Atmospheric Environment 152 (2017) 431–454

Contents lists available at ScienceDirect

Atmospheric Environment

journal homepage: www.elsevier.com/locate/atmosenv

Heating with Biomass in the United Kingdom: Lessons from New Zealand

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HIGHLIGHTS

- Residential wood combustion (RWC) currently accounts for >10% of renewable energy and >50% of renewable heat generation in the UK.
- Models predict UK RWC to increase by a factor of 14 between 1990 and 2030, with heating stoves and fireplaces dominating.
- Wood consumption per person in New Zealand is twice that of the UK, with significant air quality and climate impacts.
- Black carbon has surpassed carbon dioxide to become the most important component of RSF radiative forcing.

GRAPHICAL ABSTRACT

ARTICLE INFO

Article history:
Received 2 September 2016
Received in revised form
21 December 2016
Accepted 22 December 2016
Available online 26 December 2016

Keywords:
Residential solid fuels
Emissions inventories
Climate

ABSTRACT

In this study we review the current status of residential solid fuel (RSF) use in the UK and compare it with New Zealand, which has had severe wintertime air quality issues for many years that is directly attributable to domestic wood burning in heating stoves. Results showed that RSF contributed to more than 40 μg m^{-3} PM_{10} and 10 μg m^{-3} BC in some suburban locations of New Zealand in 2006, with significant air quality and climate impacts. Models predict RSF consumption in New Zealand to decrease slightly from 7 PJ to 6 PJ between 1990 and 2030, whereas consumption in the UK increases by a factor of 14. Emissions are highest from heating stoves and fireplaces, and their calculated contribution to radiative forcing in the UK increases by 23% between 2010 and 2030, with black carbon accounting for more than three quarters of the total warming effect. By 2030, the residential sector accounts for 44% of total BC emissions in the UK and far exceeds emissions from the traffic sector. Finally, a unique bottom-up emissions inventory was produced for both countries using the latest national survey and census data for the year 2013/14. Fuel- and technology-specific emissions factors were compared between multiple inventories including GAINS, the IPCC, the EMEP/EEA and the NAEI. In the UK, it was found that wood consumption in stoves was within 30% of the GAINS inventory, but consumption in fireplaces was substantially higher and fossil fuel consumption is more than twice the GAINS estimate. As a result, emissions were generally a factor of 2–3 higher for biomass and 2–6 higher for coal. In New Zealand, coal and lignite consumption in stoves is within 24% of the GAINS inventory estimate, but wood consumption is more than 7 times the GAINS estimate. As a result, emissions were generally a factor of 1–2

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http://dx.doi.org/10.1016/j.atmosenv.2016.12.042

higher for coal and several times higher for wood. The results of this study indicate that emissions from residential heating stoves and fireplaces may be underestimated in climate models. Emissions are increasing rapidly in the UK which may result in severe wintertime air quality reductions, as seen in New Zealand, and contribute to climate warming unless controls are implemented such as the Ecodesign emissions limits.

Abbreviations

AGWP	Absolute Global Warming Potential
BC	Black Carbon
CAU	Census Area Unit
DECC	Department for Energy and Climate Change (UK). *Note this department has recently become the Department for Business, Energy & Industrial Strategy*
DEFRA	Department for Environment, Food and Rural Affairs (UK)
DUKES	Digest of United Kingdom Energy Statistics
EC	Elemental Carbon
EECA	Energy Efficiency and Conservation Authority
EMEP/EEA	European Monitoring and Evaluation Programme/ European Environment Agency
EPA	Environmental Protection Agency (USA)
EPC	Energy Performance Certificate
GAINS	Greenhouse gas – Air pollution Interactions and Synergies
HAPINZ	Health and Air Pollution in New Zealand study
IPCC	Intergovernmental Panel on Climate Change
LPG	Liquefied petroleum gas
MBIE	Ministry of Business, Innovation and Employment (NZ)
MC	Moisture Content
MSF	Manufactured Solid Fuel
NAEI	National Atmospheric Emissions Inventory (UK)
NES	National Environmental Standard (NZ)
NMVOC	Non-Methane Volatile Organic Compounds
NSPS	National Source Performance Standard (USA)
NZFFA	New Zealand Farm Forestry Association
NZHHA	New Zealand Home Heating Association
OC	Organic Carbon
OECD	Organization for Economic Cooperation and Development
OGC	Organic Gaseous Carbon
PAH	Polycyclic Aromatic Hydrocarbons
PM	Particulate Matter
RHI	Renewable Heat Incentive (UK)
RSF	Residential Solid Fuels
RWC	Residential Wood Combustion
SIA	Stove Industry Alliance
SSF	Solid Smokeless Fuel
TPES	Total Primary Energy Supply
UNFCCC	United Nations Framework Convention on Climate Change

1. Introduction

Globally, 9.18 $GtCO_2eq$ was emitted from the residential and commercial buildings sector in 2010; accounting for approximately 19% of global greenhouse gas emissions and 33% of black carbon (BC) emissions (Lucon et al., 2014). A significant proportion of emissions in this sector are attributable to inefficient combustion in cookstoves, heating stoves and open fires. Approximately 3 billion people worldwide, mostly in developing nations, rely on biomass and other solid fuels as their primary source of energy (Bonjour et al., 2013), which has significant health impacts due to exposure to air pollutants (Butt et al., 2016; Lelieveld et al., 2015). Within the OECD, energy used for heating accounted for 37% of final energy consumption in 2009 (Beerepoot and Marmion, 2012) and is expected to grow by 79% over the period 2010–2050 (Lucon et al., 2014). Despite this, the residential and commercial buildings sector above all others was highlighted as having the greatest potential for the most cost-effective emissions reductions through energy efficiency measures and renewable space heating technologies (UNEP, 2009; IEA, 2013).

Biomass (mainly wood logs and pellets) has been identified as a key option to decarbonise the residential sector and consumption has been increasing in recent years, largely owing to a combination of bioenergy support initiatives, higher energy prices, aesthetics, and climate change consciousness (Eisentraut and Brown, 2014). Consequently there has been an impact on health due to deteriorated air quality in many areas, particularly in wintertime. For example, an estimated 20,000 and 9200 premature deaths occurred in Western Europe and high-income North America in 2010 due to residential heating with wood and coal; an increase of 23% and 18% respectively on 1990 estimates (Chafe et al., 2015). Fuel switching from oil and gas fuels to residential solid fuels (RSF) can also exacerbate air quality issues, particularly at a local scale. Moshammer et al. (2009) estimated that if all homes in an Upper Austria study region switched from oil to wood-fired heating systems, there would be an increase in the annual average PM_{10} concentration of 3–5 $\mu g\ m^{-3}$, leading to approximately 170 additional premature deaths per year.

Small scale combustion of solid fuels in heating stoves and fireplaces is often uncontrolled and unabated, leading to high emissions factors for gaseous and particulate pollutants. Methane (CH_4) and non-methane volatile organic compounds (NMVOCs) are byproducts of too low combustion temperatures or lack of available oxygen in the combustion chamber (Van Loo and Koppejan, 2007). Emissions are generally much higher for biomass fuels than for coal, but also depend on combustion conditions which are characterised by the modified combustion efficiency (MCE). A high value of MCE denotes efficient flaming combustion and low carbon monoxide (CO) to carbon dioxide (CO_2) ratios. A low value of MCE denotes inefficient smouldering combustion, with high levels of CO and organic carbon (OC). The latter which may contain tars, phenolics, acetic acid, aldehydes and polycyclic aromatic hydrocarbons (PAH). Low values of MCE are common in older log wood stoves or where there are poor operating procedures such as overloading or poor inlet air control. Nitrogen oxides (NO_x) and to a lesser extent nitrous oxide (N_2O) and ammonia (NH_3) are in the most part formed via the conversion of fuel-bound nitrogen and proteinaceous compounds at the low temperatures observed in stoves and fireplaces (Williams et al., 2012). Hence they are proportional to the nitrogen content of the fuel. The same is true of sulphur dioxide (SO_2) emissions which are dependent on the levels of sulphur, calcium, potassium and chlorine in the fuel. The sulphur content of wood is typically very low (<0.1%), so coal-based sources are more significant. Particulate matter below 10 μm in diameter (PM_{10}) and below 2.5 μm in diameter ($PM_{2.5}$) are among the most useful indicators of the health impacts of RSF use (Naeher et al., 2007; Straif et al., 2013). Many studies have shown that PM from RSF combustion is predominately in the fine and ultrafine fraction, which penetrate deep into the lungs and can cause cardiopulmonary disorders and cancer (Allan et al., 2010). The constituents of $PM_{2.5}$ include black carbon (BC), organic carbon (OC) and ash. BC is characterised by strong absorbance of visible light, insolubility in water and a microscopic appearance of aggregated carbon spherules. Radiative forcing (the net change in irradiance causing either cooling or warming) via BC arises both directly, via light absorption, and indirectly via darkening of ice and snow. There is also a cooling effect via cloud interaction, but this is uncertain and direct absorption of radiation in the atmosphere is the largest term (Bond et al., 2013; Boucher et al., 2013; Seinfeld and Pandis, 2006). Organic carbon aerosol can be primary (POA) or secondary (SOA) formed in the atmosphere by VOC oxidation products. Recent research has shown that the contribution of residential wood burning to organic aerosol loadings may be up to a factor of 3 higher when SOA is included (Bruns et al., 2015). The organic fraction is often adsorbed to the surface of BC or ash particles and is among the most harmful to health, having irritant, carcinogenic, mutagenic, teratogenic qualities (Naeher et al., 2007; Jones et al., 2014). POA has a net negative radiative forcing in the atmosphere and in clouds, with a slight positive effect on ice and snow. There is also a slight positive radiative forcing from the small fraction of OA that absorbs radiation, especially in the UV range, which is termed 'brown carbon' (Saleh et al., 2014). Interestingly, the negative radiative forcing of fossil fuel POA is almost twice that of biomass (Bond et al., 2013), which may be linked to the higher degree of oxygenation of biomass soot (Jones et al., 2005). Finally, inorganics are present in the ash fraction of PM, mainly as alkali salts (KCl, K_2SO_4 and K_3PO_4) with smaller amounts of trace elements and heavy metals including Zn, Pb, Cd and aluminosilicates (Molnár et al., 2005). Small scale unabated burning of waste wood and treated timber may also release arsenic. Elevated As concentrations have been attributed to this source in New Zealand (Ancelet et al., 2015) and the USA (Peters et al., 1984).

Residential wood burning is often assumed to be carbon neutral and one of the cheapest ways to reduce greenhouse gas emissions. In this study we assume that wood burning is indeed CO_2 neutral, and investigate the emissions and climate impacts of other pollutants, given that assumption. A comparison is made between the United Kingdom, where residential wood burning is being promoted and growing rapidly; and New Zealand, where wood burning stoves have been widely used for many years and are causing severe wintertime pollution in some areas.

2. Review of residential solid fuel (RSF) use in the UK and New Zealand

2.1. RSF in the UK

The UK has legally binding targets to ensure 15% of energy comes from renewable sources by 2020, and to reduce greenhouse gas emissions by 80% by 2050, relative to 1990 levels. For the residential and heating sectors, the Renewable Energy Strategy 2009 set a target of 12% of heat to come from renewables by 2020 (corresponding to approximately 260 PJ). Fuel switching to electricity and biomass was identified as a key pathway to achieve this (DECC, 2012a), but residential biomass use was noted to have the potential for significant air quality impacts (DECC, 2012b). The UK's greenhouse gas emissions have reduced by approximately 30% since 1990, but residential sector emissions have reduced by just 20% (DECC, 2015a). Hence the residential sector share of total GHG emissions has increased from 21% to 24%.

Official figures show that in total, RSF consumption in the UK has reduced by 87% since 1970. This reduction has been driven by a move away from coal-fired boilers to more efficient and less polluting gas & electric heating central heating systems, as shown in Fig. 1a and Fig. 1b.

Fuel switching from coal to gas has been driven by increased availability of North Sea gas and associated national grid infrastructure, as well as national policy aimed at reducing the number of smog events such as those seen in the 1950s, 1960s and 1970s. Air quality legislation such as The Clean Air Act of 1956 (revised 1993) has dramatically reduced the demand for coal since its inception, by prohibiting the emission of visible smoke.

In the year 2014, natural gas accounted for 83.6% of total residential energy consumption. Although solid biomass contributed just 5.1% of total UK non-electric energy consumption, it dominated the RSF category and represents the largest renewable energy source in the sector. Biomass use has increased more quickly in the EU28 residential sector, having increased from 929 PJ in 1990 to 1606 PJ in 2014, an increase of 73% (EUROSTAT, 2016). Other technologies such as solar thermal, biogas and air & ground source heat pumps are gaining popularity, thanks in part to government incentive schemes such as the domestic renewable heat incentive (RHI), but biomass heating systems are the largest contributor to renewable heat production. Biomass produced 55% of renewable heat paid for under the domestic RHI between April 2014 and February 2016 (DECC, 2016b). Of the total number of accreditations for biomass systems, 58% replaced oil/kerosene fired heating systems which are among the most expensive to run. It should be noted, however, that log heating stoves are not eligible for and hence not included in the RHI statistics. Pellet stoves and boilers are eligible, but must meet emissions, sustainability and metering criteria; and the home must provide an Energy Performance Certificate (EPC) or a Green Deal Advice Report.

Woodfuel for household heat is one of the major drivers of bioenergy uptake in the UK, and is strongly correlated to gas and oil prices (Adams et al., 2011). However due to relatively high capital costs and a need to develop supply chains, UK policies supporting biomass have, until recently, mostly targeted medium and large scale applications. Sites with relatively high heat demands that are not connected to the national gas grid were found to be the most likely to implement biomass for heat within the residential/commercial sector. This includes agricultural buildings, hotels and schools/higher education institutions (Carbon Trust, 2012). Such schemes are generally 100–1000 kW biomass boilers using pellets or wood chip which can be delivered in bulk. Larger systems also commonly have combustion optimisation features such as lambda sensors for oxygen feedback, secondary/tertiary air injection and flue gas abatement technologies. In the most part, heating stoves and fireplaces do not feature such control technologies which leads to more inefficient combustion and higher emissions of pollutants per unit fuel input.

Very little data is available on heating stoves and household RSF consumption in the UK, primarily due to difficulties in monitoring and regulating such small scale emissions sources. In an attempt to

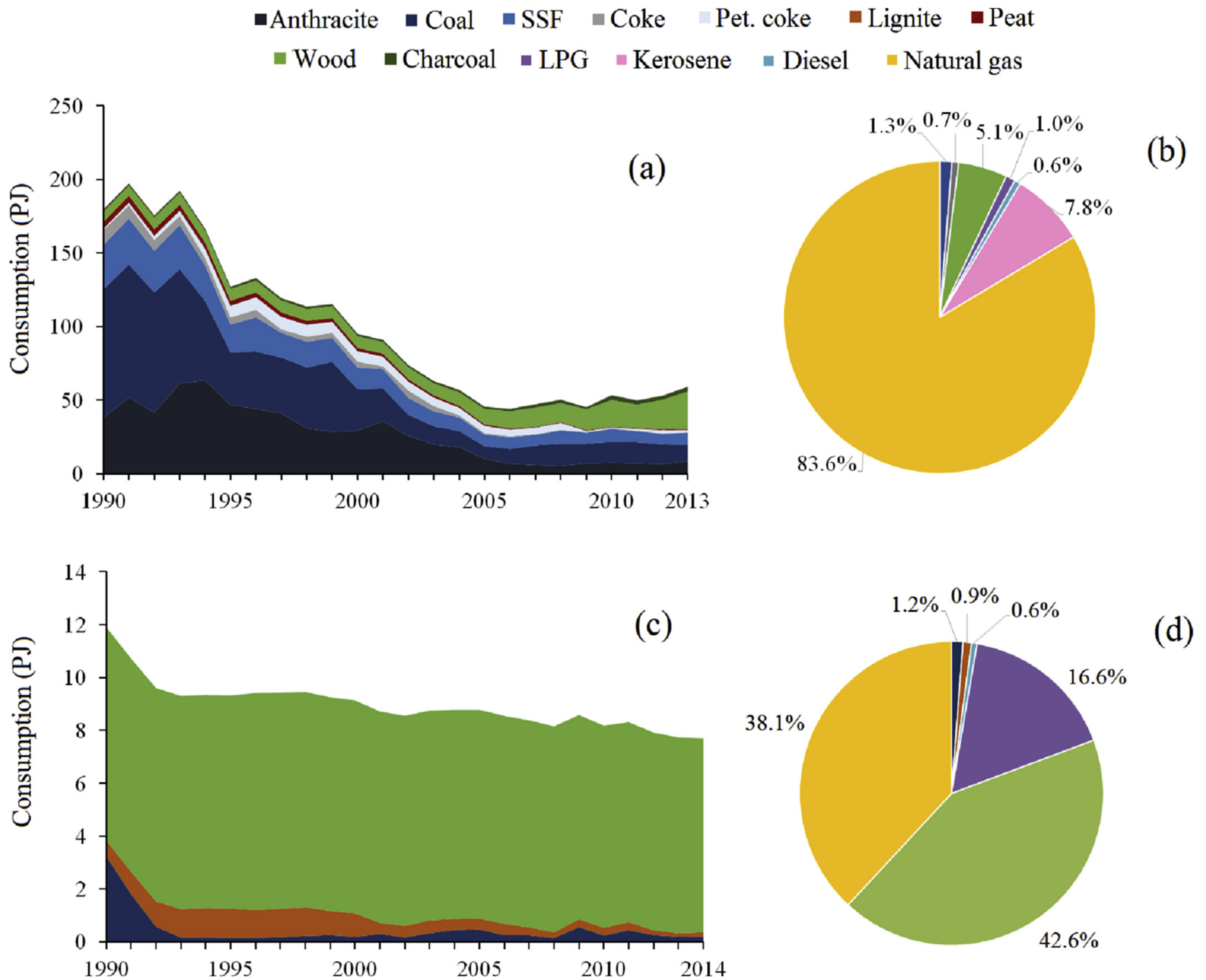

Fig. 1. Breakdown of officially reported RSF consumption in the residential sectors of the UK (a) and New Zealand (c). Consumption of all fuels including gas and liquid is shown in (b) for the UK and (d) for New Zealand. Data sources: NAEI, DUKES (DECC, 2015a), EUROSTAT (2016), MBIE (2015).

better understand the consumption of wood in UK homes, the Department for Energy and Climate Change (DECC) conducted a nationwide survey in 2015 (DECC, 2016a). In summary, the survey found that 7.5% of respondents used wood fuel, and over 90% of those used logs in heating stoves and fireplaces, rather than pellets, chips or briquettes. A similar trend was found across Europe, where 90% of residential biomass used is in the form of hardwood logs (Wöhler et al., 2016). The DECC survey also found that previous estimates of domestic wood consumption were a factor of 3 lower than the 68 PJ total for 2013. It should be noted that the data shown in Fig. 1 do not include these revisions. According to data from the Stove Industry Alliance (SIA), sales of heating stoves were 200,000 in 2014, up 21% on 2005 levels (SIA, 2016). Approximately two thirds of these were multi-fuel stoves, although research showed that 77% were used to burn wood only. Sales growth was strongest for low emission DEFRA exempt appliances, which are approved for use in smoke control areas (see Section 2.3). In the future, sales are expected to grow for stoves which meet the European Ecodesign emissions limits, which emit up to 80% less particulate matter than older stoves.

