Up in smoke

The contribution of domestic outdoor burning to UK particulate matter emissions

CLEANSTOVE consultants

Customer:

This work has been prepared by Clean Stove Consultants Ltd. for the Stove Industry Alliance with the support of CEFACD.

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Date:

November 2021

Suggested citation:

Cottom, J.W. and Mitchell, E.J.S. (2021). *Up in smoke: the contribution of domestic outdoor burning to UK particulate matter emissions.* Report by Clean Stove Consultants Ltd. for Stove Industry Alliance. Leeds, UK.

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Executive summary

Background

This document has been produced to determine the contribution that domestic outdoor burning makes to total particulate matter (PM) emissions in the United Kingdom.

Domestic burning has been the subject of numerous media reports and policy documents recently, including the Clean Air Strategy 2019 which set out clear goals to reduce emissions from this source. In the last two years, several key documents, surveys and reports have been published which have improved our understanding of indoor domestic burning habits and practices. This has enabled us to improve emission models with better data on:

- The types of fuel burned (e.g. dry wood briquettes or wet wood logs),
- The type of appliance used (e.g. open fire or modern Ecodesign stove)
- The pollutant emission factors for each fuel/appliance combination

The National Atmospheric Emissions Inventory (NAEI) is the body responsible in the UK for modelling atmospheric emissions. Emissions models such as that used by the NAEI rely on a simple concept whereby total emissions to the atmosphere are the product of source activity data and emission factors, as shown below:

Annual atmospheric emissions	=	Activity data	×	Emission factor
(tonnes of pollutant)		*(amount of fuel burned each year)*		*(tonnes of pollutant emitted per unit of fuel burned)*

In this work, each term above has been thoroughly explored for all domestic burning sources, but with a particular focus on domestic outdoor burning which has had much lower attention in modelling efforts due to it being far less understood than indoor domestic burning.

What is outdoor burning?

Outdoor burning in residential settings is an important and growing source of air pollution, which includes fuels burnt in outdoor appliances for heating, cooking or waste disposal:

Outdoor heating	Outdoor cooking	Waste disposal
• Chimeneas • Fire pits • Braziers • Wood-fired hot tubs	• Barbecues • Smokers • Pizza ovens	• Bonfires • Garden incinerators

Instances of outdoor burning are believed to have increased in recent years, particularly in the summer 2020 and 2021 due to the global coronavirus pandemic meaning people spent more time at home. Outdoor burning appliances are less regulated than indoor appliances and therefore they are generally cheaper, less efficient and more polluting. In addition, much less heat is used usefully outdoors compared to indoor heating, which, if burning biomass can act as a low carbon alternative to traditional fuels.

What are the current estimates of the contribution of outdoor burning to particulate matter emissions?

Currently the NAEI provides a sector total for domestic burning under NFR/CRF sector code 1A4bi – residential stationary combustion. However, there is a lack of distinction between indoor and outdoor domestic burning. Instead, the only sources of outdoor domestic burning are those under the separate categories of small-scale waste burning and bonfire night. In the 2019 inventory, these contributed only 5% of domestic $PM_{2.5}$ emissions, however, this is believed to be due to the NAEI currently underestimating outdoor burning due to a lack of reliable activity data. The implications of this are that domestic PM emissions have largely been attributed so far to indoor burning. This has led to regulations largely focussing on these indoor sources, whilst outdoor sources have received almost no scrutiny.

What we did

The aim of this work was to improve the model estimates of domestic $PM_{2.5}$ emissions from combustion of solid fuels from both indoor and outdoor sources. Through a multi-faceted approach, activity data and emission factors have been carefully evaluated to produce a new domestic solid fuel burning emissions model for the UK.

The data sources which have been drawn upon in construction the model are summarised below:

Sources of activity data	Sources of emission factors
Kantar survey (2020) A DEFRA-commissioned survey of more than 46,000 people across the UK spread over 10.5 months (Kantar, 2020) which helped to understand indoor and outdoor burning regularity, fuel types and mass of fuel burned. The accompanying Defra analysis was used as a basis for the central estimates of both indoor and outdoor activity data.	**EMEP/EEA Inventory Guidebook** The 2019 iteration of the Guidebook is the latest set of emission factors which have much more granular detail than the NAEI. For example, it includes $PM_{2.5}$ emission factors by fuel type and appliance type with 95% confidence intervals.

Industry survey (SIA, 2019) Over 10,000 households that undertake domestic burning were surveyed which helped to understand burning habits, fuel types and appliance types. This was a significantly greater number than the 2014-15 Domestic Wood Use Survey. The results of this survey were used to cross-check the Kantar results. **Sale and trade data** Sales and trade data was used to verify figures in the above sources and to provide uncertainty ranges. **Peer-reviewed literature** Recent literature publications were consulted to help inform uncertainty estimates of activity data.	**NAEI** Many of the values used in the NAEI are based on EMEP/EEA Guidebook values but where they have deviated due to UK-specific factors, we have endeavoured to account for this. **USEPA AP-42** The USA emissions inventory toolkit. It has mostly been used in this work for cross-checking purposes. **Peer-reviewed literature** A number of recent journal articles have been cited in this work, offering more up-to-date and granular data than the EMEP/EEA Inventory Guidebook.

A bespoke emissions model has been constructed with Monte Carlo analysis used to account for uncertainty propagation. This relied on assigning probability density functions to each source activity data and emission factor. Within this work, triangular distributions were used for all inputs in order to account for the often skewed nature of input estimates.

What we found

The key findings in this work were that domestic solid fuel burning releases 47,643 tonnes of $PM_{2.5}$ a year, of which 25,692 tonnes (54%) arise from indoor burning and 21,951 (46%) tonnes arise from outdoor burning. This dramatic increase in the contribution of outdoor burning from the previous estimate of 5% is primarily due to two reasons. Firstly, significant quantities of fuel were found to be burnt outdoors, much of which is unregulated and highly polluting based on their emission factors, for instance bonfires. Secondly, the estimate for indoor wood fuel use is much lower than the estimate used in the NAEI.

This is due to key methodology differences. In this work we made two important changes – firstly we used activity data from Kantar (2020), supported by other sources, which is more up to date than that used in the NAEI, and secondly we considered the different moisture contents, fuel types and appliance types.

When analysing the contribution of outdoor burning and indoor burning for just wood fuels (including green waste and waste/grey sourced wood), outdoor wood burning was found to account for 51% (17,518 tonnes/year) of all domestic wood burning $PM_{2.5}$ emissions (total of 34,465 tonnes/year). These results therefore signify the importance of accounting for these outdoor sources and has implications for the future design of regulations which currently largely neglect outdoor domestic burning.

Further results indicate that the largest contributor to domestic outdoor burning emissions are those from green waste, waste wood and to a lesser extent rubbish. Combined, these account for 90% of all outdoor burning emissions, or equivalent to 41% of total domestic $PM_{2.5}$ emissions (indoor and outdoor). Although some of these fuels may be burnt in appliances such as firepits, it is believed the majority are likely to be burnt on garden bonfires. This therefore suggests that bonfires are one of the major sources of domestic $PM_{2.5}$ emissions, possibly contributing as much as all wood burnt indoors. As such, it is recommended that bonfires deserve significantly more scrutiny than at present whereby no national regulations exist except those that prohibit statutory nuisance from burning as defined under the Environmental Protection Act of 1990. Moreover, as policies such as Ecodesign and the Domestic Solid Fuel Standards come into effect, indoor emissions are likely to continue to reduce in the future meaning the importance of outdoor burning as an emission source will possibly increase. This is particularly the case considering outdoor burning is believed to have increased in popularity recently based on more demand for outdoor burning appliances such as barbeques, firepits and chimineas.

Lastly, it was found that Ecodesign stoves contribute just 2.7% of $PM_{2.5}$ emissions from the burning of wood logs (including waste wood and briquettes) despite using 9% of the fuel. On the other hand, open fires contribute 39% of the emission from burning of wood logs whilst burning 26% of the wood. These results therefore highlight the substantial improvements in emissions from indoor sources that can be made by switching to modern Ecodesign stoves.

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1. Introduction

Recent key policy documents such as the DEFRA Clean Air Strategy 2019 have identified residential solid fuel combustion as a key contributor to emissions of particulate matter (PM) in the United Kingdom.

In fact, owing to flawed assumptions and inaccurate data, DEFRA's interpretation of the National Air Emission Inventory (NAEI) was that domestic burning is the single largest contributor to national PM emissions at 38%, compared with 16% for industrial combustion and 12% for road transport. However, more recently it was demonstrated that the fundamental assumptions and knowledge base underpinning the calculation of domestic solid fuel burning emissions in the NAEI was likely incorrect (Mitchell et al., 2019) with the amount of wood burned being significantly overestimated.

Two years on, our understanding of domestic burning type, frequency and emissions has improved significantly thanks to a number of surveys, reports and policy documents, including:

- A comprehensive review of domestic solid fuel burning emissions and the underpinning science (Mitchell et al., 2019)
- An industry survey of more than 10,000 households that undertake domestic burning (SIA, 2019) which helped to understand burning habits, fuel types and appliance types
- A DEFRA-commissioned survey of more than 46,000 people across the UK spread over 10.5 months (Kantar, 2020) which helped to understand indoor and outdoor burning regularity, fuel types and mass of fuel burned
- The Kantar survey corroborated the results of the industry survey, finding that activity data had been considerably over-estimated in the NAEI, thanks mainly to the flawed *Domestic Wood Use Survey* (BEIS, 2016)
- DEFRA has commissioned a laboratory study to accurately measure emissions from typical UK solid fuel appliances using typical UK fuels (ongoing at the time of writing)
- The Air Quality (Domestic Solid Fuels Standards) (England) Regulations 2020 came into force in May 2021
- The Ecodesign Regulations come into force next year (2022) forcing manufacturers to come up with ever cleaner burning designs. The clearSkies mark ecolabel has also been launched.
- Devolved administrations have published air quality strategies including:
  - Clean Air Plan for Wales: Healthy Air, Healthy Wales (August 2020)
  - Cleaner Air for Scotland 2 - Towards a Better Place for Everyone (July 2021)
  - A Clean Air Strategy for Northern Ireland (awaiting publication)

These documents have led to a much better understanding of a sector where emissions have proven to be very uncertain and difficult to regulate. In addition, user habits have changed during the coronavirus pandemic with more people spending more time at home and becoming more aware of their neighbours' burning practices, as well as their own.

It has also become apparent that recent policy and regulation developments have focussed largely on indoor burning sources only. This is understood to be largely due to historical policy prohibiting dark smoke emissions from chimneys i.e. the Clean Air Act, originally published in 1956. Consequently, many outdoor domestic burning sources have been overlooked and allowed to proliferate without regulation.

Outdoor domestic burning includes bonfires, chimeneas, fire pits, barbecues and braziers which are usually associated with much higher PM emission factors (grams of PM per kilogram of fuel burned) than indoor burning sources. These sources are most often completely unregulated and the only form of control, restriction or enforcement is through statutory nuisance legislation. Outdoor burning is also distinguishable from indoor burning in that very little of the heat generated outdoors is used usefully, whereas wood burned in the home provides low carbon heat and displaces more carbon-intensive heating.

As well as releasing greater concentrations of air pollutants, outdoor burning sources also emit at ground level with no chimney to provide thermal buoyancy and dispersion at height. The distance between source and receptor is therefore minimal. With greater numbers of these appliances being sold, it is crucial to understand their impacts on air quality.

The aim of this work is therefore to better understand the outdoor domestic solid fuel burning emissions sources and quantify their impact on national particulate matter emissions.

All definitions used are derived from the Cambridge English Dictionary (Cambridge University Press, 2021). Prices stated are correct as of May 2021.

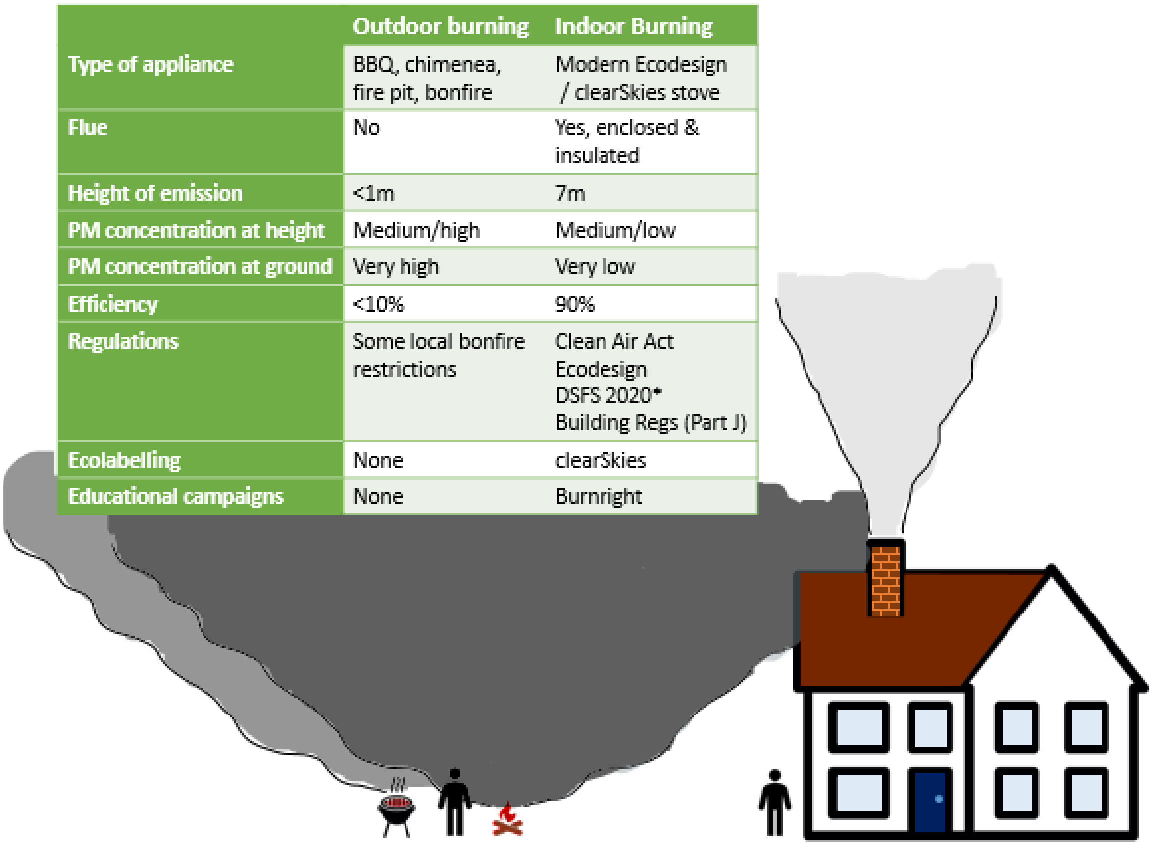
2. Regulation

This section discusses what regulations apply to outdoor burning sources. As shown below and in Figure 1, there is a need for more regulation of this sector.

In general, there are very few regulations prohibiting the use of high-smoke appliances or high-smoke fuels in outdoor domestic settings. The current regulations that do exist are summarised below:

- In general there is no law against having a bonfire, and there are no set times during which bonfires cannot be lit. Some local byelaws exist but vary depending on the local authority.
- The Environmental Protection Act 1990 prohibits a statutory nuisance being caused by smoke, fumes, gases or odour. Whether a statutory nuisance is caused depends on how often the problem occurs, the amount of smoke produced, and how the smoke affects the person complaining. To be a nuisance the smoke or fumes must either be a cause of material harm or must substantially interfere with the enjoyment of land.
- Under the Highways (Amendment) Act 1986 the police can prosecute anyone who allows smoke from a fire which they have lit to drift across a road. The maximum penalty for this is £5,000
- The D7 waste exemption from the Environment Agency allows plant tissue, sawdust, shavings and cuttings from untreated wood to be burnt in the open air by limited non-domestic persons (e.g. landscape gardeners)
- The Clean Air Strategy (2019) and Environment Bill (2020) promised greater powers for local authorities to tackle emissions from domestic combustion and to make smoke control legislation easier to enforce
- Building Regulations (Part J) section 2 specifies draught and flue height requirements for solid fuel combustion appliances up to 50 kW, including minimum distances from windows, roofs and adjacent buildings. However this does not apply to outdoor appliances.
- Under the Clean Air Act (1993), anyone emitting smoke in a Smoke Control Area without due cause is guilt of an offence unless the appliance or fuel is exempt (shown to emit less than 5 grams per hour of smoke under BS PD 6434 1969). However, this only applies to smoke emitted from a chimney in an indoor environment.

Statutory nuisance is arguably a poor method of controlling emissions as the responsibility lies on the receptor to report severe air pollution incidences to their local authority and provide evidence. Often neighbours may not wish to report a nuisance for fear of reprisals and due to the administrative burden. Amendments to clean air policy could instead extend the prohibition of smoky fuels and appliances in outdoor settings. Additional pressure could be placed on manufacturers to meet emissions criteria such as Ecodesign or to ensure outdoor appliances are fitted with flues of at least two metres. This is particularly important due to the potential difference in plume types and ground level concentrations with outdoor burning, as shown in Figure 1. For the basis of this work, only the direct atmospheric emissions are analysed and not the potential exposure from these. However, it is recommended this be a point for further research, as is a thorough review of outdoor burning regulation.



	Outdoor burning	Indoor Burning
Type of appliance	BBQ, chimenea, fire pit, bonfire	Modern Ecodesign / clearSkies stove
Flue	No	Yes, enclosed & insulated
Height of emission	<1m	7m
PM concentration at height	Medium/high	Medium/low
PM concentration at ground	Very high	Very low
Efficiency	<10%	90%
Regulations	Some local bonfire restrictions	Clean Air Act Ecodesign DSFS 2020* Building Regs (Part J)
Ecolabelling	None	clearSkies
Educational campaigns	None	Burnright

Figure 1. Comparison of residential indoor and outdoor burning key characteristics

3. Overview of outdoor burning emissions sources

This section introduces the key outdoor burning appliance types and their basic function.

3.1 Garden bonfires, incinerators and allotments

Bonfires have a very simple definition as "*a large fire that is made outside to burn unwanted things, or for pleasure*". The latter is generally associated with commercial bonfires, discussed in the following section. The focus here is on the former, whereby bonfires are used as a waste disposal method for green and garden waste.

Such material is generally of a very high moisture content, containing freshly cut vegetation. The Environment Agency (2009) suggested that poorly constructed bonfires (with wet wood and poor air flow) are thought to comprise half of all bonfires. The use of contaminated materials (wood coated with pesticides or preservatives), damp wood, wet ground and low supplies of air results in smouldering, which is associated with the highest emissions factors of pollutants.

Apollo galvanised incinerator bin 80LTR, £19.99, Screwfix

Figure 2. Example of garden incinerator

Bonfires were once commonplace on allotments throughout the UK, as a quick and easy method of disposing of green waste. However this led to a number of complaints relating to smoke and odour and consequently many local authorities have now banned or heavily restricted bonfires on allotments. Some are only permitted for the burning of diseased plant material under specified restrictions (LGA, 2009), or during specific autumn/winter periods such as 1 November to 31 March.

Despite this, anecdotal evidence suggests that allotment bonfires remain reasonably common. Depending on the size and population, local authorities may receive 150-300 complaints per year relating to smoke from bonfires, with around 30,000 complaints being received per year (DEFRA, 2006).

In 1996, there were an estimated 296,923 allotment plots in England occupying an area of 25,393 acres (UK Parliament, 1996). A typical plot size is 10 poles, equivalent to 253 m^2 (LGA, 2009). Member surveys by the National Society of Allotment and Leisure Gardeners (NSALG, 2021) suggested that 40% of allotments still have bonfires. Assuming that each plot actively burns three times per year, there are an estimated 357,000 small bonfires per year on allotments in England.

