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Development of a Probabilistic Multi-Zone Multi-Source Computational Model for Indoor Air Pollution Exposure Assessment

A thesis submitted to the National University of Ireland, Galway

For the degree of

Doctor of Philosophy

by

James McGrath

School of Physics and CCAPS
National University of Ireland, Galway

Supervisor: Dr. Miriam Byrne
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Abstract

Airborne Particulate Matter (PM) is a major environmental concern because of its known impacts on human health, and since the developed world population spends approximately 89% of its time indoors, PM in the indoor environment deserves particular attention. The use of computational models in predicting exposure to gaseous or particulate indoor air pollutants has been well documented. However, due to the existence of large uncertainties surrounding the parameterisation of exposure models, probabilistic approaches are necessary, and these type of models have had limited development to date. In this thesis, the development of an existing probabilistic model, the INDAIR model, into a state of the art model, IAPPEM, is described, and IAPPEM’s ability to fully assess the distribution of particulate air pollutants in dwellings is demonstrated through analysis of the results of a wide range of simulations.

Adaptations to the original code from which IAPPEM was developed include an increase in time resolution to one minute, the incorporation of 12 simultaneously operating emission sources, and up to 15 interconnecting rooms. Additionally, the model, which originally calculated airflow on a time-weighted average basis, was adapted to include a variable airflow rate, and the results of simulations demonstrate that without this feature, under-prediction of PM$_{2.5}$ concentrations by up to 28% occurs. Further, a modified PM$_{10}$ deposition rate, which accounts for the variability in PM$_{2.5}$/PM$_{10}$ ratios, was incorporated in IAPPEM, and the results of simulations show that this results in prediction of mean concentrations
that are 58% higher than those calculated using the unmodified model. Simulations carried out with a one minute time-resolution compared with a fifteen minute time-resolution resulted in the estimation of peak PM concentrations that were 20% higher.

Using IAPPEM, a detailed analysis of overall PM contribution from multiple different emission sources, in a variety of different internal locations in a dwelling, has been carried out for the first time, and the effect that both emission source location and internal household configuration has on PM transfer throughout a dwelling has been quantified.

IAPPEM combines a time activity model (which describes how individuals move through different zones in a dwelling) with the physical pollutant model, to create an air pollutant exposure model. The results of the simulations conducted in this study found that calculating exposure based on time averaged profiles is a poor substitute when compared with calculating exposure based on time activity profiles. In each simulated scenario, the time-averaged approach under-predicts mean exposure, in some cases by up to 74.56%. Additionally, the time-averaged approach fails to provide any information on peak exposure, whereas the time activity profile approach provided key information on this aspect.
Declarations

The work in this thesis is based on research carried out at the Centre for Climate and Air Pollution Studies (C-CAPS), School of Physics, National University of Ireland, Galway, Ireland. No part of this thesis has been submitted elsewhere for any other degree or qualification and is all my own work unless referenced to the contrary in the text.

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“The copyright of this thesis rests with the author. No quotations from it should be published without the author’s prior written consent and information derived from it should be acknowledged”.

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Dedication

This thesis is dedicated to my family. To my parents, Seamus and Rosemary, my sister Clairemaire, my aunt and uncle, Zita and Padraig, and of course Nanny and Nora. We have always been a very close family, and at times nearly too much so, but without your endless support and encouragement, I would not be where I am today. You have always encouraged me to simply do my best and that’s all I can do. There is too much to say and not enough words, without you this would not have been possible.

A lifetime is too short, and friends are all too few, to tell you all the reasons that I love you as I do. So all I can simply say is, Thank You.
Philosophy

“Imagination is more important than knowledge. For knowledge is limited to all we now know and understand, while imagination embraces the entire world, and all there ever will be to know and understand.”

Einstein, Albert

“The learning and knowledge that we have, is, at the most, but little compared with that of which we are ignorant.”

Plato

“Live as if you were to die tomorrow. Learn as if you were to live forever.”

Mahatma Gandhi
Acronyms

AER  Air Exchange Rate
EU   European Union
ETS  Environmental Tobacco Smoke
GIS  Geographic Information Systems
IAPPEM  Indoor Air Pollutant Passive Exposure Model
INDAIR  INDoor AIR
PM   Particulate Matter
PM$_1$ Particulate Matter with a particle size fraction of less than 1$\mu$m
PM$_{2.5}$ Particulate Matter with a particle size fraction of less than 2.5$\mu$m
PM$_{10}$ Particulate Matter with a particle size fraction of less than 10$\mu$m
SHS  Second Hand Smoke
UFP  Ultra-Fine Particle
VOC  Volatile Organic Compound
WHO World Health Organisation


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Acknowledgements

First and foremost, I would like to express my deepest and sincerest appreciation to my supervisor, Miriam Byrne. I have benefited hugely from her advice, guidance and inspiration over the last four years. Her open door has been a constant source of encouragement, knowing I could always call in regardless of how simple or complicated the question. In the last couple of weeks especially, her support and dedication was above and beyond the call of duty. I would not have finished when I did without this help, and for this I cannot express enough gratitude or thanks.

A special thanks to Ann, Aoife, Catherine, and Natalie. I would like thank to you all for the knowledge and technical expertise provided over the last number of years. For the help provided in statistical analysis, I want to thank Jerome Sheahan in the School of Mathematics, Statistics and Applied Mathematics.

I would like to acknowledge both Mark Lang and Andy Shearer, who were supportive in providing teaching especially in the last year. A special thanks to Tess Mahoney; whenever I need an answer to a question, she was always able to provide one. To the rest of the staff within the School of Physics for the random chats in the corridors, or advice in the tea room, it has always been appreciated.

To the Anglin, Delaney and Hosey families who’s homes got invaded on more than one occasion, so I could obtain some experimental data for the conferences. They all accommodated me at there own inconvenience.

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To the “Maynoothians”, all who I have known since 1st few weeks of my undergraduate degree, look at us now eh. You provided me with the advice when applying for the PhD. Although there were a few times when I regretted that you ever gave me that advice, but its all been worth it. So thank you Georgina, Aidan, Emma, Eavan, Grainne and Dean. I think we all had many a rant and great nights out together and this is what helped keep me sane over the last four years.

To the “Galwegian” crew, Ann, Catherine, Max, Natalie, Clare, Leah, Cormac, Arlene, Ronan, Gillian, Aonghus, Mags, Mike, Joe, Romain, Conor and Colum. In the last month, the support from some of you (and you know who you are ) has been incredible to say the least, I feel truly privileged to call some of you my friends. To the rest, I honestly think I might have submitted my thesis sooner, if I had not met some of you, whether it was coffee breaks, chats in canteen or tearoom, or the many nights out. That being said, I wouldn’t change any of it. Getting to know and spend time with all of you has made doing a PhD in Galway so enjoyable. I can honestly say I will never forget the week of the Volvo Ocean Race, the two Postgrads Balls or the Galway Races. So thanks guys.
Chapter 1

Introduction and General Literature Review

1.1 Air Pollution

“The presence of contaminant or pollutant substances in the air that interfere with human health or welfare, or produce other harmful environmental effects” (EPA, 2009).

Air pollution had been reported as a health concern as far back as 61AD when Roman philosopher Seneca reported on the conditions in Rome after the invention of the chimney. “As soon as I had escaped the heavy air of Rome and the stench of its smoky chimneys, which when stirred poured forth whatever pestilent vapours and soot they held enclosed, I felt a change in my disposition.” (Miller, 1998).

In 1285, Edward I issued the first “law” (a royal proclamation at the time) prohibiting the use of sea coal in furnaces. However by 1329, it is believed that the ban had either been lifted or had lost its effectiveness (Brimblecombe, 2012).
Pretty et al. (2007) summarised numerous air pollution complaints throughout the 15th-19th from neighbours of bakers, breweries and lime burners. In the early part of the 20th century, the term “smog”, a contraction of the words “smoke” and “fog” was first introduced by Harold Antoine Des Voeux in 1905 at a public health conference in London. The term became popular following a report Des Voeux submitted to the Manchester Conference of the Smoking Abatement League.

On Thursday, the 4th of December 1952 in London, due to a combination of factors, but the most important being an anticyclone halting over London, smoke concentrations of 14,000 $\mu g\ m^{-3}$ were recorded. Over the course of the following four days an estimated death toll of 4,000 was recorded (Fenger, 2009). The estimated death toll from December 1952 to February 1953 exceeded 12,000 deaths because of acute and persisting effects of what become known as the 1952 London Smog (Bell and Davis, 2001). The London Smog was not an isolated incident; similar but less severe events occurred in Donora, Pennsylvania 1948 (Snyder, 1994) and in Engis, Belgium 1930 (Sanford, 2004). These episodes paved the way for modern air pollution legislation.

Both the United Kingdom and the United States introduced Clean Air Acts in 1956 and 1963 respectively. In 1980, the European Commission adopted its first of many directives ((80/779/EEC), 1980) on ambient air quality. These directives cover multiple pollutants. While the first among them examined only smoke pollution and sulphur dioxide, subsequent directives later encompassed many more pollutants, including nitrogen oxide, ozone, particulate matter and carbon monoxide.

In recent years, the World Health Organisation (WHO, 2005) reported that more than two million premature deaths each year can be attributed to the effects
of air pollution from both outdoor and indoor sources. The Thematic Strategy on Air Quality (European Commission, 2005) along with Clean Air for Europe Programme (CAFE) have highlighted particulate matter as the pollutant of most concern. The World Health Organisation (WHO, 2006) concluded that long-term exposure to Particulate Matter (PM) claims an average of 8.6 months from the life expectancy of every European. Every year, more than 280,000 premature deaths are attributed to long-term exposure to PM in 25 countries across the European Union (EU).

1.2 Particulate Matter

1.2.1 Definition of Particulate Matter

Particulate Matter (PM) is a complex mixture of extremely small particles and liquid droplets comprised of a number of components, including soil, acids, metals, organic chemicals and dust particles. These particles range in size from a few angstroms to several hundred micrometres (\(\sim 0.0001 \text{ to } 100\mu\text{m}\)) (Seinfeld and Pandis, 2006). From a health perspective two distinct particle size ranges, PM\(_{10}\) and PM\(_{2.5}\) are of the most concern (WHO, 2005, 2006). The European Directive (European Parliament, 2008) on Ambient Air Quality and Cleaner Air for Europe defines these ranges as follows:

"PM\(_{10}\) shall mean particulate matter which passes through a size-selective inlet as defined in the reference method for the sampling and measurement of PM\(_{10}\), EN 12341, with a 50 % efficiency cut-off at 10 \(\mu\)m aerodynamic diameter."
“PM\textsubscript{2.5} shall mean particulate matter which passes through a size-selective inlet as defined in the reference method for the sampling and measurement of PM\textsubscript{2.5}, EN 14907, with a 50 % efficiency cut-off at 2.5 \(\mu\) m aerodynamic diameter.”

Figure 1.1 demonstrates the different size fractions of PM, compared with the size of a human hair and a grain of sand.

Figure 1.1: The particle diameter of PM\textsubscript{10} and PM\textsubscript{2.5} compared with the size of a human hair or a grain of sand (United States Environmental Protection Agency, 2011b)

The 50% cut off point in the above definitions is due to the fact that samplers are incapable of a precise size differentiation. Particle sizes are relative to a 50% cut off point at a particular aerodynamic diameters (Chow, 1995). However, in practise, most literature only refers to PM\textsubscript{10} or PM\textsubscript{2.5} as particles below the cut off point. For example, PM\textsubscript{2.5} comprises of particles with aerodynamic diameters of 2.5 \(\mu\)m or less (United States Environmental Protection Agency, 2011a).

Inhalable coarse particles are larger than 2.5 \(\mu\)m and smaller than 10 \(\mu\)m in diameter. Fine particles are 2.5 micrometres in diameter and smaller. Coarse and fine particles derive from different sources; coarse particles are formed by mechanical disruptions from sources, such as construction and demolition, farming,
mining, biological sources and the wearing of tyres and brake pads. Fine particles form through chemical reactions or vaporisation from combustion sources, such as coal, oil and wood (Wilson and Suh, 1997).

The main concern to human health is particles with diameters 10 micrometres or smaller because these particles can pass through the throat and nose entering the lungs. Once inhaled, these particles can affect the heart and lungs and cause severe adverse health effects. Inhalable particles are capable of depositing in the upper respiratory tract. Fine particles are capable of penetrating down to the alveolar region of the lungs as seen in Figure 1.2. The particles’ sizes are directly linked to their potential for causing health problems (Vallero, 2007).

