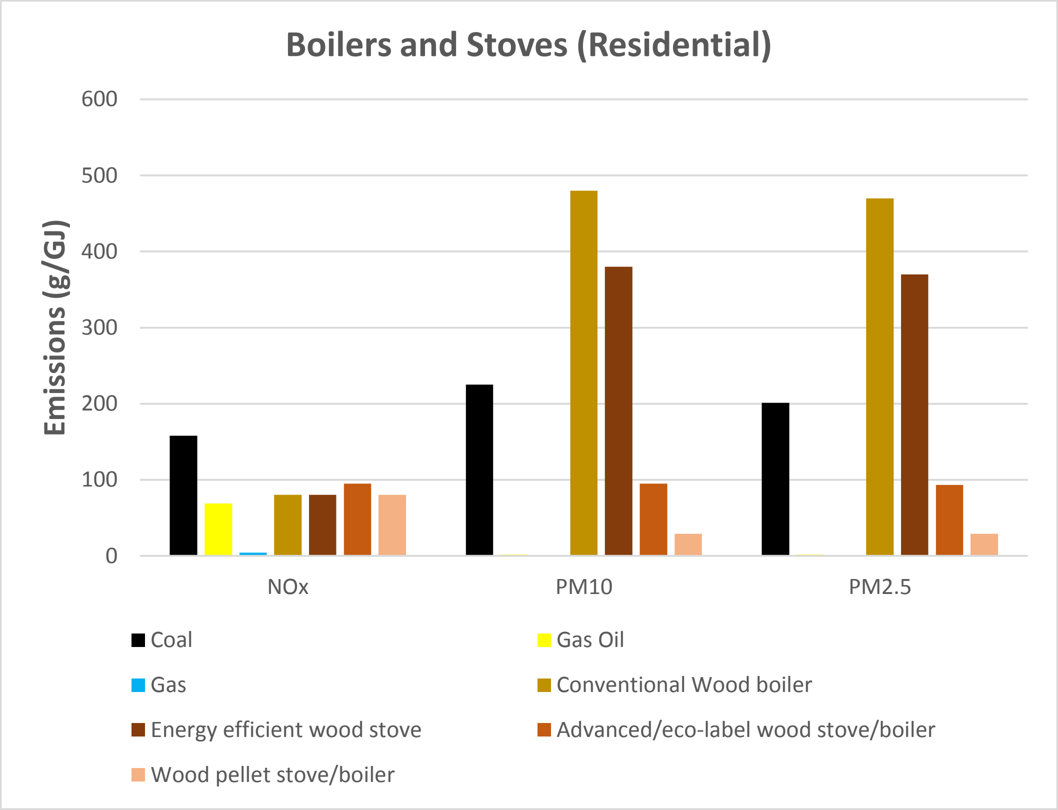


Figure 14: *Emissions of* NO_x, *PM_{10} and $PM_{2.5}$ from residential boilers and stoves utilising different fuel types*

For emissions of PM_{10} and $PM_{2.5}$ the technologies of the wood fired boiler/stove are particularly important in determining the level of emissions. Conventional boilers and even energy efficient wood stoves emit nearly twice as much PM as coal appliances. Whereas advanced or eco-labelled appliances and wood pellet stoves/boilers emit less than half of the PM_{10} or $PM_{2.5}$ compared to coal. However, even the lowest emitting wood burning appliance emits an order of magnitude more PM than a gas oil appliance, and two orders of magnitude more than a gas appliance (PM emissions from oil and gas appliances are small, and difficult to see on Figure 14). The impact of fuel switching to wood is therefore substantial, even when the most modern equipment is utilised.

For NO_x, the dis-benefit of switching from gas to biomass is smaller than for PM and the comparative factors are actually lower than for oil and coal. This may partly reflect the poor combustion conditions and low temperatures implied by the biomass emission factors used by the NAEI. For NO_x, there may be the possibility of emission factors on an energy basis increasing with more efficient combustion unless other steps in the design of the modern appliance are taken to control NO_x.

7.3 Future Emission Projections

The NAEI makes estimates of future emissions from all sources using DECC's projections on UK energy and fuel consumption for different sectors and assumptions on future emission factors taking account of future emission regulations and changes in technologies.

DECC usually revise their energy projections annually. However, the projections cover sources at a fairly coarse level of detail so that emissions cannot be forecast at the same level of sector detail as the historic emission trends. The energy projections do not split biomass from other forms of renewable energy, except for those incentivised through the RHI (DECC, 2011b).

Based on the latest projections from DECC, the NAEI has estimated significant increases in the quantity of biomass used for power stations and industrial/commercial combustion and smaller increases in wood used for domestic combustion from 2012 to 2030. This is expected to lead to an increase in future biomass emissions of NO_x, PM and BaP over this time-period while emissions from other sources are generally expected to decline. This means that the biomass share of total UK $PM_{2.5}$ emissions is estimated to increase from 25% in 2012 to 31% in 2030 and for NO_x from 0.8% in 2012 to 4.5% in 2030.

There is, however, very high uncertainty in these emission projections, partly due to uncertainties of future biomass consumption for different sources and uncertainties in future emission factors, particularly the type and performance of biomass burners and stoves. The levels of uncertainties cannot be quantified, but will be significantly higher than the uncertainty estimates for current inventory years. Predicting future emissions is inherently more uncertain than estimating current emissions, but the uncertainties could be reduced if the switch to biomass could be predicted at higher levels of sector detail, if the types of domestic and small-scale industrial burners and boilers could be predicted with greater confidence and their emission performance better characterised. Clearly, getting better, more robust information on current and near-term trends in sales and usage of different types of domestic and small-scale wood burners would help provide a stronger basis for predicting future emissions. Predicting the impacts on current and future air quality would also benefit from knowing the spatial distribution of current and future biomass consumption.

Other studies have also considered future emissions from biomass combustion and the impacts on air quality. The Renewable Heat Incentive impact assessments (DECC 2011b, DECC 2013c) project a total RHI incentivised biomass uptake in 2020 of 30 TWh from the non-domestic scheme and around 1.8 TWh from the domestic scheme. Assessments of the non-domestic scheme, based on the emissions limits in Tables 3 and 4, demonstrate fairly modest lifetime social costs to air quality (DECC, 2011b). However, were the emissions limits to be relaxed, or not to be achieved in a real world setting, then the costs would be substantially higher. The domestic RHI scheme has been modelled as having a net lifetime air quality benefit, although it should be pointed out that this is balanced across all technologies included in the RHI and not just biomass combustion devices. In the latest assessments, negative air quality impacts associated with biomass uptake are counteracted by the assumption that more coal and oil heated households will take up renewable technologies than those using gas or electricity (DECC, 2013c).

It is worth pointing out that the NAEI estimated $PM_{2.5}$ emissions from domestic wood burning for 2012 of 10.6 kt are already quite close to the estimated projections of Apsimon and Oxley (2013) for total $PM_{2.5}$ emissions from biomass heating measures by 2030 of 13 kt. Even at estimates of 13 kt, the report of Apsimon and Oxley (2013) raises concerns with respect to the impact of increasing use of biomass for domestic heating on compliance with the national ceiling for total UK emissions for 2020. Based on the revisions to the Gothenburg Protocol, which specifies a 30% reduction in UK $PM_{2.5}$ emissions relative to 2005 levels by

2020, this equates to a ceiling for 2020 emissions of 65kt when applied to the most recent NAEI estimate for 2005.

8 Measurements of particulate matter from wood and wood product burning

8.1 Measurement techniques

Smoke arising primarily from the domestic combustion of wood and wood products is likely to be a significant contributor to atmospheric concentrations of $PM_{2.5}$, and as such there is a strong imperative to devise reliable measurement methods. There are no absolutely specific chemical tracers for the burning of wood, but concentrations may be estimated approximately from chemical tracers, or by using other methods.

The main products of combustion of wood and wood products are elemental (black) carbon and organic matter. As both these components arise from all combustion sources, and organic matter is emitted also from non-combustion sources, these are not reliable tracers for woodsmoke. Consequently, most research has been based upon the analysis of relatively minor constituents characteristic of woodsmoke, the most commonly measured being the organic compound levoglucosan and the element potassium in the fine fraction of particles. Levoglucosan is a product of the decomposition of cellulose and consequently arises from not only the burning of wood but also of grass, straw, garden bonfires etc. Potassium also has sources in sea spray and soil making estimation of the woodsmoke-generated potassium typically rather imprecise. Other organic compounds arising from wood burning have also been used as tracers but are typically most useful for differentiating between the burning of hardwood and softwood rather than being used as alternatives to levoglucosan.

