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Contribution of solid fuel, gas combustion or tobacco smoke to indoor air pollutant concentrations in Irish and Scottish homes

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Abstract

There are limited data describing pollutant levels inside homes that burn solid fuel within developed country settings with most studies describing test conditions or the effect of interventions. This study recruited homes in Ireland and Scotland where open combustion processes take place. Open combustion was classified as coal, peat or wood fuel burning, use of a gas cooker or stove, or where there is at least one resident smoker. 24-hour data on airborne concentrations of particulate matter less than 2.5 microns in size (PM_{2.5}), carbon monoxide (CO), endotoxin in inhalable dust and carbon dioxide (CO₂), together with 2–3 week averaged concentrations of nitrogen dioxide (NO₂) were collected in 100 houses during the winter and spring of 2009–2010.

The geometric mean of the 24-hour time-weighted-average (TWA) PM_{2.5} concentration was highest in homes with resident smokers (99 µg/m³ – much higher than the WHO 24-hour guidance value of 25 µg/m³). Lower geometric mean 24-hour TWA levels were found in homes that burned coal (7 µg/m³) or wood (6 µg/m³) and in homes with gas cookers (7 µg/m³). In peat-burning homes the average 24-hour PM_{2.5} level recorded was 11 µg/m³. Airborne endotoxin, CO, CO₂ and NO₂ concentrations were generally within indoor air quality guidance levels.

Keywords

indoor; pollution; solid-fuel; biomass; environmental tobacco smoke; homes; health

INTRODUCTION

There is increasing interest in the effects of Indoor Air Quality (IAQ) on respiratory and cardiovascular health, with many recent studies suggesting associations between poor IAQ and exacerbations of asthma, cough and cardiovascular events including admission to hospital for acute coronary syndrome (Mackay et al., 2010a; Kurmi et al., 2010; Sundell, 2004; Blanc et al., 2005; Bernstein et al., 2008; Mackay et al., 2010b). In more economically developed countries people tend to spend between 80–90% of their time within enclosed indoor environments with a large proportion of this time at home (Klepeis et al., 2001;

Schweizer et al., 2007). Time spent indoors at home is likely to be even greater for vulnerable groups such as the elderly, the very young and the chronically ill. Outdoor air pollution levels clearly influence IAQ (Monn et al., 1997; Pekey et al., 2010) but when specific indoor sources of air pollution are present within the home environment then the contribution from these sources dominates personal exposures (Harrison et al., 2002).

There is a growing literature on exposure to Indoor Air Pollutants (IAPs) generated by open combustion in homes in less economically developed countries (e.g. Malawi, India, Mexico). Studies on biomass fuel combustion in the developing world have indicated very high fine particulate matter less than 2.5 microns in size (PM_{2.5}) and carbon monoxide (CO) exposures are linked to acute and chronic health conditions such as asthma and other respiratory illnesses, and can have an adverse effect on children's development (Fullerton et al., 2008; Fullerton et al., 2009; Khalequzzaman et al., 2007; Kilabuko et al., 2007). The World Health Organisation (WHO) estimates that exposure to IAPs from the burning of biomass fuels is responsible for two million deaths each year making this one of the world's main environmental health issues (Pruss-Ustun et al., 2008).

In more affluent countries the main sources of PM_{2.5} in homes are likely to be from the combustion of fuel such as coal, wood, peat and gas for cooking or heating, and the combustion of tobacco in the form of smoking (Lai et al. 2006). Some short duration irregular peaks from activities such as vacuuming, candle or incense burning may also occur, but these are likely to make little contribution to 24-hour average PM levels (He et al., 2004). There are limited data on the contribution of open combustion to indoor air quality in the developed world. A small number of studies have looked at IAP levels in homes with wood stoves but these have often been under experimental conditions (Traynor et al., 1987) or linked to intervention studies (Allen et al., 2009). Data on IAP levels in coal burning homes in the developed world (Moriske et al., 1993) are limited with none on IAP concentrations in peat-burning homes. Some data on concentrations of combustion gases and particulate from the use of gas fireplaces (Francisco et al., 2010) and gas cooking (Dennekamp et al., 2001) exist with various data available on IAP concentrations produced from smoking activity in homes (Hunt et al., 2011; Wallace et al., 2003).

Fuel for heating and cooking in most EU countries tends to be electricity or gas based, with efficient stoves and flues in most kitchens or heating appliances. However there are still a considerable percentage of homes where solid or biomass fuel is used within areas of Ireland and Scotland. Peat is still used in large areas of rural Western Ireland (139,308 houses, representing 9.3%) and the Highlands and Islands of Scotland. Estimates of residential fuel use in Ireland, in 2009, indicate that coal, peat and other biomass fuels account for over 18% of household energy consumption (by kilo tones of oil equivalent) with up to 17% of households using solid fuel for heating, and 20% of households having gas in their accommodation (CSO, 2007). Data from Scotland indicate that about 1% of homes use solid fuels for heating, and about 77% of households use mains gas as their primary heating fuel with a subset of this population having either gas cooking or gas fires in the main living spaces (SHCS, 2007).