It has been known for many years that RSF combustion contributes to UK air pollution, particularly in rural communities (Lohmann et al., 2000; Lee et al., 2005). Yet there are very few studies on biomass burning source apportionment compared with other countries in Europe and North America, for example. Several studies have recently found that domestic wood burning is an increasingly important source of particulate matter. Fuller et al. (2014) estimated the contribution of wood burning to annual PM_{10} in London to be 1.1 $\mu g\ m^{-3}$ and Crilley et al. (2015) estimated the contribution to black carbon to be 15–30%. Young et al. (2015) found the contribution to organic aerosol to be up to 38% during the winter. These emissions rival those of the traffic sector, causing dangerous air pollution and counteracting traffic emissions reduction policies in London (Robinson, 2015).

2.2. RSF in New Zealand

New Zealand is traditionally viewed as a good example of a low carbon economy, particularly regarding electricity and heat supply. The contribution of renewables to total primary energy supply

(TPES) in New Zealand was 38.3% in 2012, the third highest in the OECD. In contrast, the contribution in the UK was 4.5%; the fifth lowest in the OECD (OECD, 2014). Of the renewable contribution to TPES, 80% came from geothermal and hydro power in 2014. Nationwide, woody biomass supplied 58.3 PJ, up 52% since 1990 and of this, 13% (7.34 PJ) was consumed in the residential sector.

In contrast to the UK, RSF consumption in New Zealand has been relatively constant since 1990, and the fuel mix is dominated by wood, as shown in Fig. 1c and d. In comparison to the UK, there is a greater reliance on LPG (16.6%) and low grade coal/lignite, as well as wood (42.6%). There is also comparatively low uptake of kerosene/heating oil and patent fuels (manufactured solid fuels, including smokeless fuel and coke). Coal consumption is constrained largely to the west and south of the country where it is mined. The RSF mix has remained largely unchanged for many years, as shown in Fig. 1c, although total consumption has been reducing gradually at an average rate of 85 TJ $year^{-1}$ between 1995 and 2014. New Zealand's Bioenergy Strategy 2010 (BANZ, 2010) set out targets for 25% of consumer energy to come from bioenergy by 2040 (currently 8.5%), as well as a 60% increase in the country's use of biomass for heat. This includes substitution of coal or gas heating.

Both UK and New Zealand homes are often highly energy inefficient in comparison to other OECD countries, due to relatively poor insulation and heating patterns (Howden-Chapman et al., 2009). In New Zealand there is a tradition of heating just one room of the house using unflued gas and electric heaters, as well as open fires and heating stoves burning RSF. Homes using solid fuel heating stoves were found to be warmer on average than homes using other heating methods (French et al., 2007). Wood heating is also one of the cheapest options for homeowners due to the plentiful supply. New Zealand has a large domestic source of wood fuel, mainly as Radiata pine from the forestry industry. The bioenergy strategy, together with the New Zealand Home Heating Association (NZHHA), NZ Farm Forestry Association (NZFFA) and the Energy Efficiency and Conservation Authority (EECA), are pushing to increase the supply of wood fuels for export. A consequence of this surplus is lower prices for home owners. However, fuel poverty and excess winter mortality are similar in both the UK and NZ at 10–14% and 18–19% respectively (Howden-Chapman et al., 2009). Energy used for space heating accounts for the largest share of residential energy consumption in both countries. The share is 34% in New Zealand (Isaacs et al., 2010), but is much higher in the UK at 62% (Palmer and Cooper, 2014). Although total consumption of biomass in the residential sector is higher in the UK, proportionally it is much higher in NZ, as shown in Table 1.

As shown in the table, average residential biomass consumption per dwelling is over twice as high in New Zealand as the UK. However, accurate reporting of RSF consumption in both countries is confounded by huge uncertainties and variation in the data, especially in comparison to metered fuels such as gas, electricity and LPG (Isaacs et al., 2010). Daily wintertime wood consumption estimates vary from 277 MJ day^{-1} in Christchurch to 486 MJ day^{-1} in Nelson, Rotorua and Taumarunui (Wilton, 2012). An average value of 360 MJ day^{-1} was used by Kuschel et al. (2012). The calculated wood fuel use in the DECC survey is 154 MJ day^{-1} for an open fire and 128 MJ day^{-1} for a heating stove; significantly lower than the New Zealand estimates. Analysis of data from the U.S finds that the average household wood consumption in homes that use wood as their primary source of heating is 238 MJ day^{-1} versus 76 MJ day^{-1} in homes where wood is only used for secondary heating (USEIA, 2014). Despite the uncertainty, the officially reported consumption of woody biomass in the NZ residential sector reduced by approximately 9% from 1990 to 2014, as shown in Fig. 1c. This is arguably a result of efficiency improvements and new emissions limits for heating stoves.

2.3. Emissions limits and standard test methods

Three key standards exist for the testing of heating stoves in Europe, NS 3058/NS 3059 in Norway, DIN-plus in Germany and BS PD 6434 in the UK. There are significant differences in the test procedures used in these standards (Seljeskog et al., 2013), as shown in Table 2. In addition, RHI emissions limits apply to eligible boilers in the UK, which include an efficiency of 75%, CO concentrations of less than 1% (ref 13% O_2), and emissions factors of 30 g GJ^{-1} for PM and 150 g GJ^{-1} for NO_x (approx. 0.54 g kg^{-1} and 2.7 g kg^{-1} respectively). The European standard EN 13240 also requires appliance efficiency to be greater than 50% and CO emissions to be less than 1.0% (ref. 13% O_2). However, emissions of PM, NO_x and OGC are left to national legislation. Recently, the Ecodesign of Energy-related Products Directive (2009/125/EC) regulation 2015/1185 was published which has the specific aim of reducing emissions of PM, OGCs and CO from this source by 27 kt $year^{-1}$, 5 kt $year^{-1}$ and 399 kt $year^{-1}$ respectively by 2030. This will be done via the implementation of emissions limits for open- and closed-fronted heaters from the year 2022, as shown in Table 2.

As the table shows, there are significant differences in the requirements of standard test methods around the world. Historically, regulation has emphasised total (non-size segregated) particulate matter emissions, although in recent years CO and thermal efficiency have been added, followed by NO_x and organic gaseous carbon (OGC). There are significant differences in the test procedures used in these standards which complicates comparative studies. Key differences include the draught, fuel, reporting units, dilution, filter temperatures, and sampling durations & equipment. One of the highest impact variables is the use of a dilution tunnel, whereby a greater proportion of the condensable organic fraction is captured compared to hot-sampling. This alone can increase PM emissions factors by orders of magnitude (Seljeskog et al., 2013; Coulson et al., 2015). In addition, emissions factors may be increased further if atmospheric ageing of emitted smoke is taken into account (Bruns et al., 2015, 2016), though it may be argued that OGC measurement may be used as a proxy for SOA formation.

New Zealand's National Environmental Standards (NES) feature five standards for ambient air quality. The NES standards for CO, NO_2 and PM_{10} are 10,000 μg m^{-3} (8 h mean), 200 μg m^{-3} (1 h mean) and 50 μg m^{-3} (24 h mean) respectively. Most breaches of this standard are attributed to domestic heating with wood; with 24 h PM_{10} concentrations of more than 200 μg m^{-3} having been recorded in some towns (Coulson et al., 2013). Hence New Zealand

Table 1
Comparison of residential biomass consumption in the UK and NZ, 2014.

	NZ	UK	Unit	Ref
Solid biomass consumption in residential sector	7.34	54.67	PJ	(EUROSTAT, 2016); (MBIE, 2015)
Number of dwellings (million)	1.781	27.914		(DCLG, 2016); (StatisticsNZ, 2016)
Population (million)	4.509	64.596		ONS, (StatisticsNZ, 2015)
Average biomass consumption per dwelling	4.12	1.96	GJ $household^{-1}$	
Average biomass consumption per person	1.63	0.85	GJ $person^{-1}$	

Table 2
Comparison of standard test conditions for heating stoves in different countries. Expanded from Seljeskog et al. (2013).

	Country	Europe	Europe	Germany/Austria	Norway	UK	USA	Australia/New Zealand
Test parameters	Standard	Ecodesign regulation 2015/ 1185	EN 13240	DIN-plus	NS 3058	BS PD 6434/BS 3841	NSPS/ASTM E2515, E2780 – 10/EPA Method 28WHH	AS/NZS 4012, 4013 and 4014
	Location		Chimney	Chimney	Dilution tunnel	ESP/Dilution tunnel	Dilution tunnel	Dilution tunnel
	Draught		Forced 12 PA	Forced 12 PA	Natural	<1.25 Pa (natural)	<1.25 Pa (natural)	<1 Pa (natural)
	Sampling temp		70 °C	70 °C	35 °C	70 °C	<32 °C	15-32 °C
	Fuel	Range of Biomass/fossil fuels. Wood logs must be beech, birch or hornbeam	Range of Biomass/fossil fuels. Wood logs must be beech, birch or hornbeam	As specified in EN 13240	Dimensioned spruce (49 × 49 mm), 16–20% MC	Coal, lignite, patent fuels, peat and wood	“Crib wood” dimensioned (38 × 89 mm) Douglas Fir, 15–25% MC. Cordwood alternative available	Dimensioned (100 × 50 mm) Radiata pine, 16–20% MC in New Zealand. Hardwood in Aus
	Weight of test fuel	Dependent on choice of PM measurement method	As per manufacturer's instruction	As specified in EN 13240	112 ± 11 kg m^{-3} firebox volume	15 kg	112 ± 11 kg m^{-3} firebox volume	
	Test condition	Dependent on choice of PM measurement method	3 categories: Nominal, slow and safety tests	As specified in EN 13240	4 burn rate categories	2 burn rate categories: nominal and low (plus intermediates if necessary), repeated 5 times	3 Method 28 burn rate categories	3 burn rates: high, medium and low
	Test duration	Dependent on choice of PM measurement method	Min. refuelling interval 1.5 h for wood at nominal	30 min		Time between first re-fuel and a trough in radiation heat output	Load time 1060 s m^{-3} firebox volume	
	Include ignition/start-up?	Dependent on choice of PM measurement method	No	No	No – 1 h pre-test	No – provided no “undue trouble to the user”	No – kindling, newspaper and pre-burn fuel	No
	Units	mg m^{-3} at 13% O_2	mg Nm^{-3} at 13% O_2	mg Nm^{-3} at 13% O_2	g kg^{-1}	g hour^{-1}	g hour^{-1}	g kg^{-1}
Emissions limits	PM	2.4/5.0		75	10	5	4.5 reducing to 2.0	1.5
	CO	1500	<1%	1500			Optional?	
	OGC/THC (as C)	120		120				
	NOx (as NO2)	200/300		200				
	Efficiency	65%	50%	75%			63% (non-catalytic)	65%

has introduced a design standard for wood burners installed in urban areas. The NES standard for wood burners centres on PM_{10} emissions and an emissions limit of 1.5 g kg^{-1} dry fuel burned is required when tested to AS/NZ 4013. An efficiency of 65% is also required when tested to AS/NZ 4012 using fuels certified under AS/NZ 4014. AS/NZS 4013:2014 is a revised version of AS/NZS 4013:1999, and initial tests showed that the revised method is more representative of real-world conditions and gave emissions factors 2.5 times larger than the previous method (Todd and Greenwood, 2006).

A comprehensive review of particulate emissions due to RSF burning in New Zealand was carried out by Wilton (2012), who noted that real-world emissions of NES compliant appliances were typically twice as high as those determined under laboratory conditions as described above. Real-world emissions have been found to be substantially higher in New Zealand (Ancelet et al., 2010; Xie et al., 2012), as well as in Europe (Wöhler et al., 2016) and the USA (USEPA, 2016); primarily due to user operating conditions such as start-up, fuel properties, overloading and fluctuating burn rates. A statistical analysis of PM_{10} emissions factors from in-situ wood stove tests in New Zealand was carried out by Coulson et al. (2015). The study found that geometric mean emission factors for older and low-emission stoves were 9.8 ± 2.4 g kg^{-1} and 3.9 ± 3.8 g kg^{-1} (dry wood) respectively. The distribution was found to be log-normal and hence the use of geometric, rather than arithmetic, mean emission factors are recommended.

A new standard for PM emissions from wood stoves was introduced in the city of Nelson in 2006, requiring 1 g kg^{-1} rather than the NES standard of 1.5 g kg^{-1}. As a result of this implementation, PM_{10} and BC were found to be decreasing at an average rate of 0.5 μg m^{-3} and per year and 100 ng m^{-3} per year respectively (Ancelet et al., 2015). Stove replacement programs have been found to achieve similar benefits in other countries. For example, Noonan et al. (2011) noted a 70% reduction in indoor $PM_{2.5}$ concentrations in a rural community in the USA, due to replacing old and inefficient wood stoves. Rule 4901 was passed in the San Joaquin Valley, California, in 1992 which limited emissions from RSF burning during periods of poor air quality, and required new wood burners to meet EPA/NSPS certified. As a result, $PM_{2.5}$ concentrations reduced in the area by 11–15% (Yap and Garcia, 2015). In Europe, it is estimated that replacing current RSF technologies with more efficient wood pellet stoves could reduce concentrations of OC and EC by more than 50% in large parts of the continent (Fountoukis et al., 2014).

Due to regular breaches of NES air quality standards by RSF burning, a number of health impact studies have been carried out in New Zealand. Perhaps the most comprehensive was the Health and Air Pollution in New Zealand (HAPINZ) study (Kuschel et al., 2012). It found that RSF burning was attributable to 56% of premature deaths due to anthropogenic PM_{10} in 2006, making it the leading cause. This equated to 655 premature deaths, 334 admissions due to cardiac and respiratory illness, and 817,600 restricted activity days. The estimated cost due to these impacts was NZD $2.385 billion. In addition, it was noted that basing the report on PM_{10} rather than $PM_{2.5}$ led to an underestimate of the attribution of health impacts to transport and RSF emissions because these sources make a greater contribution to fine PM. For example, studies have shown that over 90% of the mass of emissions from wood combustion are below $PM_{2.5}$ (Bond et al., 2004; Nussbaumer, 2003; Mcdonald et al., 2000; Young et al., 2015).

3. Methods

The New Zealand national census is a useful means of collecting data on qualitative RSF use. Question 16 requires the resident to "*mark as many spaces as you need to show which of the following are ever used to heat this dwelling.*" The UK census is more focussed on the type of central heating used at a property (gas, electric, oil, solid fuel, other, or no central heating). Information on fuels used for supplementary heating is limited to sub-national housing surveys and studies into fuel poverty in off-grid homes by organisations such as the Office of Fair Trading (OFT, 2011), the Office of Gas and Electricity Markets (OFGEM) and the Department of Energy and Climate Change (DECC) (Palmer and Cooper, 2014). The New Zealand census also has the advantage of being held every 5 years, whereas the UK census is held every 10 years. Additionally, data is available at three different resolutions: census area unit (CAU); ward; and territorial authority. CAU represents the finest resolution, with some urban grid cells less than 1 km^2 in area. A number of models and inventories offer activity data, emissions data and emissions factors for the residential sectors of both countries. Studies have shown that several models in Europe underestimate pollutants such as wintertime OC when compared with observations, which is most likely due to residential wood burning (Aas et al., 2012). The use of revised emissions factors for RSF combustion was found to increase total $PM_{2.5}$ emissions in Europe by 20% (Denier Van Der Gon et al., 2015).

3.1. A top-down estimate of BC concentrations in New Zealand

A top-down approach was used to estimate black carbon concentrations due to RSF combustion in New Zealand. Emissions of PM_{10} and corresponding monthly atmospheric concentrations in 2006 were taken from the HAPINZ study (Kuschel et al., 2012). BC concentrations were calculated by multiplying PM_{10} concentrations by the ratio of BC/PM_{10}. To define this ratio for New Zealand both spatially and temporally, 31 separate datasets containing simultaneous measurements of PM_{10} and BC were analysed from 10 locations across New Zealand. The wintertime BC concentrations were then calculated for each census area unit (CAU) in New Zealand and were mapped using ArcGIS.

3.2. Emissions and climate impacts using the GAINS model

In order to assess the impacts of RSF emissions, the GAINS model (http://gains.iiasa.ac.at/models/) was used to provide detailed activity and emissions data broken down by fuel and technology type, in both the UK and New Zealand. The version of the model used was ECLIPSE version 5 for UNFCCC Annex 1 nations. Several scenarios are available but here we use the current legislation (CLE) scenario (Stohl et al., 2015), which assumes efficient enforcement of committed legislation, with some deviations. For the residential sector, it is not known whether this scenario includes legislation such as Ecodesign in Europe.

The residential sector in GAINS is broken down into four key technologies: commercial boilers (<50 MW), single house boilers (<50 kW), heating stoves and fireplaces. There are minor contributions from open pits and cookstoves, but these are small in comparison to the other technologies and are not considered in this work. Each technology is then also broken down by fuel type. For the UK, fuels include hard coal (grade 1), derived coal (coke, briquettes etc.), agricultural residues and fuelwood. For New Zealand, the split is between hard coal (grade 1), brown coal/lignite (grade 1), and fuel wood.