The ratio of woodsmoke mass to that of the tracer compound or element is also variable depending upon the nature of the wood and the combustion conditions. Based upon the data presented by Puxbaum et al. (2007), Harrison et al. (2012) suggest that the relationship between biomass smoke mass and levoglucosan lies typically within a range of 4.51 to 24.6, with a mean value of 11.2. The literature presents a wide range of data on the relationship between potassium and biomass smoke mass. This is because the potassium content of biomass smoke depends critically upon the combustion conditions. Under conditions of high burn-out, potassium can make up around 50% of the mass of smoke particles in the $PM_{2.5}$ size range. Puxbaum et al. (2007) report values ranging from ratios of levoglucosan/K in $PM_{2.5}$ of 33.3 for soft wood in a wood stove (Fine et al., 2004) to 6.25 for soft wood in a fireplace (Fine et al., 2001). A much cited paper by Schmidl et al. (2008a) reported levoglucosan/K ratios ranging from 19.5 to 216 for various wood types burned in an old type of domestic stove.

There is a substantial literature which shows that the combustion of biomass, and wood in particular, is a rich source of polycyclic aromatic compounds in the gas phase and bound to PM. PAHs are of particular concern because of the known carcinogenicity of the mixture. The PAH comprise a large number of compounds which are structurally related but cover a wide range of volatilities. A particularly comprehensive review of the emissions of various PAH from a range of different wood-types is given in Fine et al. (2001). What this and other studies show is that certain PAH, such as retene, can be good markers for the combustion of specific wood-types. Retene is specific to resinous soft woods and was proposed by Ramdahl (1983) as a suitable biomass PM marker, with confirmation by Simoneit et al. (1999) of it being a fairly specific tracer of combustion of wood from resinous conifer species.

The sensitivity of emitted PAH speciation to the feedstock fuel however, creates difficulties in using these chemicals as indicators for biomass PM in a more complex urban setting where in practice an ensemble of wood-types are burned. Recent work in Birmingham has compared concentrations of 21 PAH compounds measured in the Queensway Road Tunnel in central Birmingham with concentrations measured simultaneously at a background site within the city. All compounds except retene showed an appreciable enhancement of concentration in the road tunnel, indicative of a road traffic source, while retene showed lower concentrations in the tunnel indicative of a non-traffic source outside of the road tunnel. When outdoor concentrations measured in 2012-2013 were compared with data from an earlier study using the same sampling sites in 1999-2001, the majority of PAH showed concentrations in 2012/13 around 50% lower than those in 1999-2001. Whilst the sampling periods were not extensive, this is suggestive of a reduction in emissions over that period. However, there were a few exceptions to this reduction, and by far the greatest was that of retene which had increased in concentration by 125% and 175% respectively at the two sampling sites between 1999-2001 and 2012/13. This seems likely to reflect a substantial increase in smoke from biomass combustion over this period.

PAH arise from a wide range of combustion sources, each of which emits a characteristic profile of individual PAH compounds (termed congeners). Receptor modelling methods based upon Positive Matrix Factorisation are able to disaggregate profiles of PAH compounds measured in the atmosphere such that they can be attributed to the sources responsible. In a study of UK national network data collected between 2002 and 2006 at 14 non-industrial sites, Jang et al. (2013) attributed PAH in UK cities to four different sources, i.e. diesel exhaust, unburnt petroleum, coal combustion and wood combustion. The contribution of wood burning to the total measured concentration of PAH congeners (particle plus vapour forms) ranged from 2.1 ng m^{-3} to 5.6 ng m^{-3} which represented between 7 and 23% of the sum of measured PAH. The contributions to the concentrations of the compound benzo(a)pyrene for which there is a European Union target value of 1 ng m^{-3} and a UK national ambient air quality objective of 0.25 ng m^{-3} ranged from 0.014 to 0.052 ng m^{-3} representing between 9 and 46% of the total concentration of benzo(a)pyrene at the 14 sites.

Such approaches estimate of course only the biomass contribution to airborne PAH, not PM more generally, and can be constrained by a lack of a representative ensemble woodsmoke fingerprints. On occasions where woodsmoke overwhelmingly dominates ambient PAH, better quantified apportionment can be made. A published example is in Lewis et al. (2002), where the ensemble woodsmoke PAH signature observed during Bonfire night is used to constrain a chemical mass balance model.

Other techniques used for inferring woodsmoke concentrations are also worthy of mention. Analysis of the carbon-14 isotope in atmospheric particles provides a measure of "contemporary" carbon, as opposed to fossil carbon in which all carbon-14 has decayed. Measurements of carbon-14 in the organic carbon and elemental carbon fraction of airborne particles can be used to infer an atmospheric concentration of particulate matter containing carbon derived from contemporary sources. In some locations, this may well be comprised mainly of woodsmoke but considerable caution is needed in interpreting such measurements as there are other sources of contemporary carbon in the atmosphere such as plant detritus, bioaerosol and cooking aerosol, the latter being of considerable interest as a possibly significant source of $PM_{2.5}$. The only UK study to report estimated concentrations of biomass smoke based upon carbon-14 analysis is that of Heal et al. (2011). From measurements in Birmingham they inferred contemporary elemental carbon to represent about 0.1 µg m^{-3} of $PM_{2.5}$, and assuming that woodsmoke has a 20% EC content, this equates to a smoke concentration of around 0.5 µg m^{-3}.

An instrumental technique which is being widely used to characterise non-refractory components of airborne particulate matter is aerosol mass spectrometry. The Aerodyne Aerosol Mass Spectrometer (AMS) measures mass spectra of sub-micrometre aerosol particles and application of Positive Matrix Factorisation (PMF) to the mass spectral data allows apportionment of particulate mass to different sources. One source which has been widely reported is wood smoke, although the method may have problems in distinguishing wood smoke from other solid fuel combustion emissions (Allan et al., 2010). The drawbacks of this method are the high capital cost of instrumentation, the uncertainties over attribution to specific sources, limitation to sub-micrometre particles and the difficulties that would be posed by future composition changes to wood smoke particles.

The other method which has received application involves the deployment of an aethalometer to measure the light absorption of particles collected on a filter tape at wavelengths in the visible and ultraviolet regions. Elemental carbon emissions from road traffic dominate the absorption at visible wavelengths while in the ultraviolet, woodsmoke is an additional important contributor and the method was developed to distinguish between the two sources in Swiss mountain valleys with few other sources of carbonaceous particles (Sandradewi et al., 2008a,b). A systematic study of the performance of the aethalometer in the UK atmosphere (Harrison et al., 2013) found a number of anomalous aspects to the data generated and it was recommended that great caution should be exercised in the deployment of aethalometer methods to estimate woodsmoke concentrations.

8.2 Ambient measurements

In estimating woodsmoke concentrations within the UK atmosphere, Harrison et al. (2012) used a ratio of woodsmoke mass to levoglucosan of 11.2 inferring concentrations in the UK atmosphere averaging across seasons of the order of 0.2 to 0.4 µg m^{-3} in $PM_{2.5}$. Fuller et al. (2014) used a conversion factor of 11 from levoglucosan to PM_{10} mass, and estimated an annual mean contribution of woodsmoke to PM_{10} of 1.1 µg m^{-3} in London. Both studies found strong correlations between sites across London and Crilley et al. (2014) also found correlations with rural sites outside of London. They concluded that local emissions of woodsmoke were superimposed upon a widespread regional background from many diffuse sources. By comparison with other pollutants, and examining the dependence upon wind speed, Fuller et al. (2014) concluded that the source is local to London, rather than deriving from long-range transport of emissions.

Air sampling took place in London in January and February 2012 as part of the winter campaign of the ClearfLo project (http://www.clearflo.ac.uk/). Air samples were collected for the analysis of woodsmoke tracers at the London North Kensington site and at the rural sites of Harwell to the west of London and Detling to the east of London. Concentrations of woodsmoke were estimated on the basis of measurements of levoglucosan, potassium and through the analysis of multiple wavelength data from aethalometers. Measurements of the carbon-14 content of the airborne particles which reflects the non-fossil carbon was used as a check on the estimates. Average concentrations of woodsmoke estimated for this period were 0.78 µg m^{-3}, 0.87 µg m^{-3} and 1.0 µg m^{-3} at Detling, North Kensington and Harwell respectively. Estimates of traffic exhaust $PM_{2.5}$ mass measured in ClearfLo over the same period were 1.26 µg m^{-3} and 0.61 µg m^{-3} at North Kensington and Harwell respectively (Yin et al., 2014). This suggests that woodsmoke contributes as much as half as traffic exhaust emissions to $PM_{2.5}$ in central London during winter, and more than the traffic exhaust component of $PM_{2.5}$ at Harwell. The annual mean concentrations for woodsmoke are likely to be around 50% of these values and to be primarily within the $PM_{2.5}$ size fraction. Since the annual mean for the traffic aerosol is likely to be reduced by less than 50% compared to the winter average, woodsmoke would be expected to be less dominant when considered on an annual basis. Nevertheless the data demonstrates a significant influence of wood burning on ambient $PM_{2.5}$ (Yin et al., 2014)

Complementary measurements of black and brown carbon were also made over a longer period at a height of 160 metres on the BT Tower. By comparison of periods when the atmosphere was well mixed with those when the sampling site was above the surface boundary layer top, it was possible to compare the absorptive properties of the carbon (which depend upon the relative amounts of black carbon from traffic and brown carbon from woodsmoke) in air originating from lower levels in London with regional air transported from other locations in southern England. The broad similarities between the two and the directionality associated with the woodsmoke concentrations was strongly suggestive of a widespread source of woodsmoke across southern England caused by emissions both within London itself and in the surrounding countryside. For this reason, broadly similar concentrations were seen at all three ground-level sampling locations (Harwell, North Kensington and Detling) and woodsmoke tracer concentrations at all three locations showed a strong temporal correlation (Crilley et al., 2014).