There are no health-based standards for most IAPs in homes in Scotland and Ireland. The most relevant guidance available is that produced by the Department for Health Committee on the Medical Effects of Air Pollutants (COMEAP), Guidance on the Effects on Health of Indoor Air Pollutants (Department of Health, 2004). This document provides IAQ guidance on Nitrogen dioxide (NO₂), CO, formaldehyde, benzene and polycyclic aromatic hydrocarbons (PAHs). The guidance recommends that NO₂ concentrations in homes should not exceed 150 ppb (averaged over 1 hour) and that CO concentrations should not exceed 10 ppm (averaged over 8 hours). The report does not present a guidance value for indoor

particulate matter concentrations. The more recent WHO Air Quality Guidelines for Indoor Air (WHO, 2010) provides guidance that CO concentrations should not exceed 5.7 ppm (averaged over 24 hours) or 8.1 ppm (averaged over 8 hours) and indoor NO₂ concentrations should not exceed 100 ppb (averaged over 1 hour). The WHO guidelines also note that there is no convincing evidence for a difference in the hazardous nature of particulate matter from indoor sources as compared with those from outdoors and thus goes on to recommend that the 2005 WHO outdoor air quality guidance for PM should be applied to indoor environments (WHO, 2005). This suggests that PM_{2.5} concentrations should be kept at levels below 25 µg/m³ averaged over a 24-hour time period and less than 10 µg/m³ averaged over an annual period. Similar health-based guidance for outdoor PM_{2.5} levels exists from the US Environmental Protection Agency who deem concentrations of 35 µg/m³ to be 'unhealthy to sensitive groups' and 65 µg/m³ to be 'unhealthy' for the general population (US EPA, 2011).

Airborne endotoxin is part of the cell wall of Gram negative bacteria and a potent inflammatory agent. It has been measured in airborne particulate in occupational settings, office buildings, households and ambient air with various studies linking household endotoxin levels to diagnosed asthma, asthma medication use and severity of asthma symptoms (Michel et al., 1996; Thorne et al., 2005). Occupational exposures to endotoxin have been linked to respiratory illness (Smit et al., 2008; Thorne and Duchaine 2007). Endotoxin has also been identified in tobacco smoke (Larsson et al., 2004) and in indoor environments where smoking takes place (Rennie et al., 2008; Tavernier et al., 2005). Although there are no IAQ guidance values for airborne endotoxin, the Dutch Expert Committee on Occupational Standards (DECOS) has set a limit at 90 Endotoxin Units (EU)/m³ (DECOS, 2010).

Health based guidance for indoor CO₂ levels are generally not available but guidance from ASHRAE 62-1989 "Ventilation for Acceptable Indoor Air Quality" (1989) suggests that concentrations above 1000 ppm are indicative of poor indoor ventilation conditions.

In order to be able to quantify the potential health effects of exposure to IAPs in domestic environments there is a need for good quality data characterizing the main pollutant levels generated from open combustion sources in homes in the developed world. Greater understanding of household and behavioural determinants of these pollutant concentrations is required to enable consideration of how these exposures will impact upon respiratory and cardiovascular health at a population level.

This paper describes results from a study called IAPAH (Indoor Air Pollution And Health). IAPAH aims to estimate the health burden from open combustion within households in Scotland and Ireland. This paper provides details of the first element of the project: an exposure assessment programme to collect and compare IAP measurement data from 100 homes where fuel or tobacco material was burned.

METHODS

Ethical Approval

Ethical approval for the study was given by the local College Ethics Research Board of the University of Aberdeen, Scotland and by the Research Ethics Committee of the National University of Ireland, Galway. Participants provided informed consent and a consent form was signed by both participant and the researcher in all cases before sampling began.

Recruitment

Recruitment of households took place between October 2009 and March 2010 during the winter period when fuel use would be at a peak and when ventilation levels tend to be minimised. The study was publicised in several ways, for example through advertisements in the local press in Aberdeen, Scotland and Galway, Ireland and publicity in University campuses. In addition, a website (www.nuigalway.ie/iapah) was developed which provided information about the study and how to participate. Other participants were recruited via word of mouth and snowballing techniques utilizing those already recruited to the study.

Participants/Households

Our aim was to recruit 20 households that used peat as heating fuel, 20 households that used coal, 20 that used wood, 20 that used a gas stove to cook and 20 that had at least 1 adult resident smoker (with no other combustion source present e.g. electricity used for heating purposes). Households were to be recruited in and around the city of Aberdeen and Aberdeenshire in North-east Scotland and in and around Galway city in the west-coast of Ireland. Potential participants who expressed an interest in the study were screened for eligibility using a telephone questionnaire which asked questions about solid fuel use and smoking by residents in the home. Households were excluded if they reported burning more than one type of solid fuel/tobacco source within the home as we wished to be able to classify exposures by a single combustion source.