Emissions data is available for 12 pollutants in GAINS: carbon dioxide, methane, oxides of nitrogen, carbon monoxide, non-methane volatile organic compounds, sulphur dioxide, ammonia, nitrous oxide, PM_{10}, $PM_{2.5}$, black carbon and organic carbon. For some pollutants, the full breakdown by fuel and technology was not available. These included CO_2, NO_x, CO, SO_2, NH_3 and N_2O. For these

Table 3
GAINS emissions factors for the general residential sector used to calculate technology-specific emissions where the data was unavailable.

Parameter	Net Forcings ($\mu W\ m^{-2}$) ($Gg\ yr^{-1}$)$^{-1}$	Emissions factors ($t\ PJ^{-1}$)			
		Brown Coal/lignite	Hard coal, grade 1	Derived coal (coke etc)	Biomass
CO_2	0.0917	99,500	94,300	100,000	0
CH_4	2.2				
NO_x	−6.2	70	118	110	68
CO	0.48	5000	5000	5000	4000
NMVOC	0.78				
SO_2	−9.0	1239	616	541	4
NH_3	0	8	8	0.5	8.2
N_2O	24.3	1.4	1.4	1.4	4
BC	74.3				
OC (fossil fuel)	−16.9				
OC (biomass)	−12.5				

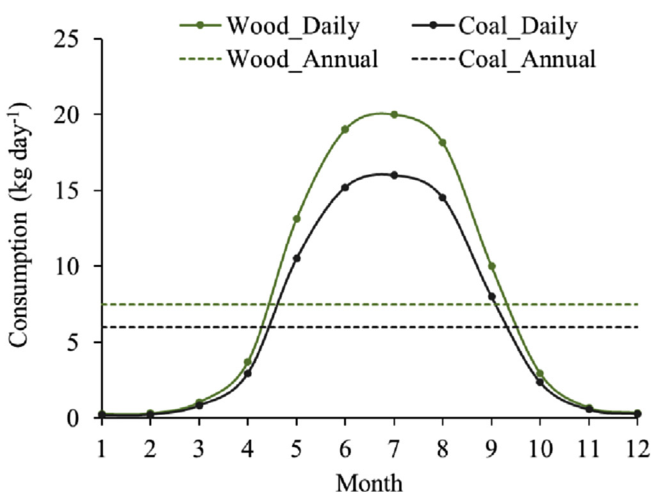

Fig. 2. Model factors for average daily and annual household consumption of wood and coal, in households using each fuel.

species, the breakdown was calculated by multiplying the GAINS activity data by the GAINS emissions factors for each fuel for the general residential/domestic sector (fuel specific but not technology specific). These are given in Table 3. The net CO_2 emissions factor is assumed to be zero for biomass, in order to investigate the climatic effects of non-CO_2 species. In the case of CO, emissions factors were not available in this version of GAINS. Therefore emissions factors were taken from the EMEP/EEA database (EEA, 2013) in this case, again using GAINS activity data. Full BC and OC emissions were available for the UK (in GAINS Europe) but not for New Zealand. The New Zealand emissions were calculated from PM_{10} emissions data, using the ratio of the GAINS BC and PM_{10} emissions factors for the UK.

The climate impacts were calculated by multiplying the emissions for each RSF source by the Absolute Global Warming Potential (AGWP) for each pollutant. The units of AGWP are radiative forcing per unit emission over one year, and are taken from (Bond et al., 2013). The values for CO_2 and N_2O were taken from the IPCC AR5 report (Myhre et al., 2013).

Table 3 shows the net radiative forcings for each pollutant, which includes direct and indirect effects on a global scale. Cloud effects for species such as BC and particulate organic carbon are included in these net factors. See Bond et al. (2013) for the full breakdown. The values used here are also central estimates. For BC, the net lower and net upper estimates are 83% lower and 144% higher than the central estimate respectively. For biomass OC, the errors are −65% to +84%.

3.3. *A bottom-up emissions inventory calculation and comparison*

Finally, a bottom-up emissions inventory was produced for both countries using unique activity data and emissions factors. This allowed the comparison of activity data, emissions, and climate impacts between this study and the GAINS model, alongside several other international climate models. An extensive review of RSF sector emissions factors was carried out. The most comprehensive and fuel/technology specific factors were found to be those of the EMEP/EEA database and these were selected for the modelling work (EEA, 2013).

Activity data in New Zealand was derived following the method of the HAPINZ study (Kuschel et al., 2012). The method uses 2006 census data for the number of homes in each census area unit, multiplied by average daily wintertime consumption factors for wood and coal, multiplied by average PM_{10} emissions factors for each species. These emissions are then constrained to inventories which have been produced for regional councils. Finally, these peak wintertime values are assigned an annual distribution in order to account for the high seasonal variability of RSF use. In this work, the updated 2013 census data (StatisticsNZ, 2015) has been used, with the same wintertime consumption factors of 20 kg day^{-1} for wood and 16 kg day^{-1} for coal. The annual distribution is presented in Fig. 2. The distinction between different grades of coal is not possible with this method, because the census does not differentiate bituminous coal from lignite or anthracite; which are known to have substantially different emissions factors (Lee et al., 2005; Mitchell et al., 2016).

Activity data in the UK was derived from the recent DECC Wood Consumption Survey for wood (DECC, 2016a) and the DECC Sub-National Residual Fuel Consumption Statistics (DECC, 2015b) for coal and derived coal/manufactured solid fuel (MSF). The former also has data on the number of homes using coal, but the focus is on wood users who use coal as well as wood.

4. Results

4.1. *Top-down calculation of 2006 BC concentrations in New Zealand*

Analysis of datasets featuring simultaneous PM_{10} and BC measurements was conducted in several wood burning communities across New Zealand in order to determine the ratio of BC/PM_{10}. The results are given in Table 4.

In addition to Table 4, a study from a suburban town near Wellington found that the contribution of wood burning to ambient $PM_{2.5}$ and BC averaged over a two year period was 2.9 $\mu g\ m^{-3}$ and 846 ng m^{-3} respectively (Davy et al., 2012). Hence the ratio of BC/$PM_{2.5}$ was 28.8%, which is similar to the BC/PM_{10} ratio observed in

Table 4
Ratio of BC/PM_{10} in urban (U), suburban (S) and rural (R) locations in the winter (W) and the summer (S) in New Zealand.

				Concentration ($\mu g\ m^{-3}$)		Ratio (%)	
Town	Class	Region	Season	PM_{10}	BC	BC/PM_{10}	Data Source
Rangiora, Waikuku, Kaiapoi and Woodend				660.0	9.4	1.9	NIWA, unpublished
	S		W	4.3	1.3	41.9	
	S		W	863.5	1.4	0.3	
	S		W	306.5	1.2	0.8	
Dunedin	U	Dnd	A	112.7	32.3	30.6	
Dunedin	U		A	99.6	25.4	29.6	
Dunedin	U		A	192.6	68.5	43.8	
Dunedin	U		A	242.7	55.6	29.6	
Dunedin	U		A	56.2	18.9	37.7	
Green Island	S		W	84.3	12.8	21.0	
Dunedin	U		W	32.9	2.9	11.0	
Dunedin	U		W	20.2	3.3	17.6	
Takapuna	S	Auk	S	14.3	1.9	13.6	GNS Science, unpublished
Takapuna	S		W	18.1	4.0	22.2	
Queen Street	U		S	17.2	3.8	22.1	
Queen Street	U		W	18.5	5.3	28.6	
Khyber Pass	U		S	17.0	4.0	23.9	
Khyber Pass	U		W	19.7	6.0	30.8	
Penrose	S		S	15.9	1.8	11.1	
Penrose	S		W	18.3	3.3	18.4	
Henderson	S		S	11.8	1.2	10.4	
Henderson	S		W	16.5	3.4	20.5	
Alexandra	R	COt	W	19	4.9	25.7	(Ancelet et al., 2014)
Alexandra	R		W	33	6.6	19.9	
Alexandra	R		W	17	4.4	25.8	
Alexandra	R		W	29	5.5	19.1	
Masterton	R	Wrp	W	25	3.1	12.6	(Ancelet et al., 2012)
Masterton	R		W	32	3.7	11.6	
Nelson	U	Nln	W			12.7	(Grange et al., 2013)
Nelson	U	Nln	W	21	2.9	12.7	(Ancelet et al., 2015)
Auckland, Masterton, Nelson, Alexandra	U	Mixed	W			14.1	(Trompetter et al., 2013)
	U	**NZ**	**W**			**24.6**	
	S	**NZ**	**W**			**16.7**	
	R	**NZ**	**W**			**19.1**	

Chc: Christchurch; DnD: Dunedin; Auk: Auckland; COt; Central Otago; Wrp: Wairarapa; Nln: Nelson.

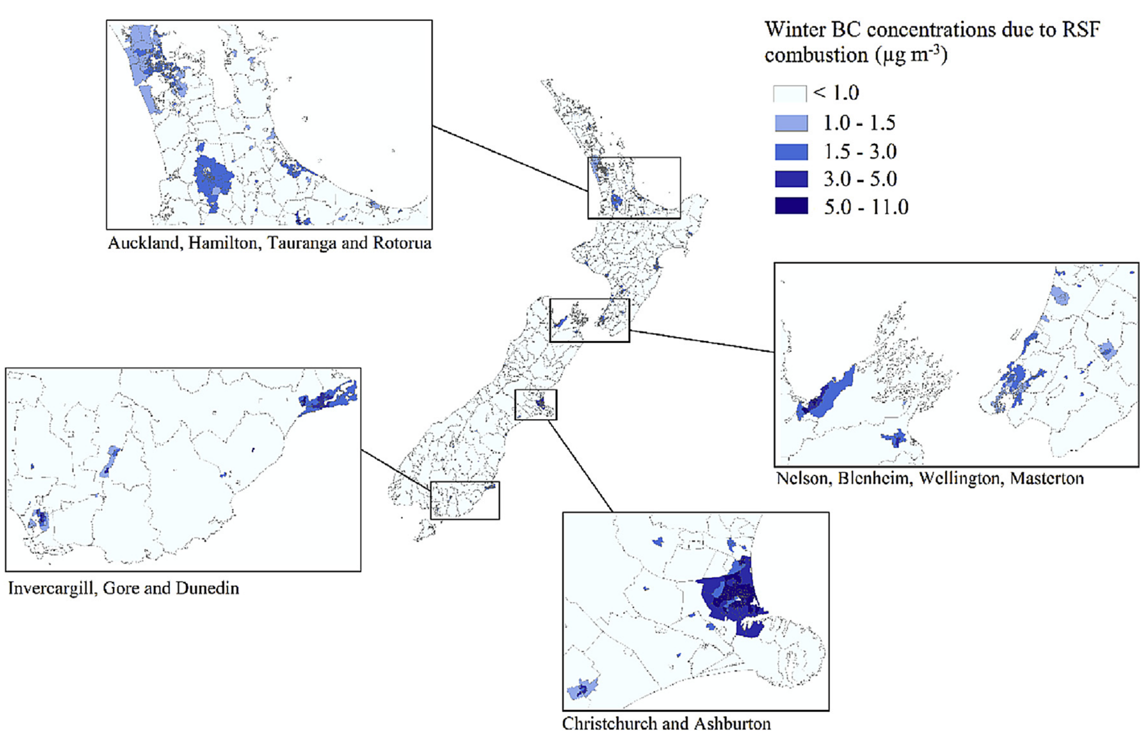

Fig. 3. Wintertime concentrations of black carbon due to residential solid fuel burning in New Zealand in 2006.

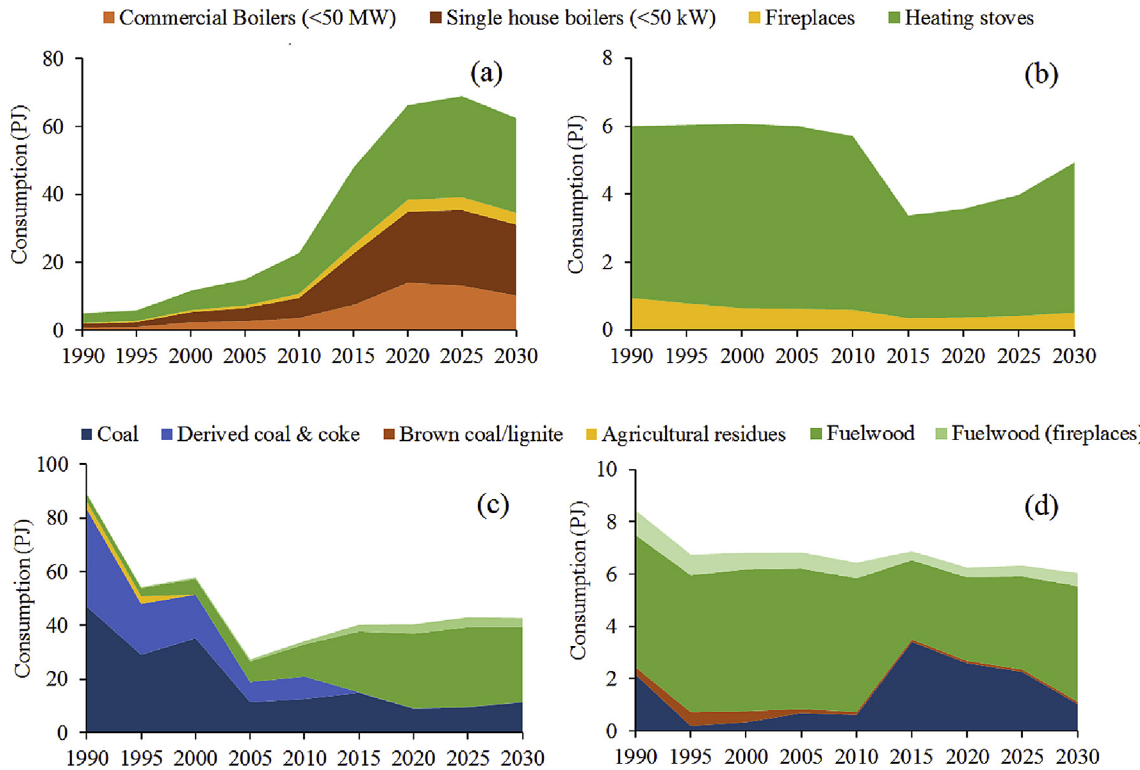

Fig. 4. Breakdown of activity data for RSF combustion by technology and fuel type according to the GAINS database, 1990–2030. Top: wood fuel consumption by technology type in (a) the UK and (b) New Zealand. Bottom: breakdown of fuel consumption in heating stoves and fireplaces in (c) the UK and (d) New Zealand.

other locations. Applying these factors to the HAPINZ data yields the wintertime concentrations of BC in New Zealand, and the results are given in Fig. 3.

The results show that the majority of the country has very low wintertime BC concentrations, typically below 1000 ng m^{-3} and below 500 ng m^{-3} in many rural areas. The highest concentrations were in the city of Nelson, specifically Toi Toi, Wahsington and Bronte districts which had mean winter BC concentrations over 10 μg m^{-3}. Also in the highest 10% were Richmond, Arrowtown, Alexandra, Milton, North beach Christchurch, Kaiapoi Christchurch. Many of these regions are known to have poor wintertime air quality as shown in Table 4.

4.2. Emissions and climate impacts using the GAINS model

Activity data for RSF combustion in the residential sector from the GAINS database is presented in Fig. 4.

In the UK, the model shows that consumption of wood in the residential sector is increasing rapidly and will continue to do so to 2025. Heating stoves account for the largest proportion of wood use (47% in 2015), and this is largely due to a switch from coal and derived coal to biomass, as shown in Fig. 4c. The model forecasts coal consumption in stoves to continue to reduce to 2030, yet wood consumption in stoves and fireplaces is estimated to increase by almost a factor of 4 between 2005 and 2030. It should be noted that GAINS only includes wood consumption in fireplaces and hence does not account for fossil fuel consumption in this technology. A small amount of agricultural residues is consumed in stoves between 1990 and 200, but is negligible compared to other fuels. In New Zealand, the model shows that consumption of wood remained comparatively constant between 1990 and 2010 at approximately 6 PJ. Wood consumption is dominated by heating stoves, with commercial and single house boilers consuming negligible amounts throughout the timeframe. Between 2010 and 2015 there is a 41% reduction in wood consumption and a six-fold increase in hard coal (grade 1) consumption, suggesting a large fuel switching programme in stoves in New Zealand. Lignite consumption remains relatively low (<0.5 PJ) throughout the period.