Source apportionment information derived from Pollution Climate Mapping (PCM) model results compiled for the 2012 compliance assessment for $PM_{2.5}$ can be used to calculate an estimate of the contribution to ambient concentrations from domestic wood (biomass) combustion for comparison with the estimates derived from ClearfLo. The modelled population-weighted annual mean $PM_{2.5}$ concentration in 2012 from domestic wood combustion is 0.114 µg m^{-3}. This has been derived from an estimate of emissions of 8.7 ktonnes per year, which has been derived from the 2011 NAEI by the application of a degree day scaling factor to take account of the difference in temperatures between 2011 and 2012. Rescaling this result to be consistent with the UK total emissions from domestic wood of 10.6 ktonnes per year from the 2012 NAEI shown in Figure 9 results in a population-weighted annual mean $PM_{2.5}$ contribution of 0.139 µg m^{-3}, which is lower than the estimate derived from the ClearfLo measurements by a factor of about 3.5. Figure 15 shows the spatial distribution of emissions from domestic wood assumed by the NAEI, which is the same as the spatial distribution assumed for domestic coal. This distribution shows zero emissions in many large urban areas for which natural gas is assumed to be the dominant fuel for domestic heating. Given the large population in these areas, this will have an important influence on the population-weighted mean. This highlights the uncertainties associated with the spatial distribution of emissions from domestic wood combustion which are in addition to the uncertainties associated with emission factors and levels of activity.

The published woodsmoke measurements referred to above, such as the ClearfLo data (Crilley et al., 2014) show woodsmoke concentrations in London which are of a similar magnitude to those at rural sites (Harwell and Detling) outside of London. This suggests an emission per unit area within London broadly similar to those in the rural areas, but a much reduced emission per head of population. Nonetheless, the urban emissions are likely to be substantially greater than those assumed by the NAEI, and consequently the population-weighted UK concentrations will similarly be higher than the estimate above.

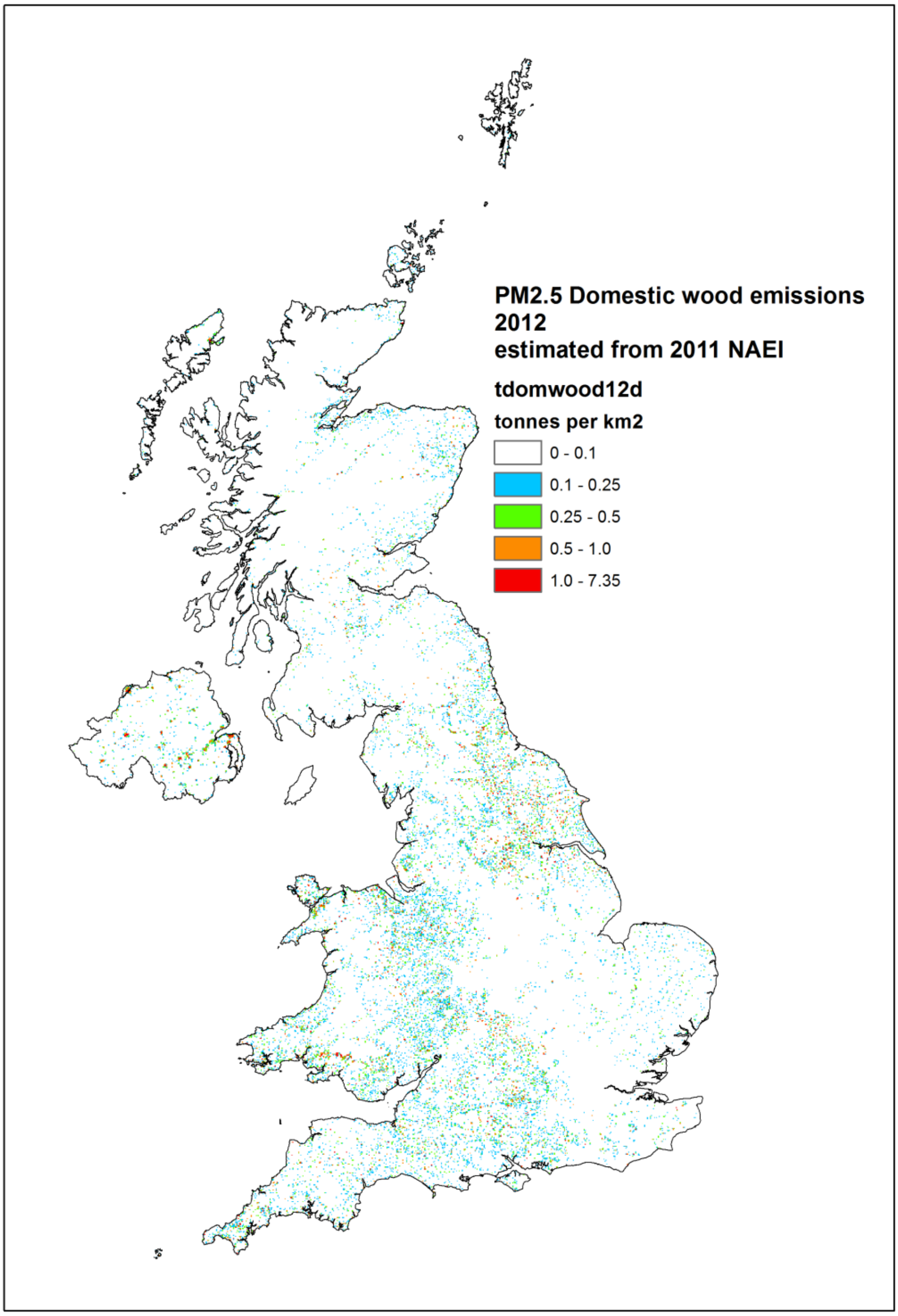


Figure 15: *Spatial distribution of $PM_{2.5}$ emissions from domestic wood combustion in 2012, derived from the 2011 NAEI (tonnes per km^2 per year)*

As part of the JOAQUIN project (www.joaquin.eu) daily samples of PM_{10} were collected using a Leckel onto aerosol filters over the period April 2013 to May 2014. Daily samples from Leicester (UK) and every six day samples from Amsterdam, Wijk-an-Zee (Netherlands), Antwerp (Belgium) and Lille (France). The filters were analysed by GC-MS for levoglucosan and its two isomers mannosan and galactosan. The analysis method has been validated in Cordell et al. (2014).

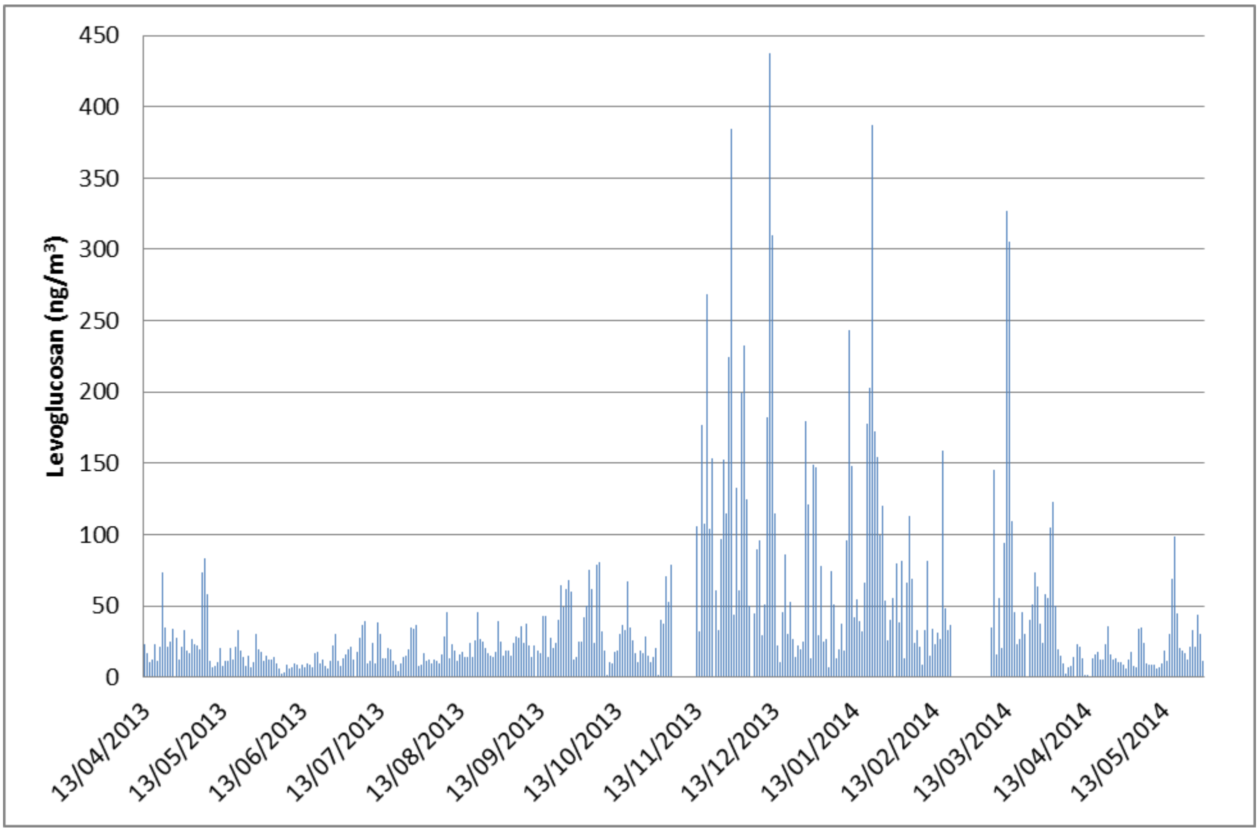


Figure 16: *Daily (24 h) mass concentration of levoglucosan as measured at the Leicester AURN, UK*