Figure 1.2: The locations at which different sized particles deposit in the human respiratory system (Center for Environment, Commerce & Energy, 2010).
1.2.2 Particle Mass and Particle Number

Particulate matter can be classified in two ways: particle number concentrations and mass concentrations. Mass concentrations are calculated in terms of the measured particulate divided by the volume of the gas filtered and measured in terms of $\mu g \ m^{-3}$. Particle number concentrations are measured in terms of the number of particles per unit volume of gas (number cm$^{-3}$) (Kulkarni et al., 2011).

An increasing number of studies highlight that the measurement of Ultra-Fine Particles (UFPs) (particles with a diameter less than 100 $nm$) might provide a better indication of the health effects. This is due to the higher concentrations of these small particles, which have a larger area of contact within the body. The research suggests that UFPs might have greater toxicity and are capable of penetrating deeper into the lungs than larger particles (Health Effects Institute, 2013). However, Kittelson et al. (2002) reported that the lifetime of ultra-fine particles in urban conditions is only on the order of a few minutes. At present all legislation only refers to mass concentrations.

1.2.3 PM Exposure Guidelines

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<th>Annual Mean</th>
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<td><strong>World Health Organisation (WHO, 2005)</strong></td>
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<tr>
<td>PM$_{10}$</td>
<td>50 $\mu g \ m^{-3}$</td>
<td>20 $\mu g \ m^{-3}$</td>
</tr>
<tr>
<td>PM$_{2.5}$</td>
<td>25 $\mu g \ m^{-3}$</td>
<td>10 $\mu g \ m^{-3}$</td>
</tr>
<tr>
<td><strong>European Parliament (2008)</strong></td>
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<td></td>
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<tr>
<td>PM$_{10}$</td>
<td>50 $\mu g \ m^{-3}$</td>
<td>40 $\mu g \ m^{-3}$</td>
</tr>
<tr>
<td>PM$_{2.5}$</td>
<td></td>
<td>25 $\mu g \ m^{-3}$</td>
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Table 1.1: The recommended guidelines for PM$_{10}$ and PM$_{2.5}$ exposure for 24-hour mean and annual mean concentrations.

Table 1.1 summaries the European guidelines at present for PM exposure.
PM\textsubscript{10} values are already in force, while PM\textsubscript{2.5} were set as a target value in 2010 and are becoming limit values in 2015. The annual European limits are still a factor of two higher than recommended by the World Health Organisation (WHO, 2005). In Ireland, the European limits were placed into legalisation (S.I. No. 180/2011, 2011) on 19th April, 2011. The World Health Organisation (WHO, 2010) reported that the World Health Organisation (WHO, 2005) guidelines can also be applied to the indoor environment.

Studies have been unable to identify a threshold, below which, PM concentrations have no adverse health effects on humans (Kappos et al., 2004; Daniels et al., 2000; Brunekreef and Holgate, 2002).

1.2.4 Health Effects

Human exposure is defined as follows: “exposure of person (i) to pollutant concentration (c) is viewed as two events occurring jointly: person (i) is present at a particular location, and concentration (c) is present at the same location” (Ott, 1982). This demonstrates how both the concentration level and the time the individual spends in contact with the pollutant are essential factors.

Exposure to particulate matter has been linked to significant health problems: premature mortality, chronic respiratory disease, increased emergency room visits and hospital admissions, aggravated asthma, acute respiratory symptoms, and decrease in lung function (Trasande and Thurston, 2005; C.A. et al., 2011; Hart et al., 2011; Slaughter et al., 2005).

Pope et al. (2011) reported that lung cancer is associated with high PM\textsubscript{10} exposure (from active cigarette smoking), while cardiovascular deaths are related to low PM exposure (consistent with ambient air pollution and SHS). After analysing
macrophages and cells, Schins et al. (2002) reported finding significant toxicity in coarse particles but not in fine particles. Similarly findings were reported by Kleinman et al. (2003), who found that coarse and fine particles can have significantly different chemical compositions. Dennekamp et al. (2010) reported that a 4.26 \( \mu g \ m^{-3} \) increase in PM\( _{2.5} \) concentrations over 2 days was associated with an 3.6% increase in risk of out-of-hospital cardiac arrest.

Ebelt et al. (2005) suggested that separating total personal PM exposure into their ambient and non-ambient components might provide a better estimation of the impact on human health. Although, the World Heath Organisation WHO (2010) concluded that to date, there is no convincing evidence that PM arising from indoor sources differs in its hazardous nature from outdoors sources.

Short-term increases in PM concentrations raise emergency room admissions from cardiovascular and respiratory symptoms (Schwartz et al., 2003; COMEAP, 2006) and additionally, short-term exposure to PM\( _{10} \) and PM\( _{2.5} \) has been associated with percentage increases in mortality. A 20 \( \mu g \ m^{-3} \) increase in PM\( _{10} \) was associated with an increase of between 0.6 - 0.8% for all causes of daily mortality; Schwartz (2000) analysed the relationship in 10 U.S. cites, Anderson et al. (2005) analysed previous published time series data on single city and multi-city studies, Levy et al. (2000) reanalysis time series PM mortality studies in literature using Empirical Bayes (EB) meta-analysis model. A 10 \( \mu g \ m^{-3} \) increase in PM\( _{2.5} \) was associated with increased daily mortality from 0.6 - 1.2%; Ostro et al. (2006) investigated the relation in nine California, Klemm and Mason (2003) reanalysis data from the Harvard Six cities study and Burnett and Goldberg (2003) investigated the relationship in eight Canadian cities.

Long-term PM exposure was shown to have greater effects on mortality; a 10
\( \mu g \ m^{-3} \) increase in PM\(_{2.5} \) was associated with an increase in the relative risk of mortality of 6.2 - 17\% (Pope et al., 2002; Jerrett et al., 2005).

When comparing the increase in relative risk associated with increased PM concentrations, a literature review by Pope et al. (2011) reported large variability between epidemiological studies. However, all studies highlighted a negative impact on human health with a corresponding increase in PM concentrations. Therefore, in order to accurately assess the health implications, exposure to PM\(_{10} \) and PM\(_{2.5} \) for both short-term and long-term durations are required.

### 1.2.5 Outdoor PM sources

Outdoor PM concentrations are generated from a number of sources, ranging from sea to road salt, crystal and mineral dust, road traffic, point sources (e.g mainly emissions from power plants and industrials emissions) and biomass burning (Belis et al., 2013).

In a review of literature in America and Europe, Pant and Harrison (2013) reported that road traffic contributes 5 - 80\% to outdoor PM concentrations depending on the site and location. The contribution includes the abrasion of brakes, roads and tyre wearing along with the resuspension of material from roads. These processes contribute as much to PM concentrations as emissions from vehicle exhausts (Harrison et al., 2001). However, vehicle exhaust emissions can be considerable higher in developing countries, due to an ageing vehicle fleet and lower levels of fuel efficiency (Colbeck et al., 2011).

Querol et al. (2004) found that PM\(_{2.5} \)/PM\(_{10} \) ratios vary widely depending on the site and regions across Europe. Querol et al. (2004) also reported that kerbside monitoring sites had PM\(_{10} \) concentrations higher than regional and urban...
monitoring sites. Polidori et al. (2009) reported on outdoor monitoring sites failing to account for proximity to outdoor sources. Meteorology conditions such as wind velocity, temperature, relative humidity and precipitation all affect outdoor PM concentrations (Vardoulakis and Kassomenos, 2008).

Outdoor modelling overcomes some of the unexplained variability in outdoor PM concentrations. Moore et al. (2007) predicted ambient PM$_{2.5}$ concentrations across Los Angeles using the Land Use Regression (LUR) model based on a Geographic Information System (GIS) approach explaining 69% of the variance between traffic density, industrial land area and government land. Pilla (2012) developed a GIS model to predict PM$_{10}$ exposure for Dublin city commuters, incorporating background concentration levels, road traffic, point and area sources and dispersion of the pollutants.

1.2.6 Indoor Environment

Koistinen et al. (2004) reported that population exposure assessment of PM$_{10}$ based on outdoor fixed-site monitoring, does not account for the contribution of indoor sources. Ramachandran et al. (2000) studied the Minneapolis - St. Paul metropolitan area and found significant variability within-day and day-to-day in both indoor and outdoor PM$_{2.5}$ concentrations.

Figure 1.3 (Chen and Zhao, 2011) shows that for homes where smoking was present; in all cases indoor/outdoor ratios exceeded 1.0, in all cases highlighting the presence of indoor emission sources. Wallace et al. (2003) analysed PM measurements taken over two week periods in 294 homes across seven cities and their study showed that outdoor particles contribute only 25% of the mean indoor PM concentration. This supports the findings of Monn et al. (1997), who reported
Figure 1.3: Figure from (Chen and Zhao, 2011), that summarises the current literature on the PM$_{2.5}$ indoor/outdoor ratios in studies where more than 20 homes were measured. The numbers in brackets represent the number of homes in each study. The * indicates that smoking was recorded in the study.

that for homes without any indoor emission sources, indoor/outdoor PM$_{10}$ ratios were approximately 0.7, while in homes with indoor emission sources, the indoor/outdoor ratios were greater than 1.8.

Klepeis et al. (2001) reported that most people in North America spend 89% of their time indoors, with 69% spent in the residential environment. Schweizer et al. (2006) reported similar values in European cities, where individuals spend between 56% to 66% of their time in the residential environment. Therefore, the residential environment deserves particular attention.

Outdoor PM concentrations, indoor PM emission sources, internal and external airflow, and PM deposition all influence indoor PM concentrations. (Ferro
et al., 2009; Singer et al., 2002; Ott, 1999).

### 1.2.7 Indoor PM sources

#### 1.2.7.1 Smoking

Numerous studies have highlighted the effects that smoking has on the generation of particulates in indoor environments (Afshari et al., 2005; Koistinen et al., 2001; Brauer et al., 2000). There are two forms of contribution that need to be taken into consideration; the emissions from the main-stream and from the side-stream of cigarettes (Charles et al., 2007; Klepeis et al., 2003; Harrison et al., 1997). Main-stream emissions are inhaled directly from the cigarettes into the lungs, while side-stream emissions are emitted directly into the air from the end of a burning cigarette. Side-stream emissions are the main component of Second Hand Smoke (SHS) and are often referred to as Environmental Tobacco Smoke (ETS).

Invernizzi et al. (2004) compares emissions from cigarettes with an ideally running diesel engine, and showed that three cigarettes smoked in half an hour emitted higher levels of PM$_{10}$, PM$_{2.5}$ and PM$_{1}$ than that of the diesel engine. Brauer et al. (2000) reported on the difference in ETS between emissions from cigarettes and cigar. The cigarettes ranged from 20 - 100 $\mu g \ m^{-3}$; while the cigars ranged from 0 - 70 $\mu g \ m^{-3}$. Hunt et al. (2011) reported PM$_{2.5}$ and PM$_{10}$ concentrations of 26.3 and 37.7 $\mu g \ m^{-3}$ respectively in smoking homes compared with 12.7 and 21.2 $\mu g \ m^{-3}$ concentrations in non-smoking homes. Wallace et al. (2003) reported smoking as the major indoor emission source when examining 101 homes with smokers, increasing indoor PM$_{2.5}$ concentrations by 37 $\mu g \ m^{-3}$.

Additional factors that need to be taken into consideration is the effect of passive smoking and how these particles interact with the room and the sur-
roundings. Studies have shown that it is not just the smoker who is at risk, but also any individual present in the enclosed environment (Koistinen et al., 2001; Ott and Siegmann, 2006). U.S. Department of Health and Human Services (2006) reported that the chances of a nonsmoker developing lung cancer increase by 20-30%, if they live with a smoker. This is of major concern for children living with an individual who smokes (Lofroth, 1993). The influence of the decay rate is another important factor with one study showing that the levels of ETS, remain between 40 - 70% for an hour after one cigarette has been smoked (Singer et al., 2003; Ott et al., 2003). Lofroth (1993) found this is to be of particular importance especially in houses with low ventilation rates.

Studies in Scotland showed how changes in legislation, e.g. the introduction of the smoking ban, reduce levels of PM$_{10}$ by up to 86% in pubs (Semple et al., 2007). Similar studies in Ireland have shown how the levels of benzene dropped dramatically after the introduction of the ban, reaching up to levels of 90% (McNabola et al., 2006). Mulcahy et al. (2005) examined nicotine levels in Irish bars following the smoking ban in workplaces and reported an 83% reduction in airborne nicotine concentrations.

1.2.7.2 Cooking

Studies have highlighted a wide range of emission factors associated with cooking events in the home, depending on the type of food cooked. See and Balasubramanian (2006b) investigated a typical Chinese food stall, reporting that average PM$_{2.5}$ concentrations increased by a factor of 12 from 26.7 $\mu g m^{-3}$ to 312.4 $\mu g m^{-3}$ during cooking hours. McDonald et al. (2003) reported PM$_{2.5}$ emissions rates for charbroiling meats ranged from 4.4 to 11.6 $g/kg$; emission rates varied according
to the cooking conditions, type of meat and the fat content in the meat. In the study, the highest PM$_{2.5}$ emissions occurred while cooking high fat hamburger on an under-fired charbroiler.