Fuel- and technology-specific emissions data is available in the GAINS database for certain pollutants in the RSF sector, but not all. The missing values have been calculated using GAINS emissions factors and the activity data given in Fig. 4c and d, as detailed in Section 3.2. The results for heating stoves and fireplaces are given in Fig. 5 for the United Kingdom and Fig. 6 for New Zealand. The UK results show that emissions are highly dependent on the type of fuel used and the activity data for each. Emissions generally follow the same trend as the activity data in Fig. 4c, whereby the total reduces to a low in 2005 as coal consumption reduces, before increasing to 2030 as wood consumption increases. CO_2 and SO_2 emissions are negligible for biomass burning compared to fossil fuel burning and reduce considerably over the period. NO_x and CO emissions are also dominated by fossil fuel combustion and increase by just 27% and 42% respectively from 2005 to 2030. CH_4 emissions are more strongly correlated with wood burning and increase from 3 kt year^{-1} in 1990 to 7 kt year^{-1} in 2030. NMVOCs are also highly dominated by wood combustion throughout the period and total residential sector emissions increase by a factor of 3.3 between 2005 and 2025. This is the largest increase of all pollutants. In 2015, heating stoves accounted for 74.6% of NMVOC emissions from wood combustion in the UK residential sector. Organic carbon (OC) emissions followed a similar trend, except for negligible emissions from derived coal. Particulate emissions are also dominated by wood combustion from the year 2001 onwards. PM_{10} emissions from wood combustion increase in by a factor of 10 in heating stoves and 14 in fireplaces respectively from 1990 to 2030. Similar trends are found in single house boilers and

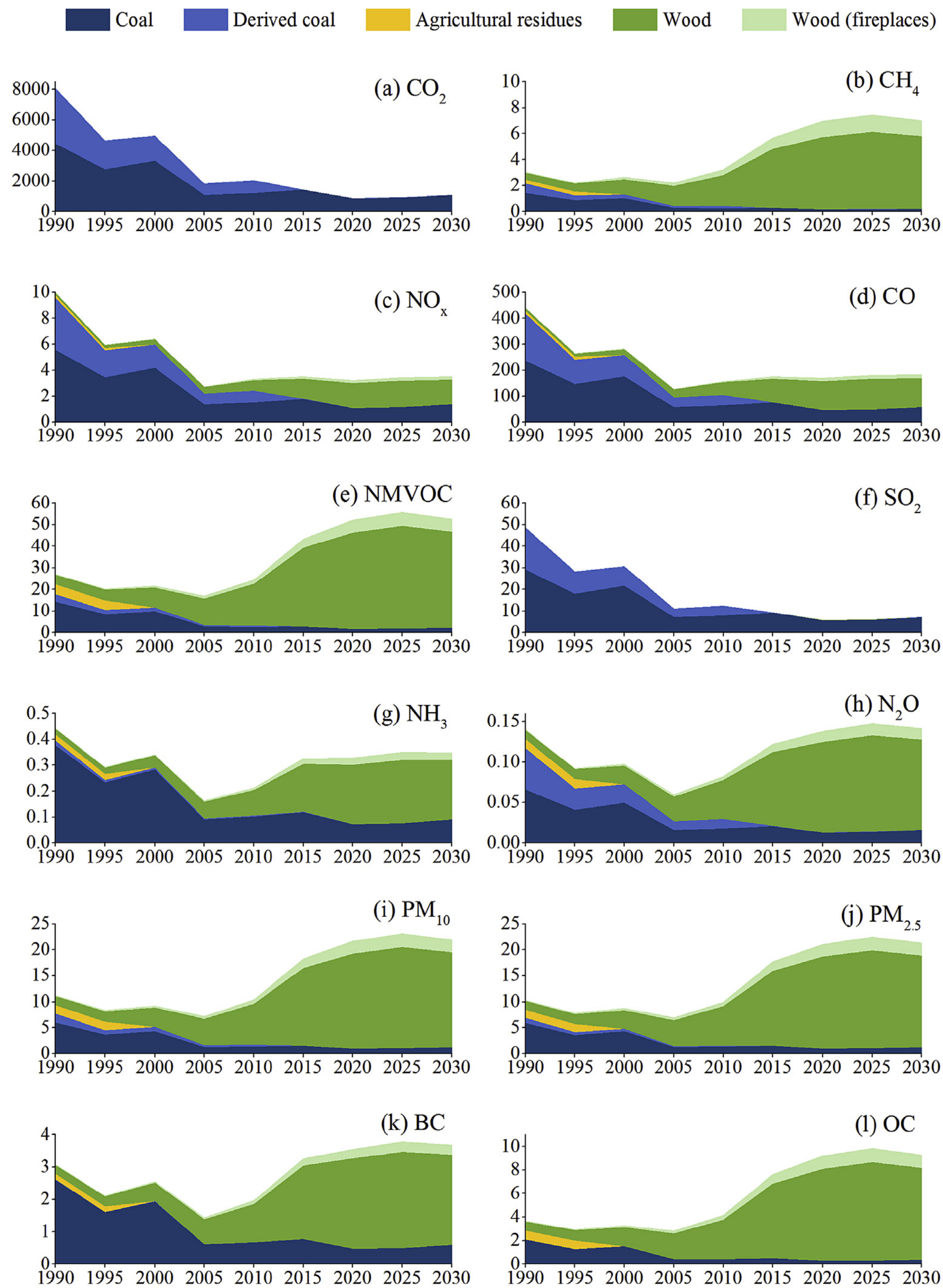

Fig. 5. Emissions of selected climate-relevant species (kt year^{-1}) from heating stoves and fireplaces in the UK, 1990 to 2030. (a) CO_2; (b) CH_4; (c) NO_x; (d) CO; (e) NMVOC; (f) SO_2; (g) NH_3; (h) N_2O; (i) PM_{10}; (j) $PM_{2.5}$; (k) BC; (l) OC.

commercial boilers over the period. $PM_{2.5}$ emissions account for more than 96% of PM_{10} emissions, indicating that the majority of the emitted particles are in the fine fraction. Black carbon emissions are shown in Fig. 5k. BC emissions from wood combustion in stoves increased from 0.27 kt year^{-1} in 1990 to 2.8 kt year^{-1} in 2030. Emissions from coal reduced over the period and fell below those of wood in the year 2004.

Emissions in New Zealand also follow the same trend as the activity data, shown in Fig. 4d. Coal consumption peaks at 3.4 PJ in 2015, with corresponding emissions peaks of 331 kt year^{-1} for CO_2 and 2.2 kt year^{-1} for SO_2. Although consumption of lignite remains low over the modelling period, the fuel contributes significantly to SO_2 emissions, peaking at 0.65 kt year^{-1} in 1995; 82% of total emissions from stoves and fireplaces. Emissions of CH_4 and NMVOCs are more dominated by wood combustion and reduce by a factor of 3 between 1990 and 2030. Emissions of CO, NH_3 and N_2O are relatively evenly split between fossil fuels and biomass and stay largely consistent at 30 kt year^{-1}, 0.5 kt year^{-1} and 0.025 kt year^{-1} respectively. Emissions of $PM_{2.5}$ and OC emissions reduce linearly at rates of 68 t year^{-1} and 23 t year^{-1} respectively. The increased coal consumption has a greater impact on BC emissions, becoming the leading source of BC between 2014 and 2027. Despite this, BC emissions reduce by 42% over the modelling period. A summary of the activity and emissions data for heating stoves and fireplaces in

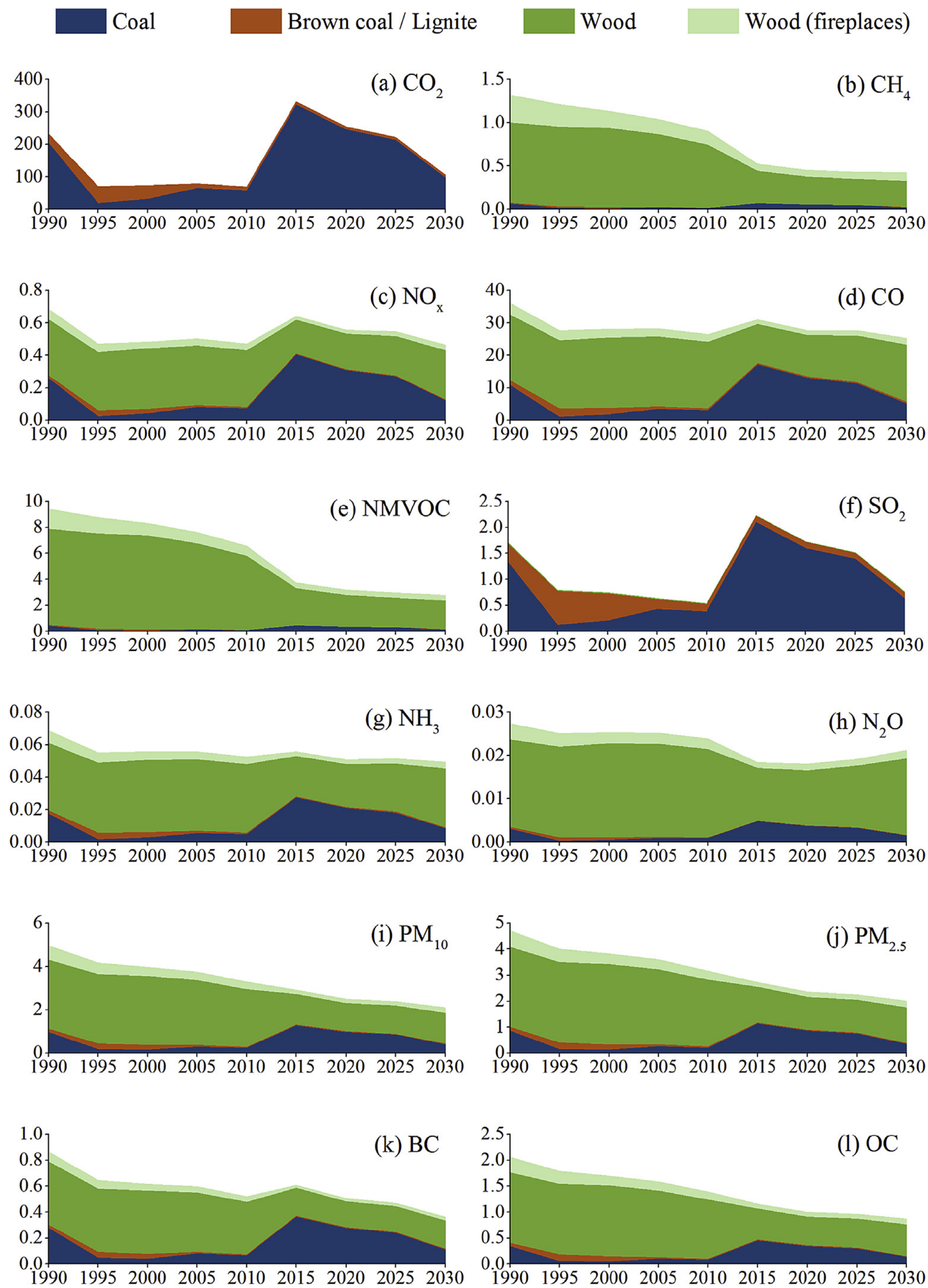

Fig. 6. Emissions of selected climate-relevant species (kt year^{-1}) from heating stoves and fireplaces in New Zealand, 1990 to 2030. (a) CO_2; (b) CH_4; (c) NO_x; (d) CO; (e) NMVOC; (f) SO_2; (g) NH_3; (h) N_2O; (i) PM_{10}; (j) $PM_{2.5}$; (k) BC; (l) OC.

the year 2015 is given in Table 5 for both New Zealand and the UK. Total emissions of black carbon in stoves and fireplaces in 2015 were 3.26 kt in the UK and 0.60 kt in New Zealand. This equates to 0.117 kg dwelling^{-1} and 0.337 kg dwelling^{-1} respectively.

The climate impacts of the emissions profiles given in Figs. 5 and 6 were then calculated for the years 2010 and 2030 and the results are presented in Fig. 7. The total radiative forcing for each fuel type over the entire modelling period is presented in Fig. 8.

The results show that carbon dioxide from fossil fuel combustion was the largest contributor to radiative forcing in the UK residential sector in 2010. The contribution from biomass burning was approximately half that of fossil fuel, with black carbon being the most important warming species. SO_2 from coal and derived coal combustion offset some of the warming by -110 μW m^{-2}, giving a net positive radiative forcing of 218 μW m^{-2} for the UK in 2010. In contrast, by 2030 biomass has a larger warming impact than fossil fuel combustion. Black carbon from wood burning in stoves and fireplaces causes a radiative forcing of 97 μW m^{-2} in 2010. Despite some offset by organic carbon, the total net radiative forcing increases by 23%–268 μW m^{-2}. In New Zealand, net radiative forcing reduces by 21% between 2010 and 2030. Forcing due to biomass burning in stoves and fireplaces is a factor of 4.3 lower than that of

Table 5
GAINS pollutant emissions inventory for RSF combustion in stoves and fireplaces in the United Kingdom and New Zealand, 2015.

Parameter	Unit	UK, 2015				NZ, 2015			
		Biomass		Fossil fuel		Biomass		Fossil fuel	
		Fireplace	Stove	Fireplace	Stove	Fireplace	Stove	Fireplace	Stove
Activity data	PJ	2.52	22.71		15.01	0.35	3.03		3.50
CO_2	kt year^{-1}				1416				331
CH_4	kt year^{-1}	0.88	4.54		0.27	0.09	0.37		0.07
NO_x	kt year^{-1}	0.17	1.55		1.78	0.02	0.21		0.41
CO	kt year^{-1}	10.10	90.86		75.04	1.39	12.11		17.52
NMVOC	kt year^{-1}	4.29	36.34		2.73	0.42	2.88		0.45
SO_2	kt year^{-1}	0.011	0.10		9.23	0.001	0.013		2.22
NH_3	kt year^{-1}	0.02	0.19		0.12	0.003	0.025		0.03
N_2O	kt year^{-1}	0.010	0.09		0.02	0.001	0.012		0.005
PM_{10}	kt year^{-1}	1.82	14.89		1.55	0.21	1.41		1.31
$PM_{2.5}$	kt year^{-1}	1.76	14.42		1.53	0.20	1.36		1.17
BC	kt year^{-1}	0.22	2.27		0.77	0.02	0.21		0.37
OC	kt year^{-1}	0.81	6.35		0.49	0.09	0.60		0.47

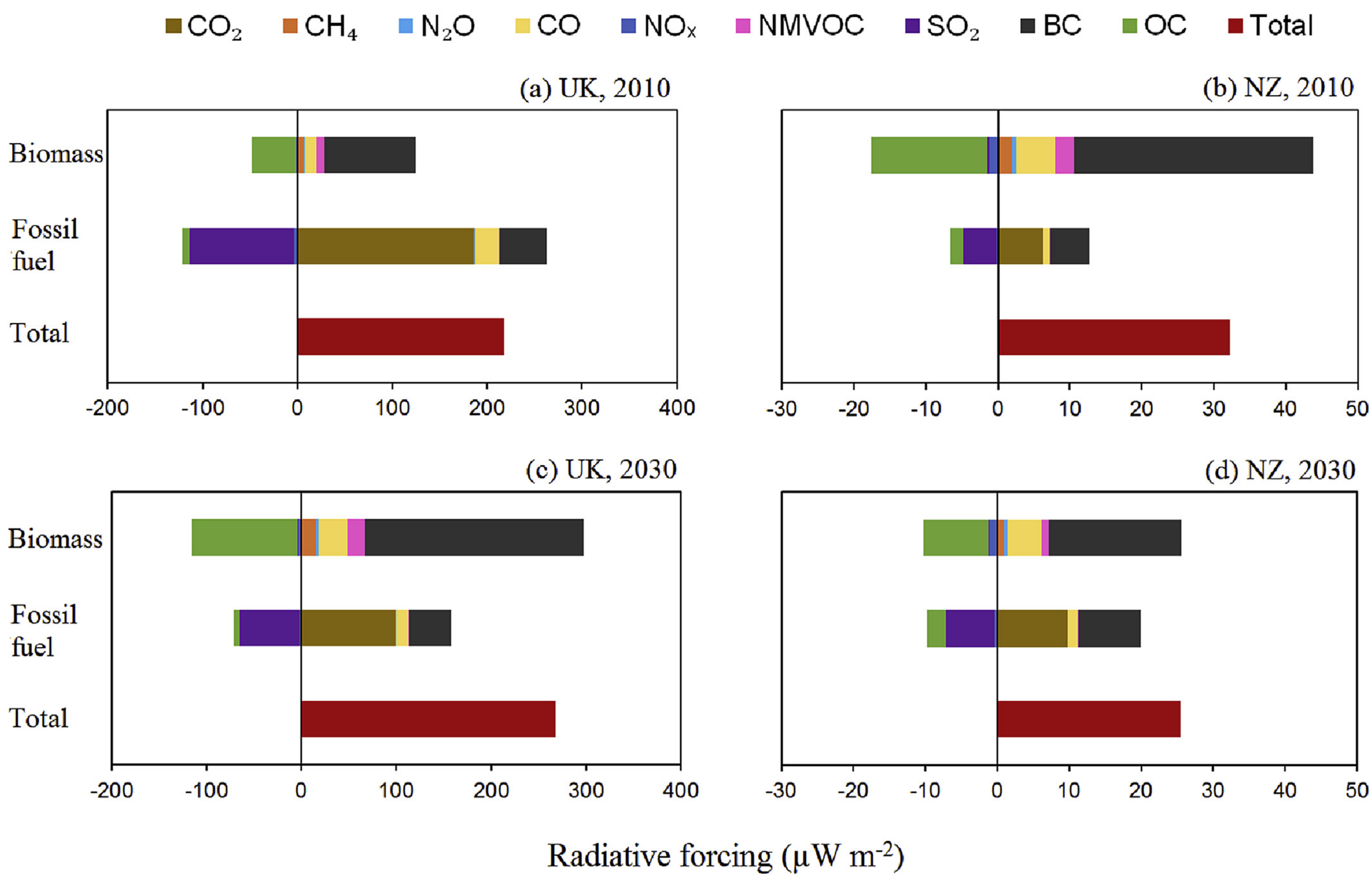

Fig. 7. Breakdown of radiative forcing due to biomass and fossil fuel RSF combustion in heating stoves and fireplaces in: (a) UK in 2010; (b) NZ in 2010; (c) UK in 2030; (d) NZ in 2030.

fossil fuel burning in 2010. By 2030, net forcing due to coal burning has increased by 40% relative to 2010, and is just 33% lower than that of biomass burning. Black carbon remains the most important forcing agent in both scenario years, accounting for 77% of the total warming effect of combined biomass and fossil fuel burning in 2010; and 76% in 2030. However, in the intervening years, forcing due to coal combustion exceeds that of biomass combustion by a factor of 2.4, due to a surge in coal consumption. This results in a slight increase in total net forcing (shown in red) in 2015, but an overall reducing trend across the modelling period. In the UK, total net forcing reduces rapidly from 1990 to 2005 but then increases at an average rate of 3.6 μW m^{-2} due to increased wood burning.