Indoor Air Pollutant Sampling

Sampling instruments were placed in the main living area of each participating home and generally located in close proximity to each other at a height of about 1.0–1.5m. Where possible devices were placed at a distance of at least 1.0m from windows, doors and the heating/cooking source under study. A total of five IAPs were measured - PM_{2.5}, airborne endotoxin within the total inhalable dust fraction, CO, CO₂ and NO₂. The sampling was performed between 1st October 2009 and 31st March 2010, with a small number of NO₂ tubes collected into April 2010. For the real-time data collected for PM_{2.5}, CO and CO₂ our analysis protocols removed the first 20 minutes of data in order to remove the period when the researcher was in the house setting up the instrument and collecting questionnaire data. A similar procedure was used to remove the final 20 minute period of the 24-hour data.

1. TSI SidePak AM510 Personal Aerosol Monitors (TSI Inc., Shoreview, MN, USA) fitted with a PM_{2.5} impactor was used to collect and log real-time data in $\mu\text{g}/\text{m}^3$ on airborne PM_{2.5} levels over a 24-hour period. The monitor was set to log at 1 minute intervals and allowed assessment of peaks and daily variability in exposure to be determined. A correction factor for combustion-generated PM_{2.5} of 0.3 was applied to the data derived from the Sidepak device (Repace, 2006). The instruments were within 1-year of factory calibration and were zero-calibrated after each use.
2. Telaire® 7001i Data loggers (Edinburgh Instruments Ltd, Livingston, UK), were used to log CO₂ levels in ppm with a data logging kit (H08-007-02 Hobo data logger Onset Computer Corporation, Bourne, MA, USA). These devices were factory calibrated and less than 1 year old. Regular two-point calibrations (0 and 380 ppm) were carried out throughout the measurement campaign.
3. Assessment of airborne endotoxin was carried out using total inhalable dust sampling following the UK Health and Safety Executive's 'Methods for the Determination of Hazardous Substances' 14/3 (HSE, 2000). 25mm glass fibre filters (Pall Gelman Sciences) previously heated at 200°C for 28 hours were loaded into IOM inhalable dust sampler cassettes (Institute of Occupational Medicine, Edinburgh, UK). Prior to loading, cassettes and holders were washed with E-Toxa

clean (soaked in 1% E-Toxa clean overnight, rinsed 10 times with tap water, 5 times with DI water, one time with nanopure water and dried at room temperature). SKC Sidekick pumps (SKC Inc., Dorset, UK), with the volumetric flow rate set at 2 l/min were used to draw air through the filters. Flow rates were checked before and after sampling using a DryCal[®] DC-Lite Primary Flow Meter Air Flow Calibrator (Brandt Instruments Inc., Prairieville, LA.). One field blank for every 10 samples was also collected. After sampling and appropriate storage at 4°C the filters were transported to the USA for analysis at the University of Iowa.

4. Average indoor NO₂ levels were measured over a period of 2–3 weeks using passive diffusion tubes (Gradko International, Winchester, UK). A single sample tube was placed in the main living area of each home away from windows and doors, at 1–1.5 metre height. They were capped on collection, date and time noted, sealed in plastic bags and refrigerated at 4°C until they were sent in batches to the Gradko International laboratory (Winchester, England) for analysis within the 16-week recommended shelf life.
5. CO levels were measured and logged every minute over a 24-hour period using Lascar Easylogger EL-USB-CO (Lascar Electronics Ltd, Wiltshire, UK) data loggers. These devices were factory calibrated and were less than one-year old during the measurement campaign.

A sampling box, large enough to accommodate the Sidepak and SKC pump, was constructed from cardboard/wood and padded with insulating material to minimise noise disturbance. The fitted lid was similarly padded. Two holes were cut in the front panel of the box to allow access for the power cables and tygon tubing. The Sidepak and SKC pump were connected to mains electricity in each home to enable operation for at least 24 hours of sampling.

Equipment was placed in the home on the first visit. A short questionnaire was also administered to determine demographic information such as age, number of people in the household and pet ownership. All sampling equipment except the NO₂ tube was collected approximately 24 hours later. The NO₂ tube was collected at a third visit to the home some 2–3 weeks later when each household also received a store voucher to the value of £20 (€30) to compensate them for their time, diary completion and use of electricity to operate the pumps. A field protocol sheet was used to record participants' answers and to log times of equipment set up and removal.