Figure 16 shows a time series of mass concentrations of levoglucosan as measured at the Leicester Automatic Urban and Rural Network (AURN) station. A clear seasonal cycle is observed with a winter burning period. Monthly means range from 10.5 ng m^{-3} in the summer months to an excess of 126 ng m^{-3} during the winter. Time variation analysis shows an enhancement of around 10-20% on Saturday and Sunday over the rest of the weekday concentrations indicative of domestic and in particular pleasure sources in line with the results of Fuller et al. (2014). The year mean levoglucosan concentration from April 1st 2013 to March 31st 2014 is 49.2 ng m^{-3}. Using a conversion factor of 10.7 (Schmidl et al., 2008b) to estimate the contribution of levoglucosan to PM$_{10,}$ this leads to an estimated annual average contribution of woodsmoke to PM$_{10}$ of 0.53 µg m^{-3} i.e. around half that estimated for London. Whilst this conversion factor is slightly different from that of 11.2 adopted by Harrison et al. (2012), it lies within the range of uncertainty suggested in that study.

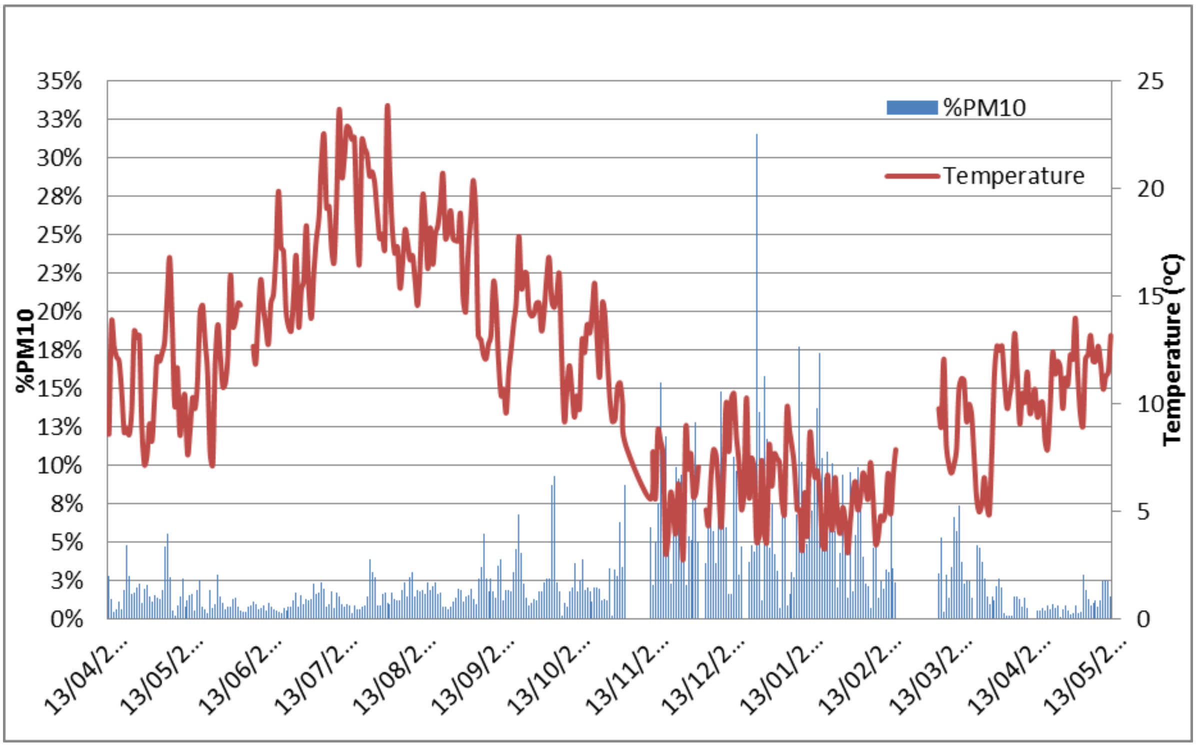


Figure 17: *Time series of the percentage contribution from wood burning to the total PM$_{10}$ and temperature variations at Leicester from April 2013 to May 2014*

Using the Schmidl et al. conversion factor (2008b) contributions of woodsmoke to PM$_{10}$ vary between less than 1% during the summer to typically around 8% in the winter (Figure 17). Particular peaks in contribution are observed on Christmas Day (> 30%). The type of wood burning can be assessed by the Levoglucosan (L) to Mannosan (M) ratio and also the Levoglucosan to Mannosoan and Galactosan (G) ratio (Fabbri et al., 2009). For Leicester, yearly averages of (L/M) = 3.1±0.3 and (L/(M+G)) = 2.3±0.1 which is typical of soft wood burning.

Retene measurements from selected sites across the UK PAH network are shown in Figure 18 and Figure 19 and represent monthly averages from June 2010 to December 2013. The sampling method used is a Digitel DHA-80 high volume PM10 aerosol sampler in order to comply with the requirements of the EU 4th Air Quality Daughter Directive. At all sites, particle bound PAHs are captured onto glass fibre filters. The measurements show a clear seasonal trend with significantly higher concentrations in the winter months which is indicative of emissions due to the use of solid fuel combustion for heating. The highest concentrations are found at sites in Northern Ireland (Figure 19) and in South Yorkshire and Derbyshire (Royston and Bolsover, Figure 18), although the winter peak concentrations at these sites appear to have reduced since 2010. Peak winter concentrations for other cities (Figure 18) show an increasing trend which may be indicative of increasing emissions from biomass burning. Unfortunately retene measurements from the PAH network are not available from April 2014 onwards, since a range of low mass PAHs were cut from the network. This will exclude the future use of retene as a possible tracer for wood burning from the UK PAH network as things currently stand.

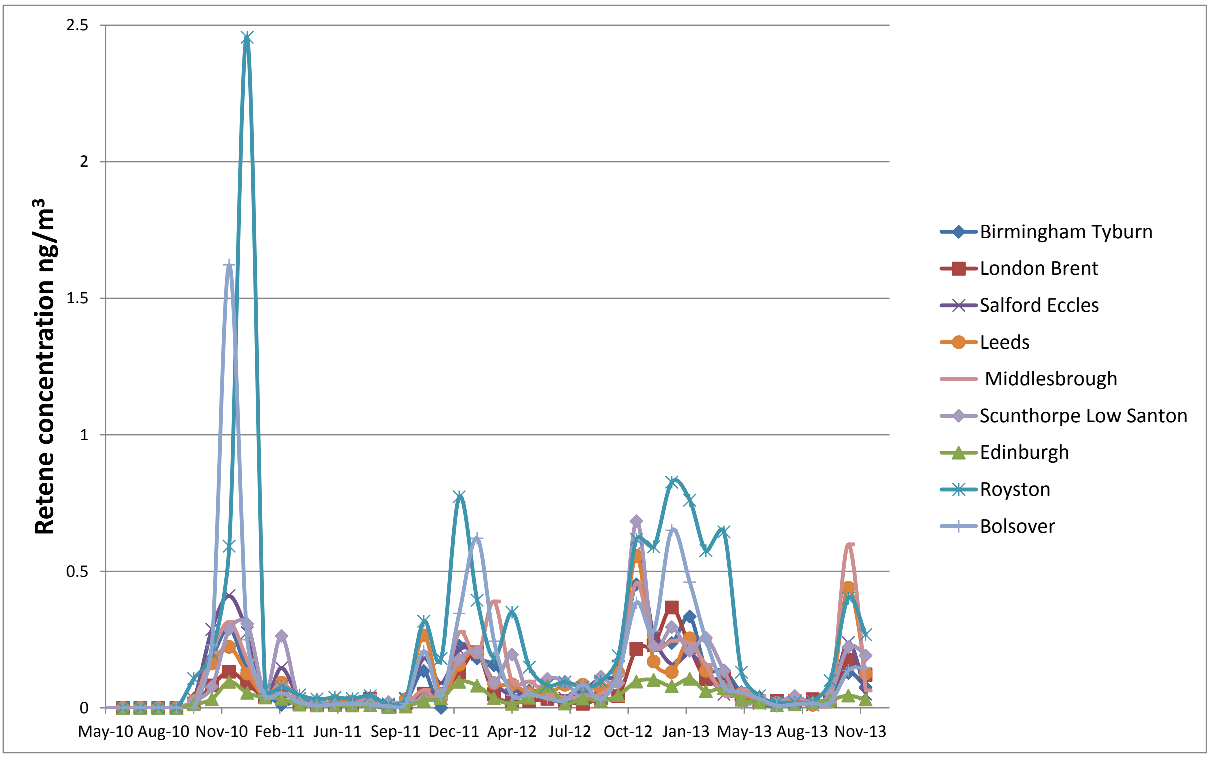


Figure 18: *Retene concentration measurements from the PAH network for selected sites around the UK (excluding Northern Ireland)*