Olson and Burke (2006) collated data on the effect of a total of 411 cooking episodes on PM$_{2.5}$ personal exposure and identified that the highest emission sources resulted from burning, grilling and frying food. The influence of cooking style on PM emissions was also seen by Lee et al. (2001), who compared emissions in a Korean barbecue restaurant, a Chinese hot pot restaurant, a Chinese du sum restaurant and a western style canteen. It was found that the Korean style barbecue has extremely high levels of PM$_{10}$ and PM$_{2.5}$, and this has been attributed to the use of frying pans for cooking food. The results showed that over 80% of the respirable particulate matter (PM$_{10}$) measured in the restaurant’s indoor air were fine airborne particles (PM$_{2.5}$).

A study by Sjaastad and Svendsen (2008) showed that frying using margarine gives off up to ten times more total particles than rapeseed oil, soybean and olive oil. Similar studies have shown that the food cooked, oil type, as well as temperature have a significant effect on cooking emissions. It has also been shown that the emission rates are higher for a gas stove than for an electric stove. In terms of deep frying chips, it has been shown that deep frying with olive oil gives off twice the concentration of particles compared with when sunflower oil is used (Buonanno et al., 2009).

See and Balasubramanian (2006a) investigated five different cooking methods; stir-frying, steaming, boiling, deep-frying and pan-frying. The study reported that the deep-frying event has the largest increase in particle number, increasing concentrations 24 times compared with the background concentration.
1.2.7.3 Heating sources

An important aspect of the air pollution levels in any indoor environment is the source of heating that is used within the environment. Most homes contain a primary heating system, that operates on either oil or gas as the fuel source. However in Ireland, many homes have an additional source of heating, consisting of an open or closed solid fuel fire. A study into the effect of domestic fireplaces as a supplementary heating source, found that for PM$_{10}$ in an environment where peat or coal was used in conjunction with smoking, 7 out of 9 houses exceeded the WHO (2005) of threshold (50 $\mu g \, m^{-3}$) between 20 - 30% of the time and, in one case, even reached up to 70% of the time (Guo et al., 2008).

Due to the prolonged winter season in 2012, Bord Na Móna (2013) reported a 28% and 22% increase in briquette and coal sales in the home heating sector, respectively. An information database available by Sustainable Energy Authority of Ireland (2013) reports that 232, 241 and 1035 ktoe (kilo tonnes of oil equivalent) are the sum of the coal products, peat products and petroleum products respectively, were consumed by the Irish residential sector in 2011.

Semple et al. (2012b) studied PM$_{2.5}$ concentrations in 100 Irish and Scottish homes and reported that the 6-hour average (6pm- midnight) was 11 $\mu g \, m^{-3}$, 13 $\mu g \, m^{-3}$ and 29 $\mu g \, m^{-3}$ for wood, coal and peat, respectively. Guo et al. (2008) reported average concentrations of 24 $\mu g \, m^{-3}$ in homes when only examining emissions from a fireplaces. The reported differences between the two studies can be attributed to varying room dimensions, airflow rates and outdoor concentrations.
1.2.7.4 Resuspension

Numerous studies have highlighted the effects of resuspension on indoor air quality, which contributes to the renewal of PM$_{10}$ particles back into the surrounding air (Dennekamp et al., 2001; Andersson, 2009). Abt et al. (2000) showed that both the human activity of cleaning (taking into account dusting, vacuuming and sweeping) and people walking around contribute significantly to indoor PM concentrations.

Vacuuming produces a significant increase in particulate matter indoors. Carpets retain more dust than bare floors, resulting in higher PM resuspension rates (Layton and Beamer, 2009). Many studies have highlighted the effects of both vacuuming and cleaning as significantly contributing to the levels of PM$_{10}$ (Baxter et al., 2007; Koistinen et al., 2001; Gehin et al., 2008).

Thatcher and Layton (1995) found normal human activity has considerable resuspension effects on particles with a diameter larger than 5 $\mu$m but particles with diameters smaller than 5$\mu$m are not readily resuspended. Particles with diameters smaller than 1 $\mu$m exhibit almost no resuspension. Ferro et al. (2004) investigated PM$_{2.5}$ and PM$_{5}$ concentrations following a number of human activities. The study found that vacuuming had the highest PM$_{2.5}$ source strength, while two people sitting on furniture and walking around the room had the highest PM$_{5}$ source strength. The maximum PM$_{2.5}$ and PM$_{5}$ source strengths contribute up to 27% and 84% respectively, when compared to the source strength from smoking a cigarette.

1.2.7.5 Additional indoor Sources

Ozkaynak et al. (1996), analysing the data obtained in the PTEAM study found that outdoor concentrations, smoking and cooking only explained 92% of PM$_{2.5}$
indoor concentrations and 85% of PM$_{10}$ indoor concentrations. (Wallace, 1996) reported similar findings.

He et al. (2004) examined mass concentration emission rates for smoking and frying compared with candles and hair dryer. The hair dryer emission rate was two orders of magnitude lower compared with cooking or smoking. Similar results were obtained for particle number concentrations from smoking, cooking, candle burning, aroma lamps, hair spray, hair dryers and incense sticks (Hussein et al., 2006; Afshari et al., 2005).

Although vacuuming contributes to resuspension, Lioy et al. (1999) reported fine particle emissions from the motor within the vacuum cleaner, ranged from 0.028 to 176 µg min$^{-1}$. Many studies have shown that when candles are extinguished, more particles are produced than during the burning phase (Fine et al., 1999; Afshari et al., 2005).

1.2.8 Deposition

Deposition is the gravitational settling of particles. From the perspective of human health, deposition is a positive benefit, reducing indoor airborne concentrations of particles. However, this only applies to inhalation of particles; Byrne (2009) summarises the additional dermal effects if the particles are radioactively contaminated, due to deposition on human skin, hair and clothing.

Howard-Reed et al. (2003) investigated deposition rates of particles ranging 0.3 to 10 µm in diameter in an occupied and unoccupied house; the results reported that the source of the particles appeared to have little influence on deposition rates, only the particles size affected the deposition rates and not their composition. Long et al. (2001) reported that indoor particle deposition is a highly variable
process, strongly dependent on particle size but also on specific site conditions such as air turbulence, temperature, surface materials and room volume.

Lai (2006) investigated the influence of electrostatic forces on particles in the size range of 3.5 to 9.0 μm in a test chamber; the results found that pre-treating a test chamber with anti-static material decreased deposition velocity onto glass by 68%. To the author’s knowledge, no studies have examined the effect of electrostatic forces on particle deposition in the home.

Lai (2002) describes Equation (1.1) use to calculate the decay rate in a rectangular chamber, based on different orientations of surface and their respective surface area.

\[
\lambda_d = \frac{V_{dw} \cdot A_w + V_{du} \cdot A_u + V_{dd} \cdot A_d}{V}
\]  

where \( \lambda_d \) is the deposition rate, \( A_w, A_u, \) and \( A_d \), are the total areas for the vertical deposition, upward-facing horizontal surface and downward facing horizontal surface, respectively. \( V_{dw}, V_{du} \) and \( V_{dd} \) are the deposition velocities for vertical wall, upward-facing and downward facing horizontal surfaces. However, most experimental studies did not measure deposition velocity onto surfaces (Lai, 2002), so Equation (1.1) is transformed into Equation (1.2).

\[
\lambda_d = \bar{V}_d \sum A
\]  

where \( \lambda_d \) is the deposition rate, \( \sum A \) is the total deposition area, \( V \) is the volume of the room and \( \bar{V}_d \) is the area-weighted average deposition velocity.

Byrne et al. (1995) measured particles in the size range of 0.7, 2.5, 4.5 and 5.4 μm in an experimental chamber with smooth walls. Byrne et al. (1995) found the larger particles had the higher deposition velocity and a greater floor deposition.
Fogh et al. (1997) reported similar results after carrying out experiments in four different real houses with particles in the range of 0.5 - 5.5 µm. Thatcher et al. (2002) reported that deposition rates were found to be higher in furnished rooms compared with unfurnished rooms.

Lai (2002) reviewed indoor particle deposition and stated that the amount of research on particle deposition indoors which is directly related to human health is far less than the amount of research that focuses on deposition in a small diameter tube channel. No additional reviews of indoor particle deposition have been conducted over the last eleven years to suggest that this view no longer holds true.

1.2.9 Ventilation

Ventilation or air exchange is the exchange of air between indoors and outdoors. This occurs through a number of processes: natural ventilation, mechanical ventilation, infiltration or exfiltration. Natural ventilation involves the movement of air through vents, windows or doors. Mechanical ventilation involves heating, ventilation and air conditioning (HVAC) devices. Infiltration and exfiltration represents undesired flow of air in and out of the structure. Air exchange influences indoor PM concentrations in two ways. Air exchange brings outdoor pollutant concentrations inside a structure if ambient concentrations are higher concentrations; while if ambient concentrations are lower the air exchange reduces the indoor concentrations (Singer et al., 2002; Baxter et al., 2007; Vallero, 2007).

Studies have shown a considerable difference in air exchange between mechanical ventilation and natural ventilation, where meteorological conditions have a greater impact on air exchange in natural ventilated homes (Niachou et al., 2005;
Evola and Popov, 2006). The effects of meteorological conditions on naturally ventilated houses, affect both infiltration and ventilation rates (Ni Riain et al., 2003); the opening of windows has been seen to double exchange rates over extended periods, while tripling air exchange rates over short periods (Howard-Reed et al., 2002; Johnson et al., 2004). Studies have shown the effect of increased ventilation on reducing PM concentration during and after cooking. This is mainly in the form of an extractor fan being turned on during the event (Sjaastad and Svendsen, 2008).

Under the European directive (2010/31/EU, 2010), all new building must adhere to minimum energy performance requirements. Irish legislation (S.I. No. 259/2008, 2008) refers to reducing heat loss in building through increasing the air tightness. Wilkinson et al. (2009) reported that changes in energy efficiencies in domestic homes may impact on indoor air quality, due to increasing air-tightness without alternative methods of reducing indoor generated pollutants. Bone et al. (2010) highlights that the driver for more energy efficiency homes will harm occupant health.

Hänninen et al. (2004) reported on external Air Exchange Rates (AERs) for residential homes in four European cities: 1.3 ± 1.1 h\(^{-1}\) in Athens, 0.83 ± 0.46 h\(^{-1}\) in Basel, 0.81 ± 0.85 h\(^{-1}\) in Helsinki and 0.75 ± 0.43 h\(^{-1}\) in Prague. Crump et al. (2005) measured AERs in 511 homes built after 1995 in the UK and found average values of 0.44 ± 0.11 h\(^{-1}\) in winter and 0.62 ± 0.19 h\(^{-1}\) in summer. In a study of 101 homes in Finland, using mechanical ventilation for heat recovery, an average AER of 0.4 h\(^{-1}\) was reported (Kurnitski et al., 2007). Dimitroulopoulou (2012) summaries that 0.5 h\(^{-1}\) is frequently used in ventilation standards and regulations across Europe.
In addition to the external airflow between the inside microenvironment and the outdoors, there is also an interzonal airflow between internal rooms. This varies considerably depending on the internal layout and the opening patterns of the doors (Ferro et al., 2009; Ott et al., 2003), and therefore interontal airflow are largely under-commented.

1.2.10 Microenvironments

Duan (1982) defined a MicroEnvironment (ME) to be a “chunk of air space with homogenous pollutant concentration”. This can be individual rooms in a building, such as the home, an indoor workplace, or a transport vehicle. In addition to assessing exposure in the residential environment, exposure should be assessed in all microenvironments where individuals are present. An individual’s exposure varies as they move through a range of different microenvironments, as seen in Figure 1.4.

![Figure 1.4: Variations in an individual’s exposure as they move through a series of microenvironments. Figure from McNabola et al. (2011)](image)

A transport vehicle (e.g. Bus, car, trains) can be one of the microenvi-
ment in which people regularly spend their time (Koistinen et al., 2001; Kaur and Nieuwenhuijsen, 2009). An important factor when dealing with buses is the stop and start times, the patterns of the windows opening and closing and the resuspension factor as people enter and exit the bus (Song et al., 2009). Studies have highlighted the variations in cycling and taking the car or bus, showing that the key factors are the time travelled and the route rather the actual mode of transport (Nasir and Colbeck, 2009; Gulliver and Briggs, 2004; Moschandreas and Saksena, 2002).