Endotoxin analysis

Endotoxin concentrations of samples were measured at four serial dilutions using the kinetic chromogenic *Limulus amoebocyte lysate* assay (Lonza, Inc., Walkersville, MD, USA) as described by Thorne (Thorne, 2000; Thorne *et al.*, 2005). Filters were extracted into pyrogen-free water and eluates were evaluated against a 12-point standard curve ranging from 0.0244 to 50 EU/ml. Samples were corrected using the average of the EU/filter value for field blanks from the batch. The analytical limit of detection (LOD) was derived by using a value equivalent to the standard deviation (SD) of the lab blanks (0.69 EU/filter). Any field-blank corrected filter with a value less than this (n=4) was assigned a value of one-half the LOD (0.34 EU/filter). Concentrations were then calculated using this blank-corrected and LOD assigned EU/filter values by dividing by the volume of air sampled to produce EU/m³. The first batch of Scottish and Irish samples were excluded due to evidence of gross contamination together with one further sample from the Irish data. Results are reported for the remaining 69 samples.

Outdoor measurements

Outdoor temperatures for Scotland were obtained from the UK Met Office for Aberdeen and for Ireland, from Met Eireann for Galway. 24-hour outdoor PM_{2.5} concentration in Aberdeen and Galway were obtained from the UK Department for Environment and the Mace Head Atmospheric Research Station in County Galway, respectively.

Statistical analysis

The associations between each of the exposure endpoints and potential explanatory variables such as fuel type, site, type of house etc., were firstly examined using cross-tabulations and boxplots. This was followed by statistical analyses of potential associations using generalised linear models, adjusting simultaneously for the effects of several explanatory variables. Where there was evidence of a systematic lack of fit for models on the original measurement scale, the exposure measurements were transformed to the log scale before the modelling was carried out. Statistical analyses were done using the Minitab v.13 and Genstat v.12 statistical software packages.

RESULTS

One hundred and two households participated in this study, 52 in Scotland and 50 in Ireland. One further household in Scotland initially agreed to take part, then withdrew before the sampling visits began. Two households in Ireland were found to have gas heating rather than gas cooking as specified in the inclusion criteria and were thus removed from the analysis.

Homes in both countries were distributed across an area of about 80km radius from the central points of Aberdeen (Scotland) and Galway (Ireland) with smoking, gas-cooking and coal-burning homes more likely to be located closer to the urban/suburban areas and peat and wood burning homes likely to be at greater distance from the city in more rural locations. The fixed site outdoor air pollution data obtained in Scotland was taken from a site located in the central urban area of Aberdeen while Irish data was generated from a rural monitoring site about 50km to the West of Galway city.

More peat burning homes were recruited in Ireland with more wood burning homes recruited in Scotland. This was due to difficulties in finding enough peat burning homes in and around Aberdeen City, and it was decided to recruit additional peat homes in Galway where peat was more commonly burned. In exchange, more homes that burned wood were recruited in Scotland as these were plentiful. Table 1 shows the number of homes in each fuel category recruited together with demographic and housing characteristics.

The solid-fuel heating systems used varied from open fires with chimneys through to more closed, airtight stoves. Most coal burning homes had open fire arrangements while wood burning homes tended to have closed stoves with a flue and a door that was opened to load wood. Peat burning homes were mixed between a traditional open fire grate or a stove similar to that described for wood. Across all homes the median number of people (including children) living in the home was 3, (range 1–6). Of the participants who reported not having central heating in their home (Ireland n=1, Scotland n=6), four had electric storage heaters in some rooms and the other three used only solid fuel to heat their entire home.

In the smoking homes the median number of cigarettes smoked across the 24-hour sampling period was 16 (range – 7 to 84) and was higher in Scotland (median = 12) than Ireland (median =7). One home in Ireland had a resident pipe smoker. Across smoking homes as a whole, 55% had only one resident smoker (Scotland 64% versus Ireland 44%), 30% had two resident smokers and 15% had three resident smokers.

Indoor air pollutant measurements

Air pollutant concentrations were measured in all 100 homes in the study. Equipment problems occurred in three homes in Scotland resulting in missing data for CO₂ in three homes and missing temperature and relative humidity data in two homes. The first batch of endotoxin samples from both Ireland and Scotland showed gross contamination and taken together with one other individual sample that was one-hundred times any other measurement this resulted in exclusion of endotoxin data from 31 homes.

The mean average PM_{2.5} level across all 100 homes in the 24-hour sampling period was 13 µg/m³. Table 2 shows results of PM_{2.5} levels found in these homes and provides details of the percentage of time when concentrations exceeded the US EPA outdoor air quality threshold deemed to be 'unhealthy for sensitive groups' (35 µg/m³) and the threshold deemed to be 'unhealthy' for the wider population (65 µg/m³).

Table 3 presents models showing the relationship between 24-hour average PM_{2.5} (µg/m³) and fuel type, site and (in smoking homes only) numbers of cigarettes smoked and contains estimates of regression coefficients, with absolute value of ratio of coefficient to standard error, z-statistic, in *italics*. Separate analyses were carried out to investigate the effects on PM_{2.5} of other potential explanatory variables e.g. number of residents, presence of kitchen ventilation, presence of pets, house type, age of residents etc. but as many of these variables were co-related with fuel type or each other and the numbers of each variable by fuel were often small, the data are not presented here.