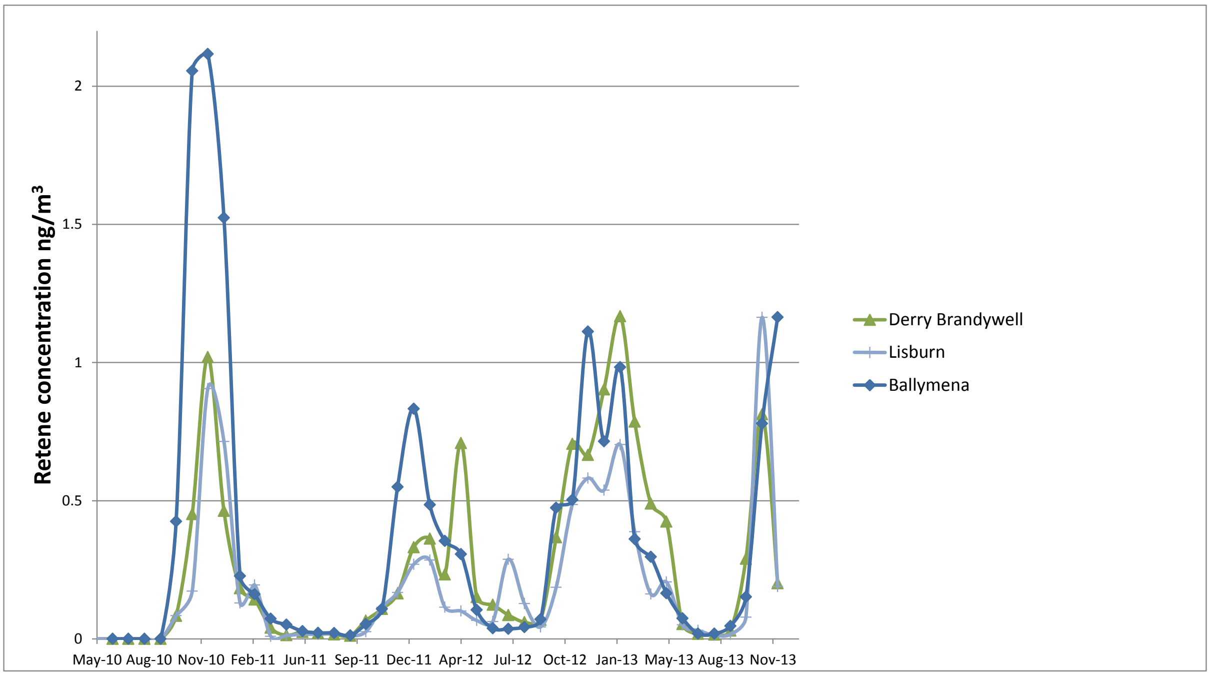


Figure 19: *Retene concentration measurements from the PAH network for selected sites in Northern Ireland*

8.3 Future outlook

Data and samples from the compliance network could potentially be used to evaluate proposed emissions factors for domestic biomass burning although there may be some concerns as to whether the location of current monitors (often close to traffic sources) would be ideal to assess trends in emissions from domestic sources. Also, the lack of future retene measurements will limit the use of the PAH network to assess wood burning tracers. So far levoglucosan measurements have been mainly obtained on a short term campaign basis. Longer term trends in levoglucosan concentrations at a wider range of sites would be extremely useful in assessing the impact of biomass burning on UK PM_{10} and $PM_{2.5}$ concentrations.

Since large uncertainties attach to emissions inventories for wood smoke, there is value in measuring temporal trends in airborne concentrations. This could most straightforwardly be achieved through measurement of either levoglucosan, or of potassium in the $PM_{2.5}$ size fraction in which interference from other sources such as sea spray and soil is minimal. There are, however, problems associated with this approach which arise from possible changes in the composition of wood smoke aerosol with time. Both the levoglucosan and potassium content of wood smoke depend critically upon the combustion conditions, with levoglucosan to potassium ratios high under conditions of poor burn-out and low under conditions of efficient combustion. Consequently, any trend towards cleaner domestic wood stoves or a change in the ratio of emissions from large utility boilers using wood pellets to those from domestic combustion in wood stoves or open fires would substantially change both the levoglucosan to potassium ratio and also the ratio of each tracer to the mass of particulate matter. Consequently, a prudent approach would be to measure both levoglucosan and potassium with a view to evaluating whether the ratio changes with time, which would then warrant further research into the specific sources and the ratio of tracer to particulate matter mass concentration. The other techniques are not considered suitable. The carbon-14 method is expensive, indirect and involves many assumptions. The aethalometer method is subject to interferences from traffic emissions and some secondary aerosol components, and the AMS method requires expensive equipment, is limited to sub-micrometre particles and requires a subjective expert data analysis.

9 The Potential Impact of Biomass Boilers and Combined Heat and Power (CHP)

In an attempt to reduce their overall carbon dioxide emissions and in response to incentives such as the RHI many local authorities (LAs) are considering proposals for the installation of biomass boilers either for heat provision within their own estate or for CHP (EPUK, 2013). They also control the planning process for large developments within their authorities which may be considering the provision of district heat or CHP from biomass plant. The air quality implications of such developments are a concern for many LAs who may have currently declared Air Quality Management Areas (AQMAs). Environmental Protection UK have produced guidance documents and a set of accompanying tools available for Local Authorities in England and Wales (EPUK, 2009, 2013) and Scotland (EPUK, 2010a), to assist with assessing individual planning applications for biomass and CHP (EPUK, 2012) installations and to help shape policy decisions with the aim of minimising impacts on local air quality. The documents give guidance on the regulatory regimes that apply to plant of different sizes as well as on mitigation measures and the potential advantages and disadvantages of different biomass fuel types. The reports conclude that potential conflicts between goals driven by climate change impacts and air quality can be avoided through the use of high quality, low emission plant. They suggest that the encouragement of larger plant, for example in conjunction with the development of heat networks, rather than a larger number of small plant is likely to result in systems which are easier to control from an air quality perspective. One reason for this is that abatement technologies become more cost effective for larger plant sizes and are currently more highly developed for larger systems than for smaller community and domestic scale boilers.

The overall impact of CHP on air quality may be complex. Since CHP provides overall efficiencies greater than those of conventional power generation through combustion, and would replace domestic scale heat provision (e.g. through domestic boilers), the expansion of CHP could potentially lead to lower overall emissions of pollutants. However, this could be combined with local increases in key pollutants such as NO_x and PM. CHP is also more likely to be located in regions of high population compared to conventional power plant. The overall uncertainties in providing projections in potential air quality benefits or dis-benefits are large, as pointed out by Apsimon and Oxley (2013).

9.1 Mitigation Strategies

The restrictions imposed by the RHI and MCPD will go some way to control potential emissions of NO_x and PM as indicated in Tables 3 and 4, but LAs also have some planning powers at their disposal which may be used to add further restrictions. However, it is worth pointing out that biomass boilers themselves do not require planning permission, except where additional building work is required e.g. for a fuel store or a completely new installation. The conversion of small boilers from gas to biomass are less likely to be captured through the planning process than larger or new installations (EPUK, 2013). The Mayor of London commissioned a report (AMEC, 2013) which investigated the impact and cost implications of a range of control measures for biomass boilers ranging from selective catalytic reduction (SCR) of NO_x to cyclones and ceramic filters for PM reduction. The report concluded that all of the additional control measures were likely to lead to a reduction in CO_2 savings (of the order of a few percent) due to overall energy costs associated with their operation, efficiency losses or direct CO_2 emissions. However, several technologies were able to provide substantial reductions in NO_x and PM which would help to mitigate local air quality impacts. These are briefly summarised in Table 8 along with a comparison of their equivalent annualised costs expressed as a percentage of the full installation cost for the unit based on 6,000 hours of operation of 10 years (AMEC, 2013). The most effective methods for control, ceramic filtration and SCR for PM and NO_x respectively, are not surprisingly also the most expensive.