Bonfire advice from the National Society of Allotment and Leisure Gardeners (NSALG, 2012)

Figure 3. Example of an allotment bonfire

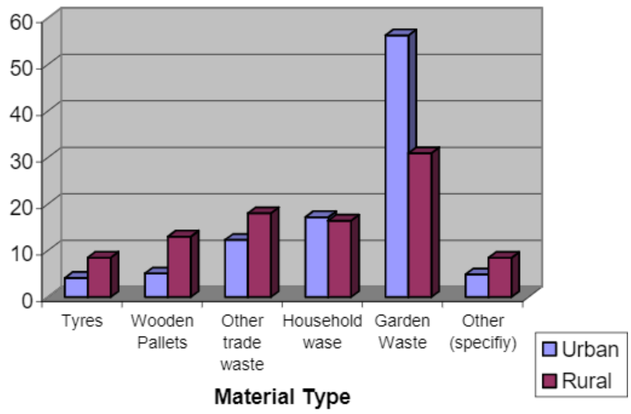
There are an estimated 27.8 million households in the UK, of which around 87% have a private or shared garden. The average garden size in Great Britain is between 188 square metres (ONS, 2020) and 196 square metres (HTA, 2018), meaning that the 24.2 million gardens occupy an area of 454,700 hectares.

In 2008, 49% of UK adults undertook gardening (Haines et al., 2010) although this is believed to have increased significantly during the global coronavirus pandemic (ONS, 2020).

A 2006 report for DEFRA reviewed the emissions and nuisance controls for bonfires within local authority areas. It found that if bonfires are lit on an infrequent basis they are unlikely to influence national air quality objectives but they may, however, cause harm locally due to the relatively high level of particulates released over a short space of time and distance (DEFRA, 2006). The report made the following assumptions about garden bonfires:

- Average bonfire size: 3 x 1 x 1 m
- Bonfire content: Green waste
- Density of green waste: 200 kg/m^3
- Average burning time for bonfire: 2 hours
- Average mass: 600 kg

The report also presented findings on the type and location of bonfires following a local authority survey. As shown in Figure 4, by far the most common type of bonfire is the burning of green garden waste in residential settings.



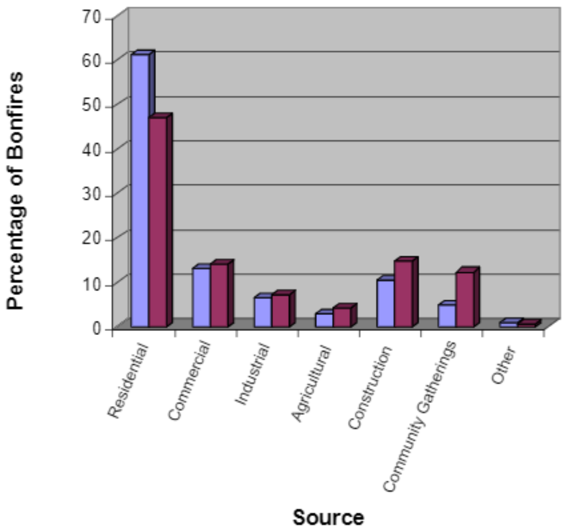


Figure 4. Information on UK bonfires by type of material burned (left) and location of incident (right). Source: DEFRA (2006).

Conservatively assuming that 5% of gardens have bonfires twice per year, there would be 2.42 million small bonfires burning 1.45 million tonnes of green waste per year. Activity data for this category is very uncertain and further discussion of garden waste arisings and burning frequency is given in section 5.

Image source: Getty via The Sun. https://www.thesun.co.uk/news/9440959/garden-bonfire-legal-uk-limit-fined/

Figure 5. Example of a garden bonfire.

3.2 Commercial bonfires

Larger scale bonfires are typically more controlled in that the fuel is whole wood logs or waste wood pallets that is kept reasonably dry, there is a clear source of ignition, and the fire is monitored. Greater quantities of dry fuel allow the fuel bed to reach higher temperatures than garden bonfires, resulting in more complete combustion and lower smoke emissions. Despite this, open burning is uncontrolled and associated with very high emission factors compared with more controlled burning in power stations and heating appliances.

During bonfire night celebrations, Adams et al. (2020) observed more than a factor of 10 increase in aerosol number concentrations, up to a factor of 10 increase in PM_{10} concentration, and more than a factor of 100 increase in black carbon (BC) mass concentration relative to pre-bonfire night levels. Pope et al. (2016) observed a similar spike, as shown in Figure 6.

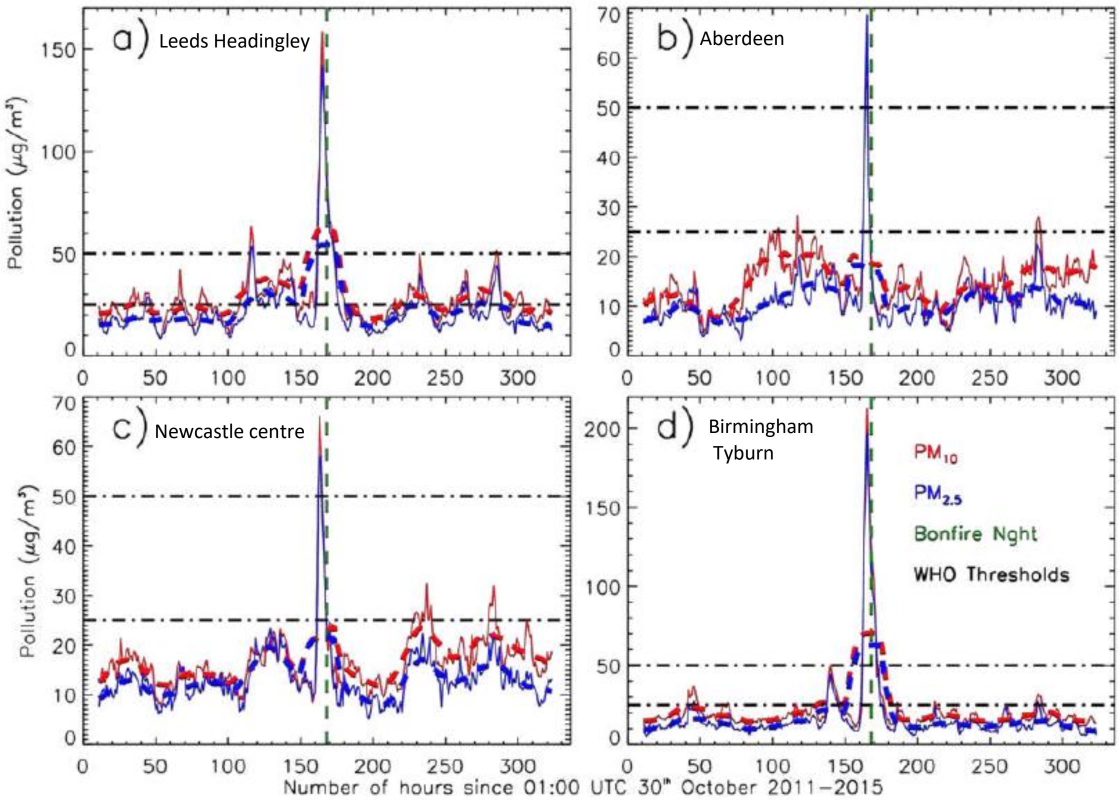


Figure 6. The impact of bonfire night on PM concentrations in four British cities. Source: Pope et al. (2016).

Bonfires generally have a very diverse fuel mix, consisting of a range of combustible materials. These fuels include waste wood, some of which is untreated while some is treated with a range of preservatives and paints; garden waste including branches, leaves, and stems of plants, with varying water content; and household waste from old newspapers and cardboard to plastic and rubber items (Adams et al., 2020).

The NAEI includes pollutants from bonfire night in the UK, with an estimated 1,288 tonnes of $PM_{2.5}$ being emitted each year from such events. However, the NAEI data shows this figure to have been fixed each year going back to 1970. This equates to 115,000 tonnes of wood and waste wood being burned each year.

A freedom of information request submitted by the authors to the Fire Statistics department of the Home Office revealed that there were 712 incidents of bonfires across England in 2019/20, as shown in Figure 7. The number also appears to have been increasing steadily since 2012/13.

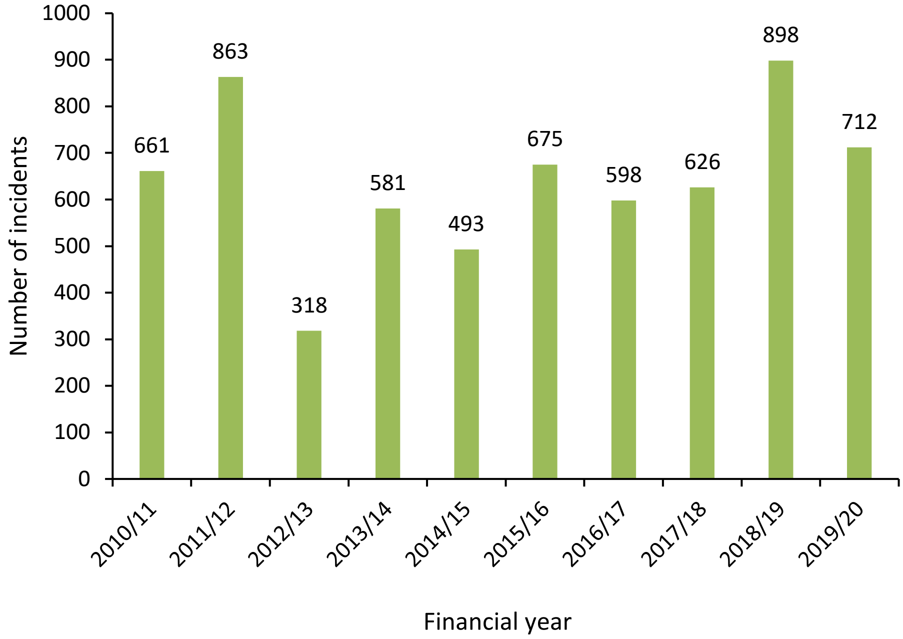


Figure 7. Number of bonfires going out of control attended by fire and rescue services in England. Data source: Home Office FOI request.

It should be noted however that this data only relates to bonfires attended by Fire and Rescue Services and it does not capture information on the type of bonfire (e.g. paid commercial bonfires versus recreational bonfires) or whether bonfires took place on public or private land).

3.3 Chimineas

Chimineas or chimeneas are simple outdoor patio heaters, defined as "*a hollow structure for holding a fire outside, usually made of clay or metal, with a narrow top where smoke can escape*".


Left: Blooma Etinas Steel Chiminea, £79, B&Q. Right: Steel chimenea square, £49.99, Aldi UK.

Figure 8. Example of chimineas.

Chimineas are most often open-fronted with a small flue section. The flue, typically up to one metre, is mostly aesthetic as it is not significant enough to induce a flue draught across the fire bed. Once the bed temperature is established, thermal buoyancy through the flue may allow for a small amount of smoke dispersion to a degree, but as this is below head height the user may still be exposed to the vast majority of emissions.

3.4 Fire pits and braziers

As with chimineas, fire pits and braziers have grown in popularity in recent years as a source of outdoor heating. These appliances are totally open to the air but raised off the ground to prevent heat damage to lawns or patios.

A firepit is defined as "*a dug out area or open container outside in which a fire can be lit*". A brazier is "*a metal container for burning coal, wood, etc., used to give warmth or to cook on*".

Left: La Hacienda Pittsburgh Medium Fire Pit, £129, Dunlem. Right: Brazier Fire Pit, £139, Notonthehighstreet Enterprises Limited

Figure 9. Example of a firepit and brazier

3.5 Wood-fired hot tubs

A relatively new product to the market, wood-fired hot tubs have risen in popularity due to the perceived 'green' benefit of using wood fuel, as well as the desire to reduce running costs. For example, it may cost £8-£10 in electricity to raise the temperature of an electric hot tub from cold to the desired level (circa 38^{o}C), plus £1.50-£2.00 per day in running costs.

Combustion in such devices is generally very inefficient with a large proportion of thermal energy being lost through the flue. The flue section does allow for some dispersal of smoke emissions but total PM emissions are high per unit fuel input. This may change as technology advances. For example, a wood pellet-fired combustor allows for greater automation and much more efficient combustion resulting in lower emissions factors.

Currently, the number of wood-fired hot tubs is thought to be low in comparison to other outdoor burning activities, but they should not be discounted from any planned outdoor burning control strategies.

Regal 190 wood-fired hot tub for 5–7 persons, £3,890, Skargards Hot Tubs.

Figure 10. Example of a wood-fired hot tub

3.6 Barbecues

Barbecues are a very common past time in UK gardens and parks in summertime. A 2010 report by Loughborough University for the then Department of Energy and Climate Change (DECC, now BEIS) used data from The Horticultural Trades Association (HTA) and found that barbecue ownership stands at 50% of household with a garden. Of these, charcoal is the dominant fuel type (62%) followed by gas (37%) (Haines et al., 2010).

Left: Grill type barbecue, Pexels. Right: Bar-Be-Quick instant barbecue, £2, Asda.

Figure 11. Examples of domestic barbecues

Barbeques come in a variety of shapes and sizes and their use is largely restricted to the summer months. On average, Americans and Canadians were found to perform outdoor barbecues approximately 23-38 times per year (Wu et al., 2015). Barbecue frequency is believed to be less in the UK, with a study by Idealo (2016) finding that on average barbecue owners light their appliances 10 times per year and 57% use charcoal. The frequency has increased during the global coronavirus pandemic; the AHDB (2020) found that there were 100 million barbecues lit between April and August 2020. This is far higher than the 60 million estimate made by Johnson (2009) eleven years earlier.

The mass of charcoal burned per barbecue event is uncertain. Vicente et al. (2018) used a mass of approximately 1 kg of charcoal with a burning time of 1.5-2.0 hours. This may be more representative of a disposable barbecue which typically contains 0.5-1.5 kg of charcoal. A larger family barbecue may burn 2-4 kg of charcoal or briquettes in a single session, lasting for 2-3 hours. A USEPA study found that on average 0.588 kilograms of meat cooked per kilogram of charcoal sold (Lee, 1999).

3.7 Smokers

Smokers or barbecue smokers are generally similar in design to garden charcoal barbecues, with the addition of a lid, an additional chamber for smoking wood chips, and a small chimney vent.

Left: American Smoker Charcoal BBQ, £75, Argos. Right: BBQ smoker barrel 3-in-1, £52.99, TecTake.

Figure 12. Example of smokers

Principally used to produce smoked meat and fish, barbecue smokers are designed to burn as inefficiently as possible thereby producing excessive levels of smoke for long periods. By closing the lid and starving the fuel bed of oxygen, the fuel smoulders; very slow combustion producing extremely high levels of pollutants per kilogram of fuel input.

3.8 Pizza ovens

Thanks in part to discount supermarkets, pizza ovens have become a popular addition to many gardens across the UK. They are available in a range of types, sizes and materials and may be fuelled by charcoal, briquettes, wood, gas or even electricity.


Left: Ooni Karu 12 Multi-Fuel Pizza Oven, £299, Lakeland. Right: La Hacienda Milano Charcoal & wood Pizza oven, £407, B&Q.

Figure 13. Examples of pizza ovens

One of the largest suppliers of pizza ovens, Scotland-based Ooni, has seen sales take off since it founded in 2012. The company accounts show shareholders funds to have increased from £2.3m in 2017 to £3.3m in 2018 to £3.8m in 2019, with record sales figures predicted in 2020 and 2021 due to the global coronavirus pandemic. There are also a range of refractory ovens available to buy from several manufacturers or to self-build as a DIY project.

As with chimineas, combustion conditions are generally not well controlled within pizza ovens but the fuel bed is enclosed and generally insulated with a small flue section. This is likely to induce a small draught and allow smoke to escape through thermal buoyancy but, again, the height is not sufficient to allow significant dilution and dispersion.

4. Current estimates of outdoor burning emissions

This section provides an overview of how UK outdoor domestic $PM_{2.5}$ emissions are currently quantified within the National Atmospheric Emissions Inventory (NAEI), including providing a comparison to indoor domestic $PM_{2.5}$ emissions.

Within the UK, the National Atmospheric Emissions Inventory (NAEI) is the body responsible for collating and quantifying emissions of greenhouse gases and air pollutants. Emissions are reported across a range of pollutants and sectors (activities), as set out by the Nomenclature for Reporting (NFR) definitions, with domestic burning emissions falling under the NRF code 1A4bi – residential stationary combustion.

The NAEI inventory is able to estimate the overall emissions of a pollutant for each activity by multiplying activity data by pollutant specific emissions factors. Here the activity rate reflects the amount of each activity, for example, the amount of a certain fuel being burnt, whereas the emissions factors represent the amount of emissions of a certain pollutant from burning a specified mass of fuel. The multiplication of both the activity data with emissions factors gives the overall emissions for that activity, which when summed across all the activities, gives the overall emissions of that pollutant:

$$\boldsymbol{Emission_{pollutant}} = \sum_{\boldsymbol{activities}} \boldsymbol{Activity\ rate_{activity}} \times \boldsymbol{Emission\ factor_{activity,pollutant}}$$

As per the above equation, particulate matter emissions for the 1A4bi – residential stationary combustion activity were calculated by multiplying activity rates (amount of fuel burnt in domestic appliances) with emission factors for each fuel.

4.1 Activity data

4.1.1 Residential stationary combustion

Activity data for the use of different fuels in residential stationary combustion comes mainly from national fuel consumption data as derived from the Digest of UK Energy Statistics, DUKES

(BEIS, 2019). Within this, wood fuel consumption in residential heating was sourced from a 2014-15 survey by the Department of Energy and Climate Change (now BEIS) named the *Domestic Wood Use Survey*. The survey found domestic wood fuel use had increased by a factor of three from 1.8m tonnes in 2005 to 4.8m tonnes in 2017, having large implications on the overall amount of emissions predicted. However, the accuracy of this survey has been called into question with wood fuel use believed to be overstated (Mitchell et al., 2019), particularly when compared to more recent industry surveys with much larger sample sizes (SIA, 2019; Kantar, 2020). Nevertheless, the DUKES estimate on wood fuel use still reports based on the *Domestic Wood Use Survey*, within the added caveat of recognising the large uncertainty in this area:

> "Activity data for this source category remain highly uncertain; the accurate assessment of wood use in the residential sector is extremely difficult due to the lack of comprehensive fuel sales data for a fuel with a substantial component outside conventional fuel markets" (Churchill et al., 2021).

Crucially, the *Domestic Wood Use Survey* primarily focussed on gaining information on wood fuel use for indoor domestic appliances such as stoves and fireplaces. Although a survey question asked about the end use for the wood fuel, the options provided did not differentiate between whether the wood was used in indoor or outdoor appliances, instead letting users give multiple responses around whether it was for heating[1] (91%), hot water (2%), cooking (1.3%) or aesthetic reasons (30%). As such, no distinction was made in the questions to enquire about wood fuel use for outdoor activities such as firepits, chimineas or pizza ovens, therefore it is unclear whether respondents will have included wood used for these purposes in their answers. In the absence of this information on the amount of wood fuel being used in outdoor burning activities, **the *Domestic Wood Use Survey* results and subsequent emission calculations have been attributed solely to burning within indoor appliances**. **In order to more accurately determine how indoor burning within stoves and fireplaces compares with the emissions of unregulated outdoor burning sources, the outdoor burning sources need to be quantified; these are currently excluded in the Domestic Wood Use Survey.**

Some quantification of residential outdoor burning sources has been attempted within the NAEI under different categories than that of the residential stationary combustion (1A4bi).

[1] 11% of respondents stated they used wood fuel to provide all their home heating, whilst 80% stated they used it to provide some home heating.

For example, some residential sources of unregulated outdoor burning included within the NAEI are:

Residential open burning sources quantified in NAEI:

- 5C2 – open burning of waste (small scale waste burning)
- 5E – other waste (bonfire night)

Additionally, a number of non-domestic open burning sources are also quantified, although these are not the focus of this work. These include:

Non-residential open burning sources quantified in NAEI:

- 3F – field burning of agricultural residues
- z 11B – forest fires (accidental and heather burning)
- 5E – Accidental fires (dwellings, other buildings, vehicles)

4.1.2 Open burning of waste

From 1990 to 2009, the activity data for 5C2 - open burning of waste was assumed static at 299 kt a year. This was derived from assuming 75% of households have a fireplace in which they burn some of their household waste (for example, 20% of paper and card, 5% of plastic waste generation etc.), with composite emission factors based on the waste types burnt as discussed in section 4.2.2. This led to an estimation for open burning of waste in fireplaces of 121 kt/year. Similarly, an estimated 37.2 kt/year of waste was assumed openly burnt in commercial, industrial and construction industries, whilst a further 141 kt/year of waste was estimated openly burnt on bonfires. This was based on the assumption that 7% of households have 13 bonfires a year with a quantity of 6.25 kg of material burnt in each and 90% of this being vegetation.