PM_{2.5} levels were higher in smoking homes and, to a lesser extent, in homes heated using peat (table 2). This was confirmed in the results of the statistical analysis (table 3), which also showed that, while PM_{2.5} levels in smoking homes were significantly higher than in other homes in both Ireland and Scotland (Model 1), the differences were significantly greater in Scottish homes (Models 2 and 3) (mean=187 µg/m³ in Scottish smoking homes, 89 µg/m³ in Irish smoking homes; p<0.01). Levels of PM_{2.5} in smoking homes were strongly related to the number of cigarettes smoked during the 24 hours during which PM_{2.5} was being measured and once number of cigarettes was adjusted for in the regression model, the difference between Scotland and Ireland was no longer significant statistically (Model 4). There was a higher proportion of smoking homes in Scotland with older residents who may be in the home, and therefore smoking, for a longer period each day. Figure 1 illustrates the relationship between 24-h average PM_{2.5} concentrations and the number of cigarettes smoked per household in the smoking homes.

Figure 2 illustrates the range of 24-hour PM_{2.5} concentrations measured in each fuel-burning or smoking home in both Scotland and Ireland. The box represents the 25th and 75th percentile values (inter-quartile range) with the dark horizontal line the median (50th percentile) value. The upper and lower whiskers are the limits of 1.5 times the inter-quartile range in each direction and the asterisks are outlying values beyond these ranges. The horizontal line is the WHO 24-hour guidance value for PM_{2.5} exposure (25 µg/m³).

For PM_{2.5} we also calculated a 6-hour evening average from our real-time data. This 6pm-midnight period was derived to better reflect personal exposure of working adults who are likely to spend a proportion of the day outside the home. Over the 100 homes this 6-hour average was 50 µg/m³, 11 µg/m³ for wood, 13 µg/m³ for coal, 12 µg/m³ for gas-cooking homes, 29 µg/m³ for peat and 197 µg/m³ for smoking homes.

Results of the other indoor air parameters measured are presented in Table 4.

All measurements for CO, NO₂ and airborne endotoxin were within the available IAQ limits. Across the 100 homes the average concentrations of CO₂ was 713 ppm. The average NO₂ concentration was 5 ppb and airborne levels averaged 5.7 EU/m³. A small number of homes (n=7) had 24-hour average CO₂ concentrations in excess of 1000 ppm suggesting poor levels of ventilation. Differences in IAP concentrations between fuel types were seen for CO₂ and NO₂. Levels of CO₂ in smoking homes were higher than for other fuel types for houses in Scotland only, there were no differences between fuel types in Irish properties. Homes in which gas cooking took place produced the highest levels of NO₂ in both countries, while smoking homes also had higher levels of NO₂ than homes heated using coal, wood or peat.

There was some evidence that endotoxin and CO₂ levels were higher in houses where dogs were kept as pets and that CO levels were higher in flats and terraced houses compared to detached and semi-detached houses.

Outdoor measurements

The average minimum outdoor temperature in Aberdeen over the monitoring period was 1.5 °C (range -9.4 – 9.8°C). Average maximum outdoor temperature was 7.6°C (range 2.3 – 17.8°C). In Galway for the same period, the average minimum outdoor temperature was 2.4°C (range -8.0 – 11.7°C), with an average maximum of 9.8°C (range 2.5 – 18.2°C) suggesting that temperatures and thus heating requirements were broadly similar in both countries.

The average 24-hour outdoor PM_{2.5} concentration in Aberdeen was 8.2 µg/m³ (Inter Quartile Range 3.7–11.9 µg/m³). The average 24-hour outdoor PM_{2.5} concentration in Galway over the project sampling period was also 8.2 µg/m³ (Inter Quartile Range 5.7–9.8 µg/m³).

DISCUSSION

Currently very little published data exist on IAQ in homes burning solid fuels in developed countries with most previous studies focusing on homes that use wood burning stoves (Levesque et al., 2001; Fine et al., 2002; Molnar et al., 2005; Gustafson et al., 2008) or gas (Garcia-Algar et al., 2004). A small number of studies have investigated IAQ within coal-burning homes (Moriske et al., 1996; Hendersen et al., 2006), or homes that burn peat (Guo et al., 2008). Several studies have shown associations between gas cookers or stoves and increased indoor concentrations of NO₂. Garcia-Algar and colleagues (2004) measured NO₂ in living rooms and found that concentrations were more than twice as high in homes with gas cookers compared to homes cooking with electricity. Gas stoves have also been shown to produce higher NO₂ levels compared to other forms of household combustion (Gilbert et al., 2006).

The combustion of tobacco during smoking is another common source of IAPs. Data from Scotland indicate that 12% of adults (Haw and Gruer, 2007) and 27% of children (Akhtar et al., 2007) are exposed to Environmental Tobacco Smoke (ETS) at home, with similar data from Ireland suggesting 21% of adults and 20% of children are exposed (Shafrir A, personal communication). Some data are available on PM levels in homes of smokers (van Deusen et al., 2009; Osman et al., 2007), but these tend to be focused on intervention studies and often describe short-term exposures during smoking episodes. The presence of a resident smoker has also been linked to higher NO₂ concentrations in homes (Cyrus et al., 2000; Raw et al., 2004).