As discussed in Section 3.2, the net AGWP factors used to create Fig. 8 are central estimates and carry a substantial uncertainty. Error bars have not been included here because the uncertainties in global radiative forcing due to anthropogenic pollution are substantial and beyond the scope of this study (Bond et al., 2013). There are also errors associated with the activity data (up to factor of 3 for the UK according to recent survey results) and with the emissions factors used. For BC and PM_{10}, emissions factors for wood burning

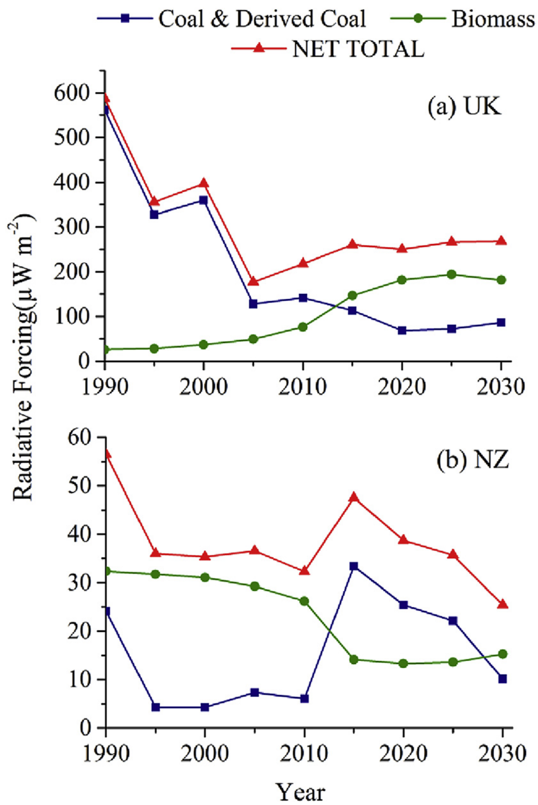

Fig. 8. Total climate forcing due to wood and coal combustion in heating stoves and fireplaces in the UK (a) and New Zealand (b).

stoves vary by ±30% between inventories (see Table 6). The combined uncertainties are substantial and hence values reported here should be treated as estimates.

4.3. A bottom-up emissions inventory calculation and comparison

A bottom-up approach was used in order to create emissions inventories for both countries, which can be compared with established inventories. In the UK, activity data for wood was derived from the DECC Wood Consumption Survey (DECC, 2016a). It found that the proportion of homes using wood for heating varies regionally. The proportion was lowest in London and the North East at 3.9% and 4.0% respectively, and highest in Northern Ireland and the South East at 18.4% and 15.8% respectively. The survey also asked wood users whether they used any additional fuels as well as wood. It found that the proportion of households using coal as well as wood was below 3% across much of the UK. The exception was in Northern Ireland where 10.1% of wood fuel users also used coal, which reflects the high consumption of mixed RSF in the region. Conversely, despite 15.8% of respondents in the South East using wood, just 1.7% of those also used coal; indicating that wood dominates the RSF mix. Activity data for coal and derived coal was derived from the DECC Sub-National Residual Fuel Consumption Statistics (DECC, 2015b). The results are shown in Fig. 9. It was found that coal consumption was highest in the East Midlands at 2.62 PJ and lowest in London at 0.22 PJ. Consumption of manufactured solid fuel (derived coal, smokeless fuel, briquettes etc) was also highest in the East Midlands at 1.98 PJ, closely followed by Yorkshire and the Humber at 1.93 PJ. Consumption in London was 0.25 PJ.

In New Zealand, activity data for both wood and coal was derived from the 2013 National Census (StatisticsNZ, 2015) using the methodology of the HAPINZ study (Kuschel et al., 2012). As shown in Fig. 10, the census data shows that the proportion of households using wood is far higher in New Zealand than in the UK. Over 90% of homes in many rural wards such as Taihape, Opuha and Glenmark use wood for heating. Coal consumption is much more dependent on location. The proportion of homes using coal for heating is below 5% across much of the country, particularly North Island. The proportion is highest in wards located in the west and south of South Island, including Northern Ward, Grey District (76%), Inangahua (69%) and Mataura (65%).

In order to produce an inventory, an in-depth review of RSF sector emissions factor inventories was carried out. Emissions factors applying to heating stoves and fireplaces were compared between the following inventories: the EMEP/EEA air pollutant emission inventory guidebook (EEA, 2013), U.S Environmental Protection Agency AP-42 (USEPA, 1995), GAINS (http://gains.iiasa.ac.at/models/), the IPCC emissions factor database (EFDP) (www.ipcc-nggip.iges.or.jp/EFDB/), and the UK National Air Emissions Inventory (NAEI) (http://naei.defra.gov.uk/). The results are shown in Table 6 for wood and coal. Inventories such as GAINS, the IPCC EFDP and NAEI offer emissions factors for other residential solid fuels such as charcoal, peat, anthracite, coke and lignite.

As the table shows, not all pollutants are accounted for in all inventories. The most extensive is the NAEI database, but these factors apply to the residential sector in general and are not technology specific. The most comprehensive fuel- and technology-specific factors were found to be those of the EMEP/EEA database and these were selected as the basis for the modelling work. EMEP/EEA emissions factors are largely consistent with other inventories. However, the PM_{10} emissions factor for wood burning in stoves in EMEP/EEA is 16% higher than in GAINS and 66% higher than in NAEI. Despite this, BC emissions are 26% lower than in GAINS for wood stoves and a factor of 4.5 lower than in GAINS for coal stoves. Also in comparison with GAINS, Table 6 shows that EMEP/EEA may overestimate emissions of cadmium, zinc and indeno[1,2,3-*cd*]pyrene from wood burning, as well as copper and total PAHs from coal burning. There may be an underestimate of emissions of arsenic, nickel, selenium and PCBs. In comparison to stoves, emissions factors for fireplaces are very similar for wood combustion in the EMEP/EEA inventory. However, for coal burning NO_x, SO_2, PM_{10}, cadmium, mercury, PAH and PCDD/F are lower for fireplaces than stoves. Furthermore GAINS does not provide emissions factors for coal burning in fireplaces, whereas EMEP/EEA does. It should be noted, however, that the EMEP/EEA factors apply to 'solid fuels other than biomass' and are not specific to a certain fuel type such as bituminous coal.

Factors for CO_2, CH_4, N_2O, OC and total PAH were not included in the EMEP/EEA inventory. The value for CO_2 was taken from the IPCC EFDP inventory. Methane emissions factors were taken from GAINS for wood burning and the NAEI for coal burning. N_2O and derived coal/MSF emissions factors were also taken from NAEI. Finally, BC and OC emissions factors were calculated from EMEP/EEA $PM_{2.5}$ emissions factors, applying the ratio of BC or OC to $PM_{2.5}$ as given in the GAINS database. Values for ΣPAH were taken from Lee et al. (2005).

Combining the activity data in Figs. 9 and 10 with the emissions factors in Table 6 yields the emission inventories for both countries. The results are presented for the UK in Fig. 11 and for New Zealand in Fig. 12. The totals for both countries are presented in Table 7.

In the UK, the results show that emissions are highly dependent on regional fuel consumption. Emissions of CO_2 and SO_2 are highest in regions with the highest fossil fuel combustion, including the North of England and Wales. All other emissions are highest in Northern Ireland and the South East, where wood fuel consumption in highest. Emissions remain consistently low in the North East,

Table 6
Summary of emissions factors applying to residential solid fuel combustion in stoves and fireplaces in five inventories, and those chosen for this study.

		EMEP				USEPA		GAINS			IPCC				NAEI		This study				
		Heating stoves		Fireplaces		Heating stove	Fireplace	Heating Stoves		Fireplaces	Heating stove	Fireplace	Residential		Residential		Heating stoves		Fireplaces		Residential
Pollutant	Unit	Wood	Coal	Wood	Coal	Wood	Wood	Wood	Coal	Wood	Wood	Wood	Wood	Coal	Wood	Coal	Wood	Coal	Wood	Coal	MSF
Carbon Dioxide	$g\ GJ^{-1}$						94,444		94,300				112,000	94,600		21,789		94,600		94,600	24,459
Methane	$g\ GJ^{-1}$					833		200	30	350	932		300	267−2650	205	476	200	476	350	476	149
Nitrogen oxides	$g\ GJ^{-1}$	50	100	50	60	78	72		80		120	110	100	100−180	49	108	50	100	50	60	113
Carbon Monoxide	$g\ GJ^{-1}$	4000	5000	4000	5000	6411	7017				10,000	11,000	5000	2000 −3600	2956	4249	4000	5000	4000	5000	3517
Non Methane VOC	$g\ GJ^{-1}$	600	600	600	600		6361	1600	300	1700			600	200	393	424	600	600	600	600	148
Sulphur Dioxide	$g\ GJ^{-1}$	11	900	11	500	11	11		726						6	785	11	900	11	500	1096
Nitrous Oxide	$g\ GJ^{-1}$						8		1.4			9	4	1.5	3	4	3	4	3	4	3
Ammonia	$g\ GJ^{-1}$	70	5	74	5				8						55	30	70	5	74	5	30
PM_{10}	$g\ GJ^{-1}$	760	450	840	330	850	961	655	455	720					458	281	760	450	840	330	56
$PM_{2.5}$	$g\ GJ^{-1}$	740	450	820	330			635	450	698					427	277	740	450	820	330	55
Black Smoke	$g\ GJ^{-1}$														56	1212					144
Black Carbon	$g\ GJ^{-1}$	74	29	57	32			100	130	86							117	130	101	95	5
Organic Carbon	$g\ GJ^{-1}$					1472	0.4	280	160	320							326	160	376	117	20
Lead	$mg\ GJ^{-1}$	27	100	27	100										51	86	27	100	27	100	76
Cadmium	$mg\ GJ^{-1}$	13.0	1.0	13.0	0.5	0.61									4.4	0.9	13.0	1.0	13.0	0.5	2
Mercury	$mg\ GJ^{-1}$	0.6	5.0	0.6	3.0										1.7	3.3	0.6	5.0	0.6	3.0	5
Arsenic	$mg\ GJ^{-1}$	0.2	1.5	0.2	1.5										1.7	14.2	0.2	1.5	0.2	1.5	16
Chromium	$mg\ GJ^{-1}$	23	10	23	10	0.03									50	27	23	10	23	10	38
Copper	$mg\ GJ^{-1}$	6	20	6	20										5.6	6.4	6	20	6	20	11
Nickel	$mg\ GJ^{-1}$	2	10	2	10	0.39									54	14	2	10	2	10	1263
Selenium	$mg\ GJ^{-1}$	0.5	2.0	0.5	1.0										5	13	0.5	2.0	0.5	1.0	19
Zinc	$mg\ GJ^{-1}$	512	200	512	200										69	75	512	200	512	200	89
Calcium	$mg\ GJ^{-1}$														525	15,856					
Tin	$mg\ GJ^{-1}$														7.5	4.2					417
Vanadium	$mg\ GJ^{-1}$														1.7	3.3					3303
Magnesium	$mg\ GJ^{-1}$														89	5145					
Sodium	$mg\ GJ^{-1}$														583	5064					
Beryllium	$mg\ GJ^{-1}$														0.36	40					4
Potassium	$mg\ GJ^{-1}$														2472	4250					
Manganese	$mg\ GJ^{-1}$					4.72															
Benzo[*a*]pyrene	$mg\ GJ^{-1}$	121	250	121	100	111									72	47	121	250	121	100	8.4
Benzo[*b*]fluoranthene	$mg\ GJ^{-1}$	111	400	111	170	167									83	2	111	400	111	170	0.4
Benzo[k]fluoranthene	$mg\ GJ^{-1}$	42	150	42	100	56									28	0.6	42	150	42	100	0.1
Indeno[123-*cd*]pyrene	$mg\ GJ^{-1}$	71	120	71	80										5	36	71	120	71	80	6.4
Benz[*a*]anthracene	$mg\ GJ^{-1}$					556									278	54					
Anthracene	$mg\ GJ^{-1}$					389									361	56					
Benzene	$mg\ GJ^{-1}$					54									14	19					
Benzo[ghi]perylene	$mg\ GJ^{-1}$					111									56	25					
Fluorene	$mg\ GJ^{-1}$					667									461	491					
Dibenz[ah]anthracene	$mg\ GJ^{-1}$														1	54					
Acenapthylene	$mg\ GJ^{-1}$					5889									4367	217					
Napthalene	$mg\ GJ^{-1}$					8000									5017	3738					
Pyrene	$mg\ GJ^{-1}$					667									406	90					
Phenanthrene	$mg\ GJ^{-1}$					2167									1356	199					
Acenapthene	$mg\ GJ^{-1}$					278									172	159					
Fluoranthene	$mg\ GJ^{-1}$					556									383	90					
Chrysene	$mg\ GJ^{-1}$					333									211	51					

(*continued on next page*)

Table 6 (*continued*)

Pollutant	Unit	EMEP				USEPA		GAINS			IPCC				NAEI		This study				
		Heating stoves		Fireplaces		Heating stove	Fireplace	Heating Stoves		Fireplaces	Heating stove	Fireplace	Residential		Residential		Heating stoves		Fireplaces		Residential
		Wood	Coal	Wood	Coal	Wood	Wood	Wood	Coal	Wood	Wood	Wood	Wood	Coal	Wood	Coal	Wood	Coal	Wood	Coal	MSF
Total PAH	mg GJ^{-1}																2389	7576	2389	7576	950
Polychlorinated biphenyls	μg GJ^{-1}	0.06	170	0.06	170										111	109	0.06	170	0.06	170	109
Dioxins	ng I-TEQ GJ^{-1}	800	1000	800	500										662	731	800	1000	800	500	742
Hexachlorobenzene	μg GJ^{-1}	5	0.62	5	0.62										4	1	5	0.62	5	0.62	
Hydrogen Chloride	g GJ^{-1}															71					

where consumption of RSF is low across all fuel types. The national totals for activity data and emissions in Table 7 may be compared with the GAINS estimates in Table 5. It can be seen that wood consumption in stoves is within 30% of the GAINS inventory estimate. However, wood consumption in fireplaces is higher by more than 30 PJ compared to GAINS. Combined fossil fuel consumption is 31.49 PJ, more than twice the GAINS estimate. The higher activity data also corresponds to higher emissions. For biomass, the majority of emissions are higher by a factor of 2–3. The exceptions are NH_3 and SO_2 which are significantly higher than in GAINS, and NMVOCs which are within 8% of the GAINS estimate. For fossil fuel, there is a greater differences between the two inventories. The majority of emissions estimates are higher by a factor of 2–6 than in GAINS. The exceptions are CH_4 and OC emissions which are significantly higher. This is because the CH_4 emissions factor for coal stoves in the NAEI is 476 g GJ^{-1} versus 30 g GJ^{-1} in GAINS.

In New Zealand, regional fuel consumption also has a large impact on emissions. CO_2 and SO_2 emissions are far higher in South Island than in North Island, particularly in Greymouth, Grey District. Emissions from wood burning are more uniformly distributed across the country, and are strongly correlated to the larger population areas. Emissions of CH_4, NMVOCs, CO, particulate matter, BC and OC are consistently high in wards such as Rotorua, Nelson and Waitakere ward which includes the Auckland suburban areas of Waitakere and Henderson. Emissions are also highest in the wards which include Invercargill and Dunedin, where BC emissions over 100 tonnes $year^{-1}$ have been calculated. This corresponds to annual BC emissions of 5.6 kg $dwelling^{-1}$ and 3.8 kg $dwelling^{-1}$ in the two wards respectively. Comparing activity data, the results show that fossil fuel consumption in the GAINS model is within 24% of the calculated consumption. However calculated national wood consumption is higher than the GAINS estimate by a factor of 7.9. This has a significant impact of total national emissions. Calculated emissions from fossil fuel combustion are in the most part higher by a factor of 1–2 than in GAINS, except for NMVOCs and N_2O which are higher by a factor of 5.8 and 4.0 respectively. Calculated emissions from biomass burning range from 4.8 times higher for NMVOCs to 67.9% higher for ammonia. Importantly, black carbon emissions are 13.5 times higher, which has significant implications for climate.

5. Discussion and implications for the UK

Analysis of HAPINZ data (Kuschel et al., 2012) found that the contribution of domestic heating to wintertime PM_{10} concentrations was highest in Alexandra, Arrowtown and Milton at 45–50 μg m^{-3}; up to 2.5 times higher than the WHO recommended annual mean. Calculated average winter BC concentrations were also highest in these areas, peaking at 10 μg m^{-3}. The nationwide average was 1.8 μg m^{-3} and typically 4–7 μg m^{-3} in urban/suburban areas which is typical of the winter concentrations reported by the studies shown in Table 4. These concentrations are comparable with those in highly polluted regions of India and Asia, which have resulted in localised radiative forcing over urban areas of up to 23 W m^{-2} (Panicker et al., 2010; Peng et al., 2016). It is therefore recommended that a full radiative transfer modelling exercise be carried out over urban areas in New Zealand in order to fully understand the climate impacts of wood burning stoves.

Emissions of NVMOCs, BC, OC and particulate matter are highly dominated by heating stoves because of the lower efficiency of combustion. This is in agreement with Denier Van Der Gon et al. (2015) who found the residential wood combustion is the largest source of organic aerosols in Europe. Lower combustion temperatures and larger fuel particle size promote pyrolysis conditions which are conducive to higher emissions of organics (Williams

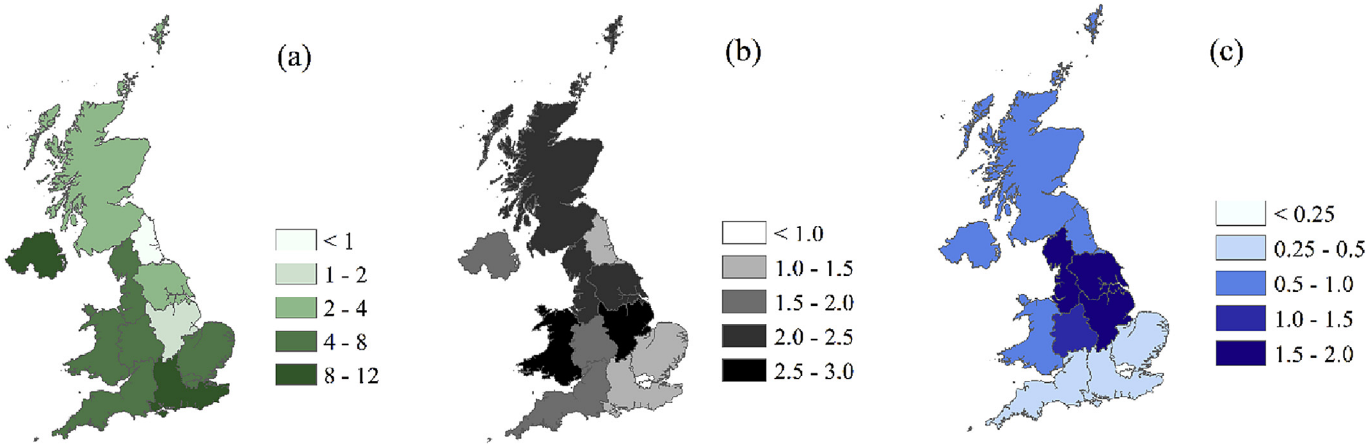

Fig. 9. UK activity data (PJ) for (a) wood; (b) coal and (c) manufactured solid fuel.