These assumptions were cross-validated in 2011 – 2012 based on a survey on UK burning habits performed by Whiting et al. (2011). It was recommended that the previous assumptions that 75% of households have a fireplace in which they burn household waste was much too high, as was the assumed amount of plastic waste burnt (previously 5% of that generated). For bonfires, Whiting et al. (2011) also recommended that whilst the number of bonfire sites seemed reasonable, the frequency of bonfires was too high and the mass burnt too low. Based on these recommendation, the NAEI activity data estimates were updated in 2011 and 2012. This resulted in a considerable drop of estimated open burning of waste down to 107 kilotonnes in 2018 (Churchill et al., 2021).

Although the exact assumptions and data currently used in this updated activity data are unclear, it is known to now be based on the survey results of Whiting et al. (2011). Some key findings in this study were that 15% of the UK adult population have had a bonfire in the last 12 months. The frequency of having bonfires (including attending community bonfires) was for the majority of people once a year or less, however, this increased for people who have allotments, as seen in Figure 14. The material found to be burnt on peoples' own garden bonfires was predominantly untreated wood (e.g. cut down trees) at 59% and garden waste at 46%, whereas for allotments, garden waste dominated and accounted for 65% [2].

For commercial, industrial and construction open burning of waste, the survey found that 15% of people who had bonfires worked in trades such as building, joinery, carpentry, plumbing, gardening, farming, or general handyman. Of these, 26% said they burnt left over materials at home, with almost half of these (47%) claiming to burn these materials at least once per month.

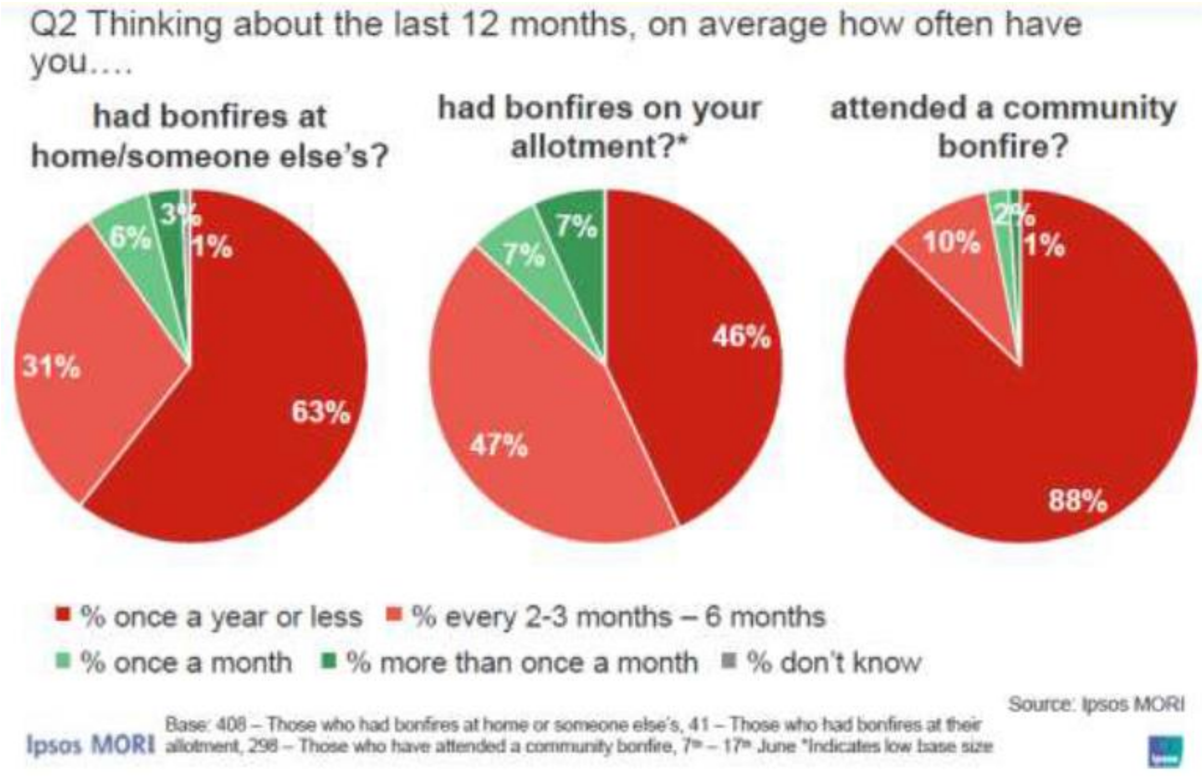


Figure 14: Frequency of bonfires in the UK (Whiting et al., 2011)

4.1.3 Bonfire night

Bonfire night is treated separately from the above due to the large-scale organised nature of events and the potential impact on air quality over a short period of time. The activity data used in the NAEI calculations for this are kept constant and are based on estimates of the

[2] Proportion of respondents saying they burnt this material, not the overall mass burnt.

quantity of material being burning in bonfires and firework displays, as sourced from Passant et al. (2004).

4.2 Emission Factors

4.2.1 Residential stationary combustion

Owing to the wide variety in the types of residential stationary combustion, different approaches were used when estimating emissions for various appliances and fuels. The simplest approach, known as the Tier 1 approach uses default emission factors that aim to reflect global average performances (e.g. of a combustion unit) for each pollutant, as reported by EMEP/EEA. These default emission factors were used throughout the residential stationary combustion calculations, expect for wood use whereby Tier 2 methods were used.

Tier 2 methods refine the emission factors to those that are specific to the UK such as from UK research. As of 2018 the NAEI uses a $PM_{2.5}$ emission factor for domestic wood burning of 480 g/GJ, which when taking the gross calorific value of wood as 16.3 GJ/tonnes (assuming 20% moisture content) equates to approximately 7.8 g/kg. This composite emission factor is calculated by taking the emission factors per appliance from the EMEP/EEA 2019 Guidebook (European Environment Agency, 2019) and weighting this by the relative proportions and assumed efficiencies of appliances as per the results of the *Domestic Wood Use Survey*. Unfortunately, the actual calculations performed in developing this composite emission factor remain confidential, although the UK Informative Inventory Report (1990 to 2019) does state:

> *"In the case of domestic wood and domestic natural gas combustion, the inventory methods aim to reflect the change in emission factors over time as lower-emitting technologies have penetrated the UK stock of combustion units. However, the methods are quite simplistic and suffer both from a lack of data on the market share of different technologies in the UK, and also a limited set of emission factors for different technologies."*

4.2.2 Open burning of waste

The $PM_{2.5}$ emission factor used for open burning of waste within the NAEI is set at 12.94 g/kg. In a similar manner to the above, the emission factor is believed to be a composite factor calculated based on emission factors from the US EPA AP-42 open burning of waste chapter, weighted by assumed open burning compositions as discussed in the activity data section of this report.

4.2.3 Bonfire night

The $PM_{2.5}$ emission factor used for the quantification of bonfire night celebrations is comparable to open burning of waste at 11.2 g/kg. As reported in the UK Informative Inventory Report (1990 to 2018), this emission factor source used for bonfire night celebrations (5th November) is based off a UK study published in 2001 (unreferenced). However, as of the 2019 reporting, the source emission factors used was updated to that of the 2016 EMEP/EEA Guidebook for domestic wood fires (in the case of CO and PM_{10}).

4.3 Uncertainty

As can be expected from the above approach, uncertainties in either the activity data or emissions factors can lead to overall uncertainty in the total amount of emissions being reported. As set out by the EMEP/EEA air pollutant emissions inventory guidebook 2019 (Churchill et al., 2021) uncertainty in activity data is usually derived by statistical methods from the source data. However, if this is not possible, default uncertainty ranges based on the data source can be combined with expert opinions to arrive at an estimate.

Likewise, the uncertainties around emission factors are also often difficult to quantify, therefore qualitative descriptions on the number of measurements, facilities/sources and operating conditions tested are related to typical error ranges (Churchill et al., 2021). For PM emissions from commercial, institutional and residential combustion, a qualitative rating of D is given in the EMEP/EEA guidebook, relating to emission factor uncertainty ranges on the order of 100% to 300%.

Once the uncertainty around the activity data and emission factors is quantified, these uncertainties need to be aggregated. This is achieved in the NAEI by applying so called Tier 1 or Tier 2 uncertainty aggregation methods (not to be confused with the EMEP/EEA Tier system explained previously).

The Tier 1 method for uncertainty aggregation is a simplified approach based on the error propagation equation. This has two rules to aggregate uncertainty based whether the uncertain quantities are combined by addition (e.g. sum of pollutant emissions from different activities) or multiplication (e.g. activity data multiplied by emission factors), details of which can be seen in the General Guidance Volume of the 2006 IPCC Guidelines (Eggleston et al. 2006). In contrast, the Tier 2 uncertainty aggregation method is based on Monte Carlo analysis. This is when uncertainties of each component in a calculation are represented by probability density functions (PDFs). Samples are then randomly drawn from these PDFs and calculated together to derive an output value, which when repeatedly iterated derive their own output PDF. This output PDF can have its properties inferred such as a mean, standard deviation and 95 percent confidence interval to inform overall uncertainty.

Within the NAEI, uncertainty analysis has been applied using both the Tier 1 for the main pollutants and Tier 2 aggregation methods at pollutants not covered by the Tier 1 assessment. PM_{10} and $PM_{2.5}$ emissions were covered by the Tier 1 assessment and were found to have large uncertainties of 46.1% for PM_{10} and 56.3% for $PM_{2.5}$ in 2018. These were slightly below that of the combined uncertainty reported by NFR code 1A (which includes residential combustion) which reported uncertainty values of 59% for PM_{10} and 67% for $PM_{2.5}$. For simplicity, these uncertainties are often reported in the NAEI as qualitative ratings. As seen in Table 1, particulate matter emissions have some of the highest uncertainty in their emissions.

Table 1: Indicative uncertainty rating for each pollutant present in the UK API (Smith et al, 2020)

Pollutant	Indicative Uncertainty Rating
Ammonia (NH_3)	Moderate
Carbon monoxide (CO)	Moderate
Nitrogen oxides (NO_x)	Low
Non-methane volatile organic compounds (NMVOCs)	Moderate
Particulate matter (PM_{10} and $PM_{2.5}$)	High
Sulphur dioxide (SO_2)	Low
Lead (Pb)	High

4.4 Approximation of outdoor vs indoor domestic solid fuel burning emissions based on current NAEI emissions

To understand the approximate current split between outdoor and indoor domestic solid fuel burning $PM_{2.5}$ emissions within the NAEI, the small-scale waste burning and bonfire night emissions can be compared to those from the 1A4bi – residential stationary combustion. For 2019, the indoor domestic burning $PM_{2.5}$ emissions are reported as 47.1 kt, whereas small-scale waste burning and bonfire night are reported as 1.34 kt and 1.3 kt respectively. This results in only 5% of domestic $PM_{2.5}$ emissions reported as originating from outdoor burning. However, this value has some large caveats including:

- Domestic burning wood fuel use is based at present on the *Domestic Wood Use Survey*, which did not distinguish between indoor or outdoor burning. As such, the current reporting seemingly assumes all the emissions originate from indoor burning.

- Outdoor burning in the domestic setting is only incorporated at present within the NAEI small-scale waste burning category (which includes bonfires) and emissions associated with bonfire night, the activity data for which is highly uncertain.

- Outdoor burning activities such as those for cooking (e.g. barbeques and pizza ovens) or outdoor heating and aesthetics (e.g. fire pits and chimineas), are currently not considered in the NAEI despite their increasing popularity and generally lower/lack of emission controls.

Based on the above, it is highly likely that current NAEI quantifications of domestic $PM_{2.5}$ emissions overstate the contribution from indoor domestic burning compared to that of outdoor domestic burning. This is explored in the remainder of this report in order to provide new quantified estimates of the split between indoor and outdoor domestic $PM_{2.5}$ emissions.

5. Review of outdoor burning activity data

This section provides a literature review of outdoor domestic burning activity data as a means to inform the subsequent Monte Carlo modelling in Section 7. Whilst the main source of data referenced is the recently released Kantar study and Defra analysis (Kantar, 2020; Defra, 2020), these values are further compared to other literature and industry estimates so as to assess the certainty of this activity data.

As mentioned in section 4.1, the NAEI is currently using activity data for domestic solid fuel burning that is derived from the 2014-15 *Domestic Wood Use Survey*. This survey increased domestic wood fuel use by a factor of three compared to previous estimations. However, when compared to more recent surveys with increased sample sizes, such as that of the Stove Industry Alliance (SIA, 2019), and Kantar (2020), this value appears to be overestimated, as detailed further by Mitchell et. al (2019). Additionally, the *Domestic Wood Use Survey* did not differentiate as to whether the wood was to be used for indoor or outdoor burning.

The Kantar report *Burning in UK Homes and Gardens* was commissioned by Defra to overcome some of these shortfalls and better quantify the amount of burning that is occurring both indoors and outdoors. The work consisted of a survey of over 46,000 people across the UK spread over 10.5 months. Of these 7,500 said they had burned (indoor or outdoor) within the last year, of which approximately 2,000 of these had burned within the last 7 days. This survey was subsequently followed up by a more in-depth phone-survey of 1,800 respondents who had burned in the last year.

The results from the Kantar survey found that 13.6% ± 0.3% of those surveyed burned outdoors compared to just 8% ± 0.3% indoors. Urban areas were slightly more likely to perform outdoor burning that rural areas, although the difference was minimal (13.9% in urban compared to 12.5% in rural). This outdoor burning was also found to be highly seasonal dependent, peaking in summer for barbeque, chimenea and fire pit use, whilst bonfires peaked in autumn, possibly due to garden clearance after the growing season or events such as bonfire night (see Figure 15).

In regards to the type of outdoor burning occurring, 68% of people who performed outdoor burning reported using non-gas fuelled barbeques. In contrast, one in five outdoor burners (19%) used a fire pit and 12% used bonfires and/or chimineas. Interestingly, the use of these chimineas and fire pits was not solely for warmth, with 38% and 35% respectively using these appliances for waste disposal. This is reflected in the results for fuel types used in outdoor burning, with garden waste, waste wood and household waste all being burnt on fire pits and chimineas by a large number of respondents (see Figure 16). Furthermore, less than half of outdoor burners made sure the wood they burnt was always dry or seasoned. Bonfires (28%) and chimineas (43%) ranked particularly low for this, although no single appliance reported higher than 60%.

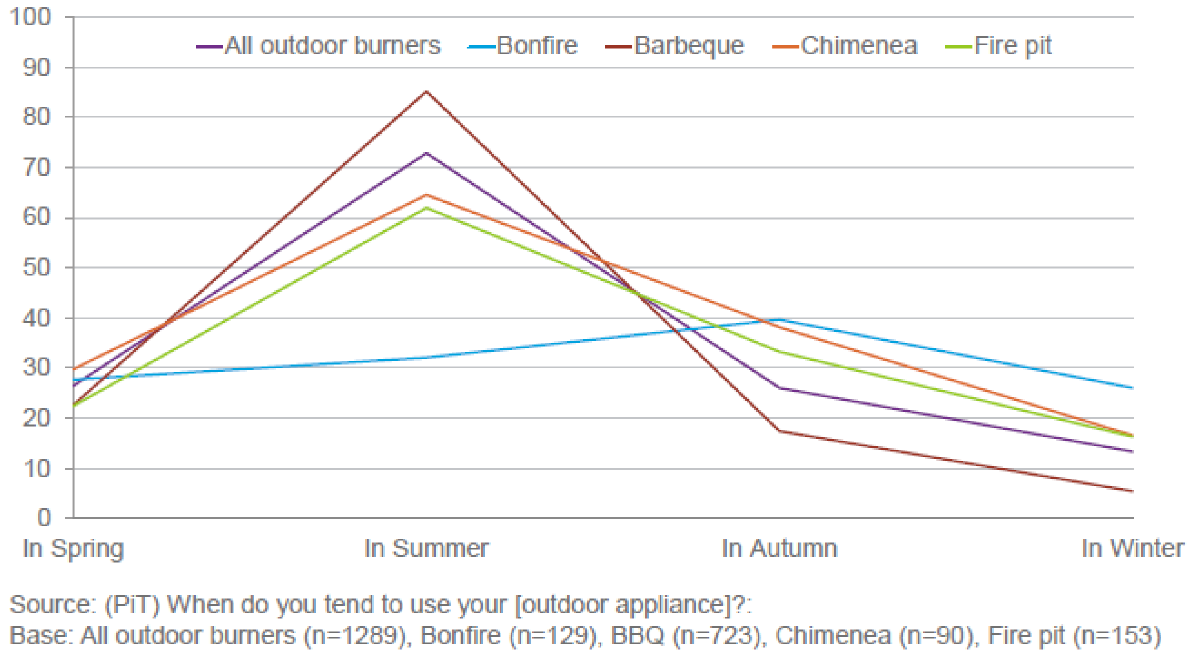
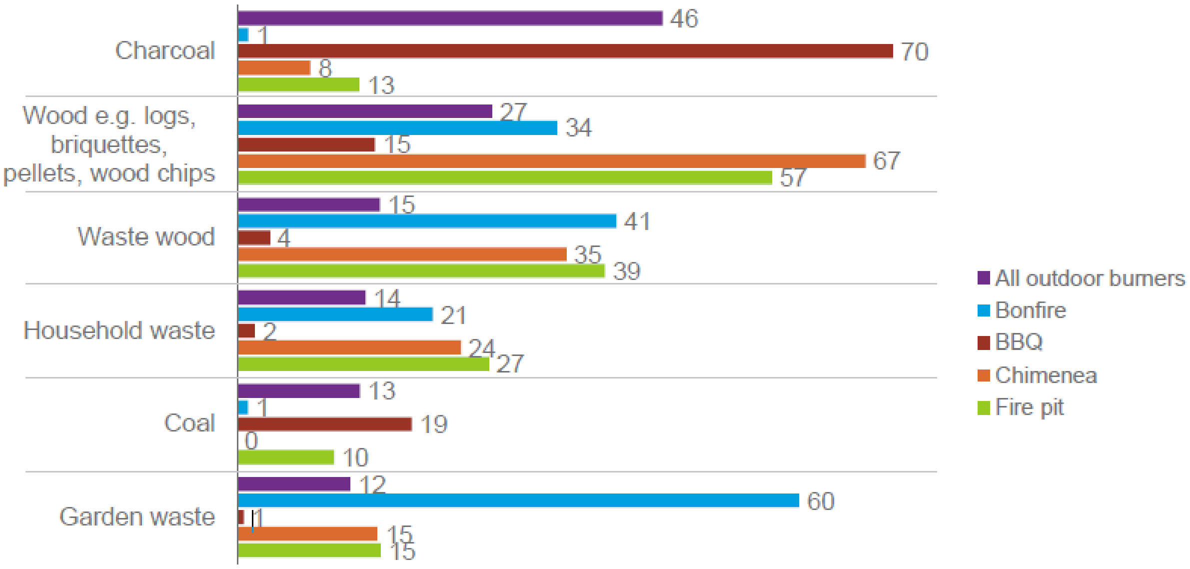


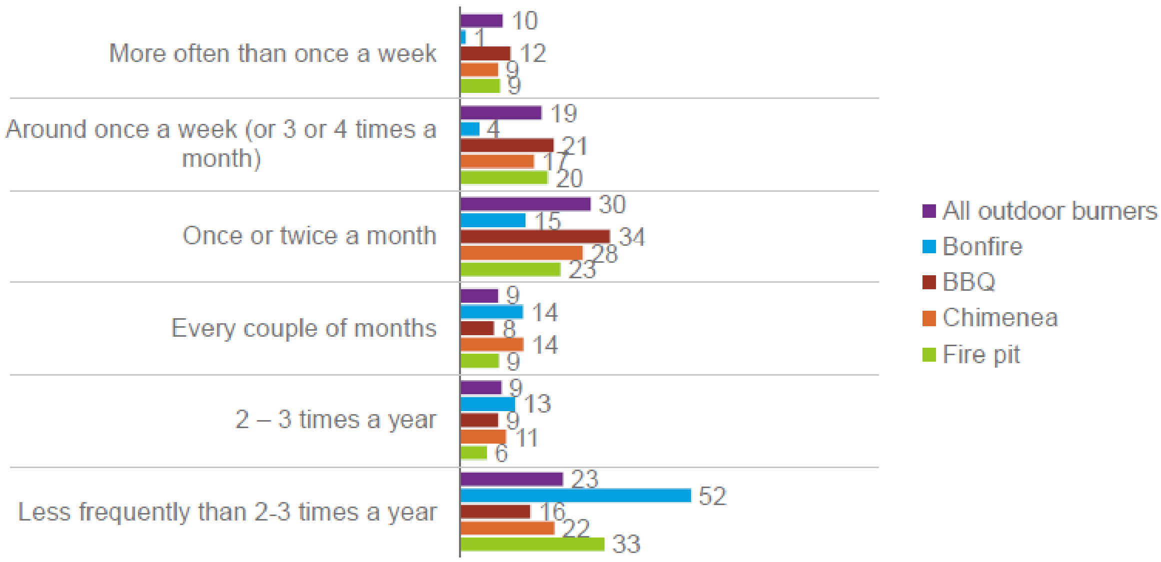
Figure 15: Seasonality of outdoor burning by fire type as a percentage of responses (multi-response allowed). Reproduced from Kantar (2020)

Despite the widespread prevalence of outdoor burning, its frequency was found to be much lower than indoor burning and differed by the type of outdoor burning (Figure 17). For example, during the period at which the respondent performed the outdoor burning the most, 31% of barbeque users burned once a week or more, whilst for chimenea and fire pit burning this was 26% and 29% respectively. Bonfires on the other hand were significantly less frequent, with only 5% burning once a week or more, and 52% burning less than 2-3 times a year.