Studies which have looked at homes that burn fossil fuels such as wood or gas in developed countries have generally found largely undetectable levels of CO, however Levesque and colleagues (2001) and Willers et al (2006) report higher levels in homes with solid fuel burning compared to homes that use electricity. Henderson measured CO in smoking and non-smoking homes and in only one case did the concentrations of CO exceed the WHO 8 hour limit of 9 ppm (Henderson et al., 2006; WHO, 1999).

The current work describes a substantial dataset of measurements of IAPs in over 100 homes across Scotland and Ireland together with detailed descriptive information on housing characteristics. The main finding is that homes using solid or fossil-derived fuels in open combustion processes have low concentrations of the main IAPs measured. Concentrations of CO, NO₂ and airborne endotoxin were well within health-based standards in all homes where measurements took place. PM_{2.5} levels were highest in smoking homes but 24-hour average levels were found to exceed the WHO 24-hour guidance level of 25 µg/m³ in one-quarter (n=25) of homes although most (n=19) of those homes exceeding this value were smoking homes. Twelve of the smoking homes (60%) had 24-hour PM_{2.5} concentrations that exceeded the US EPA 65 µg/m³ threshold deemed to be unhealthy. None of the coal, wood, peat or gas burning homes had 24-hour concentrations above this threshold. This latter point is a positive finding that suggests generally good control of IAPs in homes burning solid fuel in Scotland and Ireland. All of these homes had either chimney or flues that ensured good levels of ventilation for the burning process.

CO₂ levels of <1000ppm found in most homes in this study indicate good ventilation and air exchange. Levels of CO₂ exceeded guidance in 7 of the 102 homes- the majority of those exceeding the 1000ppm level being smoking homes (n=4). The low CO concentrations measured- the highest 24-hour value was 1.4 ppm- suggest that heating and stove apparatus was well maintained and generally producing highly efficient combustion. NO₂ levels in gas cooking homes were higher than in all other types of homes as also noted by Garcia-Algar and colleagues (2004). There were no clear differences in solid-fuel IAP levels related to household or resident characteristics and no significant differences between the two countries. The PM_{2.5} concentrations in the non-smoking homes in this study are towards the lower end of those reported in other literature. A study by Wallace and colleagues (2003) indicated daily average indoor PM_{2.5} concentrations of 18 µg/m³ across 193 non-smoking homes while Neas and co-workers (1994) report 17µg/m³ in non-smoking households during the 1980s. Both of these studies were based across several city areas in the USA where outdoor PM_{2.5} concentrations are likely to have been higher than the Aberdeen and Galway city and semi-rural areas measured in our study.

PM_{2.5} and CO concentrations in homes where smoking activity took place were markedly higher than in those homes burning coal, wood, peat or gas. The PM_{2.5} data in particular shows that mean 24-hour average levels in smoking homes was 15 to 20 times higher than those measured in the solid-fuel or gas burning homes. The mean 24-hour level in smoking homes of 143 µg/m³ is over 4 times the US EPA outdoor Air Quality Index 'unhealthy' level for sensitive groups (35 µg/m³) (US EPA, 2011) and approaching six times the WHO 24-hour guidance concentration of 25 µg/m³ (WHO, 2010). The percentage of sampling minutes when PM_{2.5} levels exceeded the US EPA 'unhealthy for sensitive groups' 35 µg/m³ threshold was typically 60% in smoking homes compared to values of <3% in coal, wood and gas burning homes and 7.3% in peat burning homes. Recent evidence suggests that removing exposure to ETS may be associated with a considerable decrease in the risk of cardiovascular and pulmonary events across the population (Mackay et al., 2010a; Mackay et al., 2010b; Oono et al., 2011). Time-series studies on the effects of exposure to outdoor PM_{2.5} suggest that there is an increase in cardiovascular and respiratory events at even quite modest increases in concentrations with a recent study in Australia by Dennekamp and

colleagues (2010) demonstrating an increase in risk of an out-of-hospital cardiac arrest of 3.6% when outdoor PM_{2.5} levels increased by 4.26 µg/m³. Using the 143 µg/m³ value generated from the present study and an exposure duration of 80% of the day (19.2 hours spent at home), with a breathing volume of 18 m³/day, this would equate to a daily PM_{2.5} dose of >2 mg. Pope and colleagues (2009) concluded this value would increase risk of cardiovascular and cardiopulmonary disease by 40–50%.