Wu et al. (2005) examined a sample of 20 asthmatic children and reported that despite children only spending 16.4% of their time in school, they receive on average 29.2% of their PM exposures at school; although this study did not distinguish between different indoor emission sources in the home environment. Additional studies have examined children’s exposure during school hours, finding that resuspension is a major factor leading to increased indoor concentrations (Fromme et al., 2008; Zwoździak et al., 2013).

Wu et al. (2012) reports on 37 small and medium commercial buildings in California; while the majority had I/O ratios for PM of less than 1.0, some buildings had I/O ratios greater than 1 indicating indoor emission sources. However, it should be noted that buildings with a I/O ratio of less than one can still contain indoor emission sources.

In addition, Burton et al. (2000) reported on PM$_{10}$ and PM$_{2.5}$ measurements for 100 office buildings; I/O ratios greater than 1.0 were found in 11 buildings for PM$_{10}$ and 9 buildings for PM$_{2.5}$. Further, a study by He et al. (2007) showed that sub-micron particles were produced by printers in an office building, with 27% of the printers emitting high concentrations.
1.2.11 Seasonal Variation

It is important to take into account the effects that seasonal variations have on PM concentrations. Studies have shown that it is important to account for more than one season; with Monn and Schaeppi (1993) measuring higher levels of PM$_{2.5}$ in winter than in summer (Mohammadyan and Ashmore, 2005) probably due to the change in lifestyle. Massey et al. (2012) reported on higher PM concentrations in winter months, explaining the increase due to more human activity and additional heating indoors, in combination with reduced wind speed reducing the decay rate of particles.

Sørensen et al. (2005) investigated PM$_{2.5}$ personal exposure in Copenhagen in a period covering seasonal variation, and recorded higher PM$_{2.5}$ concentrations in the winter months. Some of the factors that influence the variations were less time spent outdoors during the winter months and difference in air exchange rates due to temperature differences. Peng et al. (2005) reported higher PM$_{10}$ exposure in north-west U.S. cites during winter months, compared with southern cities.

1.3 Indoor Air Pollution Models

As described in the previous section, indoor PM exposure is subject to a range of factors, such as the presence of emission sources, the dynamic process of indoor deposition and resuspension; and the behaviour of individuals which cause them to progress through PM-containing microenvironments.

In epidemiological studies, where large population groups are examined, it is often impractical or too expensive to obtain indoor or personal exposure measurements. Computational models are regarded as being a valid substitute. Modelling
approaches are effective at separating out the contribution of indoor emission source from outdoor generated air pollution; this approach allows for effective exposure reduction strategies to be devised.

However, indoor exposures need to be further separated into the contributions of indoor sources and the penetration of outdoor pollution, for the proper management of air quality to be fully evaluated. Modelling approaches are essential if this is be effectively achieved (Dimitroulopoulou et al., 2000).

Whilst gravimetric sampling has proven to be a robust technique, it is unable to measure variations in PM concentrations for periods shorter than a few hours. Photometric samplers are capable of capturing real-time measurement of both indoor and outdoor air pollution, capturing information ranging from a few seconds to several days. Photometric samplers are adaptable to a range of different microenvironments, however the equipment is typically cumbersome and the experimental data are limited to specific scenarios. Additionally, calibration factors are needed to convert photometric measurement into mass concentrations (Zhu et al., 2007). However, calibration factors can only be applied to the entire datasets leading to uncertainties when estimating PM mass concentrations. The home environment represents complex scenarios, due to PM concentration arising from multiple emissions sources, therefore computational modelling is an essential tool in determining indoor PM concentrations.
Modelling approaches can be important in identifying the contribution of individual sources and there is a need for a comprehensive indoor exposure model which includes the wide range of indoor aerosol source terms and indoor aerosol transport processes (e.g. deposition and re-suspension) that exist and that can be validated by measurement data.

Modelling approaches calculate the change in indoor pollutant concentrations by considering the infiltration of outdoor air pollution, the generation of air pollution indoors and its transport between rooms, and the indoor deposition of air pollution as depicted in Figure 1.5.

In Figure 1.5 $C_0$ represents the outdoor concentration while $C_1$ represents the pollutant concentrations in Room 1. As mentioned in Section 1.2.4 pollutant concentrations forms the basis for determining exposure and Figure 1.6 demonstrates...
how both a physical pollutant model must be combined with a time-activity model in order to predict exposure to air pollutants.

Figure 1.6: A representation of how a pollutant exposure model is composed of a physical pollutant transport model and a personal activity model.
1.4 Physical Pollutant Models

Modelling approaches can be categorised into five main groups and relevant applications of these approaches are reviewed below.

a) **Statistical regression/ Measurement based microenvironments.** The temporal and spatial resolution of this modelling approach is based on experimental data. The experimental data is typically direct measurements or adjusted ambient concentrations. The application of such models is typically large scale exposure studies. Baxter et al. (2007) gathered indoor and outdoor measurements over 3-4 days for 43 homes in Boston. Collecting home characteristics and individual behavioural habits via a questionnaire, the data were analysed using regression analysing to quantify key factors that affected concentrations. However, this approach cannot examine any conditions outside of the experimental data and is therefore limited.

b) **Mass balance models** are based on physics principles incorporating flow and dispersion between indoors and outdoors. Mass balance models assume uniform distribution and instantaneous mixing throughout the microenvironments. Typically whole buildings are represented as a single microenvironment. These models use ambient concentrations, air exchange rates and indoor emission sources to predict concentrations.

Hayes (1989) developed the Personal Air Quality Model (PAQM) to estimate exposure for population groups moving through different microenvironments, including the office, home and vehicle; the home environment was represented as a single microenvironment. A Sequential Cigarette Exposure Model (SCEM) was used to predict ETS from cigarette smoke in a single
microenvironment (Ott et al., 1992). Klepeis et al. (1996) used a mass balance model to predict minute-by-minute ETS concentrations in the smoking lounge at two separate international airports. Despite these modelling efforts, it is still regarded that mass balance model are too simplistic to represent the complexity of the home environment.

c) **Multi-zone models** are based on similar principles to mass balance models. The main difference is that a large number of rooms are present in the building, incorporating additional information room dimensions, household layout and building operation schedules.

Feustel (1999) reported on the development of the COMIS model, which was designed to calculate airflows in both single and multi-zone buildings. The model calculates the airflow based on temperature differences, ventilation systems, wind direction and velocity and shape of the building. This model only examined airflows within a building and did not predict pollutant concentrations. Later work by (Sohn et al., 2007) combined COMIS with an indoor aerosol dynamics model (MIAQ4) to predict the transport of ETS in a three room chamber. The study obtained excellent agreement between observed and predicted concentrations in all three rooms.

Ott et al. (2003) predicted carbon monoxide and respirable suspended particles from ETS emissions in two interconnecting compartments. A number of models have simulated the effects of SHS in two and three zone models (Miller and Nazaroff, 2001; Ott, 1999), and while these model are accurate at predicting SHS concentration, they still ignore contributions from other indoor sources. Klepeis and Nazaroff (2006) developed a model for simulating SHS, nicotine and carbon monoxide in two hypothetical houses. The first
was divided into a typical four-room layout and the second was dominated by a single large space. SHS concentrations on a minute by minute basis were simulated. This model encompassed door positions, windows positions, locations of the smoker and non-smokers and the duration of the smoking period.

d) **Sub-zonal models** are similar to both mass balance models and the multi-zone models but with additional sub-zones in each room, for greater capture of within-room gradients. This modelling approach has scales ranging from 1m to the size of a room, and requires detailed information of sub-room air flows. The application of this type of model is to simulate large internal spaces or strong localised emission sources.

Stewart and Ren (2006) enhanced the COMIS model to create COwZ (COMIS with sub-Zones). COwZ divides a single room into 30 - 200 smaller sub-zones. The model predicted air flows, temperature and concentrations for each individual zone, and the model examined volatile organic compounds (VOCs) emissions from carpet, painted walls and ceilings. The model did not examine any particle distributions. It was noted that the program execution time is linked directly to the number of zones and sub-zones been modelled. Stewart and Ren (2003) reports that while COwZ has improved the accuracy of predictions of the transport contaminants in a room, the model requires additional input data and setup time and requires the specific number, shape and size of each sub zone, although the study report fails to detail how extensive is the additional time required.

e) **Computational Fluid Dynamics (CFD) models** operate on grid sizes of 1cm - 1m with temporal resolution of a few seconds. CFD models cap-
ture important information on airflow and concentrations, requiring specific building information. They are useful for studying fine scale concentrations but not for population exposure studies.

Stamou and Katsiris (2006) simulated indoor airflow and heat transfer in an office room using a CFD model. The time step was set to 1.5 s and for a total of a 25 minute simulation. The total computational time was approximately 23 hours. Zhang and Zhang (2007) investigated VOCs under re-circulated ventilation systems, and the time resolution was set at 0.1 s for a single room. The computation time for a single simulation was approximately 240 hours on a supercomputer.

CFD can provide very detailed information of contaminant transport in a zone. Boris (2002) used CFD models to simulate the dispersion of a contaminant in downtown Portland, Oregon providing very high spatial resolution of contaminant concentrations. However, compared to multi-zone or mass balance models, CFD calculations take hours or even days of computing time (Settles, 2006). Steskens et al. (2013) investigated the transport of heat, air and moisture transport in a single room, reporting that the computational time of the simulations varied between several hours up to a few days.

Multi-zone models, at present, represent the “Goldilocks” of modelling approaches for estimating pollutant concentrations. Multi-zone models can accurately simulate indoor air pollutant concentrations for realistic homes without the drawbacks of extensive computational run-time and the complexity needed in sub-zonal or CFD models.

CONTAM (NIST, Gaitherburg, MD, USA) is a multi-zone, airflow and transport pollutant model. CONTAM is based on a graphical user interface that allows
the user to draw floor plans and use icons to represent airflow paths, ventilation system and emission sources (NIST, 2011). CONTAM has been used in over 54 published applications (NIST, 2010) making it the most widely used indoor air pollutant model, and allowing it to be used as a baseline for comparison with other models.

Lai (2004) used CONTAM to simulate ETS and cooking emissions in zones ranging in number from a one to a seven zone apartment, examining particle exposure and engineering control strategies. In addition, Fabian et al. (2012) using CONTAM predicted NO$_2$ and PM$_{2.5}$ concentrations in a low income family home in Boston, for use in health based intervention. The study included stove use, smoking, cooking, building leakiness and exhaust fans. Furthur, Shrubsole et al. (2012) using CONTAM, predicted PM$_{2.5}$ indoor exposure in London dwellings, comparing “present day” and 2050 conditions. Adjustments were made for variations in outdoor PM concentrations, changes in smoking habits and reduced airflow between indoor and outdoors to meet reductions in greenhouse gas emissions.
1.5 Exposure Models

Ashmore and Dimitroulopoulou (2009) summarise exposure models as being of three main types:

a ) The first approach forms the basis of population-based epidemiological studies; these studies mainly estimate exposure based on outdoor air pollutant concentrations and ignore differences between indoor and outdoor concentrations and the individual’s behavioural habits.

b ) The second approach, which is also used in epidemiological studies, combines exposure measurements in empirical models to predict the exposure for individuals or groups. This is analogous to the statistical regression/ measurement approach for indoor concentrations, but in this case it is applied to exposure instead of concentrations. These models are limited to specific scenarios and do not adjust well with changes in time or location.

c ) The third approach is mechanistic modelling rather than empirical modelling. The mechanistic models are based on the microenvironment approach where an individual’s exposure is the combination of the concentrations in various microenvironments, factoring in the time spent in each microenvironment. Mechanistic models combine time activity profiles with the concentrations in microenvironments, where concentrations can be based on data from measured or modelled studies.

(Kousa et al., 2001) used a regression model based on indoor, outdoor and workplace NO$_2$ concentrations using time-weighted averages for durations spent in
each microenvironment and explained 74% of the exposure variation. Burke et al. (2001) simulated daily PM$_{2.5}$ exposures for the population living in Philadelphia using a probabilistic model to predict daily averages for the population. Although, this model included cooking as an emission source in the residential environment it only calculated a 12 hour average for each microenvironment.

While the regression analysis approach is acceptable for large scale population exposure estimations, it can not be applied to the development of exposure reduction strategies as it can only provide exposure estimations based on previously collected data.

Ott (1983) used the above approach in developing the SHAPE model designed to calculate the exposures to carbon monoxide to a given population, and highlighted how human activities influence exposure. Dimitroulopoulou et al. (2001b) predicted nitrogen dioxide exposure in the UK. The model calculates exposure using a similar modelling approach in homes and workplaces, linking to a time-activity model for three different populations groups, representing the homemaker, a schoolchild and an office worker. Klepeis and Nazaroff (2006) developed a model for simulating SHS, nicotine and carbon monoxide in two hypothetical houses. This allowed for the examination of a smoker and a non-smoker in the home and estimated exposure of an individual who follows a smoker around the house. The approach allows for time series analysis of individual exposure, indicating the times and locations where highest exposure occurred.
1.5.1 The Rationale for Probabilistic Modelling of Indoor PM Concentrations

The main question that must be addressed is that, if CONTAM is user friendly and validated, whether there is a need for further model development in indoor air pollutant exposure assessment. The rationale in the present work for future development is based on the premise that in order to accurately assess the health effects to any population group, accurate concentrations in each microenvironment must first be known. This is beyond the scope of the CONTAM model.