Source: (PiT) What do you burn in your [outdoor appliance]?
Base: All outdoor burners (n=1289), Bonfire (n=129), BBQ (n=723), Chimenea (n=90), Fire pit (n=153)

Figure 16: Fuels used for outdoor burning by appliance type (% of outdoor burners, multi-response allowed). Reproduced from Kantar (2020)



Source: (PiT) Roughly how frequently do you use your [outdoor appliance], during the time of year you use it most? All outdoor burners (n=1289), Bonfire (n=129), BBQ (n=723), Chimenea (n=90), Fire pit (n=153)

Figure 17: Frequency of outdoor burning by fire type. Reproduced from Kantar (2020)

These Kantar survey results were subsequently taken forward by Defra in 2020 (Defra, 2020) to derive estimates of the domestic solid fuel consumption within the UK and compare these to those reported in DUKES based on the *Domestic Wood Use Survey*. Modifications were included in this more recent quantification to avoid some of the pitfalls of the previous survey. For example, outdoor burning was quantified separately from indoor burning, and respondents were asked about their burning habits within the last week instead of year to allow for improved recollection, in particular concerning quantities burnt.

The method of quantification for indoor burning focussed on apportioning the reported burn time of an appliance by the fuels a user stated they used for indoor burning. This was then converted to energy use per fuel using assumed bulk densities (when bulk estimates were provided) and net calorific values of each fuel. Following extrapolation of the appliance running time across the whole year accounting for seasonal dependencies, the energy use per fuel is once again used to convert this back into the weight of each fuel burnt.

In contrast, the quantification of outdoor burning quantities did not have available information on appliance burning times. As such, the weight and bulk estimates provided by respondents for outdoor burning are used to calculate a ratio between the indoor and outdoor burning for each fuel. This ratio is then applied to the indoor burning estimates to derive the overall amount of fuel used in outdoor burning. Despite the great lengths gone in this analysis to quantify outdoor burning fuel consumption, this figure is reported as highly uncertain by Defra.

The analysis found that approximately 31% of domestic solid fuel use is for outdoor burning activities; totalling 1,176,500 tonnes per year (see Figure 18). In comparison, indoor burning fuel use was found to be 2,632,767 tonnes per year. Direct comparison of this with the value used in DUKES is challenging due to the latter reporting consumption in terms of tonnes of oil equivalent and using an underlying blanket moisture content of wood of 20%. Defra performed additional analysis to compare these on equal terms, reporting that the DUKES estimate for 2018 is 26 GWh (gross) or 5.76 Mt of domestic wood fuel use. This was attributed solely to indoor burning. In contrast, the Defra estimate based on the Kantar data found that domestic wood fuel use (indoor and outdoor) for 2018-19 is 10,886 GWh (gross) which is equivalent to 2.4 Mt on a weight basis. Of this 1.73 Mt was attributed to indoor domestic burning, with the remaining 0.67 Mt from outdoor domestic burning. **As such, the revised estimate of domestic wood fuel use based on the Kantar survey is 58% less than that reported in DUKES, and 70% less for indoor burning. As of March 2021, it was preliminary proposed that this new Kantar-based estimation be incorporated into the DUKES energy balances (and therefore NAEI activity data), with a provisional decrease from 2.2mtoe to 0.7mtoe (BEIS, 2021). This has potentially large implications with regards to the amount of emissions associated with different sources, as examined further in section 7.**

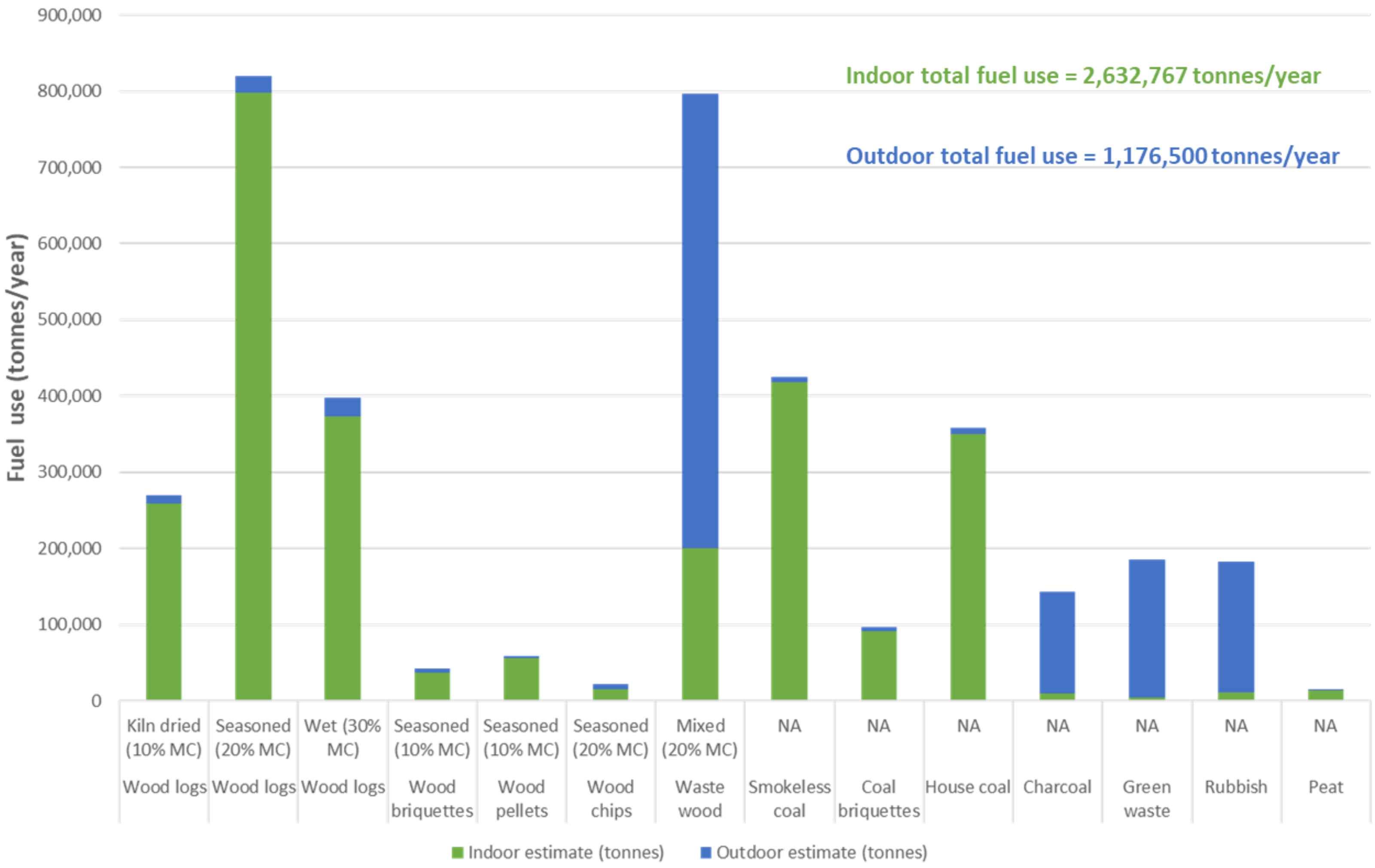


Figure 18: Fuel use in tonnes per year as estimated based on the analysis by Defra on Kantar survey results (Defra, 2020).

Further validation of the Kantar based wood fuel use estimates can be found by comparing the wood use to that of a recent large survey by the Stove Industry Alliance (unpublished). The results of this survey suggested wood fuel use at 1.87 million tonnes per year based on appliance type, run time, months used and fuel use per appliance, or 2.51 Mt based on users own estimates[3]. As such, these values are very similar to the 1.73 Mt indoor wood fuel use estimated from the Kantar data suggesting reliability in the data and methods used.

The relevant proportions of fuel burnt outdoors compared to indoors based on the Defra analysis of the Kantar data can be seen in more detail in Figure 18. Wood logs are by far the most likely to be used indoors as would perhaps be expected due to their prominent use within stoves, as is also coal. Only 3.9% of wood logs were found to be used outside, with unseasoned logs in particular having a higher proportion burnt here (6.2%). However, it should be noted that the Kantar/Defra analysis was conducted before the Covid-19 pandemic, and as such this activity data for outdoor burning of wood logs does not account for the widely reported increase in popularity of outdoor burning appliances such as barbeques, fire pits, chimineas and pizza ovens during this time. For instance, Figure 19 shows data from Google of the interest over time[4] for different outdoor appliance search terms. From this, it is evident that a large increase in interest was found during the spring of 2020 for all outdoor burning appliances compared to previous years. Furthermore, data from the spring of 2021 also suggest that this popularity was maintained despite Covid-19 restrictions easing.

[3] User estimates are deemed more unreliable as users tend to overestimate their wood use.
[4] Numbers represent search interest relative to the highest point on the chart for the given region and time. A value of 100 is the peak popularity for the term. A value of 50 means that the term is half as popular. A score of 0 means that there was not enough data for this term.

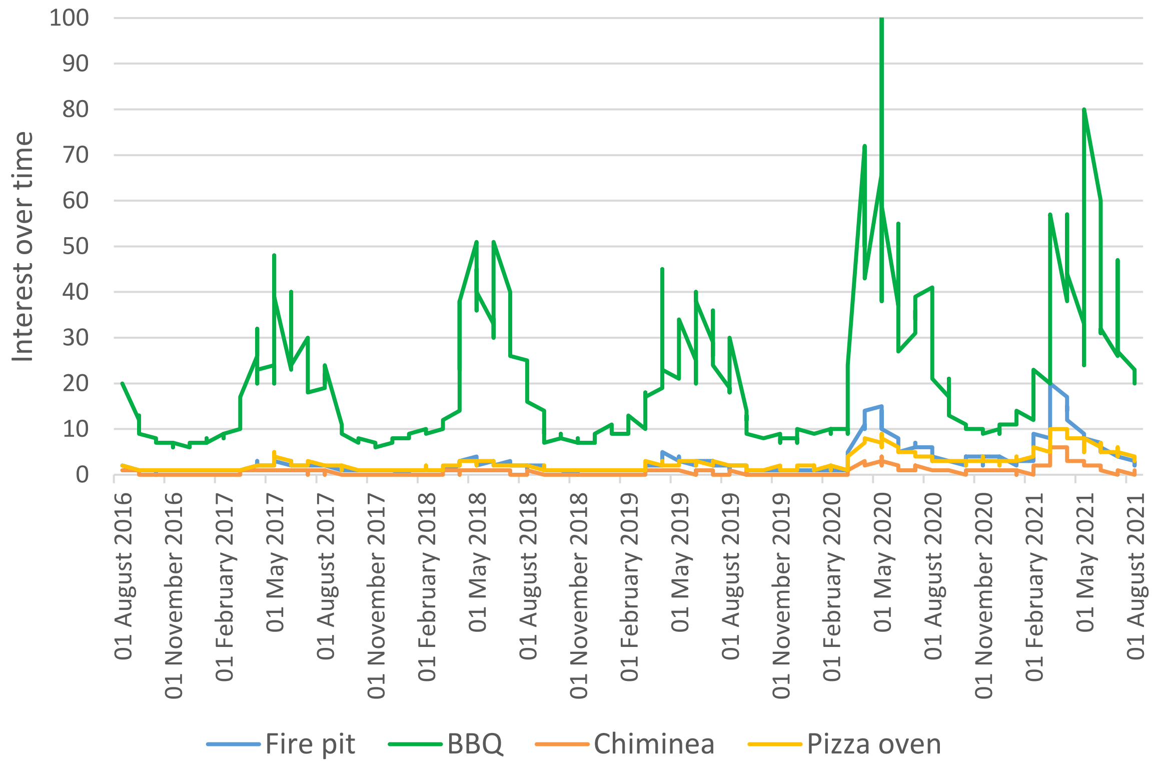


Figure 19: Google search interest over time for outdoor burning appliances.

Charcoal, green waste and rubbish were all fuels that primarily took place only in outdoor settings due to their use in barbeques and bonfires respectively. Interestingly, small amounts (generally less than 7%) were also burnt indoors. Whilst this is perhaps conceivable for rubbish due to people using paper for starting fires, the use of charcoal in indoor settings is confusing and perhaps is due to misreporting from users, for instance, selecting charcoal instead of another form of coal, or using charcoal briquettes incorrectly indoors.

The overall amount of charcoal burnt from the Kantar/Defra analysis is estimated at 143,376 tonnes/year, of which 133,650 tonnes/year is used in outdoor settings. This total amount can be cross checked by comparing it to charcoal import trade figures from UN Comtrade given the majority of charcoal is used domestically. Additionally, as UN Comtrade only reports on imported charcoal, any domestic production also has to be added on this value, estimated in 2016 as 5,000 tonnes per year (Forestry Commissions, 2017). This is charcoal that has been produced and consumed within the UK and therefore not captured in trade data. As UN Comtrade shows no charcoal was exported from the UK, this therefore represents an estimate of the total charcoal availability, as shown in Figure 20.

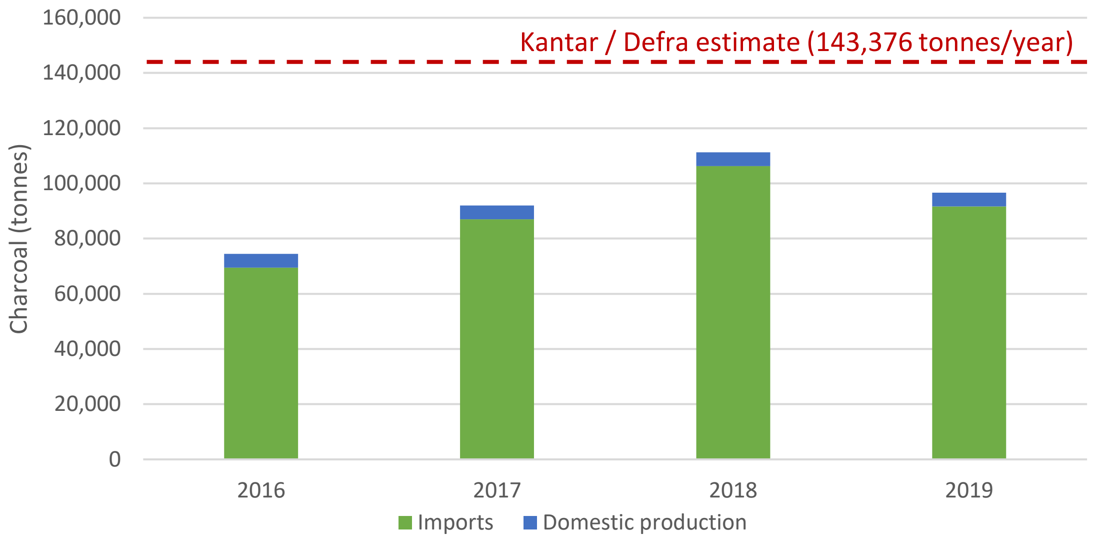


Figure 20: UK Charcoal import (UN Comtrade) and domestic production (Forestry Commissions, 2017) by year

It can be seen in Figure 20 that charcoal availability is smaller than that estimated by the Kantar/Defra analysis, reaching a value of almost 97,000 in 2019, but peaking in 2018 at 111,000 tonnes. On the other hand, data from the UN Food and Agriculture Organisation (FAO, 2020) suggests charcoal imports into the UK peaked in 2018 at 146,278 before decreasing in 2019 and 2020. Adding the 5,000 tonnes on for domestic production would therefore give a total value of charcoal availability for 2018 of 151,000 tonnes. As such the Kantar charcoal consumption is seen to fall within similar bounds.

The second highest fuel use in Figure 18 is that of waste wood with a total of 796,454 tonnes consumed, of which 597,000 tonnes (75%) is burnt in outdoors. Whilst high, the definition used in the Kantar analysis is very broad, defining this as:

> *"Waste wood includes pallets, salvaged wood (meaning wood that has been discarded, for example, from building sites or skips), old furniture, fence posts and other items from the home, all of which are likely to be treated and therefore contain contaminants that may be released on burning.*
>
> *In addition, for this research, the waste wood category also includes fallen wood from trees that is untreated and may be gathered or given for free, distinct from garden waste (a separate response category)."*

This broad definition therefore perhaps explains why the value of waste wood is reported as being so high. Unfortunately, data on wood sourced through these grey markets is not readily available and therefore should be treated with high uncertainty. However, the survey by (Whiting et al. 2011) into UK burning habits does suggest that waste wood is a prominent material burnt both on bonfires and particularly by those burning trade waste at home. Since that study was published in 2011, many local authorities may have improved brown bin garden waste collections, which may affect these results. As no new data on the prevalence of bonfires is available, the Whiting et al. (2011) data is used in this study; however, it is recommended that a new survey on bonfires be carried out.

Similarly, the amount of green waste burnt is also difficult to cross check again due to this not being a traded commodity. The survey results of Whiting et al. (2011) suggest that green waste is the second most prevalent material burnt on bonfires at home with 53% of respondents claiming to do so. For reference, this is only slightly less than the value found for waste wood burnt at home (58%), however when using the definition proposed in the Kantar study, this increases to 94% of respondents[5]. In allotment bonfires, garden waste more closely matches the prevalence of waste wood with 65% of respondents claiming to burn garden waste compared to 66% for waste wood (using Kantar definition).

Unfortunately, as the above figures are as a percentage of respondents who burn the different materials, it is difficult to extrapolate this to overall quantities burnt. The work of Eades et al. (2020) overcame this shortfall by performing a household questionnaire where they asked respondents to estimate the volume of garden waste produced and the disposal methods they used, including backyard burning. They found that open burning of garden waste was significant within their study area of the Test Valley Borough Council (Hampshire, UK), particularly in rural areas as shown in Figure 21.

[5] Combing the categories of 'wood that has not been treated', 'wooden furniture', composite materials e.g. MDF, and 'fencing / wood'.