The 24-hour PM_{2.5} concentrations in the homes where tobacco smoking took place are considerable and the average value of 143 µg/m³ appear to be higher than several other studies. Neas and colleagues (1994) reported mean PM_{2.5} concentrations of 48 µg/m³ in 'consistently smoking households', Wallace and colleagues (2003) measured PM_{2.5} levels of 46 µg/m³ in 101 smoking homes across 7 US cities, while more recent measurement of 24-hour PM_{2.5} concentrations in homes of smoking mothers with a child <1 year of age reported a geometric mean of 26 µg/m³ (Hunt et al., 2011). The mean value in our study can be compared to similar measurements made in a range of public space environments where smoking takes place. The average of PM_{2.5} measurements in 106 bars across the UK prior to the introduction of smoke-free restrictions was 200 µg/m³ (Semple et al., 2010) while a recent study of PM_{2.5} concentrations in 66 US casinos where smoking is permitted reported a geometric mean value of 54 µg/m³ (Repache et al., 2011). One-minute peak levels measured in this study were also considerable with values reaching above 250 µg/m³ in 90% (n=18) of homes sampled. The US EPA Air Quality Index rates PM_{2.5} concentrations at this level and above as 'Very Unhealthy'.

Airborne endotoxin concentrations measured in this study are similar to those reported in previous studies in domestic environments. Concentrations were broadly similar in coal (5.78 EU/m³), peat (5.12 EU/m³), and smoking (5.38 EU/m³) homes but were higher in wood burning (7.63 EU/m³) homes. Household data of airborne endotoxin levels indicate that levels are generally less than 10 EU/m³. A large study of the homes of 332 children in Canada (Dales et al., 2006) presented mean concentration of 0.49 EU/m³ while Thorne and Duchaine's (2007) data describing endotoxin levels in a number of environments indicate geometric mean (GM), inhalable fraction endotoxin in homes of rural asthmatic children of 5.8 EU/m³ (n=326). A further small study measuring airborne endotoxin in 10 homes in northern California (Chen et al., 2009) again suggested mean concentrations of <1 EU/m³. From our data there is little evidence that different fuel types, smoking activity or presence of pets influenced airborne endotoxin levels in the homes we measured although we do not have a control group of homes with no open combustion with which to compare our measurements.

Strengths and limitations of the study

Our study characterizes a range of indoor air pollutant concentrations in homes where open combustion takes place. It has a relatively large sample size with over 100 homes sampled across two neighbouring countries in Northern Europe. By the nature of our selection process to include homes with a single fuel type we have been able to examine if there are differences in IAP concentrations between different combustion materials. This is a particular strength of this study.

We acknowledge that it is difficult to be certain of the representativeness of our study sample. No data exist on the type and demographics of solid fuel burning or tobacco using homes in Scotland and Ireland so we are unable to compare our recruited group with the overall population from which they were sampled. The lack of a control group is another potential weakness of the study. Future work should examine IAP levels in a sample of homes where neither solid fuels nor smoking takes place in order to be able to compare the levels found with those reported in this work.

Sampling was not carried out simultaneously in all homes due to the limited amount of equipment we had and so there are likely to have been temporal variations in outdoor pollutant concentrations over the 6-month measurement program. This will have resulted in different contributions to indoor pollutant levels from outdoor pollutants on different days. We believe this effect would have been small and, although we only have data for outdoor PM_{2.5} levels in the Aberdeen and Galway areas, we note that the inter-quartile range for outdoor PM_{2.5} was <10 µg/m³ in Aberdeen and <5 µg/m³ in Galway.

As with all observational studies and exposure measurement it is possible that the act of measurement has influence the parameters under study. The possibility of modification of behavior in heating, cooking, ventilation and smoking is possible.

CONCLUSIONS

From this study IAQ in homes burning solid-fuels or using gas stoves appears to be generally good in homes in Ireland and Scotland. 24-hour fine particulate matter levels were broadly similar to those found in outside air in coal, wood and gas burning homes. Peat burning homes had particulate concentrations levels that were closer to the WHO annual guidance level for PM_{2.5}. CO and CO₂ levels indicate good levels of ventilation and generally well-maintained combustion apparatus. NO₂ levels were well within health-based guidance values in all homes.

The main cause for concern in terms of indoor air pollutants related to combustion in homes in Scotland and Ireland is from smoking activity. PM_{2.5} concentrations in homes with a resident smoker are, in general, an order of magnitude higher than those found in homes burning coal, wood, peat or gas. The concentrations of PM_{2.5} are comparable with those measured in smoky bars prior to smoke-free legislation being introduced in the UK and exposure to these levels is likely to be damaging to the health of residents within these homes.

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Practical Implications

Little is known about indoor air quality in homes that burn solid or fossil-derived fuels in economically developed countries. Recent legislative changes have moved to improve Indoor Air Quality at work and in enclosed public places, but there remains a real need to begin the process of quantifying the health burden that arises from indoor air pollution within domestic environments. This study demonstrates that homes in Scotland and Ireland that burn solid fuels or gas for heating and cooking have concentrations of air pollutants generally within guideline levels. Homes where combustion of cigarettes takes place have much poorer air quality.