***Table 8**: Effectiveness of post combustion emissions reduction strategies for solid biomass boilers (AMEC, 2013)*

Proposed post combustion control technology	**Reduction Efficiency %**	**Indicative Emissions Factor g/GJ**	**Equivalent limit in RHI g/GJ**	**Equivalent annualised costs as % of initial investment**
Selective Non Catalytic Reduction (SNCR) for NO_x	40	70	150	
Selective Catalytic Reduction (SCR) for NO_x (>1 MWth)	80	23	150	
Multi-cyclone	50	17	30	1-2%
Ceramic filter	99	2	30	<10%

In order to find a balance between cost, potential CO_2 savings and mitigation of air quality impacts, the report suggests a tiered approach to proposed emissions limits for biomass boilers in London. The required limits are suggested to be based on Air Pollution Exposure Criteria (APEC) bands which specify how close current air pollution levels are with respect to annual air quality standards. Hence if an area is currently exceeding or close to failing standards then more stringent controls would be required. The proposed emissions limit for PM for areas within 5% or exceeding the annual limit for PM_{10} is 6 g/GJ which is 5 times lower than the limit proposed in the RHI and would necessitate the use of controls which are more effective than commonly used cyclones such as filtration or electrostatic precipitation methods. Note that concentrations close to or exceeding the annual mean limit value for PM_{10} are not expected across locations in the UK and hence the 24 hour limit is likely to be more relevant here (Defra, 2014). For NO_x, a value of 70 g/GJ is suggested for such areas, which is likely to require the use of staged combustion and or SCR. Note that concentrations exceeding the annual mean limit value for NO_2 are more likely than for PM_{10}, especially close to busy roads (Defra, 2014).

AEA (2010) also produced a detailed review of potential abatement technologies for particulate emissions from wood-burning boilers including their cost effectiveness. They concluded that a large range of boilers are expected to meet the requirements of the RHI without the need for more expensive add on particle abatement technologies. A range of emissions factors for different boilers are provided in the report which vary from 3 g/GJ to 29 g/GJ, and hence within the limits set by the RHI (30 g/GJ) there could be a wide range of impacts depending on technology choice. The report suggests that more expensive particle abatement techniques, such as high efficiency ceramic or fabric filters, are only likely to be considered by Local Authorities in cases where Local Air Quality Management Areas or sensitive receptors exist, in line with the proposals by AMEC (2013) for the Greater London Authority. The AEA report suggests that current regulatory development is focussing research on small-scale abatement measures which may result in a lowering of the size range of application for technologies such as electrostatic precipitation and high efficiency particle filtration. This could improve the impacts of <50 kW devices but may require support from appropriate emissions limits since these technologies carry a significant capital cost (e.g. around 15% of the installation cost for a ceramic filter (AEA, 2010)).

The AEA report also suggests that gasification and pyrolysis techniques offer the best potential to lower particulate emissions since they are based on converting solid fuels into more uniform gaseous (or liquid) fuels, thus improving mixing and lowering the potential for particle formation. However, the report concludes that whilst small-scale gasification has a long history, current technologies are largely used in plant that is larger than 2 MW. Further information on small scale gasification boilers and comparative PM emissions factors relative to direct biomass combustion boilers would be useful.

9.2 Air Quality Impact Studies of Proposed Biomass Plant

As part of planning application procedures for proposed biomass heating and CHP plant, a number of air quality impact studies have recently been performed. It should be pointed out that the requirement for such studies is dependent on the particular planning authority and not all planning applications for biomass boilers have been accompanied by detailed impact studies. The guidance given by EPUK (2013) is that neither the HMIP technical guidance note D1 (HMIP 1993) nor the third edition of the 1956 Clean Air Act Memorandum are suitable for estimating stack heights for biomass boilers and that detailed dispersion studies should be undertaken. One such example is the assessment for a proposed biomass boiler for energy provision for Greenfaulds Secondary School in Cumbernauld, North Lanarkshire carried out by Ricardo-AEA (2014) on behalf of Morrison Construction. The proposed boiler would have a maximum output of 450 kW and would be fuelled using woodchip. Air quality impacts in the surrounding area were calculated using ADMS 5.0, incorporating 5 years of meteorological data (2008-2012), with chimney heights varied between 6.85 and 9.85 m above ground level. A “worst-case” approach was adopted with emission rates assumed to be close to the limit imposed by the RHI.

With a stack height of 7.85 m, the maximum stack contribution to the annual mean predicted NO_2 concentrations at any point model domain was 14.2 µg m^{-3}. For PM_{10}, the maximum stack contribution to the annual mean predicted at any point model domain was 3.0 µg m^{-3}. A number of sensitive receptor points were also included but as expected, their modelled concentrations were lower than the maximum grid values. The contributions of the boiler emissions at the worst case receptor point represent 19% and 9% of the annual mean objective concentrations of 40 µg m^{-3} and 18 µg m^{-3} for NO_2 and PM_{10} respectively (note that the location is in Scotland). For short-term objectives, the equivalent figures represent 25% and 18% respectively of the hourly objective concentration of 200 µg m^{-3} for NO_2 and daily objective of 50 µg m^{-3} for PM_{10}. When combined with local background concentrations, 64% and 76% of the annual objectives, and 42% and 66% of the short-term air quality objectives, are reached for NO_2 and PM_{10} respectively at this stack height. The highest contributions at any point on the receptor grid to the annual mean objectives for NO_2 and PM_{10} were 36% and 17% respectively, and 26% and 19% respectively of the hourly/daily objective concentrations for NO_2 and PM_{10}. When combined with local background concentrations, this represents 80% and 83% of the annual objectives and 44% and 67% of the short-term air quality objectives for NO_2 and PM_{10}.

The effect of stack height was investigated within the study and increasing the stack height from 6.85 to 9.85 m was found to reduce the predicted annual mean contributions of stack emissions to PM_{10} and NO_2 by almost a factor of 2. A similar factor of 2 reduction was also shown for the percentage modelled contribution of the stack emissions to 1 hr average objectives for NO_2. For PM_{10}, at the lowest stack height, the modelled concentrations at the worst case receptor, when combined with background levels, led to concentrations that were 98% of the annual mean objective, and thus could lead to cause for concern from the perspective of compliance with the Scottish air quality objective. Hence, for the proposed biomass plant, designing the installation with an appropriate stack height was clearly critical.

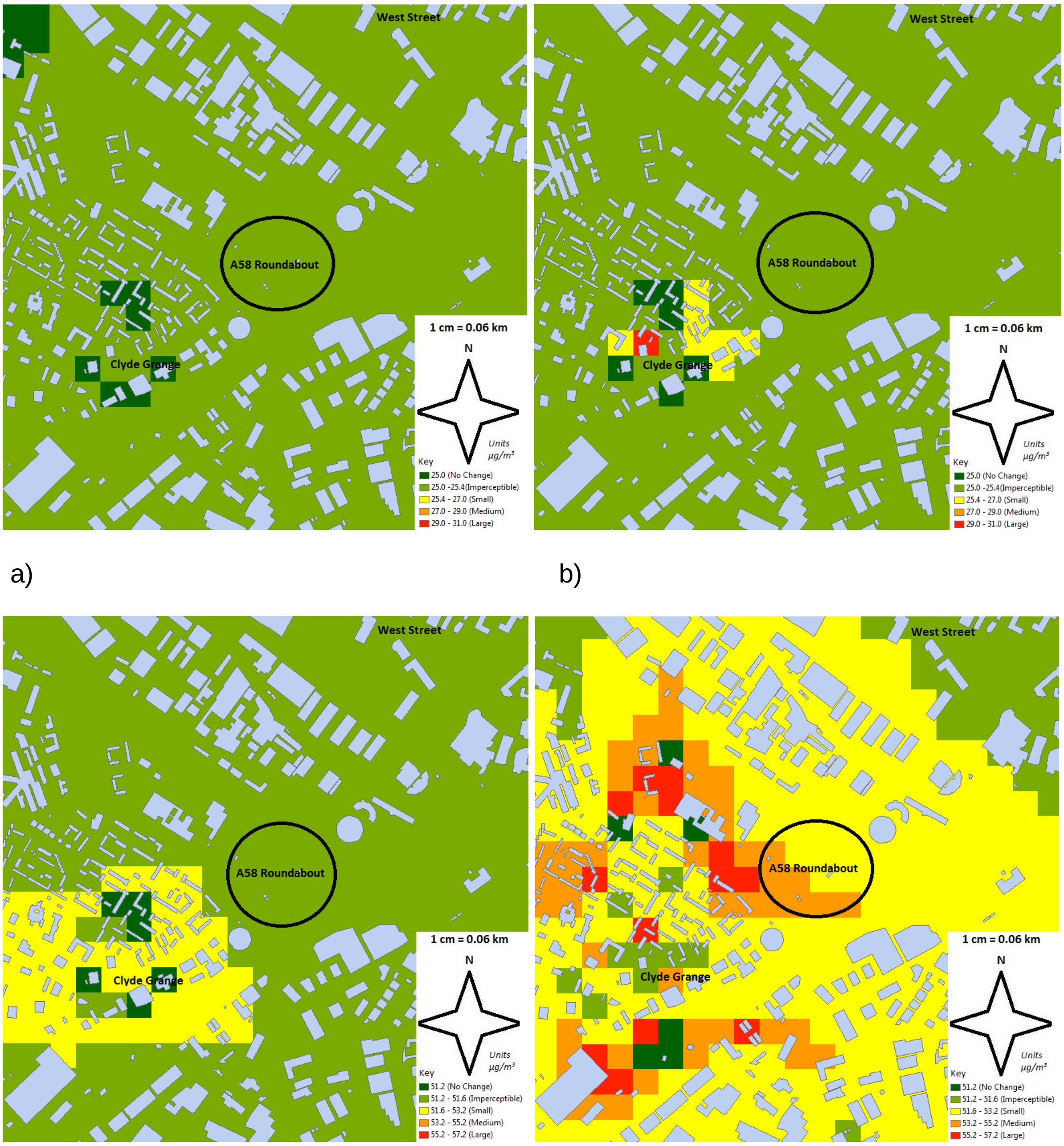
Using EPUK guidance (EPUK, 2010b) the change in modelled NO_2 annual mean concentrations using a stack height of 7.85 m was described as 'slight adverse' at several receptor locations and 'negligible' at most others. For PM_{10}, the change in modelled annual mean concentrations was described as 'negligible' at all modelled receptors and 'slight adverse' at the point of maximum impact. On this basis, a stack height of 7.85 metres was adopted.