An important point made by Fabian et al. (2012) is that there are challenges imposed on simulations due to the large variation in emission strengths. CONTAM is a deterministic model, allowing the user to only select a single emission rate for any one simulation. For example, He et al. (2004) calculated a PM$_{2.5}$ emission for frying of $2.68 \pm 2.18 \text{ mg min}^{-1}$, over several minutes; this would amount to large variations in concentration. CONTAM fails to consider any variation in modelling parameters and this severely limits its use in representing complex indoor environments as every input parameter in an air pollution model has a variation associated with it.

Probabilistic modelling is one approach that overcomes the above issue (Dimitroulopoulou et al., 2008). Probabilistic models uses probability density functions to simulate a range of possible values for each parameter, assessing the range of likely outcomes when specific details are unavailable. This can encompass uncertainties in experimental obtained data, but can also encompass uncertainties in the selection of appropriate modelling parameters between studies.
The INDIAR model is a probabilistic model designed to simulate the frequency distribution within population groups of indoor exposure to air pollutants (Dimitroulopoulou et al., 2001b) and its development and main features are reviewed in the next section. Figure 1.7 highlights the difference between a deterministic approach and a probabilistic approach.

1.5.2 The History of the INDAIR Model’s Development

The INDAIR model originated as a dynamic multi-compartment computer model (Dimitroulopoulou et al., 2001a) designed to simulate the physical processes determining indoor pollutant concentrations as a function of outdoor concentrations, indoor emission rates and building characteristics. The model was originally parameterized for UK homes and workplaces, linking to a time-activity model to calculate the personal exposure of the individual with respect to NO₂. It combined two types of modelling. Firstly a physical model, was used to calculate
hourly indoor air pollutant concentrations for different microenvironments (ME) by solving a set of differential equations using a fourth-order Runge-Kutta scheme. Secondly, personal exposures were calculated by combining the movement of individuals through a series of microenvironments with ME concentrations. This was done by using a linear combination of concentrations in MEs, weighted by the time spent in each of those individual MEs. The physical model was broken down into individual MEs, the domestic ME used a two compartmental approach stimulating the kitchen and living room (which used the assumption that the kitchen and living room were directly connected together). The office, classroom and outdoors MEs were each represented by a single compartment. The exposure model was based upon the activity profiles that were constructed to represent the individual. This model was coded in Visual Basic.

The above model later became the INTAIR model (Dimitroulopoulou et al., 2001b), which was parameterized to estimate the contribution of passive smoking to PM$_{10}$ concentrations in typical UK homes. The smoking was assumed to take place when the room was unoccupied and the occupant was present once the smoking has ceased. Again the model, used a two compartment approach to represent the domestic setting of the kitchen and living room.

Later, the model was enhanced to become the INDAIR (Dimitroulopoulou et al., 2006) model in 2006. It was used to predict air pollutant concentrations in the home ME in the UK, using probability functions for four different pollutants. These pollutants were NO$_2$, CO, PM$_{10}$ and PM$_{2.5}$, under three different emission based scenarios. These encompassed a scenario where there were no sources present, a scenario where there was cooking taking place and thirdly where smoking took place. This model added an additional ME as an extension of the previous
models, the bedroom. The model was run for a combination of different residential locations, depending on the season and whether it was during the week or at the weekend.

The INDAIR model provided an internationally advanced probabilistic modelling tool which could assess the contribution of indoor and outdoor sources, to the concentrations in the indoor environment. It could simulate the upper percentiles of the population exposure to air pollutants, or the proportion of the population exposed to concentrations above critical health thresholds.

The INDAIR model still has a number of limitations, which prevent its full application to representative indoor environments. The review of the literature has highlighted the large variation in emission rates, deposition rates and air exchange rates that are possible, and while the probabilistic approach accounts for these, the temporal and spatial limits restrictions of INDAIR limit its scope. Specifically,

- The INDAIR model has a 15 minute time resolution, which fails to capture accurate details on peak concentrations during short emissions durations.
- The INDAIR model has only three rooms in the home environment, which provides only a limited representation of a realistic home environment.
- The INDAIR model incorporates a time-weighted average airflow, which fails to account for any short-term variations in airflow; these can cause significant fluctuations in indoor air pollutant concentrations.
- The INDAIR model incorporates a single emission source, and cannot therefore account for the combination effect of multiple indoor emission sources, which have been found to be present in homes.
• The INDAIR model includes only three additional microenvironments. To accurately assess exposure, a realistic number of additional microenvironments is needed to include individual’s movements over a 24-hour period.

• The INDIAR model does not include a time activity model, and therefore cannot predict individual exposure.
1.5.3 Objectives of the Present Work and Thesis Layout

The overall objective of this thesis was the development of a comprehensive probabilistic model capable of simulating more realistic representations of a home environment, with a minute-by-minute time resolution to fully capture details on peak and mean concentrations incorporating a number of indoor emission sources.

This thesis is presented in the format of a series of scientific papers. Chapters 2, 3 and 4 provide details on the specific adaptations made to the INDAIR model. Chapter 5 focuses on experimental data for model validation. Chapter 6 includes a general discussion, additional modelling applications and proposals for potential air pollutant reduction strategies that can be explored.

Chapter 2 details the adaptations made to the physical pollutant model; specifically, improving the time resolution, increasing the number of rooms in the household and incorporating 12 simultaneously-operating emission sources. Simulations are provided to highlight the contribution from outdoor PM concentrations and indoor emission sources as well as demonstrate the combination effect of multiple indoor emission sources.

Chapter 3 compares simulations based on a time-weighted averaged airflow with a variable airflow, simulating airflow variations by the opening and closing of internal doors. The simulations are validated based on an experimental comparison; examining airflow variations at 1, 2, 5, 10, 15 and 30 minute intervals.

Chapter 4 details the combination of the physical pollutant model with a time
activity model to create an accurate air pollutant exposure model, illustrating exposure variability for four individuals present in the same household. Each individual experiences different PM$_{10}$ and PM$_{2.5}$ peak and mean exposure depending on their movements through the dwelling.
Chapter 2

Development of a Multi-zone Multi-source Computational Model to Evaluate Passive Exposure to Particulate Matter Indoors
Development of a Multi-zone Multi-source Computational Model to Evaluate Passive Exposure to Particulate Matter Indoors

J. A. McGrath\textsuperscript{a}, M. A. Byrne\textsuperscript{a}, M.R. Ashmore\textsuperscript{b}, A. Terry\textsuperscript{b}, C. Dimitroulopoulou\textsuperscript{c}

\textsuperscript{a}School of Physics, National University of Ireland Galway, Galway, Ireland  
\textsuperscript{b}Environment Department, University of York, York  
\textsuperscript{c}Department of Mechanical Engineering, University of West Macedonia, Siatista and Bakola, Kozani, Greece

Abstract

This paper highlights the development of the probabilistic model (IAPPEM), which predict PM\textsubscript{10} and PM\textsubscript{2.5} concentrations in residential environments. A number of features are detailed and justified through simulated comparison, which are shown to be necessary when modelling indoor PM concentrations. A one minute resolution predicts up to 20\% higher concentrations compared with a 15 minute resolution. A modified PM\textsubscript{10} deposition method, devised to independently analyse the PM\textsubscript{2.5} fraction of PM\textsubscript{10}, predicts up to 58\% higher mean concentrations compared with the previous method. A comparison between different indoor emission sources was made here analysing the total PM contribution in terms of emission source and duration, in combination with the effect of source location has on concentrations throughout the environment. A 24 hour sample profile is simulated based on sample data, designed to demonstrate the combined functionality of the model, predicting PM\textsubscript{10} and PM\textsubscript{2.5} peak concentrations up to 1107.76 ± 175.1 and 649.58 ± 84.9 μg m\textsuperscript{-3} respectively, while predicting PM\textsubscript{10} and PM\textsubscript{2.5} mean concentrations up to 242.16 ± 20.16 and 151.38 ± 11.96μg m\textsuperscript{-3} respectively.

Keywords: Modelling, Indoor Air Quality, PM\textsubscript{10}, PM\textsubscript{2.5}, Emissions
2.1 Introduction

Particulate Matter (PM) has become a major environmental concern because of its known impacts on human health; ranging from aggravated asthma, chronic respiratory disease, increased emergency room visits and hospital admissions, acute respiratory symptoms, decrease in lung function and even premature mortality (Abbey et al., 1995; Burnett et al., 1999; Dockery et al., 1992; Fairley, 1999). Michaels and Kleinman (2000) highlighted that brief exposure to high levels of pollution can have more severe health effects than long term average exposure, and this evidence suggests that greater risk reduction could be achieved by controlling the PM average concentrations on an hourly scale rather than simply tightening control of the 24h averages.

Indoor PM concentrations are affected by infiltration of outdoor particles, meteorological parameters and seasonal effects, indoor activities of occupants, emissions from indoor PM sources, removal of particulates by deposition, the dilution of indoor PM through external and internal air exchange and internal house layout (Ferro et al., 2009; Singer et al., 2002; Ott, 1999).

Ebelt et al. (2005) showed that the health effects associated with indoor-generated particles differ from those arising from particles of outdoor origin. Research has highlighted that indoor activities contribute to PM concentrations through combustion events such as smoking, frying, solid fuel fire and use of candles and incense (Ott and Siegman, 2006; Ozkaynak et al., 1996; He et al., 2004) and resuspension actives; such as walking, dusting or vacuuming (Ferro et al., 2004). The indoor environment deserves particular attention since people spend approximately 89% of their time there (Klepeis et al., 2001).
Chapter 2: Development of a Multi-zone Multi-source Computational Model to Evaluate Passive Exposure to Particulate Matter Indoors

Whilst gravimetric sampling has proven to be a robust technique, it is unable to measure variations in PM concentrations for shorter than a period of a few hours. Photometric samplers are capable of capturing real-time measurement over timescales of a few seconds, but the equipment is typically cumbersome and therefore limited to specific scenarios. However, photometric samplers can overestimate PM concentrations if a proper calibration is not applied (Zhu et al., 2007) and therefore the accurate comparison of PM concentration from multiple sources in the home environment is complex and requires computational modelling as an essential tool.

Previous studies have modelled air pollution concentrations in the home microenvironment (Klepeis and Nazaroff, 2006; Miller and Nazaroff, 2001; Ott, 1999), in the context of second-hand smoke (SHS), with simulations ranging from a single zone to four compartments. Fabian et al. (2012) predicted NO\textsubscript{2} and PM\textsubscript{2.5} concentrations for a four room home scenario in Boston using CONTAM (NIST, Gaithersburg, MD, USA), but highlighted the challenges imposed due to the large variation in emission strengths. Dimitroulopoulou et al. (2006) developed a probabilistic model to calculate PM\textsubscript{10} and PM\textsubscript{2.5} concentrations, which overcomes some of these issues. Burke et al. (2001) simulate daily PM\textsubscript{2.5} exposures for the population living in Philadelphia using a probabilistic model to predict daily averages for the population. The model does include cooking and emission sources in the residential environment but it only calculated a 12 hour average for each microenvironment.

As PM exposure varies substantially with an individual’s personal behaviour, it is imperative that a computational model is developed to simulate a variety of individuals’ behaviour patterns and hence predict indoor PM concentrations.
While current models have demonstrated the capability to predict indoor PM$_{10}$ and PM$_{2.5}$ concentrations, the need still exists for a comprehensive computational model capable of simulating more realistic representations of a home environment. The model should encompass a full range of possible emission sources situated in a range of different room locations, and simulated on a sufficiently short time scale so as to capture details on peak and mean concentrations and accurately determine the time duration for emission concentrations to fully decay.