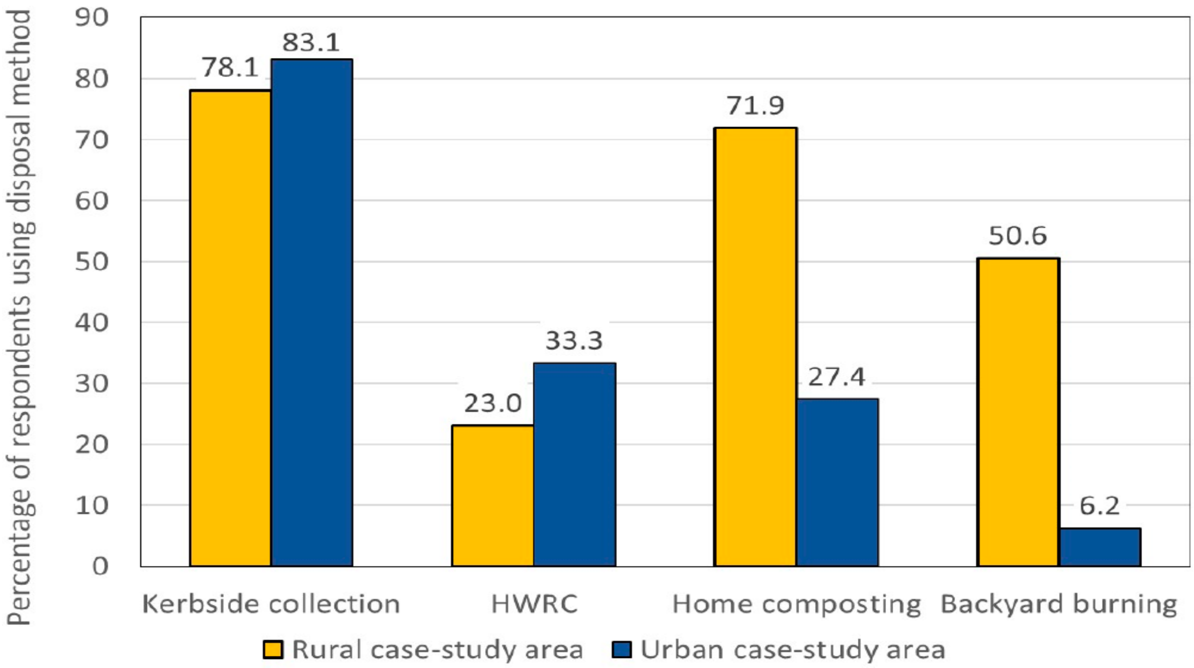


Figure 21: Percentage of survey participants using different disposal option (source: Eades et al., 2020)

Eades et al (2020), further extrapolated their findings across England accounting for the share of rural and urban households, and the proportion of households with no garden. They concluded that garden waste per household in England amounted to approximately 0.79 ± 0.67 kg of garden waste per household per day. This was slightly higher than previous estimates by Parfitt (2002) and Defra (2003) of 0.68 kg household^{-1} day^{-1}, however, this was justified by Eades et al. (2020) because these studies did not account for the backyard burning. This backyard burning was calculated as 0.09 kg household^{-1} day^{-1}, or equivalent to 12% of garden waste arisings.

Whilst Eades et al. (2020) warns against extrapolating the average waste generation values across the UK due to differences in waste policy and regulation, here we extrapolate only the burning amount to give a sense of overall quantities to compare to the Kantar/Defra analysis. As such, taking the total number of UK households as 27.8 million and applying the backyard burning of garden waste rate of 0.09 kg household^{-1} day^{-1}, (note this figure already accounts for households without gardens) this results a total backyard burning amount of 913,230 tonnes/year. This is five times higher than the 184,123 tonnes/year of green waste burnt outdoors as estimated in the Kantar/Defra analysis. A reason for this is possibly due to the backyard burning of garden waste value in Eades et al. (2020) accounting for both waste wood and green waste, however, even making this adjustment the total is still over 4x higher (778,000 tonnes per year).

6. Review of outdoor burning emission factors

This section provides a literature review of outdoor domestic burning emission factor data as a means to inform the subsequent Monte Carlo modelling in Section 7.

There are fundamental differences in the combustion conditions between indoor and outdoor burning, which affect particle formation and emission. The key stages of combustion when solid fuels are heated are drying, pyrolysis (devolatilisation), volatile combustion and char oxidation. The duration of the drying stage is clearly determined by the moisture content of the fuel, but this also affects the proceeding phases. Higher moisture fuels, such as wet garden waste, may have moisture contents in excess of 35% and a significant amount of energy is required to liberate and vaporise the free water and cell water. This energy penalty lowers the temperature of the fuel bed, impeding devolatilisation and leading to more incomplete combustion and higher smoke emissions.

During the char burn-out phase or smouldering, the lack of airflow and high heat losses observed in open burning often lead to extinguishing of the fire. In contrast, the char bed in modern stoves is supplied with directly primary air as well as secondary or tertiary air for more complete combustion of volatiles and smoke. The stove walls also store and re-radiate heat back into the fuel bed, maintaining the higher temperatures which allow for more complete combustion of the char. Outdoor burning is also more variable due to the blowing wind causing rich and lean areas as well as a lack of radiative heating. Consequently, PM emission factors are generally much higher for uncontrolled outdoor burning and the chemical composition of the emitted aerosols show a higher degree of unburned and partially burned wood constituents compared to more confined indoor wood burning (Simoneit et al., 2000).

6.1 Inventory emission factors

6.1.1 The National Atmospheric Emissions Inventory (NAEI)

Emissions factors used in the NAEI are broadly applied to domestic sector combustion, broken down by fuel type. Sector 1A4bi represents fuel combustion in residential or domestic settings but there is no differentiation between indoor burning and outdoor burning.

Table 2 shows the emission factors for fine particulate matter ($PM_{2.5}$) used in the NAEI in 2018. Note that the units have been converted from mass of pollutant per unit fuel energy to mass of pollutant per unit fuel mass, using the net calorific values for fuels.

Table 2. Domestic sector emission factors for $PM_{2.5}$ used in the NAEI in 2018.

Fuel Name	EF	Unit	NCV	Unit
Anthracite	1.84	g/kg	32.6	GJ (net) / tonne
Burning oil	0.0834	g/kg	43.9	GJ (net) / tonne
Charcoal	7.56	g/kg	29.5	GJ (net) / tonne
Coal	9.14	g/kg	28.6	GJ (net) / tonne
Gas oil	0.0809	g/kg	28.3	GJ (net) / tonne
LPG	1.29	g/GJ	40.7	GJ (net) / tonne
Natural gas	1.33	g/GJ	42.6	GJ (net) / tonne
Peat	9.58	g/kg	0.098	PJ (net)/Mtherm
Petroleum coke	1.92	g/kg	0.095	PJ (net)/Mtherm
Solid Smokeless Fuel	1.75	g/kg	12.2	GJ (net) / tonne
Wood (20% MC ar)	7.05	g/kg	14.7	GJ (net) / tonne

For the purposes of this work, the emission factors of wood and charcoal are particularly noteworthy. The $PM_{2.5}$ emission factor stated for domestic wood combustion is 479.5 g/GJ, or 7.05 g/kg assuming wood at 20% moisture with a net calorific value of 14.71 MJ/kg. The value for charcoal is 30.7 g/GJ or 7.56 g/kg assuming a net calorific value of 29.5 MJ/kg.

The figure of 7.05 g/kg is a weighted average of appliance-specific values as detailed in the EMEP/EEA Guidebook. The weightings (for open fires / closed stoves etc.) are based on the 2014 Domestic Wood Use Survey (BEIS, 2016) which found, in table 2.3a, that the average appliance split across the UK is as shown in Figure 22.

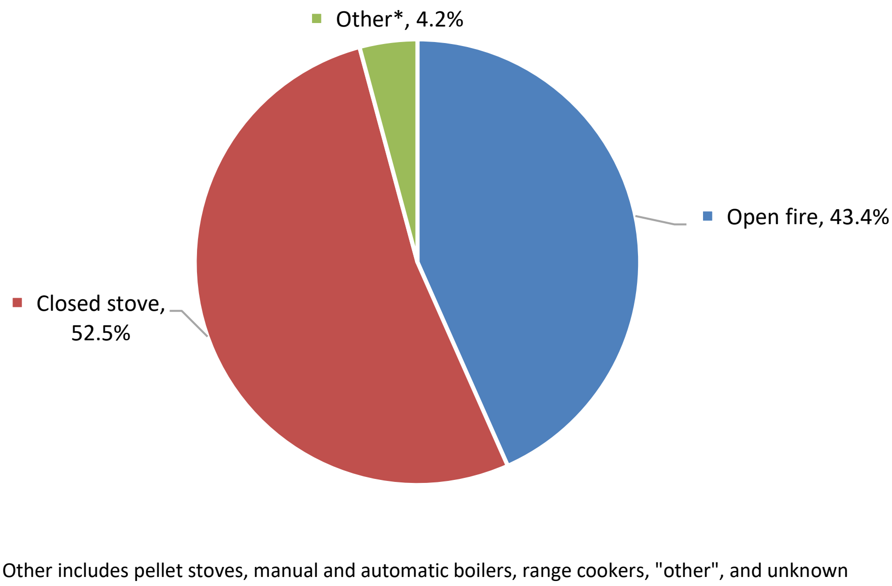


Figure 22. Average appliance type split by number of appliances. Data source: Domestic Wood Use Survey 2014 (BEIS, 2016).

The NAEI also comprises emission factors for other relevant outdoor burning sources such as accidental fires, bonfire night, and charcoal production, which are summarised below.

Table 3. Other relevant sector emission factors for $PM_{2.5}$ used in the NAEI in 2018.

Source	Fuel Name	Emission factor	Unit
Accidental fires - vehicles	Mass burnt	46.45	g/kg
Small-scale waste burning	Waste	12.94	g/kg
Accidental fires - dwellings	Mass burnt	7.43	g/kg
Accidental fires - other buildings	Mass burnt	7.43	g/kg
Accidental fires - vegetation	Area burnt	17.7	kg/Ha
Accidental fires - forests	Area burnt	265.1	kg/Ha
Accidental fires - straw	Mass burnt	9.0	g/kg
Bonfire night	Mass burnt	11.2	g/kg
Charcoal production	Charcoal produced	124.2	g/kg charcoal

6.1.2 The EMEP/EEA Inventory Guidebook

The European Environment Agency (EEA, 2019) contains a large number of emission factors for different pollutants for use in emission inventories. Chapter '1.A.4 Small combustion' details the emission factors for use in domestic combustion applications, reproduced in Table 4. Unlike the NAEI, the EMEP/EEA Inventory Guidebook offers a range of emission factors for the same fuel type, to reflect the importance of appliance type on total emissions. This is particularly true for indoor burning, but the evidence base for outdoor burning is significantly weaker. The emission factors for waste combustion are given in Table 5.

Table 4. EMEP/EEA Inventory Guidebook emission factors for $PM_{2.5}$ from domestic combustion, sector 1A4bi. Source: EEA (2019)

Table	Tier	Technology	Fuel	Value	CI lower	CI upper
3-15	2	Small (single household scale, capacity <=50 kWth) boilers	Solid Fuel (not biomass)	201	100	300
3-14	2	Stoves	Solid Fuel (not biomass)	450	216	480
3-39	2	Open fireplaces	Wood	820	410	1640
3-6	1	NA	Biomass	740	370	1480
3-49_03	2	Residential - High-efficiency stoves	Wood	140		
3-49_04	2	Residential - Advanced/ecolabelled stoves and boilers	Wood	47		
3-49_05	2	Residential - Conventional boilers < 50 kW	Wood	140		
3-49_02	2	Residential - Conventional stoves	Wood	140		
3-12	2	Fireplaces, Saunas and Outdoor Heaters	Solid Fuel (not biomass)	330	198	462
3-19	2	Advanced coal combustion techniques <1MWth - Advanced stove	Coal Fuels	220	72	230
3-40	2	Conventional stoves	Wood and similar wood waste	740	370	1480
3-13	2	Stoves, Fireplaces, Saunas and Outdoor Heaters	Natural Gas	2.2	1.3	3.1
3-49_01	2	Residential - Open fireplace	Wood	240		
3-41	2	Energy efficient stoves	Wood	370	285	740
3-44	2	Pellet stoves and boilers	Wood	60	30	120
3-15	2	Small (single household scale, capacity <=50 kWth) boilers	Solid Fuel (not biomass)	201	100	300
3-49_06	2	Residential - Pellet stoves and boilers (burning pellets)	Wood	30		
3-42	2	Advanced / ecolabelled stoves and boilers	Wood	93	19	233
3-3	1	NA	Hard Coal and Brown Coal	398	72	480

Units are **grams** of $PM_{2.5}$ emitted per **gigajoule** of fuel consumed based of net calorifc value
CI: 95% confidence interval upper and lower bounds

Table 5. EMEP/EEA Inventory Guidebook emission factors for $PM_{2.5}$ from waste combustion. Source: EEA (2019)

Sector	Table	Technology	Value	CI lower	CI upper	Reference
Open burning of waste	3-2	Forest residues	3.76	1.25	11.28	Jenkins et al (1996a)
Open burning of waste	3-3	Orchard crops	4.61	1.54	13.83	Jenkins et al (1996a)
Open burning of waste	3-1	NA	4.19	1.4	12.56	Jenkins et al (1996a)
Forest fires	3-7	Shrubland	9	2	80	averaged from US EPA (1996)
Forest fires	3-4	Boreal forest	9	2	80	averaged from US EPA (1996)
Forest fires	3-6	Mediterranean forest	9	2	80	averaged from US EPA (1996)
Forest fires	3-5	Temperate forest	9	2	80	averaged from US EPA (1996)
Forest fires	3-8	Grassland / Steppe	9	2	80	averaged from US EPA (1996)

Units are **grams** of $PM_{2.5}$ emitted per **kilogram** of fuel consumed
CI: 95% confidence interval upper and lower bounds

As shown, there is little to distinguish between indoor and outdoor burning. However, the accompanying documentation states that "*Chimeneas and barbecues are outdoor appliances which burn wood and charcoal solid fuels. They are little different from an open fire in operation. Other types of outdoor solid fuel appliances include wood-fired pizza and other ovens which also tend to have very limited controls. The emission factors associated with the equipment detailed here can be found in Table 3.12 for solid fuels excluding biomass and in Table 3.39 for wood fuels in open fireplaces*."

From this interpretation, all outdoor combustion appliances using wood would be allocated the emission factors stated in Table 4, which for $PM_{2.5}$ is 820 (410-1640 95%CI) g/GJ (net). Assuming a NCV of 14.71 MJ/kg for wood at 20% moisture content, this equates to 12.1 (6.0 – 21.1 95%CI) g/kg.

For charcoal, the values in table 3-12 of the Guidebook apply which for $PM_{2.5}$ is 330 (198-462 95%CI) g/GJ (net). Assuming a NCV of 29.5 MJ/kg for charcoal, this equates to 9.7 (5.8-13.6 95%CI) g/kg.

6.1.3 US EPA AP-42

In North America, the most commonly used emission factor inventory is the U.S. Environmental Protection Agency's AP-42 Compilation of Air Pollutant Emissions Factors (USEPA, 2020). The current version is the Fifth Edition, first published in 1995, and volume one relates to Stationary Point and Area Sources.

Within volume one, chapter one lists emission factors for external combustion sources broken down by fuel type. Subchapters 1.9 and 1.10 list emission factors for residential wood stoves and fireplaces which are reproduced in Table 6.

Table 6. Emission factors for wood combustion in US residential stoves and fireplaces. Source: USEPA (2020).

Pollutant	Wood in residential fireplaces (Table 1.9-1, Oct 1996)	Wood stoves (table 1.10-1, Oct 1996)		
		Conventional	Noncatalytic	Catalytic
PM_{10}	17.3	15.3	9.8	10.2
CO	126.3	115.4	70.4	52.2
SOx	0.2	0.2	0.2	0.2
NOx (as NO_2)	1.3	1.4	-	1
EF rating	B	B	B	B

Emission factors expressed in grams of pollutant per kilogram of dry wood burned. Source chapters are dated 1996.

It should be noted that the source chapters for this data are dated October 1996 and therefore these emission factors may not be applicable to modern stoves using the latest technology. Catalytic stoves are also more common in the USA than the UK, where honeycomb catalysts are more readily available at retailers.

The USEPA AP-42 data does not provide an uncertainty range or 95% confidence within the individual subchapters, but does provide an emission factor rating. AP-42 EF ratings (A to E) do not imply statistical error bounds or confidence intervals, but instead provide indications of the robustness, or appropriateness, of emission factors for estimating average emissions for a source activity. A description of the bandings is given below:

A *Excellent*. Factor is developed from A- and B-rated source test data taken from many randomly chosen facilities in the industry population. The source category population is sufficiently specific to minimize variability.

B *Above average*. Factor is developed from A- or B-rated test data from a "reasonable number" of facilities. Although no specific bias is evident, it is not clear if the facilities tested represent a random sample of the industry. As with an A rating, the source category population is sufficiently specific to minimize variability.

C *Average*. Factor is developed from A-, B-, and/or C-rated test data from a reasonable number of facilities. Although no specific bias is evident, it is not clear if the facilities tested represent a random sample of the industry. As with the A rating, the source category population is sufficiently specific to minimize variability.

D *Below average*. Factor is developed from A-, B- and/or C-rated test data from a small number of facilities, and there may be reason to suspect that these facilities do not represent a random sample of the industry. There also may be evidence of variability within the source population.

E *Poor*. Factor is developed from C- and D-rated test data, and there may be reason to suspect that the facilities tested do not represent a random sample of the industry. There also may be evidence of variability within the source category population.

Volume I Chapter 2 of AP-42 presents emission factors for solid waste disposal, of which subchapters 2.1 refuse combustion and 2.5 refuse burning are relevant to this work and reproduced in Table 7 and Table 8.

Table 7. Emission factors (g/kg) for uncontrolled refuse burning and open burning. (USEPA, 2020).

Pollutant	Uncontrolled refuse combustion (Table 2.1-12)		Open burning (Table 2.5-1)	
	Wood	Municipal refuse	Municipal refuse	Automobile components
PM_{10}	6.5	18.5	8	50
CO	-	-	42.5	62.5
SOx	0.05	1.25	0.5	-
NOx (as NO_2)	2	-	3	2
EF rating	D	D	D	D

Table 8. Emission factors for open burning of selected fuel types. (USEPA, 2020).

Type of burn	Fuel type	PM (g/kg)	CO (g/kg)	Fuel loading factors (tonne/Ha)	EF Rating
*	Field crops (unspecified)	11	58	4.5	D
*	Barley	11	78	3.8	D
*	Grasses	8	50	-	D
Headfire	Wheat	11	64	4.3	D
Backfire	Oats	11	68	3.6	D
*	Weeds	8	42	7.2	D
*	Orchard crops	3	26	3.6	D
*	Forest residues	8	70	157	D
*	Leaf burning	19	56	-	D

*Burning techniques not significant. Source Table 2.5-5

Particularly noteworthy are the PM emission factors for uncontrolled wood burning and open burning of forest residues which, at 6.5 g/kg and 8 g/kg respectively, are significantly lower than the PM emission factors for wood burning stoves and fireplaces presented in Table 6. This is likely due to Table 7 and Table 8 being expressed in grams of pollutant per unit mass of material burned (i.e. wet basis), whereas the Table 6 is expressed in grams of pollutant per kilogram of dry wood burned (i.e. dry basis). Assuming a wood moisture content of 20%, the PM_{10} emission factors in Table 6 become 13.8 g/kg for fireplaces and 7.8 g/kg noncatalytic stoves, which is more in keeping with the NAEI and EMEP/EEA factors.

Emission factors for barbecue grilling over charcoal were the subject of a later USEPA project (Lee, 1999). Approximately 5-6 kg of meat was cooked atop 6 kg of charcoal over a test period of 3 hours. The emission factors determined for particulate matter, carbon monoxide and nitric oxide for marinated and unmarinated meats are reproduced in Table 9.

Table 9. Emission factors for charcoal grilling in the USA. Source: Lee (1999)

Test condition	Mari-nade	g/kg of meat plus charcoal			g/hour		
		PM	CO	NO	PM	CO	NO
Beef	Y	8.9	188.9	2.6	18.1	385.6	5.2
Chicken	Y	10.8	179.4	7.9	22.7	376.3	16.5
Beef	N	7.7	182.1	3.3	19.5	462.9	8.5
Charcoal only	-	0.6	198.1	6.0	1.3	435.7	13.2
Charcoal only	-	1.8	311.5	10.3	2.8	494.3	16.3
Beef	Y	10.0	148.7	4.4	32.5	484.3	14.2
Chicken	Y	10.1	160.2	1.9	34.9	556.7	6.6
Beef	N	7.9	134.8	1.4	30.4	518.1	5.2
Beef (with control screen)	N	5.7	138.5	1.5	23.7	574.5	6.3

The results show significant variation in PM emission factors between tests, averaging 9.2 g/kg of meat plus charcoal for all cooking tests and 1.2 g/kg for charcoal only. This stark difference demonstrates the impact of meat cooking on the formation of PM during barbecuing.