Overall this particular study demonstrated that the proposed boiler should not lead to exceedances of the air quality standards in the local area under worst case scenarios. This finding is very dependent on the background air quality at the particular site as well as the size and nature of the plant and the stack height chosen, and therefore the outcomes cannot easily be generalised. Such studies are non-trivial to undertake, but would be necessary in areas where the existing air quality is close to or above current legislative limits.

A similar study was undertaken for a proposed biomass boiler installation in the area of Wortley, Leeds (Conibear, 2014). The application is to supply a district heating system at residential flats: the Clydes, and will replace the current electrical storage method with a biomass boiler (170 kW), in addition to three back-up gas boilers to be used for peak demand situations (3 x 500 kW in total). In terms of emissions, the system will add a local source of PM and NO_x, since electrical heating has no local emissions but rather would be associated with emissions from a distant power generation source.

A similar approach to air quality assessment was used to that of Ricardo-AEA (2014) described above; namely that emissions limits equivalent to those stipulated by the RHI were used, and a worst case approach was adopted. ADMS 5.0 was used for the dispersion assessment based on hourly meteorological data from 2011-2013. 2013 was the worst case year for annual average assessments and is used in the presentation of results. Downwash from buildings within 5 stack heights of the boiler flue were modelled in ADMS using idealised building shapes based on data from Landmap (2014). Background air quality was estimated using maps provided by the Defra Local Air Quality Management System (http://uk-air.defra.gov.uk/data/laqm-background-maps?year=2011).

The main difference between the Clydes system and that of Greenfaulds Secondary School is the proposed height of the emissions source. At the Clydes, the proposal is to integrate the flue into one of the high rise buildings, terminating 2 m above the roof height which is 53 m above ground. For comparison, a case study assessment was also performed assuming that the flue was to terminate 2 m above the proposed boiler housing i.e. 6 m above ground. The study therefore serves to highlight the importance of chimney height on the resulting local air quality impacts. Figure 20a,c shows that for the 53 m chimney, changes in both annual average and maximum hourly average NO_2 concentrations are classified as either small or imperceptible according to the EPUK Air Quality Impact Assessment Magnitude of Change criteria (EPUK, 2010b). However, had the design used a short chimney attached to the boiler housing, the impact would have been raised to large in some regions close to the stack where downwash played a significant role (Figure 20b,d). Incorporating the flue within a high rise building therefore presents a low impact application of energy from biomass in Leeds. However, as biomass fuelled district heating systems are rolled out to other areas of the city in response to the RHI, it will be essential to perform air quality impact studies to determine appropriate chimney heights or required mitigation measures.



a)

b)

c)

d)

Figure 20: *Predicted concentrations and magnitude of change assessments for ground level NO_2 concentrations in the region surrounding the proposed boiler installation at the Clydes in Leeds a) Annual average concentrations for 53 m chimney, b) Annual average concentrations for 6 m chimney c) Maximum hourly concentrations for 53 m chimney d) Maximum hourly concentrations for 6 m chimney*

9.3 Future Projections

A collaborative project between Imperial College and Ricardo-AEA examined the possible impact on air quality of a future energy scenario for London with extensive deployment of CHP and district heating networks. This work, which was part of the APRIL network (http://www.april-network.org/emissions/) included the compilation of an emission inventory and air dispersion modelling for a “coordinated energy scenario” developed in a GLA study towards achievement of the Mayor’s objectives of 25% of London’s energy from decentralised energy (DE) sources by 2025. This DE scenario gives an estimated saving of 0.8 Mt CO_2 per year, but involves siting combustion plants in highly populated areas of London. The focus of this study was on NO_x emissions. Biomass CHP and Biomass district heating plants were included in this scenario, along with several other technologies.

Table 9 shows a summary of the key emission assumptions for this study. Ambient concentration results calculated using the PCM model are shown in Table 10, with the results expressed as population-weighted mean annual mean NO_x increments for London for the different technologies. The modelled NO_x concentration increments for biomass CHP and biomass district heating for this scenario are shown in Figure 21.

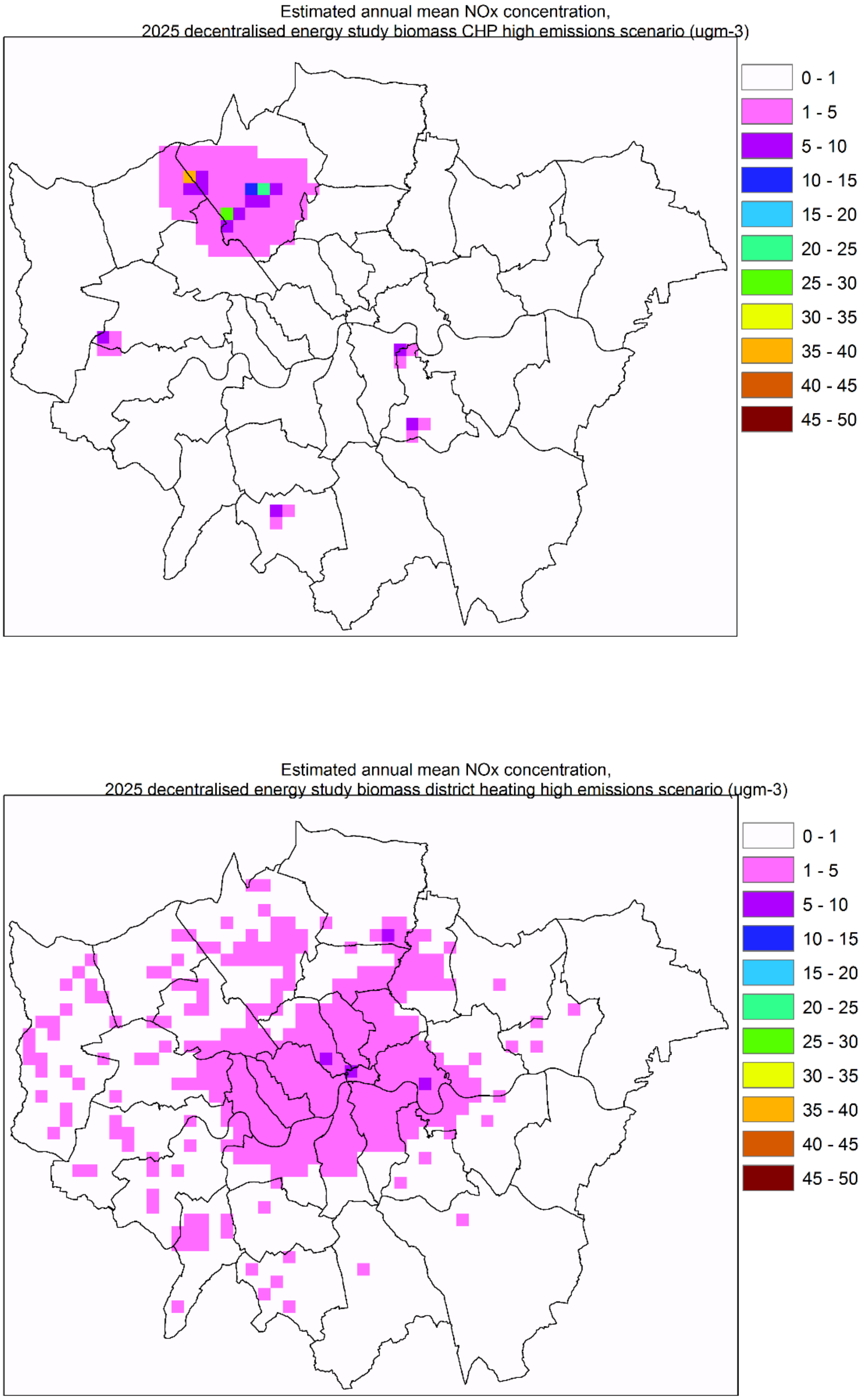
Table 9: *Emission assumptions for London distributed energy scenario in 2025*