2.2 Model Description

2.2.1 History of the Model

The current model, titled Indoor Air Pollutant Passive Exposure Model (IAPPEM), is based on the INDAIR model described by Dimitroulopoulou et al. (2006), which is an advanced probabilistic modelling tool that can evaluate the contribution of indoor and outdoor sources to the air pollution concentration in the indoor environment.
IAPPEM calculates the change in indoor pollutant concentrations by solving the differential Equation 2.1; considering the infiltration of outdoor air pollution, the generation of air pollution indoors and its transport between rooms, and the indoor deposition of air pollution.

\[
\frac{dC_k}{dt} = \frac{\lambda_{0k}}{V_k} (f_k * C_0 - C_k) - v_g \left( \frac{A_k}{V_k} \right) C_k + \frac{Q_k}{V_k} + \sum_{i=1}^{n} \lambda_{ik} \left( C_i - C_k \right) \tag{2.1}
\]

Equation 2.1 is solved for each \( k \), where \( k \) represents each individual room. Subscripts of 0, 1 and 2 are used to represent outside, room 1 and room 2 for different parameters. \( C_k \) represents the concentrations of the pollutant in that room \((\mu g m^{-3})\), where \( C_0 \) represents the outdoor concentration \((\mu g m^{-3})\). \( f_k \) represents the building filtration factors between the outdoor and that room. \( v_g \) is the deposition velocity of the pollutant \((m hr^{-1})\). \( \lambda_{(ik)} \) is the interzonal airflow between internal rooms, e.g. \( \lambda_{(12)} \) represents the airflow of pollutants from room 1 into room 2, and \( \lambda_{(0k)} \) the airflow from outside into room \( k \) \((m^3 hr^{-1})\). \( A_k \) is the surface area of room \( k \) \((m^2)\). \( V_k \) is the volume of room \( k \) \((m^3)\). \( Q_k \) is the indoor emission rate of the pollutant in room \( k \) \((\mu g hr^{-1})\).

### 2.2.2 Adaptations

To date, in the domestic building context, the INDAIR model has been applied to the simulation of PM concentrations arising from (i) passive smoking in a two-room dwelling (Dimitroulopoulou et al., 2001a) and (ii) to the generation of \( \text{NO}_2 \), CO, \( \text{PM}_{10} \) and \( \text{PM}_{2.5} \) concentrations in a three-room dwelling, with the pollutants arising from separately-occurring smoking and cooking events (Dimitroulopoulou et al., 2006). In developing the IAPPEM model, where the aim is
to demonstrate capability in simulating PM concentrations arising from a number of simultaneously-operating sources in a number of different rooms of a multi-room building, several adaptations were made to the INDAIR approach, and these are

a) Modifying the approach for calculating the PM deposition rate. Indoor air pollution models to date, calculate PM\textsubscript{10} decay rates independently of PM\textsubscript{2.5}. However, for combustion sources, PM\textsubscript{2.5} can constitute up to 90% of PM\textsubscript{10} by particle mass concentrations (as discussed in Section 2.2.4.3) which implies that the PM\textsubscript{10} deposition velocity needs to encompass the entire particle size fraction. The approach taken in this paper is to separate the contribution of PM\textsubscript{2.5} from PM\textsubscript{10}, calculate the deposition rate for both size fractions independently, and then recombine for the calculation of the deposited PM\textsubscript{10} concentration. This approach is expressed in Equation 2.2, where $\gamma$ represents the deposition rate.

\begin{align*}
((PM_{10} - PM_{2.5}) \times \gamma_{PM_{10}}) + (PM_{2.5} \times \gamma_{PM_{2.5}})).
\end{align*}

(2.2)

2.2.3 Simulations

Each scenario is simulated with a one minute time resolution over a 24-hour period, starting at 08:00 am in a 11 room dwelling (the layout is discussed in Section 2.2.4.2). The outdoor PM concentrations, room dimensions and the external air exchange rate were held constant throughout all the simulations. Emission rates were varied depending on the emission source, and internal air exchange
rates varied depending on whether the doors were open or closed. The separate simulations that were carried out are detailed below:

i ) To demonstrate the effect of modifying the deposition rate calculation in the model, and also the effect of the improved time resolution, a smoking event at 12:00 was simulated in the kitchen with the internal doors closed.

ii ) Six independent emission events, each starting at 12:00, were simulated in the kitchen with both the doors opened and the doors closed. Analysis of the PM concentrations in each case yielded information on the overall PM contribution from each source and permitted a calculation of the time required for an emission event to result in an exceedance of an internationally recognised exposure limit value.

iii ) To demonstrate the effect that source location and household layout has on PM concentrations, and to highlight the importance of a realistic representation of the home environment, six independent emission events were simulated with each emission source in a different location. Emission periods commenced at 12:00 for each simulation.

iv ) A sample 24-hour activity profile was simulated, to demonstrate the combination of all the adaptations to the model.
2.2.4 Model Parameterisation

To simulate the scenarios mentioned above, it is necessary to assign values for the input parameters in the model. It should be noted that while Irish data was selected solely for demonstration purposes, this does not reflect any geographical limitations in the model.

2.2.4.1 Outdoor PM Concentration

Ambient outdoor PM$_{10}$ and PM$_{2.5}$ concentrations were estimated from data supplied by the Irish EPA (2013). The data corresponded to hourly concentrations from a background air monitoring station, collected over a two week period in September 2012, located 3 km from Dublin city centre. Geometric mean and standard deviations were extrapolated from this dataset to generate a 24-hour profile. 24-hour mean PM$_{10}$ and PM$_{2.5}$ concentrations were $17.14 \pm 0.33 \mu g \ m^{-3}$ and $9.77 \pm 0.24 \mu g \ m^{-3}$ respectively.

2.2.4.2 Room Dimensions

Data on room dimensions were collected from a sample of 50 houses from an online estate agent (Daft, 2013), in accordance with property classification on the Irish Census (Irish Central Statistics Office, 2013). Room dimensions are listed in Table 2.1. The house layout is as follows: the hallway was connected to each bedroom, the bathroom, the kitchen and the living room. The kitchen alone was connected to a utility room and dining room, and Bedroom 1 alone was connected to an en-suite bathroom. The household layout can be seen in Figure 2.2.
Figure 2.2: A representation of the household layout.

<table>
<thead>
<tr>
<th>Location</th>
<th>Wall Surface Mean (StDev) (m²)</th>
<th>Floor Surface Mean (StDev) (m²)</th>
<th>Volume Mean (StDev) (m³)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Utility Room</td>
<td>22.8 (4.7)</td>
<td>5.1 (1.8)</td>
<td>12.7 (4.4)</td>
</tr>
<tr>
<td>Kitchen</td>
<td>46.8 (11.5)</td>
<td>21.9 (11.4)</td>
<td>54.7 (28.6)</td>
</tr>
<tr>
<td>Dining Room</td>
<td>40.5 (5.6)</td>
<td>16.4 (4.6)</td>
<td>40.9 (11.5)</td>
</tr>
<tr>
<td>Hallway</td>
<td>35.2 (9.8)</td>
<td>9.7 (4.8)</td>
<td>24.1 (11.9)</td>
</tr>
<tr>
<td>Living Room</td>
<td>44.6 (8.7)</td>
<td>19.9 (7.5)</td>
<td>49.7 (18.8)</td>
</tr>
<tr>
<td>Bathroom</td>
<td>26.2 (5.6)</td>
<td>6.5 (2.4)</td>
<td>16.3 (6.0)</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>37.4 (6.4)</td>
<td>14.0 (5.0)</td>
<td>35.1 (12.5)</td>
</tr>
<tr>
<td>Bedroom 2</td>
<td>35.6 (6.1)</td>
<td>12.7 (4.7)</td>
<td>31.8 (11.7)</td>
</tr>
<tr>
<td>Bedroom 3</td>
<td>38.8 (5.6)</td>
<td>12.1 (3.8)</td>
<td>30.3 (9.6)</td>
</tr>
<tr>
<td>Bedroom 4</td>
<td>34.4 (6.5)</td>
<td>11.9 (4.8)</td>
<td>29.7 (12)</td>
</tr>
<tr>
<td>En Suite bathroom</td>
<td>18.4 (2.1)</td>
<td>2.9 (0.6)</td>
<td>7.3 (1.6)</td>
</tr>
</tbody>
</table>

Table 2.1: Summary of the arithmetic mean and standard deviations (in brackets) of the room dimensions as an input parameter for the model.
2.2.4.3 Indoor Sources of PM

Table 2.2 summarises PM$_{10}$ and PM$_{2.5}$ indoor emission rates, together with the corresponding emission durations; emission values and durations have been compiled from literature and if multiple references were found, mean and standard deviations were calculated. When insufficient information was found in literature for PM$_{10}$ emission rates, they were extrapolated based on a relationship between PM$_{10}$ and PM$_{2.5}$.

<table>
<thead>
<tr>
<th>Source and duration in minutes</th>
<th>PM$_{10}$ Emission Strength (mg h$^{-1}$)</th>
<th>PM$_{2.5}$ Emission Strength (mg h$^{-1}$)</th>
<th>References</th>
<th>Comments</th>
</tr>
</thead>
<tbody>
<tr>
<td>Single Cigarette (9)</td>
<td>145.5 ± 42.7</td>
<td>96.0 ± 28.2</td>
<td>(Brauer et al., 2000; He et al., 2004; Ozkaynak et al., 1996; Ott, 1999; Ott et al., 2003; Jiang et al., 2011)</td>
<td>Ozkaynak et al. (1996) found that PM$<em>{2.5}$ comprised 66% of PM$</em>{10}$.</td>
</tr>
<tr>
<td>Frying (8)</td>
<td>189.3 ± 163.5</td>
<td>160.8 ± 130.8</td>
<td>He et al. (2004)</td>
<td>Lee et al. (2001) estimate that PM$<em>{2.5}$ encompasses 80% of PM$</em>{10}$ for cooking events.</td>
</tr>
<tr>
<td>Incense Stick (40)</td>
<td>108.9 ± 118.0</td>
<td>98.0 ± 106.9</td>
<td>(See and Balasubramanian, 2011; Lee and Wang, 2004; Jiang et al., 2011; Jetter et al., 2002).</td>
<td></td>
</tr>
<tr>
<td>Solid Fuel Fire(210)</td>
<td>39.6 ± 37.8</td>
<td>12.0 ± 12.6</td>
<td>(Guo et al., 2008).</td>
<td></td>
</tr>
<tr>
<td>Candles (15)</td>
<td>5.3 ± 2.8</td>
<td>5.3 ± 2.8</td>
<td>(Fine et al., 1999; Pagels et al., 2009).</td>
<td>PM$<em>{2.5}$ emission were assumed to be the same for PM$</em>{10}$ (Zai et al., 2006).</td>
</tr>
</tbody>
</table>

Table 2.2: PM$_{10}$ and PM$_{2.5}$ emission rates, with corresponding emission duration for each source. Comments are provided when extrapolating a PM$_{10}$ emission rate based on PM$_{2.5}$ data.
2.2.4.4 Deposition Velocities & Penetration Factors

In test chamber studies, Byrne et al. (1995) found that approximately 60 - 65% of the particles between 0.7 to 2.5 µm deposited on the floor and that 35-40% deposited on the walls, while 75 - 80% of the larger size particles (up to 5.4 µm) deposited on the floor with the remainder depositing on the walls. Byrne et al. (1995) found that particles with a diameter of 0.7 and 2.5 microns had deposition velocities of $4.1 \times 10^{-5}$ and $6.2 \times 10^{-5} \text{ m s}^{-1}$ respectively. Based on these values, a PM$_{2.5}$ deposition velocity of $5.15 \times 10^{-5} \text{ m s}^{-1}$ was selected. This was shown to be in agreement with simulations by McGrath et al. (2013b).

In the study by Byrne et al. (1995) the largest particles present were 5.4 µm and for that reason, a PM$_{10}$ deposition velocity value of $3.9 \times 10^{-4} \text{ m s}^{-1}$ was selected from the PTEAM study (Ozkaynak et al., 1996).

Ozkaynak et al. (1996) and Thatcher et al. (2003) found, in natural ventilated dwellings, a penetration factor of 1.0 for PM$_{10}$ and PM$_{2.5}$.

2.2.4.5 Airflow Rates

All the rooms are naturally ventilated with external windows and external doors assumed to remain closed for the entire duration. Crump et al. (2005) reported external air exchange rates ranging from $0.44 \pm 0.11$ air changes per hour, which were converted into volumetric air flow rates ($\text{m}^3 \text{ h}^{-1}$) for each room. Internal doors were simulated as either all closed or all open. In the absence of a detailed Irish study, U.S. volumetric air flow rates were used as a substitute, with airflow rates for the closed door condition ranging from 0.4-5.1 $\text{m}^3 \text{ h}^{-1}$ and with airflow rates for the open door condition ranging from 60-245 $\text{m}^3 \text{ h}^{-1}$ (Ott et al., 2003; Miller and Nazaroff, 2001; Ferro et al., 2009).
Chapter 2: Development of a Multi-zone Multi-source Computational Model to Evaluate Passive Exposure to Particulate Matter Indoors

2.3 Results

2.3.1 Time Resolution

The importance of choosing an appropriate model time resolution is illustrated by analysing the comparison between a one minute resolution and a longer time resolution. A 15 minute time resolution was selected for demonstrative purposes. Each resolution independently simulating the kitchen smoking event described in Section 2.2.3 was independently simulated.