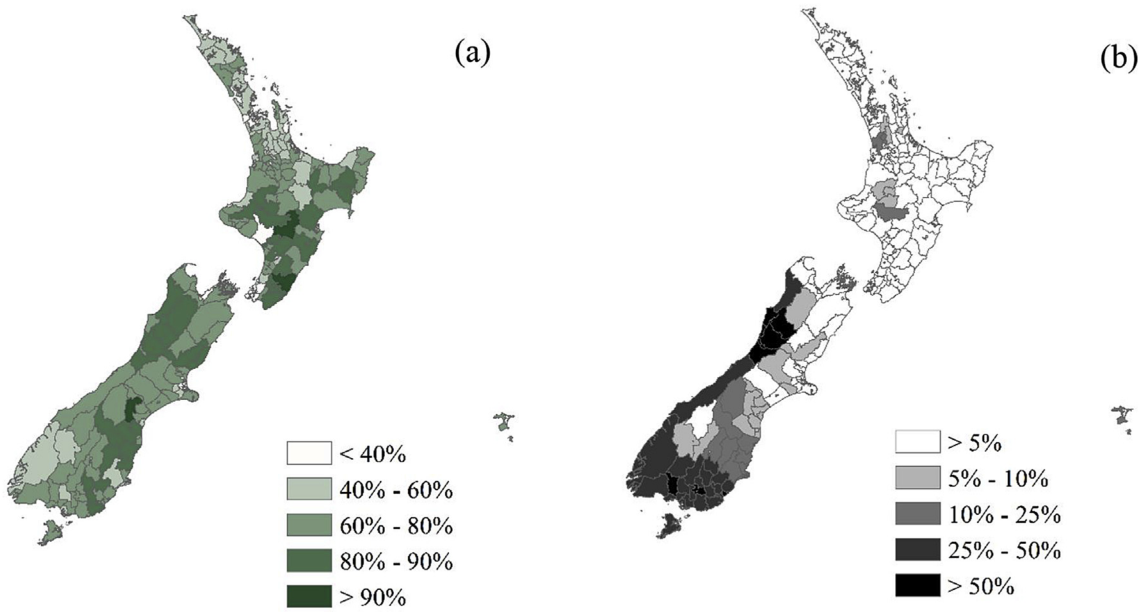

Fig. 10. Proportion of households in New Zealand using a) wood; and b) coal; in 2013. Data Source: (StatisticsNZ, 2015).

et al., 2012; Jones et al., 2014). The NVMOC emissions factor for coal combustion in heating stoves (300 g GJ^{-1}) is more than a factor of 5 lower than for wood (1600 g GJ^{-1}) in the GAINS database. In contrast, the factor is the same (600 g GJ^{-1}) for both wood and coal combustion in the EMEP/EEA database, and very similar in the NAEI database. Specific NO_x emissions factors by technology were not available in GAINS but the factor for biomass in the general residential sector is almost half that of coal, as shown in Table 3. NO_x emissions are influenced by the nitrogen content of the fuel (Mitchell et al., 2016) and the temperature of combustion (Jones et al., 2014). The same is true of SO_x emissions. Fuel-bound sulphur is typically very low in wood and biomass fuels, but can be as high as 2% in manufactured solid fuel (Van Loo and Koppejan, 2007). However, the use of binders or additives such as calcium carbonate during the production of MSF briquettes can help retain a proportion of the sulphur in the ash. Fig. 6f shows that lignite contributes to SO_2 emissions, particularly between 1995 and 2000. The GAINS emissions factor for lignite in heating stoves is 558 t PJ^{-1} versus 616 t PJ^{-1} for hard coal, which is consistent with the relative sulphur contents reported by Beamish et al. (2001). New Zealand has several billion tonnes of lignite resources in the Southland and Otago regions which may contribute to RSF emissions in the future.

Emissions of PM_{10} and $PM_{2.5}$ increase substantially from 2005 to 2030 in the UK, largely due to the increase in wood burning. The PM_{10} emissions factor for wood burning in heating stoves is 44% higher than that of coal burning in the GAINS database. This is corroborated by the EMEP/EEA and NAEI databases which find PM_{10} emissions from wood burning are 64% and 63% higher respectively than coal, on an energy basis. However, PM_{10} emissions are higher for coal on a mass basis. For example, the NAEI reports emissions factors of 9.3 g kg^{-1} and 8.2 g kg^{-1} for coal and wood respectively. This is in good agreement with Coulson et al. (2015) who found emissions factors from in-situ wood stoves exhibit a log-normal distribution with a mean of 9.8 g kg^{-1} (±2.4 g kg^{-1}). The 95% confidence interval for PM_{10} emissions from conventional heating stoves burning wood and similar wood waste in the EMEP/EEA database is 6.8–27.3 g kg^{-1} (380–1520 g GJ^{-1}) with a mean of 13.7 g kg^{-1}. The range of the 95% confidence interval

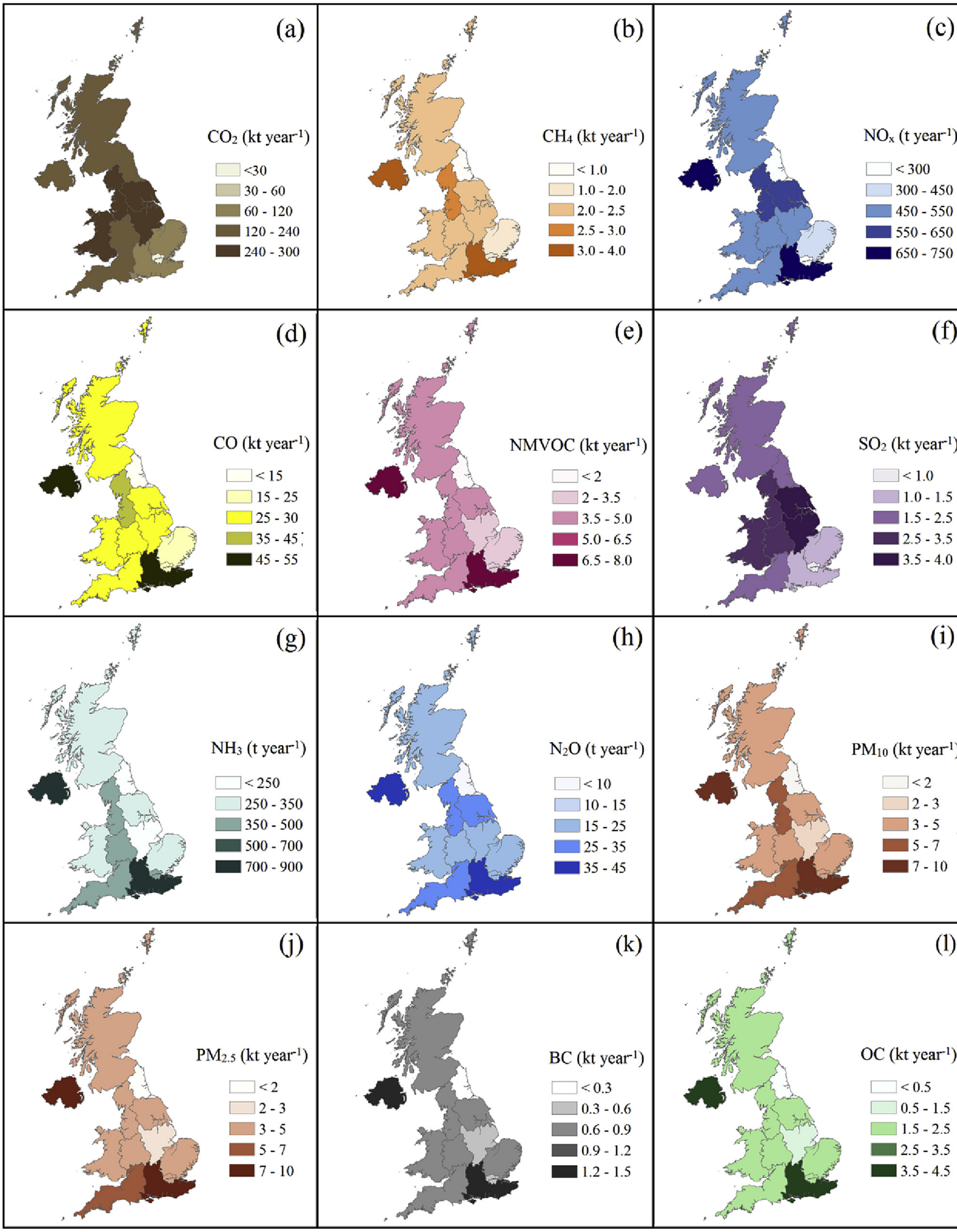

Fig. 11. Distribution of emissions from stoves and fireplaces in the UK in 2014.

is lower for fossil fuel at 7.5–15.8 g kg^{-1}. The HAPINZ study used factors of 8 g kg^{-1} for wood and 25 g kg^{-1} for coal (Kuschel et al., 2012). The most important component of particulate matter for climate change is black carbon and this is presented at a percentage

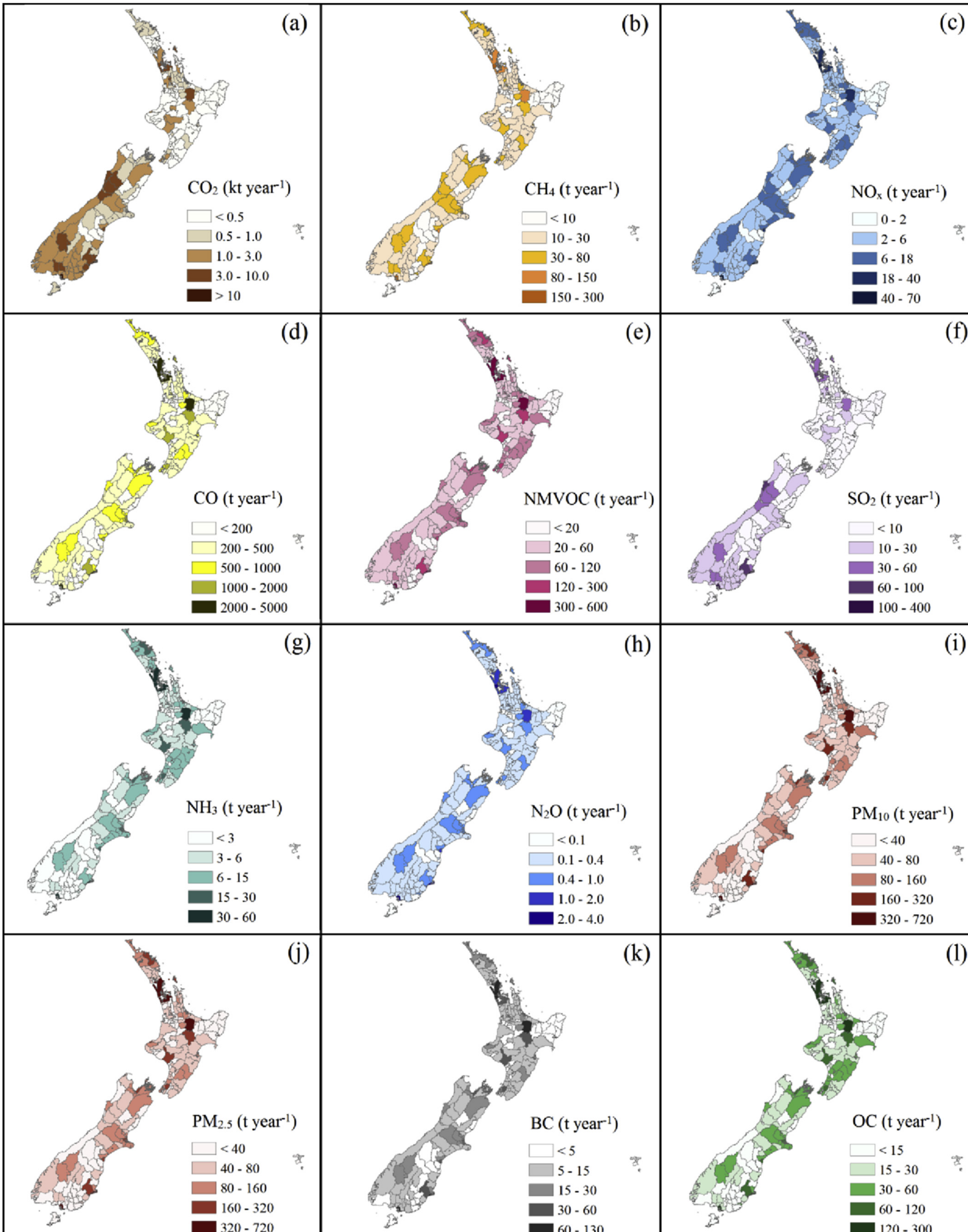

Fig. 12. Distribution of emissions from stoves and fireplaces in New Zealand in 2013.

Table 7
Pollutant emissions inventory for RSF combustion in the United Kingdom and New Zealand, 2013/14.

		UK, 2013/14					NZ, 2013	
		Wood		Coal		MSF	Wood	Coal
Parameter	Unit	Fireplaces	Stoves	Fireplaces	Stoves	Stoves & Fireplaces	Stoves & Fireplaces	Stoves & Fireplaces
Activity data	PJ	32.80	29.51	10.79	9.71	10.99	26.72	4.34
CO_2	kt year^{-1}			1021	918	269		415
CH_4	kt year^{-1}	11.5	5.9	5.1	4.6	1.6	5.3	0.1
NO_x	kt year^{-1}	1.6	1.5	0.6	1.0	1.2	1.3	0.4
CO	kt year^{-1}	131.2	118.0	53.9	48.5	38.6	106.4	21.9
NMVOC	kt year^{-1}	19.7	17.7	6.5	5.8	1.6	16.0	2.6
SO_2	kt year^{-1}	0.4	0.3	5.4	8.7	12.0	0.3	3.9
NH_3	kt year^{-1}	2.4	2.1	0.05	0.05	0.3	1.9	0.02
N_2O	kt year^{-1}	0.10	0.09	0.04	0.04	0.03	0.08	0.02
PM_{10}	kt year^{-1}	27.6	22.4	3.6	4.4	0.6	20.2	2.0
$PM_{2.5}$	kt year^{-1}	26.9	21.8	3.6	4.4	0.6	19.7	2.0
BC	kt year^{-1}	3.3	3.4	1.0	1.3	0.05	3.1	0.6
OC	kt year^{-1}	12.3	9.6	1.3	1.6	0.2	8.7	0.7
Lead	t year^{-1}	0.89	0.80	1.08	0.97	0.84	0.72	0.44
Cadmium	t year^{-1}	0.43	0.38	0.01	0.01	0.02	0.35	0.004
Mercury	t year^{-1}	0.02	0.02	0.03	0.05	0.05	0.01	0.02
Arsenic	t year^{-1}	0.01	0.01	0.02	0.01	0.18	0.01	0.01
Chromium	t year^{-1}	0.75	0.68	0.11	0.10	0.42	0.61	0.04
Copper	t year^{-1}	0.20	0.18	0.22	0.19	0.12	0.16	0.09
Nickel	t year^{-1}	0.07	0.06	0.11	0.10	13.88	0.05	0.04
Selenium	t year^{-1}	0.02	0.01	0.01	0.02	0.21	0.01	0.01
Zinc	t year^{-1}	16.79	15.11	2.16	1.94	0.98	13.62	0.88
B[*a*]P	t year^{-1}	3.97	3.57	1.08	2.43	0.09	3.22	1.10
B[*b*]F	t year^{-1}	3.64	3.28	1.83	3.88	0.004	2.95	1.76
B[k]F	t year^{-1}	1.38	1.24	1.08	1.46	0.001	1.12	0.66
I[123-*cd*]P	t year^{-1}	2.33	2.09	0.86	1.16	0.07	1.89	0.53
ΣPAHs	t year^{-1}	78.4	70.5	81.7	73.5	10.4	63.5	33.2
PCB	g year^{-1}	2.0	1.8	1834	1650	1199	1.6	746.1
Dioxins	g I-TEQ year^{-1}	26.2	23.6	5.4	9.7	8.1	21.3	4.4
HCB	g year^{-1}	164.0	147.5	6.7	6.0		133.0	2.7

B[*a*]P: Benzo[*a*]pyrene; B[*b*]F: Benzo[*b*]fluoranthene; B(k)F Benzo[k]fluoranthene; I[123-*cd*]P: Indeno[123-*cd*]pyrene.

in the EMEP/EEA database. The 95% confidence interval is 2–20% for wood (average 10%) and 2–26% for coal (average 6%). In comparison, fractions reported in GAINS are 16% for wood and 29% for coal. Analysis of several studies by Winther and Nielsen (2011) found the BC fraction to vary from 10% in wood fireplaces to 15% in wood stoves and 35% in wood boilers. The fraction was much higher for coal at 45%.

The results show that the net impact on climate of heating stoves and fireplaces in both the UK and New Zealand is strongly warming, and black carbon is the most important component of radiative forcing, particularly where consumption of wood exceeds that of coal. A comparison of the BC emissions reported here is made with several international climate models, and is shown in Fig. 13. The figure also shows projected emissions under different scenarios from RSF combustion until the year 2100. The suffix _calc denotes that BC has been calculated from PM_{10} data. In the UK, most scenarios predict a gradual reduction in BC emissions over the period. However, the GAINS and NAEI data show that after 2004 there has been a significant increase in BC emissions, which will continue until 2025. In New Zealand, all model scenarios suggest a large reduction in BC emissions from 2010 onwards. The BC emissions estimate of this study is approximately 40% higher than the highest estimate made by the PEGASOS model, but significantly higher than all other models. The BC emissions factors used here are similar to that of the GAINS database so it is most likely the activity data which carries the largest uncertainty.