Technology	**GWh/y**	**gNO_x/kWh**	**Number of units**
CCGT	16954	0.17	30
Anaerobic digester	249	0.94	6
Waste gasification	2972	0.73	60
Small gas engine	964	1.75	60
Biomass CHP	1308	0.38	17
Biomass district heating	1596	0.55	305
Sum	24043		

Table 10: NO_x *concentration results for London distributed energy scenario in 2025*

Technology	**London population-weighted annual mean** NO_x **concentration (µgm^{-3})**	**1000 * concentration/GWh/y**
CCGT	0.957	0.056
Anaerobic digester	0.152	0.609
Waste gasification	1.704	0.573
Small gas engine	2.215	2.298
Biomass CHP	0.454	0.347
Biomass district heating	1.070	0.671
Sum	6.552	

The largest contribution to ambient NO_x concentrations per unit of energy generated was predicted to be for small gas engines. The lowest contribution was predicted to be for the technology with the largest plant: Combined Cycle Gas Turbine (CCGT). Biomass CHP and Biomass district heating were predicted to have an impact intermediate between these two extremes. These findings are consistent with many of the other conclusions within this report that the impact on ambient concentrations per unit of activity is generally lower for larger plant.



Estimated annual mean NOx concentration, 2025 decentralised energy study biomass CHP high emissions scenario (ugm-3)

a)

Estimated annual mean NOx concentration, 2025 decentralised energy study biomass district heating high emissions scenario (ugm-3)

b)

Figure 21: *Modelled* NO_x *increment in London for 2025 distributed energy scenario. a) Biomass CHP, b) Biomass district heating*

10 Summary and Recommendations

A number of key questions were raised at the start of the report and the main findings in relation to these questions will now be addressed along with recommendations.

Is there is a well-developed evidence-base with respect to the current range of, and likely changes in, source distributions from biomass burning?

- With respect to larger biomass burning technologies, there is a strong evidence base.
- Domestic wood burning is a challenging issue owing to the difficultly of assessing the numbers, trends, operational characteristics and performance of domestic wood burning appliances across the UK.

Recommendations

1. Investment should be made in survey data, to provide substantially improved information on trends in domestic stove installation numbers, geographical distribution, usage and operational patterns. This is essential in allowing better quantification of emissions.
2. Further research into the development of suitable cost effective PM control methods for the domestic sector should be undertaken, and given a high priority. Particular emphasis should be given to developing control methods which could be retro-fitted to current stock.

Are biomass sources well represented within current UK inventories and is there any evidence for significant inaccuracies?

- The UK emission inventory is considered to have a high standard of completeness in terms of source types and categories with respect to biomass burning sources. The quantification of emissions from some of these sources is however subject to high uncertainties in terms of real world emissions factors.
- Emissions from large-scale sources such as from co-firing within power stations can be estimated with significantly lower uncertainties than for smaller scale sources.
- For biomass boilers below 1 MW, emissions factors and activity data are uncertain because they are more variable and may differ under real world conditions compared to regulatory tests.
- Biomass consumption for domestic combustion is highly uncertain, as it is currently informed by national scale data such as fuel supply rates and limited survey data.
- Real world emission factors from small scale devices depend on a number of operational factors which are highly variable between different appliances, users and test regimes.
- Currently, the spatial disaggregation of biomass related emissions is based on previous estimates for coal combustion, and is therefore highly uncertain and neglects growing domestic usage, particularly in urban locations.

Recommendations

3. Real world emissions data for biomass boilers and CHP units that fall under the Renewable Heat Incentive over a range of operating conditions are needed. The current availability of these data should be reviewed to determine whether additional

emissions measurement is required to deliver comprehensive information on emissions from these sources.

4. A programme of real world emissions measurements from domestic wood stoves should be undertaken, as these data are critical in establishing suitably accurate emissions factors for use in the NAEI.
5. The spatial disaggregation of biomass combustion related emissions needs to be improved within UK inventories to better reflect domestic and community scale sources, particularly within urban areas. Predicting impacts in terms of population weighted mean exposure would also benefit from knowledge of the spatial distribution of current and future biomass consumption. The recent DECC survey data could be used to provide information on the location and usage patterns of different appliances but surveys of this type will need to be repeated in order to provide information on trends.

Will the impact of future levels of biomass burning, with proposed policy measures in place, lead to a significant change in ambient concentrations of major pollutants and further in population weighted mean exposure?

- Biomass burning activity data used in the NAEI suggests that there are increasing emissions of PM_{10}, $PM_{2.5}$, NO_x, BaP and dioxins from this source category at a range of scales of combustion.
- Domestic emissions of wood smoke have an especially high impact on ground-level concentrations relative to those from elevated sources. If current trends in domestic biomass burning continue upwards, it is inevitable that there will be an increase in airborne concentrations of wood smoke which will consequently lead to increased population exposures. Wood smoke is already a significant contributor to population weighted $PM_{2.5}$ exposures and is liable to make an increasingly important contribution in the future for this reason, and as emissions from some other sources decline.
- Based on estimated national emissions trends, the largest impact of biomass burning is likely to be on national emission totals of PM, PAHs and dioxins, and less so for NO_x, particularly if increases in biomass burning observed in recent years represent a fuel switching from gas. However, increasing the number of installations of biomass boilers for CHP may lead to local increases in NO_x within urban areas as emissions are displaced from large power generation sources.

Recommendations

6. Emissions from newer appliances are likely to be lower than for legacy devices (e.g. domestic stoves, boilers) due to developing legislation. Therefore capturing information on device replacement and fuel switching is critical to assessing net changes in emissions, and it is recommended that data are sourced on this, either through surveys or collecting data via processes associated with the purchase or installation of new appliances.
7. In general the turnover of domestic combustion devices is likely to be much slower than for emissions sources in the domestic road transport sector. Incentives to encourage the replacement of older devices with newer ones which meet required regulatory constraints would be beneficial, as would improving guidance on solid fuel

stove operation. It is recommended that consideration is given to the feasibility of supporting these actions.

How strong is measurement evidence in relationship to air pollution from biomass burning?

a. ***Does measurement evidence currently suggest an increasing trend in the contribution of biomass burning to PM?***
b. ***Will the current set up of the monitoring networks allow us to assess future trends in the impact of biomass burning on air quality?***

- The current measurement evidence is insufficient to suggest any long term trends.
- Short term measurements suggest a significant contribution of biomass burning to PM during winter in urban areas.
- Current long term monitoring activity does not address the need for specific tracer measurements for compounds such as levoglucosan. Retene analysis has also recently ceased.
- It is unclear whether the current location of monitoring stations is optimal for capturing the impact of biomass sources since, particularly within urban areas, many sites are located with the aim of assessing the influence of traffic emissions.
- Biomass combustion also contributes to concentrations of organic compounds such as PAH and dioxins which are currently monitored at very few locations.

Recommendation

8. Long term measurements (initially 5 years) of biomass burning tracers such as levoglucosan and/or potassium should be established in order to provide a picture of the changing contribution of biomass sources to atmospheric PM and for inventory evaluation. The physical location of measurements should reflect the predominant sources, which will include domestic heat provision or district heating systems and CHP.

Are the* air quality impacts *well assessed with respect to the introduction of biomass-based heat provision and combined heat and power (CHP) systems into the urban environment within UK?

- The number of biomass boilers registered under the Renewable Heat Incentive (RHI) scheme is carefully monitored and reported by Ofgem each quarter. The number of accredited installations and overall installed capacity is growing in both the domestic and non-domestic sectors. What is not captured is the heating system that the new biomass installations replace (i.e information on fuel switching) and therefore it is difficult to assess whether the net impact on air quality will be positive or negative. There has not to date, been a comprehensive study on the UK wide impact of the RHI on air quality.
- A range of abatement technologies are available which could substantially reduce the impacts of biomass boilers at the district/community scale. The most effective measures for abating NO_x and PM are not explicitly required by the RHI.
- Air quality impact studies based on dispersion modelling are emerging as planning applications are made for district heat and CHP systems and are essential to ensuring

that air quality exceedances or substantial increases in population weighted mean exposures do not occur.

- For community scale boilers (e.g. used in CHP or district heat networks), Local Authorities have some planning powers that allow them to impose stricter emissions limits than those mandated in National or European policies. These are likely to be most beneficial in regions whose current air quality is close to or exceeds current standards, but will necessitate the use of abatement technologies such as SCR (for NO_x) and particle filtration or ESP for PM_{10} and $PM_{2.5}$.

Recommendations

11. Where there is introduction of significant biomass into the local urban infrastructure, impact case studies should be undertaken to determine the potential AQ impacts.
12. Information on fuel switching should be captured where relevant as part of the RHI accreditation process.

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