To directly compare a one minute and 15 minute resolution, both smoking durations are set to 15 minutes i.e. the value of nine minutes, as specified in Table 2.2, was not used. For the one minute resolution test case, the peak PM$_{10}$ and PM$_{2.5}$ concentrations were $650 \pm 269 \ \mu g \ m^{-3}$ and $445 \pm 192 \ \mu g \ m^{-3}$ respectively, whereas for the 15 minute resolution, peak PM$_{10}$ and PM$_{2.5}$ concentrations were $569 \pm 223 \ \mu g \ m^{-3}$ and $399 \pm 177 \ \mu g \ m^{-3}$ respectively. This difference arises as the 15-minute resolution case overestimates PM decay, whereas the one minute resolution case calculates PM decay at individual minutes and gives a better estimate.

If the nine minute smoking duration, as specified in Table 2.2, is used, the effect of carrying out a simulation of 15 minutes duration is that peak PM$_{10}$ and PM$_{2.5}$ concentrations of $342 \pm 142 \ \mu g \ m^{-3}$ and $242 \pm 109$ are calculated, as compared with peak PM$_{10}$ and PM$_{2.5}$ concentrations of $410 \pm 176 \ \mu g \ m^{-3}$ and $280 \pm 123 \ \mu g \ m^{-3}$, associated with a one minute time resolution. This is because an additional six minutes of PM decay are incorporated in the 15 minute resolution case, as it averages over a period that is longer than the actual emission period.
2.3.2 Deposition

The above simulation also highlights the effect of incorporating a modified deposition rate equation in the model, as was described in Section 2.2.2. Figure 2.3 shows PM$_{2.5}$ and PM$_{10}$ concentrations, estimated by the modified method, together with PM$_{10}$ concentrations estimated by the original method.

Figure 2.3 shows how the original method results in PM$_{10}$ concentrations decaying faster than PM$_{2.5}$, as PM$_{10}$ is dominated by smaller particles for combustion sources (PM$_{2.5}$ represents 66% of the PM$_{10}$ for smoking emissions (Section 2.2.4.3)). By the new PM$_{10}$ deposition method, calculated PM$_{10}$ concentrations remain higher than PM$_{2.5}$ concentrations, which makes greater physical sense. For this simulation, the 24-hour mean PM$_{10}$ concentrations calculated by the method that incorporates the original decay equation was $20 \pm 4 \, \mu g \, m^{-3}$ compared with $32 \pm 3 \, \mu g \, m^{-3}$ calculated by the revised method; this represents a difference of 58%.

![Figure 2.3: Differences in PM$_{10}$ concentrations estimated using the modified PM$_{10}$ decay method, the original PM$_{10}$ decay method and the corresponding PM$_{2.5}$ concentration during a smoking event in the kitchen when doors are closed.](image-url)
2.3.3 The Relative Importance of Different Sources

To demonstrate the relative contributions of different sources to overall PM contribution, a series of different simulations were run in the kitchen, each one incorporating one of the following combustion events: smoking, frying, solid fuel fire, incense and candles; the duration of each event, and the appropriate emission rate, is detailed in Table 2.2. In each case, separate simulations were run with the internal kitchen door closed and open, and “no source” simulations were also run. Tables 2.3 and 2.4 show PM\textsubscript{10} and PM\textsubscript{2.5} concentrations contributed by different sources. It can be seen that PM\textsubscript{10} and PM\textsubscript{2.5} concentrations are both higher when the kitchen door is closed, compared with the open door case, and this is most evident in the 24-hour mean concentration data. Although frying has a marginally shorter emission duration than smoking, frying’s higher emission rate generates higher PM concentrations. The incense stick and the solid fuel fire both have lower emission rates than either smoking or frying, but the longer emission durations impact substantially on PM concentrations.

The highest peak PM\textsubscript{10} and PM\textsubscript{2.5} concentrations are associated with the burning of the incense stick are at least twice those associated with all other sources. Interestingly, higher PM\textsubscript{10} peak concentrations are observed for solid fuel fire usage than for frying, although frying has the higher PM\textsubscript{2.5} peak concentrations. Although the PM\textsubscript{10} incense peak concentration is twice that of the solid fuel fire peak, there is no substantial difference between the mean PM\textsubscript{10} concentration for either source.
### Chapter 2: Development of a Multi-zone Multi-source Computational Model to Evaluate Passive Exposure to Particulate Matter Indoors

<table>
<thead>
<tr>
<th>Source</th>
<th>Doors Configuration</th>
<th>Peak Concentration $\mu g m^{-3}$</th>
<th>24 Hour Mean Concentration $\mu g m^{-3}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>No Emission Source</td>
<td>Opened</td>
<td>$10 \pm 0$</td>
<td>$10 \pm 0$</td>
</tr>
<tr>
<td>No Emission Source</td>
<td>Closed</td>
<td>$11 \pm 1$</td>
<td>$10 \pm 1$</td>
</tr>
<tr>
<td>Smoking</td>
<td>Opened</td>
<td>$247 \pm 63$</td>
<td>$17 \pm 1$</td>
</tr>
<tr>
<td>Smoking</td>
<td>Closed</td>
<td>$407 \pm 181$</td>
<td>$31 \pm 3$</td>
</tr>
<tr>
<td>Frying</td>
<td>Opened</td>
<td>$329 \pm 102$</td>
<td>$19 \pm 1$</td>
</tr>
<tr>
<td>Frying</td>
<td>Closed</td>
<td>$527 \pm 247$</td>
<td>$39 \pm 4$</td>
</tr>
<tr>
<td>Incense Stick</td>
<td>Opened</td>
<td>$592 \pm 92$</td>
<td>$43 \pm 3$</td>
</tr>
<tr>
<td>Incense Stick</td>
<td>Closed</td>
<td>$1442 \pm 474$</td>
<td>$112 \pm 10$</td>
</tr>
<tr>
<td>Solid Fuel Fire</td>
<td>Opened</td>
<td>$236 \pm 19$</td>
<td>$44 \pm 2$</td>
</tr>
<tr>
<td>Solid Fuel Fire</td>
<td>Closed</td>
<td>$671 \pm 88$</td>
<td>$114 \pm 13$</td>
</tr>
<tr>
<td>Candles</td>
<td>Opened</td>
<td>$22 \pm 3$</td>
<td>$11 \pm 0$</td>
</tr>
<tr>
<td>Candles</td>
<td>Closed</td>
<td>$34 \pm 10$</td>
<td>$12 \pm 1$</td>
</tr>
</tbody>
</table>

Table 2.3: Peak and mean PM$_{10}$ concentrations in the kitchen during 12 independent emission scenarios. Six different emission sources are simulated, each with the doors open and then with the doors closed.

<table>
<thead>
<tr>
<th>Source</th>
<th>Doors Configuration</th>
<th>Peak Concentration $\mu g m^{-3}$</th>
<th>24 Hour Mean Concentration $\mu g m^{-3}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>No Emission Source</td>
<td>Opened</td>
<td>$8 \pm 0$</td>
<td>$7 \pm 0$</td>
</tr>
<tr>
<td>No Emission Source</td>
<td>Closed</td>
<td>$8 \pm 0$</td>
<td>$7 \pm 0$</td>
</tr>
<tr>
<td>Smoking</td>
<td>Opened</td>
<td>$167 \pm 43$</td>
<td>$12 \pm 1$</td>
</tr>
<tr>
<td>Smoking</td>
<td>Closed</td>
<td>$279 \pm 126$</td>
<td>$24 \pm 2$</td>
</tr>
<tr>
<td>Frying</td>
<td>Opened</td>
<td>$252 \pm 78$</td>
<td>$15 \pm 1$</td>
</tr>
<tr>
<td>Frying</td>
<td>Closed</td>
<td>$406 \pm 192$</td>
<td>$32 \pm 4$</td>
</tr>
<tr>
<td>Incense Stick</td>
<td>Opened</td>
<td>$542 \pm 88$</td>
<td>$38 \pm 3$</td>
</tr>
<tr>
<td>Incense Stick</td>
<td>Closed</td>
<td>$1327 \pm 458$</td>
<td>$108 \pm 11$</td>
</tr>
<tr>
<td>Solid Fuel Fire</td>
<td>Opened</td>
<td>$98 \pm 6$</td>
<td>$22 \pm 1$</td>
</tr>
<tr>
<td>Solid Fuel Fire</td>
<td>Closed</td>
<td>$296 \pm 35$</td>
<td>$56 \pm 4$</td>
</tr>
<tr>
<td>Candles</td>
<td>Opened</td>
<td>$20 \pm 3$</td>
<td>$8 \pm 0$</td>
</tr>
<tr>
<td>Candles</td>
<td>Closed</td>
<td>$31 \pm 10$</td>
<td>$9 \pm 1$</td>
</tr>
</tbody>
</table>

Table 2.4: Peak and mean PM$_{2.5}$ concentrations in the kitchen during 12 independent emission scenarios. Six different emission sources are simulated, each with the doors open and then with the doors closed.
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2.3.4 Exceedance of 24 hour guideline values

<table>
<thead>
<tr>
<th>Source in kitchen</th>
<th>Doors Configuration</th>
<th>PM$_{10}$ (Hours:Minutes)</th>
<th>PM$_{2.5}$ (Hours:Minutes)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Smoking</td>
<td>Closed</td>
<td>3:00</td>
<td>4:16</td>
</tr>
<tr>
<td>Smoking</td>
<td>Opened</td>
<td>1:03</td>
<td>1:50</td>
</tr>
<tr>
<td>Incense Stick</td>
<td>Closed</td>
<td>6:32</td>
<td>8:19</td>
</tr>
<tr>
<td>Incense Stick</td>
<td>Opened</td>
<td>3:51</td>
<td>5:26</td>
</tr>
<tr>
<td>Frying</td>
<td>Closed</td>
<td>3:34</td>
<td>5:03</td>
</tr>
<tr>
<td>Frying</td>
<td>Opened</td>
<td>1:25</td>
<td>2:30</td>
</tr>
<tr>
<td>Candles</td>
<td>Closed</td>
<td>0:00</td>
<td>0:26</td>
</tr>
<tr>
<td>Candles</td>
<td>Opened</td>
<td>0:00</td>
<td>0:00</td>
</tr>
<tr>
<td>Solid Fuel Fire</td>
<td>Closed</td>
<td>7:15</td>
<td>8:21</td>
</tr>
<tr>
<td>Solid Fuel Fire</td>
<td>Opened</td>
<td>4:56</td>
<td>5:36</td>
</tr>
</tbody>
</table>

Table 2.5: The duration for which PM$_{10}$ and PM$_{2.5}$ concentrations for 24-hour guidelines values are exceeded after 10 independent emission scenarios. Five different emission source are simulated, each with the doors closed and then with the doors open.

Table 2.5 presents the calculated time for which each single PM emission event, as described in Table 2.2, results in an exceedance of the WHO (2005), 24-hour mean PM$_{10}$ and PM$_{2.5}$ concentration guidelines of 50 $\mu$g m$^{-3}$ and 25 $\mu$g m$^{-3}$ respectively. Although these guidelines represent 24-hour mean concentrations and not threshold limits, they give indications of the extent of PM concentration decay in each case. In all cases, PM$_{2.5}$ concentrations exceed the guidelines for a longer time than PM$_{10}$, as a consequence of the lower PM$_{2.5}$ limit and the lower PM$_{2.5}$ deposition velocity. With doors closed between rooms, the limits are exceeded for longer times than when doors are open. While incense and the solid fuel fire have similar PM$_{10}$ mean concentrations, PM$_{2.5}$ mean concentrations for the solid fuel fire are approximately half that of the incense, but there is no noticeable difference between the time for which each exceed the WHO guidelines. This is due to
the longer duration of the solid fuel burning event, which is of a timescale such that the incense-originating PM can decay to a similar concentrations to those occurring that at the end of the solid fuel fire emission period.

### 2.3.5 The effect of source location on PM concentration in various rooms.

In this section, simulated PM concentrations in various rooms of the dwelling, according to whether they contain an emission source, or are subject to the influence of a source in another room, are examined. Table 2.6 shows estimated PM concentrations in Bedroom 1, associated with a series of separate emission events, of durations detailed in Table 2.2, in the bedroom itself, and in four other rooms; in each case, the emission event is set to commence at 12:00. When these data are compared with those shown in Tables 2.3 and 2.4, it can be seen that all PM concentrations in the bedroom are lower, except in the case where candles are burnt in the bedroom itself.

When the doors remain closed; frying, incense and solid fuel fire burning results in only a marginal increase, relative to background (no source scenario) in peak and mean concentrations in the bedroom, as the room containing the sources is some distance away. However, the smoking event gives rise to higher concentrations in the bedroom, due to the source being in the adjoining room.