6.2 Literature emission factors

6.2.1 Open burning of biomass and waste

The burning of rubbish or 'black bag waste' in domestic settings such as gardens is prohibited as it can cause extreme levels of pollution. Not only are there high emissions of $PM_{2.5}$, NOx and SO_2 but burning waste also releases persistent organic pollutants including polychlorinated dibenzodioxins and dibenzofurans (PCDD/F's) and polychlorinated biphenyls (PCB's). These chlorinated organic compounds are extremely toxic and bioaccumulate in the food chain. Consequently, combustion activities that are linked to their formation and emission are tightly regulated, with the only significant sources now being incidental fires or illegal open burning. The latter is of major concern in some developing countries where waste management practices are less advanced (Lundin et al., 2013).

Hedman et al. (2005) carried out emissions testing on a range of wastes including garden waste, paper and plastic packaging, RDF, tyres and electronic waste. The mass of the fuel charge was in the range 4-6 kg. It was found that the highest emissions factors were associated with the fuel charges with the highest level of contamination, reaching up to13,000 ng(WHO-TEQ)/kg. In contract, the emission factors for waste fuels with a low level of contamination such as garden waste were found to be in the range of 4-72 ng (WHO-TEQ)/kg. The authors noted that, for garden waste, factors associated with the combustion efficiency and other uncontrolled factors seemed to have a stronger influence on the emission levels. Similar results were observed by Wevers et al. (2004); recording 4.5 ngTEQ/kg for garden waste and 35 ngTEQ/kg for municipal waste. Heavy metals such as mercury, cadmium or lead may also be emitted depending on the source.

A report by the Environment Agency (2009) reviewed the emissions factors for incident fires and presented a particulate matter emission factor of 8 g/kg for the open burning of municipal solid waste and 4.8 g/kg for a typical 3 m^3 domestic bonfire.

Garden waste

Lemieux et at. (2004) performed a comprehensive review of open burning pollution sources, finding that emission factor for open burning are significantly higher than those from well-controlled combustion sources as shown in Table 10.

Table 10. ***Emission factors for the different types of open burning. Source: Lemieux et al. (2004) and references therein.***

Type of fire	PM emission factor (g/kg)
Prescribed burning	16.6
Agricultural burning	11.0
Land clearing	10.3
Yard waste	19.0
Household waste	8.0
Tyres	119.0
Vehicle fires	50.0

Kannan et al. (2004) burned sample of a variety of garden wastes in a simulation fire test chamber. Although the likely anomalous emission factor for grass was low, the PM emission factor for combined grass, leaves and twigs was 4.4 g/kg and over 32 g/kg for leaves.

The Environment Agency report (2009) also showed that there is significant variation in emissions factors with fuel type, with the burning of leaves emitting the highest levels of particulate matter per unit mass.

Table 11. ***Emission factors for the open burning of agricultural material and leaves. Source: Environment Agency (2009)***

Type of green waste	PM_{10} emission factor (g/kg)
Leaves	19
Unspecified field crops	11
Forest residues	8
Weeds	8
Corn	7

Alves et al. (2019) measured and characterised the particulate emissions from the open burning of garden wastes from tree pruning.

Table 12. Emissions factors (g/kg) for the open burning of garden wastes. Source: Alves et al. (2019).

	PM	OC	EC	CO
Olive branches	16.9 ± 5.1	7.4 ± 0.8	1.2 ± 0.03	87.7 ± 2.6
Vine branches	8.8 ± 5.1	2.7 ± 1.2	0.32 ± 0.17	51.0 ± 7.2
Acacia branches	13.5 ± 2.5	2.7 ± 1.3	0.32 ± 0.07	60.5 ± 19.2
Willow branches	14.4 ± 13.0	4.9 ± 2.7	0.6 ± 0.37	40.6 ± 2.3

Robertson et al. (2014) observed that $PM_{2.5}$ emission factors for open burning of pine-grasslands depended on the amount of fine fuel present in the fuel bed, with higher densities of fine material (grass, leaves, pine needles) leading to higher emissions factors.

Table 13. $PM_{2.5}$ emission factors for open burning of pine-grasslands under different conditions. Source: Roberston et al. (2014).

Season	Years since fire	Flaming $PM_{2.5}$ emissions factor (g/kg)	Smouldering $PM_{2.5}$ emissions factor (g/kg)
Dormant (Jan-Mar)	1-2 (low fines)	14.7 ± 2.5	66.2 ± 52.8
Growing (Apr-Jul)	1-2 (low fines)	22.9 ± 2.6	96.9 ± 89.7
Dormant (Jan-Mar)	3-4 (high fines)	23.6 ± 4.1	85.5 ± 104
Growing (Apr-Jul)	3-4 (high fines)	27.9 ± 3.0	191 ± 108

The results showed that open burning emission factors are extremely high at over 20 g/kg during flaming and over 85 g/kg during smouldering combustion. There is also extremely high variance, particularly in the smouldering emission factors.

The impact of wildfire or prescribed burning emissions on ambient particulate matter concentrations was reviewed by Pearce et al. (2012). It found that variation was significantly influenced by burn duration, burn size, mode of ignition, and regional background $PM_{2.5}$; indicating the importance of these for fire management. The average $PM_{2.5}$ concentration observed during prescribed burning events was 74 ± 166 μg/m^3, three times the WHO Guideline 24-hour mean.

Akagi et al. (2011) conducted a thorough review of emission factors for different types of biomass burning for use in atmospheric models. The compiled emission factors are presented in Table 14 and Table 15.

Table 14. PM emission factors for different types of biomass burning (Akagi et al., 2011)

Type	Emissions factor (g/kg)			
	$PM_{2.5}$	PM_{10}	Black Carbon (BC)	Organic Carbon (OC)
Tropical Forest	9.1 (3.5)	18.5 (4.1)	0.52 (0.28)	4.71 (2.73)
Savanna	7.17 (3.42)	-	0.37 (0.20)	2.62 (1.24)
Crop Residue	6.26 (2.36)	-	0.75	2.3
Pasture Maintenance	14.8 (6.7)	28.9 (13.0)	0.91 (0.41)	9.64 (4.34)
Boreal Forest	15.3 (5.9)	-	-	-
Temperate Forest	12.7 (7.5)	-	-	-
Extratropical Forest	15.0 (7.5)	-	0.56 (0.19)	8.6-9.7
Peatland	-	-	0.20 (0.11)	6.23 (3.60)
Chaparral	11.9 (5.8)	15.4 (7.2)*	1.3	3.7
Open Cooking	6.64 (1.66)	4.55 (1.53)*	0.83 (0.45)	2.89 (1.23)
Patsari Stoves	-	3.34 (1.68)*	0.74 (0.37)	1.92 (0.90)
Charcoal Making^	-	0.7-4.2*	0.02 (0.02)	0.74 (0.72)
Charcoal Burning	-	2.38*	1	1.3
Dung Burning	-	-	0.53	1.8
Garbage Burning	9.8 (5.7)	-	0.65 (0.27)	5.27 (4.89)

Parentheses indicate natural variation.

* Values for total suspended particulate (TSP).

^ EF reported in units of g of compound emitted per kg of charcoal produced

Table 15. Gaseous emission factors for different types of biomass burning (Akagi et al, 2011)

	CO	CH_4	HCHO	NH_3	NOx	NMVOC
Tropical Forest	93 (27)	5.07 (1.98)	1.73 (1.22)	1.33 (1.21)	2.55 (1.40)	51.9
Savanna	63 (17)	1.94 (0.85)	0.73 (0.62)	0.52 (0.35)	3.9 (0.80)	24.7
Crop Residue	102 (33)	5.82 (3.56)	*2.08 (0.84)*	*2.17 (1.27)*	3.11 (1.57)	51.4
Pasture Maintenance	135 (38)	8.71 (4.97)	*1.90 (1.11)*	*1.47 (1.29)*	0.75 (0.59)	89.6
Boreal Forest	127 (45)	5.96 (3.14)	1.86 (1.26)	2.72 (2.32)	0.90 (0.69)	58.7
Temperate Forest	89 (32)	3.92 (2.39)	2.27 (1.13)	0.78 (0.82)	2.51 (1.02)	23.7
Extratropical Forest	122 (44)	5.68 (3.24)	1.92 (1.14)	2.46 (2.35)	1.12 (0.69)	54
Peatland	182 (60)	11.8 (7.8)	1.69 (1.62)	10.8 (12.4)	0.80 (0.57)	97.3
Chaparral	67 (13)	2.51 (0.72)	0.83 (0.25)	1.03 (0.66)	3.26 (0.95)	12.1
Open Cooking	77 (26)	4.86 (2.73)	2.08 (0.86)	0.87 (0.40)	1.42 (0.72)	57.7
Patsari Stoves	42 (19)	2.32 (1.38)	0.37 (0.40)	0.03	-	5.62
Charcoal Making	255 (52)	39.6 (11.4)	3.62 (2.42)	1.24 (1.44)	0.22 (0.22)	321
Charcoal Burning	189 (36)	5.29 (2.42)	0.6	0.79	1.41	11.1
Dung Burning	105 (10)	11.0 (3.3)	-	4.75 (1.00)	0.5	97.7
Garbage Burning	38 (19)	3.66 (4.39)	0.62 (0.13)	0.94 (1.02)	3.74 (1.48)	22.6

CO: Carbon Monoxide, CH4: Methane, HCHO: Formaldehyde, NH3: Ammonia, NOx: Oxides of nitrogen as NO, NMVOC: Non-methane volatile organic compounds.

Noblet et al. (2021) determined emission factors of PM, EC, OC and 88 particle-bound organic species for green waste and residential wood burning tests under simulated real conditions. The study reported PM emission factors of 33.4 ± 5.7 g/kg for hedge trimmings and 34.7 ± 9.2 g/kg for leaves. This is however on a dry basis, where the moisture contents are 60% and 45%. This equates to 13.4 g/kg for hedge trimmings and 19.1 g/kg for leaves on an as received basis,

Interestingly, the PM emission factors for green waste burning were found to be 2 to 30 times higher than those observed for wood log combustion experiments. Noblet et al. (2021) reported PM (dry basis) emission factors of 13.3 ± 2.2 g/kg for outdoor burning of wood logs compared with 7.7 ± 1.3 g/kg for an indoor fireplace and 1.1-5.5 g/kg for a wood stove.

6.2.2 Outdoor cooking and barbecuing

As described in section 3.6, the predominant fuel used for outdoor cooking and barbecuing in the UK is charcoal (Haines et al., 2010).

As a fuel, charcoal is a very low emission option in comparison to other solid fuels. Indeed, fuel switching from wood to charcoal has been actively promoted in developing countries for years where its use can achieve PM emission factors lower than 2 g/kg; less than half that of cooking with wood (Mitchell et al., 2020).

The benefit can also be seen in modern stoves. Sevault et al. (2017) found that PM emissions factors for a typical 4.75 kW indoor wood stove could be reduced from 4.8 g/kg to 3.7 g/kg (-23%) by switching from wood logs to charcoal, although the carbon monoxide emissions factor increased by a factor 3-7 without a catalytic converter. It is understood that fuel switching to charcoal could achieve more substantial reductions in PM emissions but some modifications may be required to the combustion chamber due to the differences in fuel stoichiometry.

The reduction in PM emission for charcoal is due to its low volatile content and high fixed carbon content. Solid fuel volatile content is closely related to particulate matter emissions (Mitchell et al., 2016). This is the reason why smokeless fuel is so-called, as the majority of the volatiles in the raw material have been driven off by thermal treatment.

The quality of barbecue charcoal is therefore also related to its volatile content and fixed carbon content, and this has been standardised by BS EN 1860-2:2005, as shown in Table 16.

Table 16. Requirements for barbeque charcoal and charcoal briquettes in the UK according to BS EN 1860-2:2005

	BBQ Charcoal	BBQ Charcoal Briquettes
Fixed carbon	>75% db	> 60% db
Ash	< 8% db	< 18% db
Moisture	< 8% db	< 8% db
Fuel particle size	0-150 mm (80% > 20mm)	Suitable for BBQ equipment
Volatiles	-	-
Bulk density	130 kg m^{-3}	-
Binder	-	Must be food grade quality, no health hazards

Quality assurance and quality control are therefore critical in the manufacture and supply of barbecue charcoal. Poor quality charcoal that has not been exposed to high enough temperatures for long enough will contain a higher volatile matter content and therefore emit significantly more $PM_{2.5}$ when burned in domestic barbecues (Jelonek et al., 2020).

Significant quantities of $PM_{2.5}$, NMVOCs, PAH and malodorous compounds are also emitted from both the firelighters during ignition and from the cooking of food, particularly fatty meat products. However the pollutant that is arguably of most concern when burning charcoal is carbon monoxide (CO). CO emissions are significantly increased for high fixed carbon fuels and can reach dangerous levels when charcoal is burned in small inefficient appliances, necessitating their use in outdoor environments only.

There is a reasonable body of evidence in the literature for emission factors for outdoor cooking and barbecuing. Vicente et al. (2018) found that average $PM_{2.5}$ emission factors per unit charcoal were 7.38 ± 0.353 g/kg. It also found that anhydrosugars (e.g levoglucosan) constituted the dominant organic class in PM. Levoglucosan is an important tracer used in source apportionment studies and is produced during the partial combustion of a wide range of organic materials.

Table 17. Emission factors for barbecue charcoal. Source: Vicente et al. (2018)

Pollutant	Emission factor (g/kg)
CO	219 ± 44.8
NOx (as NO_2)	3.01 ± 0.698
$PM_{2.5}$	7.38 ± 0.353
Formaldehyde	383 ± 90.3

Huang et al. (2016) recorded much lower $PM_{2.5}$ emission factors for different types of charcoal, generally below 1 g/kg. However, the study used an artificial tube furnace for combustion experiments which may not truly reflect real world emissions. Despite this, similar carbon monoxide emission factors were reported; approximately 150-200 g/kg.

Badyda et al. (2020) suggested that during grilling, barbecues in Poland could contribute 2.3 kilotonnes of $PM_{2.5}$, 92.1 kt of CO, and 4.1 kt of NOx emissions per year.

Aside from the combustion of charcoal, significant PM emissions are associated with the grilling of meat and vegetables during barbecuing. Using a zero-emissions electric grill, Torkmahalleh et al. (2017) observed total particle emission rates of 44.9 mg/min from the grilling of ground beef meat. Given that 250 g of meat was cooked in a sampling time of 20 minutes, the equivalent PM emission factor is 3.6 g/kg of meat. Similarly, Lee et al. (2020) grilled 600 g of pork belly for 10 minutes over a butane barbecue. However, $PM_{2.5}$ emission factors were much lower at 0.75 g per kg of meat. Conversely, Macdonald et al (2003) recorded $PM_{2.5}$ emission factors of 4.4 to 11.6 g/kg of uncooked meat for natural gas fired charbroiling. During the commercial charbroiling of hamburger patties, Gysel et al. (2017) recorded PM emission factors in order of 20 g/kg meat. The study again used a natural gas fired grill and found that the particulate was dominated by organic matter.

Wu et al. (2015) sampled the fumes from a charcoal barbecue at 2 m and 10 m distances away and at a height of 1.2 m. At 2 m from the barcecue, $PM_{2.5}$ concentrations were found to be 2,400 ± 1,700 µg/m^3, reducing to 450 ± 170 µg/m^3 at a distance of 10 m. Significant concentrations of PAH were also detected in both the gaseous and particle phase, as well as benzo(a)pyrene, a common tracer for wood smoke. Badyda et al. (2018) found that exposure to PAHs emitted from barbecue charcoal briquettes was four orders of magnitude greater than that for a propane grill, and also the loading of grills with food generates significantly more PAH. Charcoal barbecues are also an import sources of VOCs, notably benzene and toluene, as well as and carbonyls such as formaldehyde (Kabir et al., 2010). There is also evidence to suggest barbecue charcoal is an important source of trace metal exposure (Susaya et al., 2010).

As well as combustion of the fuel, upstream emissions from charcoal production can be extremely high and in some cases higher than the burning of the fuel itself. As shown in Table 14, Akagi et al (2011) found PM_{10} emission factors to be 0.7 – 4.2 g/kg of charcoal produced. Fugitive emissions from charcoal production are included in the NAEI as a distinct category with an emissions factor of 124 g/kg of charcoal produced. Taking the Kantar data of 143,376 tonnes of charcoal fuel use in domestic settings and applying this NAEI emission factor, this would equate to almost 18 kilotonnes of PM emissions, although the majority of these would occur outside the UK due to the heavy reliance of charcoal imports.

7. Revised quantification of domestic $PM_{2.5}$ emissions from solid fuel combustion

This section discusses the Monte Carlo modelling used to estimate the UK domestic $PM_{2.5}$ emissions in both indoor and outdoor settings. Initially the method and input parameters of the modelling is outlined, before then discussing the results and their significance.

7.1 Method

To determine a more realistic comparison of outdoor versus indoor domestic solid fuel burning PM emissions than that presented in the NAEI, the activity data and emissions factors of previously unreported sources of outdoor burning need to be quantified. To do this, the recent data from the Kantar / Defra analysis as discussed in section 5 is used as a basis for the activity data of both indoor and outdoor burning. It is believed that this data is more reliable than the current NAEI estimates of domestic fuel use both in indoor settings, where wood fuel use is likely overstated; and in outdoor settings, which were previously largely neglected.

Despite more reliable data being used for this quantification, many of the data inputs remain highly uncertain. For example, the outdoor activity data in the Defra analysis of the Kantar data is reported as “highly uncertain”. Similarly, specifying an emission factor based on the fuel type alone is highly problematic as emissions depend not only on what is being burnt, but also the manner and appliance in which it is being burnt. For example, Ecodesign stoves have been found to have 80% lower PM emissions than traditional stoves and 90% lower PM emissions compared to open fires (SIA, undated). To add to this, there is also a great deal of variance in reported emission factors depending on the specifics of the combustion and test methods, as shown in section 6.

To account for the uncertainty in both activity data and emission factors, Monte Carlo analysis was performed. Monte Carlo analysis is a stochastic method used to propagate uncertainties in modelling and is the preferred NAEI Tier 2 approach to do this. The method involves firstly assigning all input values (activity data and emission factors in this case) a probability density function (PDF) that represents the uncertainty of the input value. In contrast to the NAEI Tier 1 uncertainty propagation method, the specified uncertainties in the PDF can be large and non-normally distributed, thereby providing better scope to representing any uncertainty. A value for the activity data and emission factor per category type is then sampled at random from the PDFs to give one estimate of an emission. This sampling is then iteratively repeated many times to give outputs that reflects a range of possible emission values. Performed enough times, these output values start resembling their own probability density function that shows the distribution of possible output values. Statistical analysis can then be performed on these output values (e.g. mean, medium, 95% confidence intervals etc.) to summarise the result and you confidence in it.

Monte Carlo analysis in this work was performed in R using 10,000 iterations and assuming triangular probability density functions for each input variable. This PDF requires three parameters of the minimum, most likely and maximum value. The associated parameter values used for the activity data and emission factors for each modelled category are discussed in the following two sections.

7.1.1 Activity data and uncertainty estimates

The Kantar/Defra data is used to estimate the use of the different solid fuels in both indoor and outdoor domestic settings. However, as mentioned in the previous section, particularly in indoor settings, the appliance type is also as crucial as the fuel used in estimating PM emissions. As such, the wood fuel use is split into three appliance categories of 'open fire', 'traditional stoves', and 'Ecodesign stoves', across which emissions are quantified.

The wood fuel use split by appliance type of open fire and stove is provided in Annexe B of the Defra analysis, however, this only has two categories of 'open fire' and 'stoves and other appliances'. To convert this in to the three appliance categories used within this work it is assumed that 15% of stoves are Eco-design[6]. The amount of wood logs in the 'stoves and other' category is then split between traditional stoves and Ecodesign stoves by scaling these according to the heat output using efficiencies of 65% for traditional stoves and 80% for Ecodesign stoves. This results in Ecodesign stoves being allocated 12.2% of all stoves wood log fuel usage (9% of total wood usage). This same logic is also used for waste wood and wood briquettes. However, whilst wood pellets and wood chips are also split in the Defra analysis between the two appliance categories, here we assume they are only used in suitable pellet/chip fed boilers and therefore these are not split by the three appliance categories. The same approach is also used for coal products due to them also being disaggregated by open fires and stoves in Annex B of the Defra analysis. The remaining solid fuels (e.g. peat, charcoal, rubbish etc.) are also not disaggregated by appliance in this work.