Fig. 13a also shows that the calculated UK BC emissions are approximately three times higher than most climate models predict. This is in agreement with the findings of the recent DECC Domestic Wood Use Survey, which found that DUKES has previously underestimated wood consumption by a factor of three (DECC, 2016a). Denier Van Der Gon et al. (2015) also found that previous inventories in Europe underestimated emissions from wood RSF by a factor of 2–3. If BC emissions were to increase at the same rate as $PM_{2.5}$, as given in the NAEI inventory between 2005 and 2013, then emissions would be over 6.7 kt year^{-1} by 2030; an increase of 84% on 2013 emissions. In context, emissions from passenger cars (UNFCCC section 1.A.3.b.i) were 1.7 kt in 2015, reducing to 0.4 kt in 2030 according to GAINS (ECLIPSE version 5, CLE scenario). The GAINS model predicts a reduction in BC emissions across most UNFCCC sectors, but an increase in the residential sector (section 1.A.4.b.i). In fact, by 2030 the residential sector accounts for 44% of total BC emissions and 40% of total OC emissions across all sectors in the UK. This is comparable to Denmark, where residential wood combustion is prevalent (Winther and Nielsen, 2011). The high contribution of RSF to BC and OC is largely due to increased use of wood in heating stoves as shown in Fig. 4. The contribution of other technologies in the residential sector to BC, OC and total $PM_{2.5}$ is comparatively low, as shown in Fig. 14a. In 2025, heating stoves and fireplaces account for 77% of BC emissions, 90% of OC emissions, and 85% of total residential sector $PM_{2.5}$ emissions. This is a result of lower combustion efficiencies, lower MCE and higher emissions factors for small scale biomass technologies. However, larger technologies such as single house biomass boilers (<50 kW) and commercial biomass boilers (<50 MW) make a larger contribution to NO_x emissions due to higher combustion temperatures and formation of thermal NO_x (Williams et al., 2012). As shown in Fig. 14b, heating stoves and

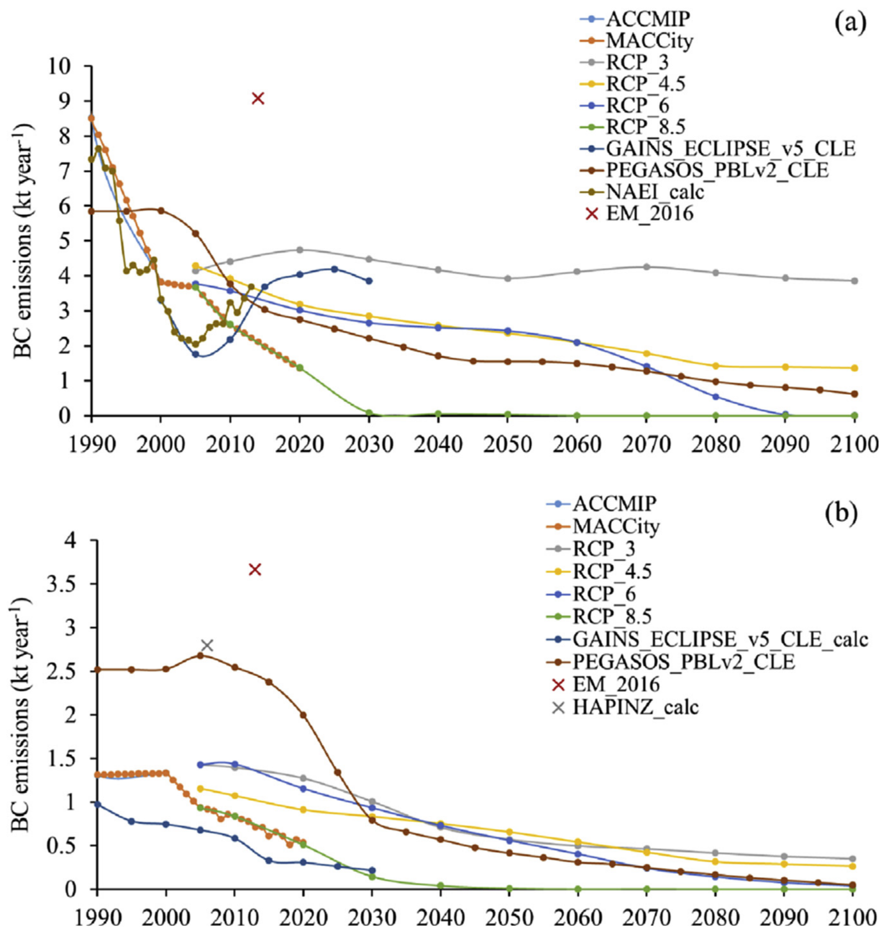

Fig. 13. Comparison of model predictions of BC emissions from the residential sector in (a) the UK; and (b) New Zealand; for the years 1990–2100.

fireplaces account for just 42% of NO_x emissions in 2025.

As discussed in Section 2.2, there is good comparability between residential heating sectors in New Zealand and the UK in terms of fuel poverty and energy efficiency of homes. However, space heating accounts for a greater proportion of residential energy consumption in the UK than New Zealand. Both average wood consumption per household, and average wood consumption per day are twice as high in New Zealand as in the UK. This may be linked to the limited availability or higher cost of alternative heating fuels, particularly as New Zealand has a large domestic supply of wood, whereas the UK does not and may rely on wood imports in the future. In addition, the climates of the two countries are comparable, but distinct. The latitude of New Zealand ranges from 34° to 47° South, whereas mainland UK covers 50°–58° North. Being closer to the equator, the far north of New Zealand has a sub-tropical climate and typical winter daytime maximum air temperatures are 12–17 °C. The South Island is generally cooler and more mountainous, with maximum winter daytime temperatures of 5–12 °C. Average winter daily maximum temperatures in the UK are similar but generally lower, ranging from 5 to 7 °C in northern Scotland to 7–10 °C in southern England. Both countries also commonly experience smog episodes during winter anticyclones and atmospheric temperature inversions (Kossmann and Sturman, 2004; Milionis and Davies, 2008). Such events are typically correlated with lower temperatures and higher emissions from home heating.

The UK also has 60 million more inhabitants and 26 million more homes than New Zealand, and currently 7.5% of UK households burn wood compared to >50% of NZ households (see Section 2.1). Due to the higher density of housing, small increases in emissions may have a greater impact in the UK. For example, a 1% increase in the number of UK homes burning wood would lead to over 30,000 extra tonnes of wood (dry basis) being burned per year, assuming the factors given in Table 1.

6. Policy implications

A high degree of uncertainty remains in RSF activity data estimates, due to inherent difficulties in monitoring this highly variable emissions source. Bottom-up inventories using the latest census, survey and sales data hold the potential to reduce uncertainty.

6.1. Implications for air quality and health

Biomass burning stoves and boilers have the potential to significantly reduce greenhouse gas (mainly CO_2) emissions from the residential sector, but care must be taken to ensure that this is not done at the detriment of air quality, particularly in the winter time. The UK is facing a number of legal challenges over European air quality breaches. Hence an increase in residential wood burning could impede efforts to reduce national emissions of NO_x, NMVOCs, NH_3, $PM_{2.5}$ and CH_4 through planned revisions to the National Emission Ceilings (NEC) Directive 2001/81/EC. The improvement of emissions inventories for residential wood burning was identified as one of the key areas for improvement in receptor modelling studies and "*substantially more information*" is needed in this area

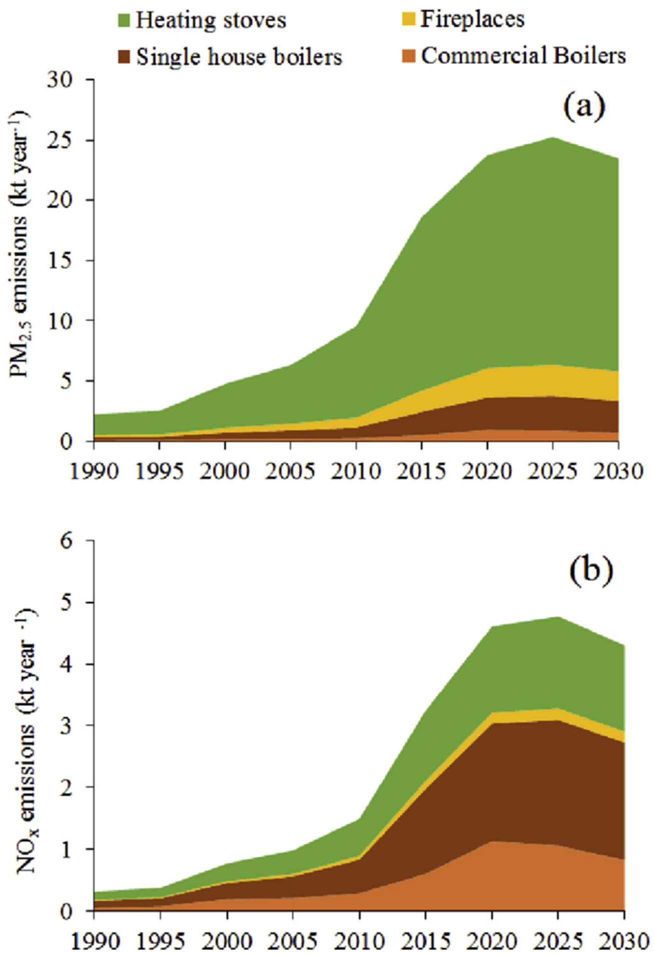

Fig. 14. Breakdown of UK residential sector emissions from wood combustion by technology for (a) $PM_{2.5}$ and (b) NO_x, according to the GAINS model, 1990–2030.

"*before abatement policies can be formulated*" (AQEG, 2012).

Although a range of low-emission appliances are available through the RHI, uptake remains low, particularly where there is an option to install a cheaper more traditional wood burning stove. The Ecodesign Regulations in Europe have the potential to increase uptake of such appliances and significantly reduce emissions in the future. The regulations also help to minimise variation between standard test methods across Europe, but significant differences remain internationally such as in standard fuels and sampling methods. Before Ecodesign is implemented, voluntary eco-labelling of new appliances such as Flamme Verte (France), Nordic Swan (Scandinavia) and Burnwise (NSPS, USA) may help to reduce emissions. If emissions from older appliances are to be reduced without replacement, then policies may target fuel switching to pellets/briquettes or pretreated fuels (torrefied biomass or washed wood), as well as 'No Burn Days' and retrofitting of abatement technologies.

6.2. *Implications for climate change*

As described in Section 2.1, the UK must achieve targets of 12% renewable heat by 2020, 15% total renewables by 2020, and 80% emissions reductions by 2050. In order to achieve this, the Committee on Climate Change (CCC) has developed a series of quinquennial 'carbon budgets' with specific targets enshrined into law. The fifth carbon budget (2015–2035) sets a target of installing 400,000 extra biomass boilers for space heating (not including district heating), equating to 36 PJ and GHG savings of 1.3 $MtCO_2$-equivalent. Current policy incentivising residential biomass uptake explicitly targets biomass boilers (CCC targets and RHI policy) and there is little or no support for stoves. This is because heat generated must be metered in order to be eligible for RHI payments. A coinciding benefit is that boilers tend to have lower emissions factors than stoves and must meet RHI emissions and efficiency criteria. Consumption of wood pellets is also more easily audited than wood logs, where there is a large 'grey' or informal market consisting of self-sourced fuel and waste wood (Bitterman and Suvorov, 2012). However, the DECC Domestic Wood Consumption Survey and subsequent revisions to DUKES highlight the importance of small scale unmetered residential wood combustion (RWC) in the renewable energy mix, as shown in Table 8.

The revisions mean that the UK moves from level 3 (RWC <10% renewables) to level 2 (RWC 10–30% renewables), according to European 20-20-20 reporting standards (Bitterman and Suvorov, 2012). As a result it is recommended that the UK conduct a RWC survey every 3–4 years instead of 5–10 years and errors in the reporting should be ±10% rather than ±30%.

7. Conclusions

Here we present one of the first detailed inventories of black carbon concentrations from RSF combustion in New Zealand. Concentrations were higher than 10 $\mu g\ m^{-3}$ in some suburban areas of Christchurch, Dunedin, and Nelson. In comparison, BC concentrations due to wood burning in London are estimated to be 0.17–0.33 $\mu g\ m^{-3}$ (see Section 2.1). This has significant implications for air quality and climate and serves as an example of the BC concentrations that can be expected in similar sized UK towns and cities, should RSF use in stoves and fireplaces continue to increase without emissions controls. As is the case in New Zealand, residential wood combustion (RWC) may become the largest source of ambient wintertime PM_{10} and BC in the UK. Model predictions show a 14-fold increase in the consumption of wood in the UK residential sector between 1990 and 2030 and heating stoves alone account for 40–55% of this. As a result, emissions of CH_4, NMVOCs, PM_{10}, $PM_{2.5}$ and OC increase significantly and total net radiative forcing increases by 23% between 2010 and 2030. Due to the reduction in coal use and the increase in wood use, black carbon surpasses carbon dioxide to become the most important component of RSF radiative forcing, with wood burning BC alone accounting for over 50% of the total positive radiative forcing in 2030.

A unique bottom-up emissions inventory was produced for both countries using the latest census data for New Zealand and survey

Table 8
Revised contributions of domestic wood combustion to renewable heat and total renewable energy generation in the UK. Data source: DUKES 2016 Chapter 6, table 6.6, (DECC, 2016a).

	DUKES 2014 (year 2013)	DUKES 2016 (year 2013)	DUKES 2016 (year 2015)
Renewable heat	35%	63%	54%
Total renewable energy	5.4%	14.2%	10.7%

data for the UK. One recommendation from New Zealand is that conducting a survey of fuels used for home heating every 3–5 years helps to reduce uncertainty in activity data which is important for renewable energy targets, emissions inventories and air quality and climate models. Activity data was multiplied by emissions factors derived from a critical analysis of 5 inventories, which highlighted the uncertainty in emissions factors in this subcategory. In order to reduce uncertainty in emissions factors, it is recommended that standard test methods be modified to replicate real-world emissions, and in-situ testing be carried out as has been done in New Zealand. More than ten years of research has been conducted on RSF emissions and associated air quality impacts in New Zealand, whereas UK research has largely focussed on other sectors. The relative success of imposing additional emissions limits on wood burners has also been demonstrated, such as in Nelson where PM_{10} and BC are reducing (see Section 2.3). In terms of BC, OC and climate, a deeper understanding of the impact of 'brown' fraction of organic carbon is required, as well as the impact of high SOA formation from aged wood smoke.

Acknowledgements

The authors wish to thank Liz Waters (DECC), Perry Davy (GNS Science), Jes Sig Andersen (DTI) and Dennis Milligan (SIA) for providing data and advice for this work. We also wish to thank Dr Jonathan Carrivick for making the project possible. The work has been funded by the EPSRC Doctoral Training Centre in Low Carbon Technologies (EP/G036608) and we are grateful to the Supergen Bioenergy Hub (EP/J017302) for further contributions.

References

Coulson, G., Wilton, E., Somervell, E., Longley, I. 2013. 10 Years of Research on Woodstove Emissions in New Zealand – a Review. In: Proceedings of the 16th IUPPA World Clean Air Congress, Cape Town, SA, October 2013.

Aas, W., Tsyro, S., Bieber, E., Bergström, R., Ceburnis, D., Ellermann, T., Fagerli, H., Frölich, M., Gehrig, R., Makkonen, U., Nemitz, E., Otjes, R., Perez, N., Perrino, C., Prévôt, A.S.H., Putaud, J.P., Simpson, D., Spindler, G., Vana, M., Yttri, K.E., 2012. Lessons learnt from the first EMEP intensive measurement periods. Atmos. Chem. Phys. 12, 8073–8094.

Adams, P.W., Hammond, G.P., Mcmanus, M.C., Mezzullo, W.G., 2011. Barriers to and drivers for UK bioenergy development. Renew. Sustain. Energy Rev. 15, 1217–1227.

Allan, J.D., Williams, P.I., Morgan, W.T., Martin, C.L., Flynn, M.J., Lee, J., Nemitz, E., Phillips, G.J., Gallagher, M.W., Coe, H., 2010. Contributions from transport, solid fuel burning and cooking to primary organic aerosols in two UK cities. Atmos. Chem. Phys. 10, 647–668.

Ancelet, T., Davy, P., Trompetter, W., Markwitz, A., Weatherburn, D., 2010. A comparison of particulate and particle-phase PAH emissions from a modern wood burner with those of an old wood burner. Air Qual. Clim. Change 44, 21.

Ancelet, T., Davy, P.K., Mitchell, T., Trompetter, W.J., Markwitz, A., Weatherburn, D.C., 2012. Identification of particulate matter sources on an hourly time-scale in a wood burning community. Environ. Sci. Technol. 46, 4767–4774.

Ancelet, T., Davy, P.K., Trompetter, W.J., Markwitz, A., Weatherburn, D.C., 2014. Particulate matter sources on an hourly timescale in a rural community during the winter. J. Air Waste Manag. Assoc. 64, 501–508.

Ancelet, T., Davy, P.K., Trompetter, W.J., 2015. Particulate matter sources and long-term trends in a small New Zealand city. Atmos. Pollut. Res. 6, 1105–1112.

AQEG, 2012. Fine Particulate Matter (PM2.5) in the United Kingdom. Air Quality Expert Group (AQEG) to the Department for Environment, Food and Rural Affairs (DEFRA).

BANZ, 2010. New Zealand Bioenergy Strategy. The Bioenergy Association of New Zealand.

Beamish, B.B., Barakat, M.A., St. George, J.D., 2001. Spontaneous-combustion propensity of New Zealand coals under adiabatic conditions. Int. J. Coal Geol. 45, 217–224.

Beerepoot, M., Marmion, A., 2012. Policies for Renewable Heat: an Integrated Approach. OECD/IEA, Paris.

Bitterman, W., Suvorov, M., 2012. Quality Standard for Statistics on Wood Fuel Consumption of Households Concerted Action to Support the Implementation of the RES Directive 2009/28/EC (CA-RES).

Bond, T.C., Streets, D.G., Yarber, K.F., Nelson, S.M., Woo, J.-H., Klimont, Z., 2004. A technology-based global inventory of black and organic carbon emissions from combustion. J. Geophys. Res. Atmos. 109, D14203.