Figure 1.
24-hour PM_{2.5} concentrations and cigarettes smoked per household

Figure 2.
24-hour PM_{2.5} concentrations by fuel type and country.

Table 1

Demographic and housing characteristics of sampled households

Demographic/Housing Characteristic	All (n=100)	Scotland (n=52)	Ireland (n=48)	P (probability)
Coal burning	22	10	12	
Peat burning	20	3	17	
Wood burning	22	17	5	
Gas cooking	16	11	5	
Smoking	20	9	11	
Age of householder giving consent (mean, years)	51	52	50	NS
Room volume (m ³)	57	56	58	NS
Central heating (%)	93	88	98	0.06
Type of house (n, %)				
Detached	51 (51%)	25 (48%)	26 (54%)	
Semi-detached *	30 (30%)	16 (31%)	14 (29%)	
Terraced *	8 (8%)	2 (4%)	6 (13%)	
Flat/apartment	11 (11%)	9 (17%)	2 (4%)	0.09
Age of house (n, %)				
Pre early 1980s	59 (59%)	42 (81%)	17 (35%)	
Post early 1980s	41 (41%)	10 (19%)	31 (65%)	0.00
Pets in household (%)	53	54	52	NS
Outdoor temp (°C)	6.0	5.8	6.2	NS
Outdoor PM _{2.5} (µg/m ³)	8.2	8.2	8.1	NS

NS=Not significant

* Terraced houses= houses in a row of similar houses that share side-walls; Semi-detached houses= consists of pairs of houses built side by side as units sharing a wall.

Table 2

24-hour PM_{2.5} levels in houses using different combustion source types

PM _{2.5} (μg/m ³)	All (n=100)	Coal (n=22)	Gas Cooking (n=16)	Peat (n=20)	Smoking (n=20)	Wood (n=22)
Mean hours sampling	23.6	24.1	24.1	24.0	23.1	23.0
Average PM _{2.5} (range)	36.2	8.9 (1–19)	8.6 (2–28)	15.6 (2–44)	143.1 (21–463)	7.7 (2–23)
Geometric Mean PM _{2.5} (GSD)	12.6 (3.7)	7.4 (2.0)	7.1 (1.9)	10.9 (2.5)	99.3 (2.5)	5.7 (2.2)
**% of sampling time >35 μg/m ³	14.7	2.3	2.5	7.3	60.8	1.7
++% of sampling time >65 μg/m ³	11.1	0.5	0.6	4.8	50.3	0.5

** % of minutes in 24-hour period sampled >24-hour average ambient air quality standard of 35 μg/m³

++ % of minutes in 24-hour period sampled > 24-hour average ambient air quality standard of 65 μg/m³

Table 3

Regression analysis for PM_{2.5} concentrations.

Terms	All homes				Smoking homes			
	Model 1		Model 2		Model 3		Model 4	
	Coeff [†]	z-value	Coeff [†]	z-value*	Coeff [†]	z-value	Coeff [†]	z-value*
Fuel category (vs Coal)								
Gas cooking	-0.3	-0.02	2.8	0.11				
Peat	6.7	0.41	6.8	0.36				
Smoking	134.2	8.16	80.5	3.68				
Wood	-1.1	-0.07	-3.6	-0.14				
Site (vs Ireland)								
Scotland			1.0	0.05	98.6	2.00	41.6	1.27
Additional fuel effect in Scotland								
Gas Cooking			-4.8	-0.14				
Peat			1.8	0.05				
Smoking			97.6	3.17				
Wood			2.8	0.09				
No of cigarettes smoked (log scale)							146.8	5.55
% variance explained	49.3%		56.0%		13.7%		67.6%	

[†]The estimate of the regression co-efficient

* A z-value of 2 or more indicates a statistically significant association.

Table 4

CO, CO₂, NO₂ and airborne endotoxin levels in houses using different combustion source types

Pollutant time-weighted average mean values	All (n=100)	Coal (n=22)	Gas Cooking (n=16)	Peat (n=20)	Smoking (n=20)	Wood (n=22)
[†] CO (ppm) (range)	0.05	0.01 (0–0.03)	0.04 (0–0.31)	0.01 (0–0.17)	0.22 (0–1.44)	0.00 (0–0.003)
[‡] CO ₂ (ppm) (range)	713	642 (480–854)	687 (450–1171)	713 (490–1097)	818 (469–1290)	708 (520–1540)
[‡] NO ₂ (ppb) (range)	5.12	4.03 (1.48–13.5)	9.01 (2.11–24.1)	3.99 (1.08–15.8)	6.82 (2.2–13.6)	2.87 (1.05–6.2)
[‡] Airborne endotoxin (EU/m ³) (range)	5.69	5.78 (0.11–25.7)	3.09 (0.72–6.9)	5.12 (0.12–24.7)	5.38 (0.92–21.7)	7.63 (0.12–16.6)

[†] 24-hour sampling period

^{*} 6 hour time-weighted average from 6pm-midnight

[‡] two week sampling period