In addition to assigning the most likely estimate for domestic burning solid fuel use, minimum and maximum values are also assigned to each modelled category in order to populate the triangular probability density functions. Due to other reliable estimates of domestic solid fuel usage being sparse, statistical analysis cannot be used to determine uncertainty. Instead the uncertainty estimates are inferred largely by expert opinion supported based on the literature

[6] Based on trade data from the SIA and correcting for non-member sales.

review of alternate activity data in section 5. The activity data and uncertainty estimates used as the input data into the Monte Carlo modelling can be seen in Table 18.

Table 18: Activity data in tonnes/year and uncertainty estimates used in the Monte Carlo modelling.

Fuel	Seasoning	Appliance type	Activity data (tonnes/year)					
			Indoor burning			Outdoor burning		
			C	LB	UB	C	LB	UB
Wood logs	Kiln dried (10% MC)	Open fire	66,050	-5%	+10%	10,944	-20%	+20%
		Traditional stove	168,841	-5%	+10%			
		Ecodesign stove	23,433	-5%	+10%			
	Seasoned (20% MC)	Open fire	204,098	-5%	+10%	22,021	-20%	+20%
		Traditional stove	521,730	-5%	+10%			
		Ecodesign stove	72,411	-5%	+10%			
	Wet (30% MC)	Open fire	95,329	-5%	+10%	24,781	-20%	+20%
		Traditional stove	243,687	-5%	+10%			
		Ecodesign stove	33,821	-5%	+10%			
Wood briquette	(10% MC)	Open fire	9,980	-10%	+10%	5,633	-20%	+20%
		Traditional stove	23,092	-10%	+10%			
		Ecodesign stove	3,205	-10%	+10%			
Wood pellets	(10% MC)	Pellet stove / boiler	56,304	-10%	+10%	2,749	-20%	+20%
Wood chip	(20% MC)	Chip boiler	14,834	-10%	+10%	6,822	-20%	+20%
Waste wood	Mixed (20% MC)	Open fire	54,887	-25%	+25%	596,937	-25%	+25%
		Traditional stove	127,003	-25%	+25%			
		Ecodesign stove	17,627	-25%	+25%			
Smokeless coal	N/A	Open fire	160,229	-10%	+10%	6,608	-20%	+20%
		Traditional stove	226,139	-10%	+10%			
		Ecodesign stove	31,386	-10%	+10%			
Coal briquettes	N/A	Open fire	34,823	-10%	+10%	6,198	-20%	+20%
		Traditional stove	49,147	-10%	+10%			
		Ecodesign stove	6,821	-10%	+10%			
House coal	N/A	Open fire	287,140	-10%	+10%	7,641	-20%	+20%
		Traditional stove	55,437	-10%	+10%			
		Ecodesign stove	7,694	-10%	+10%			
Charcoal	N/A	Open fire/stove	9,726	-33%	+6%	133,650	-33%	+6%
Green waste	N/A	Open fire/stove	3,698	-25%	+147%	181,123	-25%	+147%
Rubbish	N/A	Open fire/stove	11,158	-25%	+25%	171,180	-25%	+25%
Peat	N/A	Open fire/stove	13,038	-10%	+10%	213	-20%	+20%

Central estimate (C), 95% confidence interval lower bound (LB) and upper bound (UB)

The wood fuel use derived in the Kantar / Defra analysis can be compared to the SIA wood use survey with the latter estimated approximately 8% higher. Therefore to encapsulate the SIA value within the maximum possible range, the upper limit for wood fuel usage is assumed at +10% of the Kantar data. Similarly, the minimum amount of wood fuel usage is believed to be more certain based on the Kantar/Defra analysis being the lowest quantified value to date, therefore the minimum is assumed as -5% of the Kantar data. These assumed minimum and maximum values are applied to all wood log usage across all three appliance types.

As also discussed in section 5, comparative values of charcoal usage were also sourced from UN Comtrade data and UN Food and Agriculture Organisation (FAO, 2020) data. As these values bounded the Kantar/Defra data they were used directly as the minimum and maximum values respectively, assuming the same split between indoor and outdoor usage as proposed by the Kantar data.

The data from Eades et al. (2020) as discussed in section 5 was further used to estimate the maximum amount of green waste burnt. However, as Eades et al. quantified garden waste instead of just green waste, it can be assumed the reported figure of 913,230 tonnes/year includes waste wood logs. To overcome this, it was assumed here that green waste makes up approximately 50% of the waste burnt on bonfires, as supported by the findings of Whiting et al. (2011). As such, the upper value for green waste was taken as 456,615 tonnes/year, which was again split between indoor and outdoor combustion based on the same ratio as in the Kantar/Defra data. The lower value for green waste was assumed at -25% of the Kantar estimate.

Lastly, for other fuel types a lack of comparative data made the above kind of uncertainty estimates impossible. For these, the minimum and maximum values were assumed at ± 10% of the central Kantar/Defra data for indoor burning, and ± 20% for outdoor burning. The exception to this is for the categories of rubbish and waste wood which is due to them not being sourced through traditional markets, have increased uncertainty and therefore assigned minimum and maximum values of ± 25%. The increased uncertainty allocated to outdoor burning activity data compared to indoor activity data is due to the Kantar/Defra data having reportedly higher uncertainty for outdoor burning as previously explained.

7.1.2 Emission factors and uncertainty estimates

The emission factors and uncertainty estimates used in emission modelling in this work draw on a number of sources including the NAEI, EMEP/EEA Inventory Guidebook and published peer-reviewed articles.

The most common source used is the EMEP/EEA Inventory Guidebook 2019 which already underpins many of the NAEI emission factors. This is also the primary source of the uncertainty ranges used in the modelling exercise, providing an upper and lower bound within the 95% confidence interval. As shown in Table 19, in many cases the range is very large which reflects both the high levels of uncertainty in emission factors for domestic burning and also the inherent variability due to factors such as fuel charge, air supply, user error, etc.

Table 19. $PM_{2.5}$ emission factors used in the modelling exercise

Fuel	Seasoning	Appliance type	Emission factor (g/kg)							
			Indoor burning				Outdoor burning			
			C	LB	UB	Ref	C	LB	UB	Ref
Wood logs	Kiln dried (10% MC)	Open fire	12.3	6.1	24.5	[A]	12.3	6.1	24.5	[A]
		Traditional stove	6.3	4.9	12.7					
		Ecodesign stove	2.1	0.7	5.2					
	Seasoned (20% MC)	Open fire	12.1	6.0	24.1	[B]	12.1	6.0	24.1	[B]
		Traditional stove	6.2	4.8	12.5					
		Ecodesign stove	2.1	0.7	5.2					
	Wet (30% MC)	Open fire	17.9	8.9	35.7	[C]	17.9	8.9	35.7	[C]
		Traditional stove	9.2	7.1	18.5					
		Ecodesign stove	3.1	1.0	7.6					
Wood briquette	(10% MC)	Open fire	6.2	3.1	12.3	[D]	6.2	3.1	12.3	[D]
		Traditional stove	3.2	2.4	6.4					
		Ecodesign stove	1.1	0.7	2.6					
Wood pellets	(10% MC)	Pellet stove / boiler	1.0	0.5	2.0	[E]	1.0	0.5	2.0	[E]
Wood chip	(20% MC)	Chip boiler	1.3	0.3	3.2	[F]	1.3	0.3	3.2	[F]
Waste wood	Mixed (20% MC)	Open fire	12.1	6.0	24.1	[G]	12.1	6.0	24.1	[G]
		Traditional stove	10.9	5.4	21.8					
		Ecodesign stove	3.6	0.8	9.0					
Smokeless coal	N/A	Open fire	3.4	1.1	3.6	[H]	3.4	1.1	3.6	[H]
		Traditional stove	1.8	0.8	1.9					
		Ecodesign stove	1.1	0.5	1.2					
Coal briquettes	N/A	Open fire	3.7	1.2	4.0	[I]	3.7	1.2	4.0	[I]
		Traditional stove	1.9	0.9	2.0					
		Ecodesign stove	1.2	0.6	1.3					
House coal	N/A	Open fire	18.3	7.1	40.4	[J]	18.3	7.1	40.4	[J]
		Traditional stove	9.4	5.7	13.2					
		Ecodesign stove	5.0	3.0	6.9					
Charcoal	N/A	Open fire/stove	7.4	4.4	10.3	[K]	7.4	4.4	10.3	[K]
Green waste	N/A	Open fire/stove	12.9	4.3	38.8	[L]	30.2	15.1	60.3	[L]
Rubbish	N/A	Open fire/stove	9.8	3.3	29.4	[M]	9.8	3.3	29.4	[M]
Peat	N/A	Open fire/stove	9.6	1.7	11.6	[N]	11.5	2.1	13.9	[N]

Central estimate (C), 95% confidence interval lower bound (LB) and upper bound (UB)

[A] Values have been calculated by scaling the values in [B] based on the relationship between fuel moisture content and emissions observed by Price-Allison et al. (2019). Values for outdoor burning assumed to be the same as for an open fire.
[B] Values for open fire, traditional stove and Ecodesign stove based on EMEP/EEA Inventory Guidebook 2019 Tables 3-39, 3-41 and 3-49_03 respectively. LB based on Table_3-49_04 and UB scaled from Table_3-42. Values for outdoor burning assumed to be the same as for an open fire.
[C] Values have been calculated by scaling the values in [B] based on the relationship between fuel moisture content and emissions observed by Price-Allison et al. (2019). Values for outdoor burning assumed to be the same as for an open fire.
[D] Values have been scaled from [B] based on the findings of Mitchell et al. (2020b) that PM emission factors for wood briquettes are on average 49% lower than for wood logs. Findings corroborated by Roy and Corscadden (2012). Values for outdoor burning assumed to be the same as for an open fire.
[E] Values sourced from EMEP/EEA Inventory Guidebook 2019, Table 3-44. Values for outdoor burning assumed to be the same as indoor as the appliance is the key variable.
[F] Values sourced from EMEP/EEA Inventory Guidebook 2019, Table 3-42. Values for outdoor burning assumed to be the same as indoor as the appliance is the key variable.
[G] Values for open fire and traditional stove sourced from EMEP/EEA Inventory Guidebook 2019, Tables 3-39 and 3-40. Values for Ecodesign stove scaled based on [B]. Values for outdoor burning assumed to be the same as for an open fire.
[H] Values for smokeless fuel burning in a traditional stove taken from NAEI (2021). LB and UB scaled based on EMEP/EEA Inventory Guidebook 2019, table 3-14. Values for open fire have been scaled based on [B]. Values for Ecodesign stove are derived from the findings of Trubetskaya et al. (2021) whereby EF's are 38.5% lower than a traditional stove burning smokeless fuel. Values for outdoor burning assumed to be the same as for an open fire.
[I] Values for smokeless fuel burning in a traditional stove taken from NAEI (2021). LB and UB scaled from [H]. Values for outdoor burning assumed to be the same as for an open fire.
[J] Values for traditional stove derived from EMEP/EEA Inventory Guidebook 2019 Table_3-12. Values for open fire scaled based on [B], but upper bound is taken from Hobson and Thistlethwaite (2003) and Lee et al. (2005). Values for outdoor burning assumed to be the same as for an open fire.
[K] Values from Vicente et al. (2018). LB and UB scaled from EMEP/EEA Inventory Guidebook 2019 table 3.12.
[L] Central value for green waste from NAEI (2021) for indoor small-scale waste burning. LB and UB scaled based on EMEP/EEA Inventory Guidebook 2019 table 3-1. Values for outdoor burning assumed of green waste are a factor of 2.5 times higher than for open burning of wood logs as reported by Noblet et al. (2021).
[M] Values for rubbish derived from Akagi et al. (2011). LB and UB scaled based on EMEP/EEA Inventory Guidebook 2019 table 3-1. Values for outdoor burning assumed to be the same as for indoor burning.
[N] Central value for peat derived from NAEI (2021). LB and UB scaled based on EMEP/EEA Inventory Guidebook 2019 table 3-3. Outdoor burning values assumed to be 20% higher than indoor.

7.2 Quantification results of domestic solid fuel burning $PM_{2.5}$ emissions

The mean $PM_{2.5}$ emissions calculated from the Monte Carlo modelling across both indoor and outdoor domestic burning of solid fuels is 47,643 (+64%, -47%) tonnes/year [7].For comparison, the equivalent NAEI 2019 quantification of domestic $PM_{2.5}$ emissions (including bonfire night and small-scale waste burning) is 49,824 tonnes/year. Although these values are close, critically, the quantification in this analysis includes both the indoor and outdoor domestic burning $PM_{2.5}$ emissions, made possible by using the Kantar/Defra data as a basis. This results in significantly more $PM_{2.5}$ emissions being estimated from outdoor burning than previously thought. For example, as shown in Figure 23 **outdoor emissions contribute 46% all residential solid fuel burning $PM_{2.5}$ emissions. This is as a result of significant quantities of fuel being estimated to be burnt outdoors such as on bonfires, barbeques and in firepits / chimineas, and because the largely uncontrolled nature of these sources results in high emission factors, often multiple times higher than that of modern indoor appliances**.

[7] Uncertainty values represent the 95% confidence intervals as determined through the Monte Carlo analysis

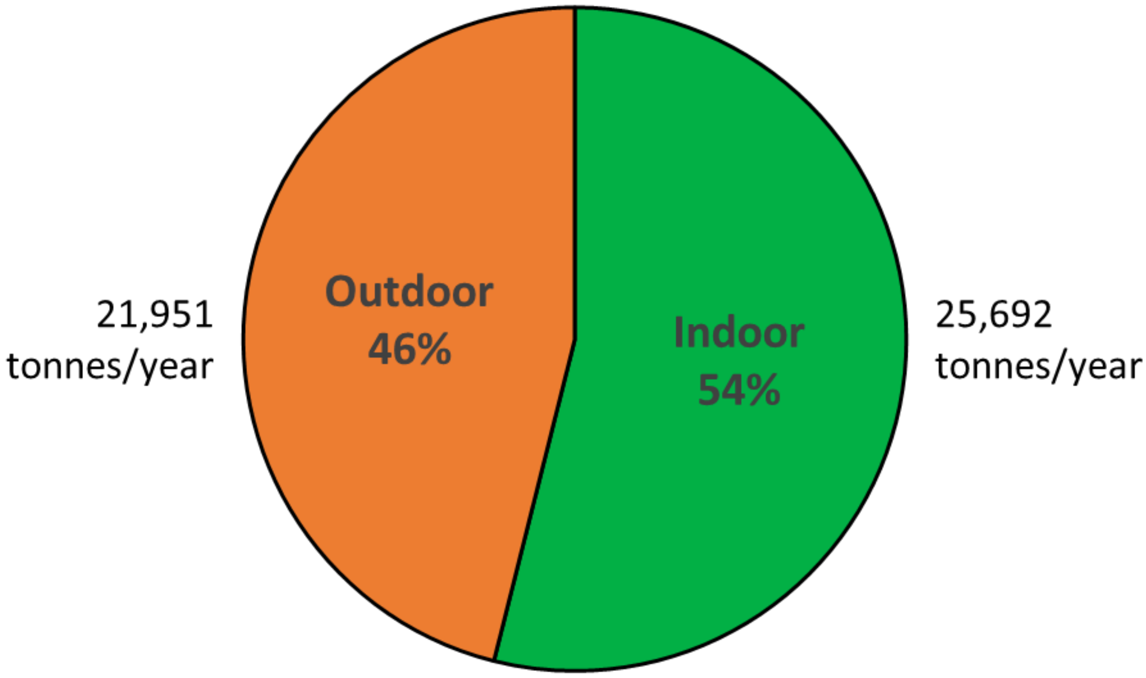


Figure 23: Contribution of indoor and outdoor sources to domestic $PM_{2.5}$ emissions.

Although the average estimate from the Monte Carlo analysis suggest 46% of domestic PM emissions are from outdoor sources, there is significant uncertainty in this estimate due to the inherent underlying uncertainty in both activity data and emission factors. For example, the 95% confidence intervals show outdoor burning could range from 42% up to 49% of domestic $PM_{2.5}$ emissions. Despite this unavoidable uncertainty, it is clear that outdoor domestic burning contributes significantly more than the 5% currently estimated in the NAEI due to both previous overestimation of the indoor domestic solid fuel burning emissions and underestimation of outdoor domestic burning emissions. This uncertainty is visualised in the histogram plots of Appendix 1 for the main indoor and outdoor fuels, with many of the figures being right skewed alluding to the potential for high emissions.

As an alternate analysis, we also consider the split in emissions between indoor and outdoor sources across just wood fuels (wood logs, wood briquettes, wood chips, green waste and waste wood). Doing so, **the emissions from wood fuel results in outdoor burning accounting for over half of all domestic $PM_{2.5}$ emissions**, as seen in Figure 24.

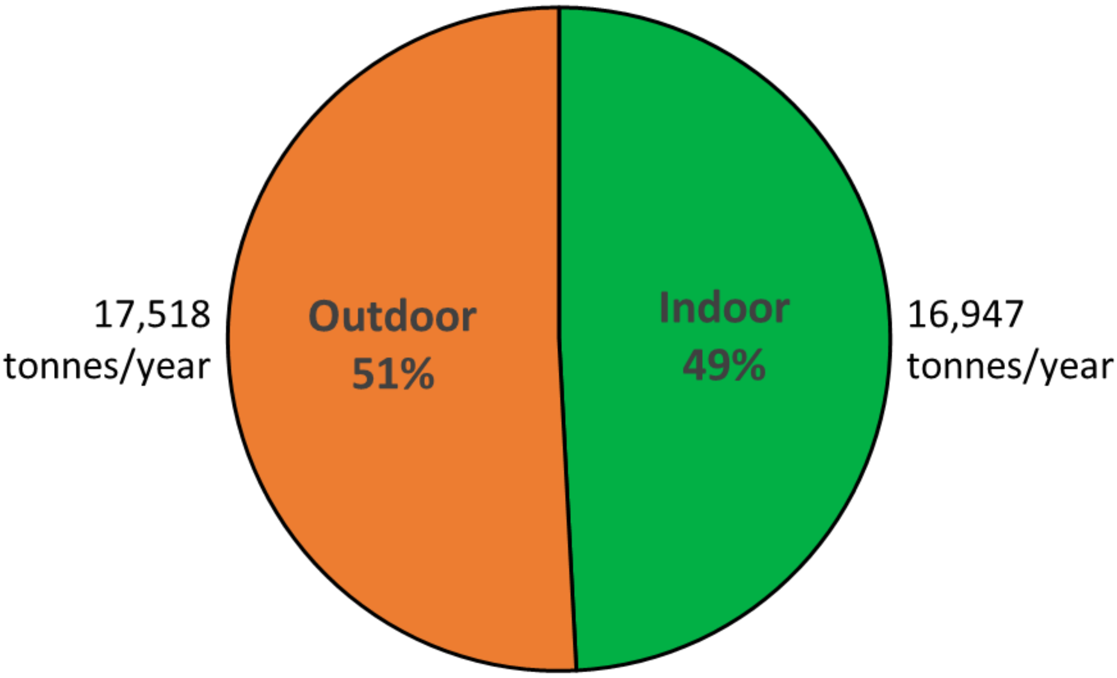


Figure 24: Contribution of indoor and outdoor wood burning sources to domestic $PM_{2.5}$ emissions

A more detailed analysis of the specific fuels and appliances contributing to the domestic $PM_{2.5}$ emissions can be seen for indoor burning in Figure 25 and outdoor burning in Figure 26. Focussing first on the indoor emissions, the majority of $PM_{2.5}$ emissions are seen to originate from the combustion of large amounts of wood logs, and in particular those seasoned to 20% moisture content or burnt largely wet (assumed 30% moisture content). This is as a result of high activity data for these fuels and relatively high emission factors, particularly for wet wood logs. In addition to wood logs, house coal is also a major contributor to $PM_{2.5}$ emissions, with the vast majority of this coming from open fires. Kiln dried wood logs and waste wood contribute the next largest share of indoor $PM_{2.5}$ emissions, whilst the remaining categories contribute just 7% of the total.

In the 2019 version of the NAEI all domestic wood burning sources contributed 38% of total UK $PM_{2.5}$ emissions. In contrast, through the analysis presented in this work, we are able to show for the first time the breakdown by fuel type and appliance type. The results show that wood burning stoves and fireplaces contribute 16.9 kt/year of $PM_{2.5}$ compared to 41 kt/year as reported in the NAEI.