Bond, T.C., Doherty, S.J., Fahey, D.W., Forster, P.M., Berntsen, T., Deangelo, B.J., Flanner, M.G., Ghan, S., Kärcher, B., Koch, D., Kinne, S., Kondo, Y., Quinn, P.K., Sarofim, M.C., Schultz, M.G., Schulz, M., Venkataraman, C., Zhang, H., Zhang, S., Bellouin, N., Guttikunda, S.K., Hopke, P.K., Jacobson, M.Z., Kaiser, J.W., Klimont, Z., Lohmann, U., Schwarz, J.P., Shindell, D., Storelvmo, T., Warren, S.G., Zender, C.S., 2013. Bounding the role of black carbon in the climate system: a scientific assessment. J. Geophys. Res. Atmos. 118, 5380–5552.

Bonjour, S., Adair-Rohani, H., Wolf, J., Bruce, N.G., Mehta, S., Pruss-Ustun, A., Lahiff, M., Rehfuess, E.A., Mishra, V., Smith, K.R., 2013. Solid fuel use for household cooking: country and regional estimates for 1980-2010. Environ. Health Perspect. 121, 784–790.

Boucher, O., Randall, D., Artaxo, P., Bretherton, C., Feingold, G., Forster, P., Kerminen, V.-M., Kondo, Y., Liao, H., Lohmann, U., Rasch, P., Satheesh, S.K., Sherwood, S., Stevens, B., Zhang, X.Y., 2013. Clouds and aerosols. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Climate Change 2013: the Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Bruns, E.A., Krapf, M., Orasche, J., Huang, Y., Zimmermann, R., Drinovec, L., Močnik, G., El-Haddad, I., Slowik, J.G., Dommen, J., Baltensperger, U., Prévôt, A.S.H., 2015. Characterization of primary and secondary wood combustion products generated under different burner loads. Atmos. Chem. Phys. 15, 2825–2841.

Bruns, E.A., El Haddad, I., Slowik, J.G., Kilic, D., Klein, F., Baltensperger, U., Prévôt, A.S.H., 2016. Identification of significant precursor gases of secondary organic aerosols from residential wood combustion. Sci. Rep. 6, 27881.

Butt, E.W., Rap, A., Schmidt, A., Scott, C.E., Pringle, K.J., Reddington, C.L., Richards, N.A.D., Woodhouse, M.T., Ramirez-Villegas, J., Yang, H., Vakkari, V., Stone, E.A., Rupakheti, M.S., Praveen, P.G., Van Zyl, P.P., Beukes, J., Josipovic, M., Mitchell, E.J.S., Sallu, S.M., Forster, P.M., Spracklen, D.V., 2016. The impact of residential combustion emissions on atmospheric aerosol, human health, and climate. Atmos. Chem. Phys. 16, 873–905.

Carbon Trust, 2012. Biomass Heat Accelerator: Overview and Summary of Output. The Carbon Trust, London, UK.

Chafe, Z., Brauer, M., Héroux, M., Klimont, Z., Lanki, T., Salonen, R.O., Smith, K.R., 2015. Residential Heating with Wood and Coal: Health Impacts and Policy Options in Europe and North America. Denmark, World Health Organisation (WHO), Copenhagen.

Coulson, G., Bian, R., Somervell, E., 2015. An investigation of the variability of particulate emissions from woodstoves in New Zealand. Aerosol Air Qual. Res. 15, 2346–2356.

Crilley, L.R., Bloss, W.J., Yin, J., Beddows, D.C.S., Harrison, R.M., Allan, J.D., Young, D.E., Flynn, M., Williams, P., Zotter, P., Prevot, A.S.H., Heal, M.R., Barlow, J.F., Halios, C.H., Lee, J.D., Szidat, S., Mohr, C., 2015. Sources and contributions of wood smoke during winter in London: assessing local and regional influences. Atmos. Chem. Phys. 15, 3149–3171.

Davy, P.K., Ancelet, T., Trompetter, W.J., Markwitz, A., Weatherburn, D.C., 2012. Composition and source contributions of air particulate matter pollution in a New Zealand suburban town. Atmos. Pollut. Res. 3, 143–147.

DCLG, 2016. Live Tables on Dwelling Stock (Including Vacants). Department for Communities and Local Government, London, UK.

DECC, 2012a. The Future of Heating: a Strategic Framework for Low Carbon Heat in the UK. Department of Energy and Climate Change, London, UK.

DECC, 2012b. UK Bioenergy Strategy. Department of Energy & Climate Change, London, UK.

DECC, 2015a. Digest of United Kingdom Energy Statistics (DUKES). Energy Consumption by Final User (Energy Supplied Basis), 1970 to 2014 (DUKES 1.1.5). Department of Energy and Climate Change, London, UK.

DECC, 2015b. Sub-national Residual Fuel Consumption in the United Kingdom, 2005-2013, 24 September 2015 ed. Online.

DECC, 2016b. Non-domestic RHI and Domestic RHI Monthly Deployment Data. February 2016. 17/03/2016 ed. Department of Energy and Climate Change, London, UK.

DECC, 2016a. Domestic Wood Use Survey. Department for Energy and Climate Change, London, UK.

Denier Van Der Gon, H.A.C., Bergström, R., Fountoukis, C., Johansson, C., Pandis, S.N., Simpson, D., Visschedijk, A.J.H., 2015. Particulate emissions from residential wood combustion in Europe – revised estimates and an evaluation. Atmos. Chem. Phys. 15, 6503–6519.

EEA, 2013. EMEP/EEA Air Pollutant Emission Inventory Guidebook 2013. European Environment Agency, Luxembourg.

Eisentraut, A., Brown, A., 2014. Heating without Global Warming: Market Developments and Policy Considerations for Renewable Heat. France, OECD/IEA, Paris.

EUROSTAT, 2016. Energy Balances. European Commission.

Fountoukis, C., Butler, T., Lawrence, M.G., Denier Van Der Gon, H.A.C., Visschedijk, A.J.H., Charalampidis, P., Pilinis, C., Pandis, S.N., 2014. Impacts of controlling biomass burning emissions on wintertime carbonaceous aerosol in Europe. Atmos. Environ. 87, 175–182.

French, L.J., Camilleri, M.J., Isaacs, N.P., Pollard, A.R., 2007. Temperatures and heating energy in New Zealand houses from a nationally representative study—HEEP. Energy Build. 39, 770–782.

Fuller, G.W., Tremper, A.H., Baker, T.D., Yttri, K.E., Butterfield, D., 2014. Contribution of wood burning to PM10 in London. Atmos. Environ. 87, 87–94.

Grange, S.K., Salmond, J.A., Trompetter, W.J., Davy, P.K., Ancelet, T., 2013. Effect of

atmospheric stability on the impact of domestic wood combustion to air quality of a small urban township in winter. Atmos. Environ. 70, 28–38.
Howden-Chapman, P., Viggers, H., Chapman, R., O'dea, D., Free, S., O'sullivan, K., 2009. Warm homes: drivers of the demand for heating in the residential sector in New Zealand. Energy Policy 37, 3387–3399.
IEA, 2013. Tracking Clean Energy Progress 2013. IEA Input to the Clean Energy Ministerial. OECD/IEA, Paris.
Isaacs, N.P., Burrough, C.M., Pollard, A., Saville-Smith, K., Fraser, R., Rossouw, P., Jowett, J., 2010. Energy Use in New Zealand Households. Final Report on the Household Energy End-Use Project (HEEP). BRANZ Ltd., Judgeford, NZ.
Jones, J., Ross, A., Williams, A., 2005. Atmospheric chemistry implications of the emission of biomass smoke. J. Energy Inst. 78, 199–200.
Jones, J.M., Lea-Langton, A.R., Ma, L., Pourkashanian, M., Williams, A., 2014. Pollutants Generated by the Combustion of Solid Biomass Fuels. Springer, London, UK.
Kossmann, M., Sturman, A., 2004. The surface wind field during winter smog nights in Christchurch and coastal Canterbury, New Zealand. Int. J. Climatol. 24, 93–108.
Kuschel, G., Metcalfe, J., Wilton, E., Guria, J., Hales, S., Rolfe, K., Woodward, A., 2012. Updated Health and Air Pollution in New Zealand Study (HAPINZ). Ministry of Transport, Ministry for the Environment and New Zealand Transport Agency. http://www.hapinz.org.nz/HAPINZ%20Update_Vol%201%20Summary%20Report.pdf. Prepared for Health Research Council of New Zealand.
Lee, R.G.M., Coleman, P., Jones, J.L., Jones, K.C., Lohmann, R., 2005. Emission factors and importance of PCDD/Fs, PCBs, PCNs, PAHs and PM10 from the domestic burning of coal and wood in the U.K. Environ. Sci. Technol. 39, 1436–1447.
Lelieveld, J., Evans, J.S., Fnais, M., Giannadaki, D., Pozzer, A., 2015. The contribution of outdoor air pollution sources to premature mortality on a global scale. Nature 525, 367–371.
Lohmann, R., Northcott, G.L., Jones, K.C., 2000. Assessing the contribution of diffuse domestic burning as a source of PCDD/Fs, PCBs, and PAHs to the U.K. Atmos. Environ. Sci. Technol. 34, 2892–2899.
Lucon, O., Ürge-Vorsatz, D., Zain Ahmed, A., Akbari, H., Bertoldi, P., Cabeza, L.F., Eyre, N., Gadgil, A., Harvey, L.D.D., Jiang, Y., Liphoto, E., Mirasgedis, S., Murakami, S., Parikh, J., Pyke, C., Vilariño, M.V., 2014. Buildings. In: Edenhofer, O., Pichs-Madruga, R., Sokona, Y., Farahani, E., Kadner, S., Seyboth, K., Adler, A., Baum, I., Brunner, S., Eickemeier, P., Kriemann, B., Savolainen, J., Schlömer, S., Von Stechow, C., Zwickel, T., Minx, J.C. (Eds.), Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK. and New York, USA.
MBIE, 2015. Energy in New Zealand. Energy Balances. Ministry of Business, Innovation & Employment, Wellington, NZ.
Mcdonald, J.D., Zielinska, B., Fujita, E.M., Sagebiel, J.C., Chow, J.C., Watson, J.G., 2000. Fine particle and gaseous emission rates from residential wood combustion. Environ. Sci. Technol. 34, 2080–2091.
Milionis, A.E., Davies, T.D., 2008. A comparison of temperature inversion statistics at a coastal and a non-coastal location influenced by the same synoptic regime. Theor. Appl. Climatol. 94, 225–239.
Mitchell, E.J.S., Lea-Langton, A.R., Jones, J.M., Williams, A., Layden, P., Johnson, R., 2016. The impact of fuel properties on the emissions from the combustion of biomass and other solid fuels in a fixed bed domestic stove. Fuel Process. Technol. 142, 115–123.
Molnár, P., Gustafson, P., Johannesson, S., Boman, J., Barregård, L., Sällsten, G., 2005. Domestic wood burning and PM2.5 trace elements: personal exposures, indoor and outdoor levels. Atmos. Environ. 39, 2643–2653.
Moshammer, H., Kaiser, A., Flandorfer, C., Haluza, D., Neuberger, M., 2009. Air pollution due to wood burning for heating: a health impact assessment. In: International Society for Environmental Epidemiology 21st Annual Conference. Epidemiology, Dublin, Ireland.
Myhre, G., Shindell, D., Bréon, F., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T., Zhang, H., 2013. Anthropogenic and natural radiative forcing. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Climate Change 2013: the Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
Naeher, L.P., Brauer, M., Lipsett, M., Zelikoff, J.T., Simpson, C.D., Koenig, J.Q., Smith, K.R., 2007. Woodsmoke health effects: a review. Inhal. Toxicol. 19, 67–106.
Noonan, C.W., Ward, T.J., Navidi, W., Sheppard, L., Bergauff, M., Palmer, C., 2011. Assessing the impact of a wood stove replacement program on air quality and children's health. Res. Rep. Health Eff. Inst. 3–37, 39–47 discussion.
Nussbaumer, T., 2003. Combustion and Co-combustion of Biomass: Fundamentals, technologies, and primary measures for emission reduction. Energy Fuels 17, 1510–1521.
OECD, 2014. OECD Factbook 2014: Economic, Environmental and Social Statistics. http://dx.doi.org/10.1787/factbook-2014-en.
OFT, 2011. Off-grid Energy: an OFT Market Study. Office of Fair Trading, London, UK (Crown Copyright).
Palmer, J., Cooper, I., 2014. United Kingdom Housing Energy Fact File. Department of Energy and Climate Change, London, UK.
Panicker, A.S., Pandithurai, G., Safai, P.D., Dipu, S., Lee, D.-I., 2010. On the contribution of black carbon to the composite aerosol radiative forcing over an urban environment. Atmos. Environ. 44, 3066–3070.
Peng, J., Hu, M., Guo, S., Du, Z., Zheng, J., Shang, D., Levy Zamora, M., Zeng, L., Shao, M., Wu, Y.-S., Zheng, J., Wang, Y., Glen, C.R., Collins, D.R., Molina, M.J., Zhang, R., 2016. Markedly enhanced absorption and direct radiative forcing of black carbon under polluted urban environments. Proc. Natl. Acad. Sci. 113, 4266–4271.
Peters, H.A., Croft, W.A., Woolson, E.A., Darcey, B.A., Olson, M.A., 1984. Seasonal arsenic exposure from burning chromium-copper-arsenate-treated wood. JAMA 251, 2393–2396.
Robinson, D.L., 2015. Wood burning stoves produce PM2.5 particles in amounts similar to traffic and increase global warming. BMJ 351.
Saleh, R., Robinson, E.S., Tkacik, D.S., Ahern, A.T., Liu, S., Aiken, A.C., Sullivan, R.C., Presto, A.A., Dubey, M.K., Yokelson, R.J., Donahue, N.M., Robinson, A.L., 2014. Brownness of organics in aerosols from biomass burning linked to their black carbon content. Nat. Geosci. 7, 647–650.
Seinfeld, J.H., Pandis, S.N., 2006. Atmospheric Chemistry and Physics: from Air Pollution to Climate Change. John Wiley & Sons, New Jersey, USA.
Seljeskog, M., Goile, F., Sevault, A., Lamberg, H., 2013. Particle Emission Factors for Wood Stove Firing in Norway. SINTEF Energy Research, Trondheim, Norway.
SIA, 2016. Stove Industry Alliance Research. Stove Industry Alliance.
StatisticsNZ, 2015. 2013 Census Data Tables. Statistics New Zealand, Wellington, NZ.
StatisticsNZ, 2016. Dwelling and Household Estimates. Statistics New Zealand, Wellington, NZ.
Stohl, A., Aamaas, B., Amann, M., Baker, L.H., Bellouin, N., Berntsen, T.K., Boucher, O., Cherian, R., Collins, W., Daskalakis, N., Dusinska, M., Eckhardt, S., Fuglestvedt, J.S., Harju, M., Heyes, C., Hodnebrog, Ø., Hao, J., Im, U., Kanakidou, M., Klimont, Z., Kupiainen, K., Law, K.S., Lund, M.T., Maas, R., Macintosh, C.R., Myhre, G., Myriokefalitakis, S., Olivié, D., Quaas, J., Quennehen, B., Raut, J.C., Rumbold, S.T., Samset, B.H., Schulz, M., Seland, Ø., Shine, K.P., Skeie, R.B., Wang, S., Yttri, K.E., Zhu, T., 2015. Evaluating the climate and air quality impacts of short-lived pollutants. Atmos. Chem. Phys. 15, 10529–10566.
Straif, K., Cohen, A., Samet, J. (Eds.), 2013. Air Pollution and Cancer. WHO. IARC Scientific Publication No. 161.
Todd, J.J., Greenwood, M., 2006. Proposed Changes to AS/NZS 4013 – Determination of Particle Emissions Factors. Eco-Energy Options Pty Ltd. and AHHA Testing Laboratory report for the Commonwealth Department of the Environment and Heritage.
Trompetter, W.J., Grange, S.K., Davy, P.K., Ancelet, T., 2013. Vertical and temporal variations of black carbon in New Zealand urban areas during winter. Atmos. Environ. 75, 179–187.
UNEP, 2009. Buildings and Climate Change: Summary for Decision-makers. UNEP DTIE, Paris.
USEIA, 2014. Table CE5.2 Household Wood Consumption in the U.S. Totals and Averages, 2009. U.S. Energy Information Administration, 2009 Residential Energy Consumption Survey (RECS) Data. https://www.eia.gov/consumption/residential/data/2009/.
USEPA, 1995. AP 42, Fifth Edition Compilation of Air Pollutant Emission Factors. Volume 1: Stationary Point and Area Sources. United States Environmental Protection Agency, Research Triangle Park, NC, USA.
USEPA, 2016. Process for Developing Improved Cordwood Test Methods for Wood Heaters. U.S. EPA Office of Air Quality Planning and Standards (OAQPS), Washington, DC.
Van Loo, S., Koppejan, J., 2007. The Handbook of Biomass Combustion and Co-firing. Earthscan.
Williams, A., Jones, J.M., Ma, L., Pourkashanian, M., 2012. Pollutants from the combustion of solid biomass fuels. Prog. Energy Combust. Sci. 38, 113–137.
Wilton, E., 2012. Review – Particulate Emissions from Wood Burners in New Zealand. Environet Limited, Christchurch, NZ.
Winther, M., Nielsen, O.-K., 2011. Technology dependent BC and OC emissions for Danmark, Greenland and the Faroe Islands calculated for the time period 1990–2030. Atmos. Environ. 45, 5880–5895.
Wöhler, M., Andersen, J.S., Becker, G., Persson, H., Reichert, G., Schön, C., Schmidl, C., Jaeger, D., Pelz, S.K., 2016. Investigation of real life operation of biomass room heating appliances – results of a European survey. Appl. Energy 169, 240–249.
Xie, S., Mahon, K., Petersen, J., 2012. Effects of fuel and operation on particle emissions from wood burners. Air Qual. Clim. Change 46, 17.
Yap, P.-S., Garcia, C., 2015. Effectiveness of residential wood-burning regulation on decreasing particulate matter levels and hospitalizations in the san Joaquin Valley air basin. Am. J. Public Health 105, 772–778.
Young, D.E., Allan, J.D., Williams, P.I., Green, D.C., Harrison, R.M., Yin, J., Flynn, M.J., Gallagher, M.W., Coe, H., 2015. Investigating a two-component model of solid fuel organic aerosol in London: processes, PM1 contributions, and seasonality. Atmos. Chem. Phys. 15, 2429–2443.