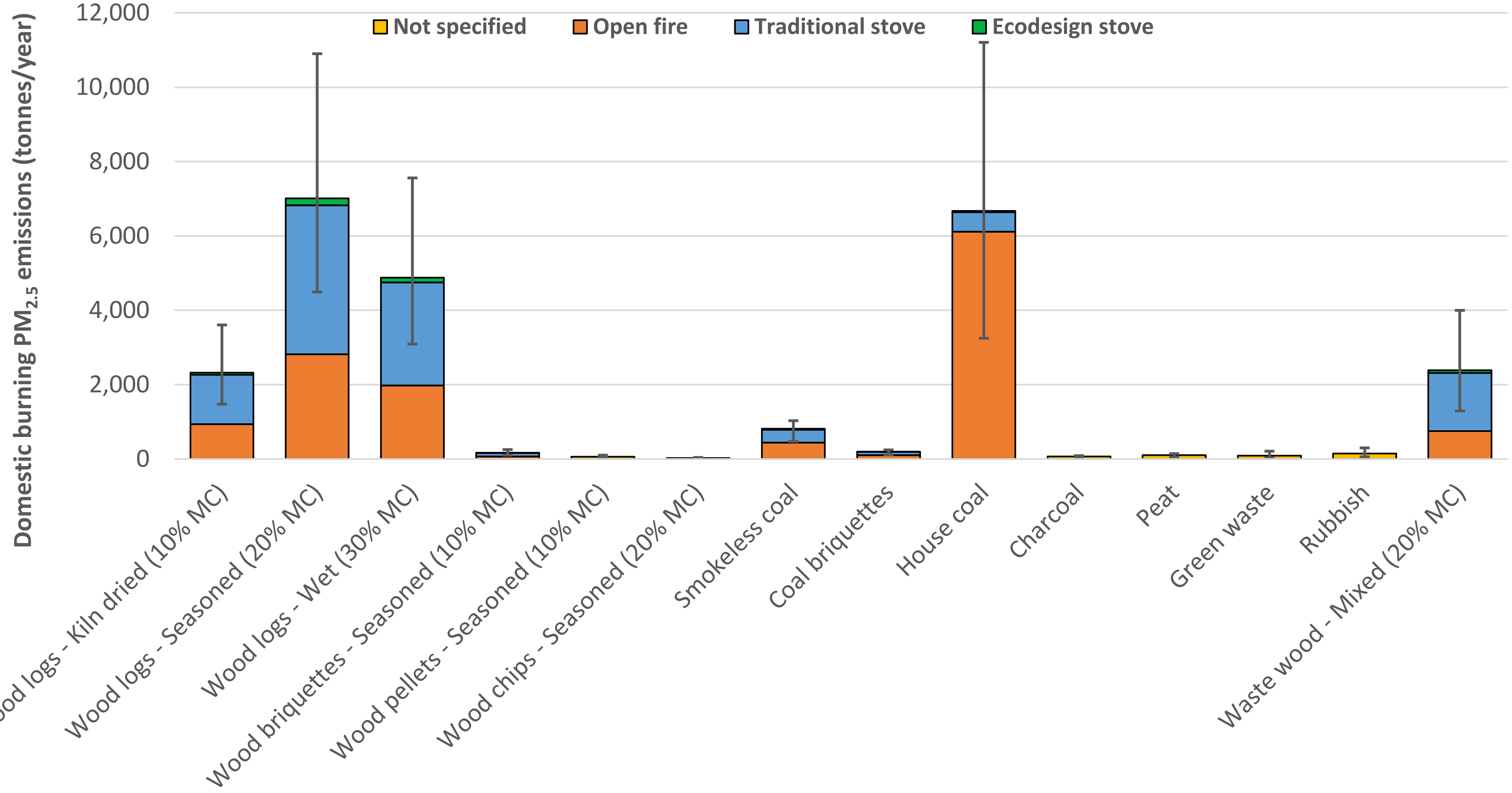


Figure 25: Domestic burning $PM_{2.5}$ emissions in tonnes/year for indoor solid fuel sources and by appliance type. Error bars relate to 95% confidence intervals for each fuel type.

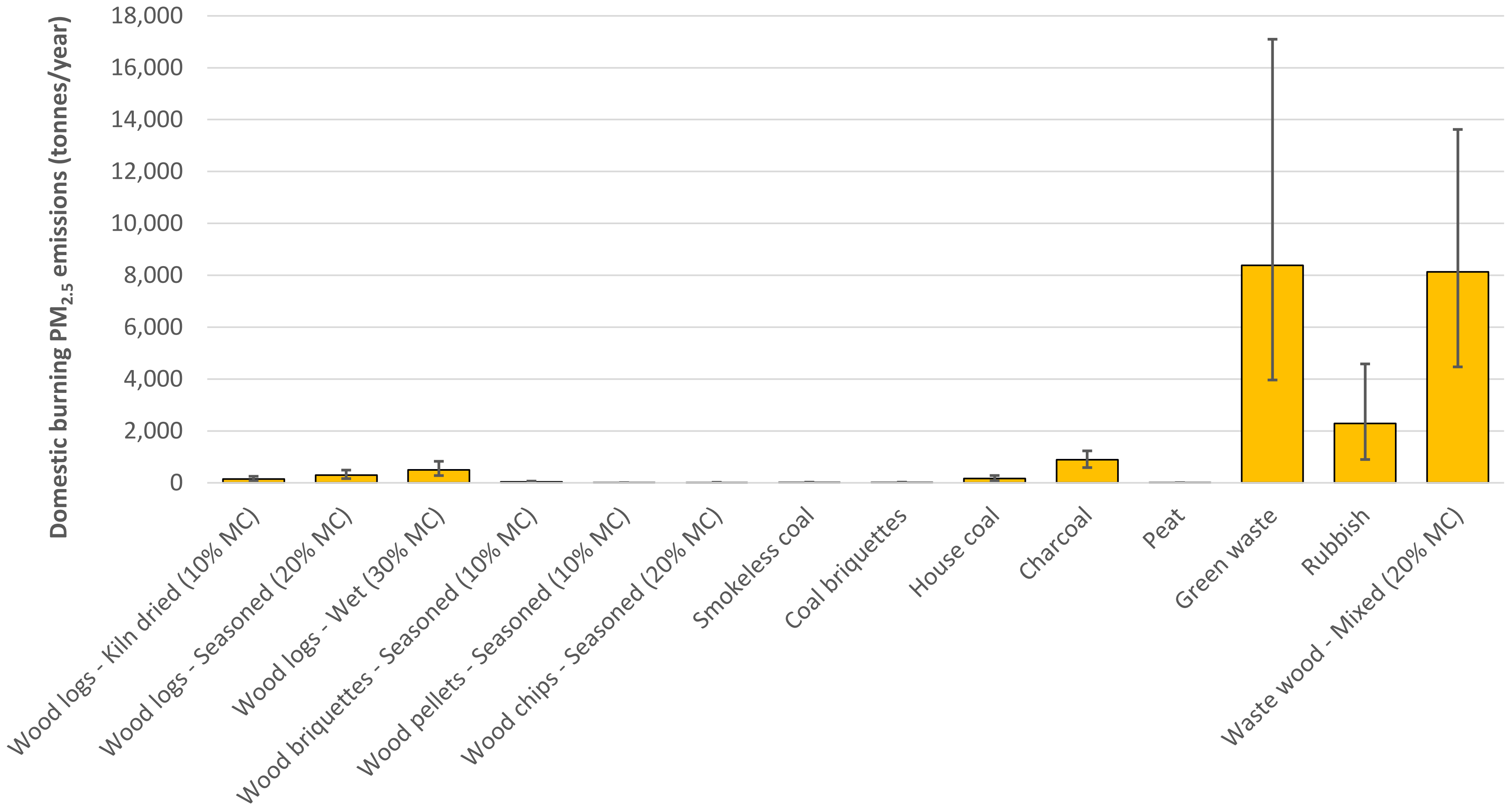


Figure 26: Domestic burning $PM_{2.5}$ emissions in tonnes/year for outdoor solid fuel sources. Error bars relate to 95% confidence intervals for each fuel type.

Although these indoor emissions largely follow the trends seen in activity data for each fuel, there are a number of noticeable exceptions. For instance, smokeless coal, and to a lesser extent kiln dried logs, produce significantly lower emissions compared to their relative use owing to their lower emission factors. On the other hand, house coal produces considerably more emissions than its activity data alone would suggest due to its prevalence for being burnt in open fires and its resulting high emission factor.

When comparing the emissions from wood fuel use across appliance types, deviations away from the trends in activity data are more noticeable. For example, **Ecodesign stoves contribute just 2.7% of $PM_{2.5}$ emissions from the burning of wood logs (including waste wood and briquettes) despite using 9% of the fuel. On the other hand, open fires contribute 39% of the emission from burning of wood logs whilst burning only 26% of the wood. These results therefore highlight the substantial improvements in emissions that can be made by switching to modern Ecodesign stoves.**

The outdoor burning emissions as shown in Figure 26 clearly highlight that **waste wood, green waste and rubbish contribute by far the largest amount of emissions at 90% of all outdoor domestic $PM_{2.5}$ emissions. Crucially, when comparing this to total domestic $PM_{2.5}$ emissions (indoor and outdoor), these fuels contribute to 41% of total $PM_{2.5}$ emissions, or equivalent to all indoor wood burning emissions. Although some of the waste wood is also likely to be burnt in firepits and chimineas, it can be assumed that the majority of this waste wood, green waste and rubbish are burnt on bonfires. This therefore suggests that bonfires are the main source of outdoor $PM_{2.5}$ emissions and deserve significantly more scrutiny to achieve emissions reduction targets**. Despite this striking result, it should be noted that these fuels have some of the highest associated uncertainty, as shown by the 95% confidence intervals error bars in Figure 26. Further research needs to be done to better quantify both the activity data and the associated emission factors. This is particularly the case given that the Kantar/Defra analysis was conducted before the Covid-19 pandemic, and as such it is possible that the activity data for outdoor burning of waste items increased due to temporary closure of household waste recycling centres (i.e. tips). Likewise, as reported in section 5, the popularity of outdoor burning appliances such as barbeques and firepits was also expected to increase during this time meaning outdoor wood log use in the Kantar/Defra analysis may now be underestimate these sources.

Aside from the aforementioned fuels, charcoal constitutes the next highest source of outdoor $PM_{2.5}$ emissions accounting for 894 tonnes/year (4% of outdoor emissions), with this likely burnt on barbeques. Lastly, wood logs and other fuels account for only 6% of outdoor $PM_{2.5}$ emissions despite the increased popularity of outdoor burning appliances such as firepits and chimineas. It is likely however that some of the waste wood category is also burnt in these appliances, however further research is needed to understand the extent of this.

8. Conclusions

NAEI quantification of outdoor domestic $PM_{2.5}$ emissions

- $PM_{2.5}$ emissions in the 1A4bi – residential stationary combustion sector of the NAEI are at present largely attributed to indoor burning due to a lack of distinction between indoor and outdoor burning sources. This has led to regulations focussing purely on indoor burning whilst outdoor burning sources have largely been ignored.
- The NAEI categories of small-scale waste burning and bonfire night represent domestic outdoor burning sources. However, in the 2019 inventory, these contribute only 5% of domestic $PM_{2.5}$ emissions. This work has shown this to be a large underestimate, with outdoor domestic burning instead potentially contributing 46%.

Review of outdoor burning activity data

- The Kantar/Defra analysis of *Burning in UK homes and gardens* gives a more reliable estimation of both indoor and outdoor solid fuel burning compared to current NAEI data, with 31% of fuel found to be burnt in outdoor settings and indoor wood fuel use decreasing considerably to be more in line with other surveys.
- Data on outdoor burning activities may have increased compared to that reported in the Kantar/Defra analysis. This is due to data collection being prior to Covid-19, when the popularity of outdoor burning seemingly grew, and HWRC's were closed.

Revised quantification of domestic PM2.5 emissions from solid fuel combustion

- Monte Carlo analysis was used to simulate domestic $PM_{2.5}$ emissions from both indoor and outdoor burning based on the Kantar/Defra data. This analysed the emissions for both fuel type and appliance type (indoor only).
- Outdoor burning is responsible for 46% of all domestic $PM_{2.5}$ emissions therefore significantly higher than the 5% currently quantified in the NAEI.
- Analysing across just wood fuel, the contribution of outdoor burning increases to over half (51%) of all domestic $PM_{2.5}$ emissions.
- Major sources of PM emissions from indoor burning include wood logs and house coal on both traditional stoves and open fireplaces.
- Ecodesign stoves contribute 2.7% of $PM_{2.5}$ emissions from indoor burning but use 9% of the fuel. Open fires contribute 39% of the emissions but use 26% of the fuel.
- Waste wood, green waste and rubbish account for 90% of outdoor burning $PM_{2.5}$ emissions, or equivalent to 41% of the total (indoor and outdoor) emissions. It is expected that these fuels are largely burnt on bonfires.
- Charcoal burning (primarily used on barbeques) is responsible for 4% of outdoor $PM_{2.5}$ emissions, whereas all wood logs burnt outdoors contribute only 6% of outdoor $PM_{2.5}$ emissions. However, increased popularity of outdoor burning appliances since the Covid-19 pandemic may mean these are underestimated at present.

9. Recommendations

- Action needs to be taken on tackling the outdoor burning sources of $PM_{2.5}$ emissions, and in particular bonfires which are currently largely unregulated and were found to be a major source of $PM_{2.5}$ emissions.
- Existing clean air legislation should be reviewed to consider all domestic outdoor burning sources. In particular, consideration should be given to how best to improve on the statutory nuisance legislation which at present is not fit for purpose as the primary control mechanism. For example, greater restrictions on outdoor burning could be combined with new regulations on smoke control areas as these are amended in the Clean Air Strategy.
- Further research should be carried out to measure emission factors for outdoor burnings sources such as bonfires, chimeneas, fire pits, braziers, pizza ovens and barbecues.
- Consideration needs to be given to the rapid increase in sales of low cost, low quality, highly polluting outdoor heaters so that these do not become a future concern for emissions.
- Educational campaigns should be extended to include outdoor burning.
- Given the high contribution to emissions, alternatives to bonfires such as home composting or garden waste collection schemes need to be promoted and expanded to more households, particularly in rural settings and allotments as these are believed to be major users of bonfires.
- A study should be carried out to determine personal exposure to outdoor burning smoke plumes. Since smoke is emitted at ground level, exposure is likely to be much higher than for indoor burning in closed-fronted appliances with a flue.

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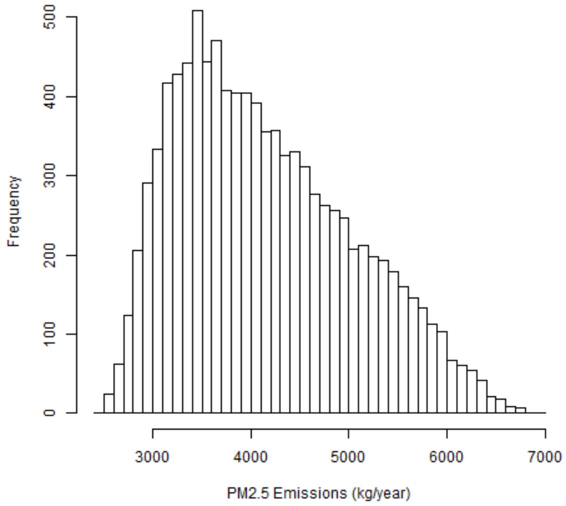
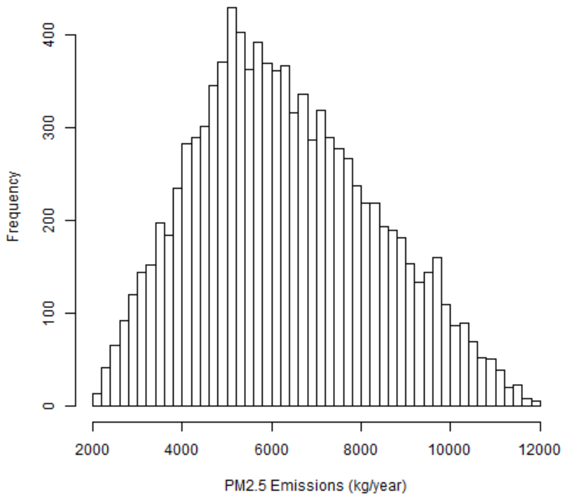
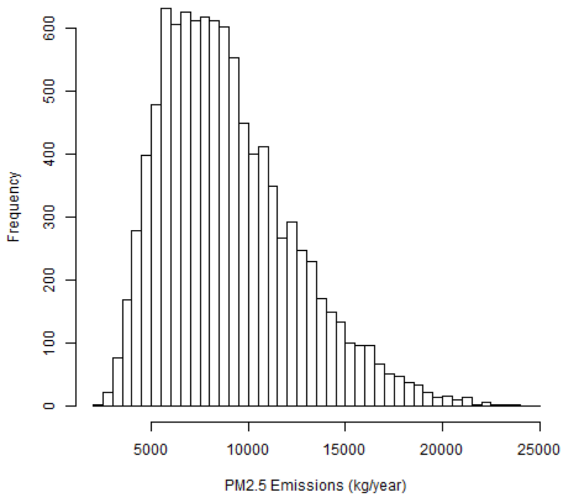
11. Appendix 1



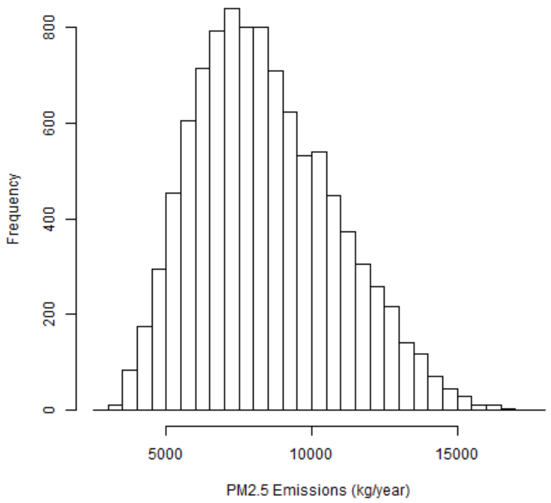
Seasoned wood (20% MC) burnt indoors on traditional stoves



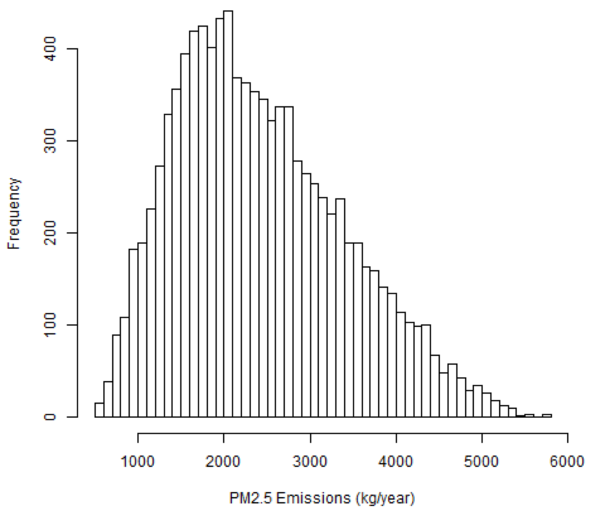
House coal burnt on indoor open fires



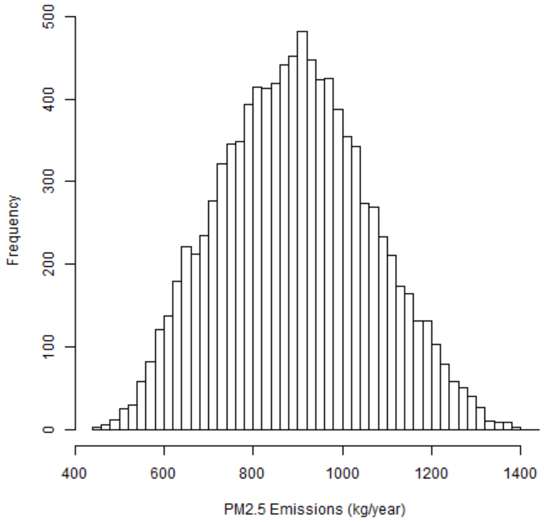
Green waste burnt outdoors



Waste wood burnt outdoors



Rubbish burnt outdoors



Charcoal burnt outdoors

Figure 27: Histograms showing calculated emissions for different fuels and settings from the 10,000 iterations used in the Monte Carlo modelling.