When the doors are opened between rooms, the concentrations shown in Table 2.6 differ substantially from the mean PM$_{2.5}$ concentrations in Bedroom 1 that were calculated based on the simulations described in Section 2.3.3, namely, smoking $10 \pm 0 \mu g m^{-3}$, frying $11 \pm 1 \mu g m^{-3}$, incense stick $23 \pm 2 \mu g m^{-3}$, solid fuel fire $15 \pm 1 \mu g m^{-3}$ candles $8 \pm 0 \mu g m^{-3}$.
Table 2.6: Peak and mean PM$_{2.5}$ concentrations in Bedroom 1 as different emission sources are independently simulated in different locations throughout the dwelling.

<table>
<thead>
<tr>
<th>Source and Location</th>
<th>Doors Configuration</th>
<th>Peak PM$_{2.5}$ Concentrations $\mu g m^{-3}$</th>
<th>Mean PM$_{2.5}$ Concentrations $\mu g m^{-3}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>No Source</td>
<td>Closed</td>
<td>7 ± 1</td>
<td>7 ± 0</td>
</tr>
<tr>
<td>No Source</td>
<td>Opened</td>
<td>8 ± 0</td>
<td>7 ± 0</td>
</tr>
<tr>
<td>Smoking in Hallway</td>
<td>Closed</td>
<td>24 ± 5</td>
<td>10 ± 1</td>
</tr>
<tr>
<td>Smoking in Hallway</td>
<td>Opened</td>
<td>56 ± 10</td>
<td>11 ± 0</td>
</tr>
<tr>
<td>Frying in Kitchen</td>
<td>Closed</td>
<td>8 ± 1</td>
<td>8 ± 0</td>
</tr>
<tr>
<td>Frying in Kitchen</td>
<td>Opened</td>
<td>37 ± 6</td>
<td>11 ± 1</td>
</tr>
<tr>
<td>Incense Stick in Dining Room</td>
<td>Closed</td>
<td>8 ± 0</td>
<td>8 ± 0</td>
</tr>
<tr>
<td>Incense Stick in Dining Room</td>
<td>Opened</td>
<td>107 ± 14</td>
<td>21 ± 2</td>
</tr>
<tr>
<td>Fire in Living Room</td>
<td>Closed</td>
<td>10 ± 1</td>
<td>8 ± 0</td>
</tr>
<tr>
<td>Fire in Living Room</td>
<td>Opened</td>
<td>56 ± 3</td>
<td>16 ± 1</td>
</tr>
<tr>
<td>Candles In Bedroom 1</td>
<td>Closed</td>
<td>44 ± 11</td>
<td>10 ± 1</td>
</tr>
<tr>
<td>Candles In Bedroom 1</td>
<td>Opened</td>
<td>31 ± 5</td>
<td>8 ± 0</td>
</tr>
</tbody>
</table>

The smoking scenario results in elevated PM$_{2.5}$ concentrations, with peak smoking concentrations being only 48% lower than those associated with burning an incense stick; previously (Table 2.4) these were 79% lower. Peak concentrations that are similar to those arising from the solid fuel fire and higher concentrations than those associated with frying are observed. Although smoking still results in the second lowest mean concentrations, there is a only a marginal difference between concentrations.

Interestingly, the solid fuel burning scenario, with the fire now situated in the living room, results in elevated concentrations in the bedroom, although still connected by the same number of rooms to the room containing the source, relative to the previous case where the fire was in the kitchen. The two additional rooms connected to the kitchen in combination with the living room’s smaller volume, results in higher concentrations in the living room, allowing greater particle infiltration into the hall and subsequently into the bedroom.
While the incense stick is in the dining room (one room further away from the bedroom than is the kitchen) decreased concentrations are observed in the bedroom. With both solid fuel fire and the incense stick in the same location (i.e. the kitchen, reference Table 2.4), the incense stick resulted in 54% higher concentrations in the bedroom than the fire (as can be seen from the data shown earlier in this section), but by adjusting only the location of both sources, increases in concentrations reduce to only 28%.

2.4 A simulation of simultaneous combustion events.

A sample 24-hour activity profile, incorporating a number of simultaneously occurring emission events, is presented in Table 2.7, and Tables 2.8 and 2.9 show the corresponding estimated PM$_{10}$ and PM$_{2.5}$ peak and mean concentrations, in the various rooms of the dwelling.

The peak living room PM concentration occurs at 19:39, prior to the end of the solid fuel fire emission, demonstrating the combined effect of this and the smoking source, and also the fact that the PM decay rate exceeds the emission rate from the solid fuel fire. While peak kitchen concentrations remain broadly similar to those shown in Tables 2.3 and 2.4 (single frying scenario), marginally increased concentrations, due to infiltration from the living room, are discernible. Although peak kitchen concentrations occur after the second frying event, concentrations differ by less than 5%, compared with the first frying event.
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<table>
<thead>
<tr>
<th>Time</th>
<th>Location</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>10:30-10:39</td>
<td>Living Room</td>
<td>Single Smoking Event</td>
</tr>
<tr>
<td>12:00-12:09</td>
<td>Living Room</td>
<td>Single Smoking Event</td>
</tr>
<tr>
<td>13:30-13:38</td>
<td>Kitchen</td>
<td>Single Frying Event</td>
</tr>
<tr>
<td>17:00-17:09</td>
<td>Living Room</td>
<td>Single Smoking Event</td>
</tr>
<tr>
<td>18:00-21:30</td>
<td>Living Room</td>
<td>Solid Fuel Fire</td>
</tr>
<tr>
<td>18:00-18:08</td>
<td>Kitchen</td>
<td>Single Frying Event</td>
</tr>
<tr>
<td>19:30-19:39</td>
<td>Living Room</td>
<td>Single Smoking Event</td>
</tr>
<tr>
<td>22:00-22:09</td>
<td>Living Room</td>
<td>Single Smoking Event</td>
</tr>
</tbody>
</table>

Table 2.7: A 24-hour profile showing a number of different emission sources present at different times and locations throughout the dwelling.

<table>
<thead>
<tr>
<th>Source</th>
<th>Time of Peaks (Hours:Minutes)</th>
<th>Peak PM$_{10}$ Concentration ($\mu g \ m^{-3}$)</th>
<th>Mean PM$_{10}$ Concentration ($\mu g \ m^{-3}$)</th>
<th>Time exceeding WHO guidelines (Hours:Minutes)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Doors Closed</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>18:08</td>
<td>556 ± 234</td>
<td>68 ± 6</td>
<td>7:29</td>
</tr>
<tr>
<td>Dining Room</td>
<td>19:11</td>
<td>30 ± 4</td>
<td>16 ± 1</td>
<td>0:00</td>
</tr>
<tr>
<td>Hallway</td>
<td>20:35</td>
<td>84 ± 10</td>
<td>33 ± 2</td>
<td>5:13</td>
</tr>
<tr>
<td>Living Room</td>
<td>19:39</td>
<td>1104 ± 172</td>
<td>260 ± 20</td>
<td>16:07</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>21:48</td>
<td>17 ± 1</td>
<td>13 ± 1</td>
<td>0:00</td>
</tr>
<tr>
<td>Doors Open</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>13:38</td>
<td>363 ± 100</td>
<td>54 ± 3</td>
<td>8:00</td>
</tr>
<tr>
<td>Dining Room</td>
<td>18:20</td>
<td>163 ± 30</td>
<td>47 ± 2</td>
<td>7:54</td>
</tr>
<tr>
<td>Hallway</td>
<td>19:47</td>
<td>205 ± 20</td>
<td>60 ± 2</td>
<td>10:34</td>
</tr>
<tr>
<td>Living Room</td>
<td>19:39</td>
<td>642 ± 83</td>
<td>105 ± 4</td>
<td>11:14</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>20:08</td>
<td>165 ± 12</td>
<td>52 ± 2</td>
<td>9:45</td>
</tr>
</tbody>
</table>

Table 2.8: The peak and mean PM$_{10}$ concentrations in different rooms after simulating the 24-hour profile scenario with doors opened and doors closed. Additional information is provided on the times of peak concentrations and duration in each room for which the WHO 24-hour guideline values are exceeded.

The highest mean PM concentrations are observed in the living room, as the solid fuel fire is the largest PM contributor modelled in this scenario (Section 2.3.3). As previously discussed (Section 2.3.5), the closing of doors between adjoining rooms can be an effective method of reducing infiltration of PM con-
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Centrations; this is evident in the current simulation set, where only marginally increased PM concentrations are observed in Bedroom 1.

In the open doors scenario, the hallway experiences higher mean PM concentrations than the kitchen, although the hallway contains no emission sources; this is more evident for PM$_{10}$ than for PM$_{2.5}$, due to the solid fuel fire’s larger PM$_{10}$ emission rate.

Table 2.9: The peak and mean PM$_{2.5}$ concentrations in different rooms after simulating the 24-hour profile scenario with doors opened and doors closed. Additional information is provided on the times of peak concentrations and duration in each room for which the WHO 24-hour guideline values are exceeded.

<table>
<thead>
<tr>
<th>Source</th>
<th>Time of Peak Concentration (Hours:Minutes)</th>
<th>Peak PM$_{2.5}$ Concentration ($\mu g m^{-3}$)</th>
<th>Mean PM$_{2.5}$ Concentration ($\mu g m^{-3}$)</th>
<th>Time exceeding WHO guidelines (Hours:Minutes)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Doors Closed</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>18:08</td>
<td>437 ± 182</td>
<td>58 ± 6</td>
<td>10:01</td>
</tr>
<tr>
<td>Dining Room</td>
<td>19:19</td>
<td>26 ± 4</td>
<td>13 ± 1</td>
<td>0:35</td>
</tr>
<tr>
<td>Hallway</td>
<td>20:34</td>
<td>56 ± 6</td>
<td>25 ± 2</td>
<td>10:55</td>
</tr>
<tr>
<td>Living Room</td>
<td>19:39</td>
<td>595 ± 100</td>
<td>167 ± 11</td>
<td>18:24</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>22:48</td>
<td>12 ± 1</td>
<td>10 ± 1</td>
<td>0:00</td>
</tr>
<tr>
<td><strong>Doors Open</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>18:08</td>
<td>282 ± 76</td>
<td>41 ± 2</td>
<td>13:20</td>
</tr>
<tr>
<td>Dining Room</td>
<td>18:21</td>
<td>130 ± 24</td>
<td>37 ± 2</td>
<td>12:26</td>
</tr>
<tr>
<td>Hallway</td>
<td>19:47</td>
<td>123 ± 12</td>
<td>42 ± 2</td>
<td>14:26</td>
</tr>
<tr>
<td>Living Room</td>
<td>19:39</td>
<td>361 ± 53</td>
<td>64 ± 2</td>
<td>14:34</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>20:08</td>
<td>104 ± 8</td>
<td>38 ± 2</td>
<td>14:11</td>
</tr>
</tbody>
</table>

In the open door scenario, the higher concentrations in the hallway also affects the kitchen’s PM decay. One hour after the second frying event, the kitchen’s PM$_{10}$ concentrations decayed to 32% of the peak concentration, compared to 18% in the single frying scenario (Table 2.3) resulting in overall higher mean concentrations in the kitchen; this effect is not as obvious in the closed doors scenario, as the kitchen is located two closed doors away from the hallway.
2.5 Sensitivity Analysis

Table 2.10 highlights the results of the sensitivity analysis, simulating a 20% independent decrease in each input parameter. The results analyse peak and mean concentrations in the kitchen were analysed while the doors were open, and concentration differences between simulations with original parameters values and simulations with the 20% decreased parameter values were compared.

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Peak PM(_{10}) µg m(^{-3})</th>
<th>Mean PM(_{10}) µg m(^{-3})</th>
<th>Peak PM(_{2.5}) µg m(^{-3})</th>
<th>Mean PM(_{2.5}) µg m(^{-3})</th>
</tr>
</thead>
<tbody>
<tr>
<td>No Source</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Deposition Velocity</td>
<td>0.63 (6.08 %)</td>
<td>0.59 (5.98 %)</td>
<td>0.24 (3.13 %)</td>
<td>0.24 (3.30 %)</td>
</tr>
<tr>
<td>Interzonal Airflow</td>
<td>0.01 (0.10 %)</td>
<td>0.01 (0.10 %)</td>
<td>0.01 (0.13 %)</td>
<td>0.01 (0.14 %)</td>
</tr>
<tr>
<td>Outdoor Concentrations</td>
<td>2.07 (20.00%)</td>
<td>-1.87 (-18.96%)</td>
<td>-1.54 (20.05%)</td>
<td>-1.36 (-18.68%)</td>
</tr>
<tr>
<td>External Airflow</td>
<td>-0.82 (-7.92%)</td>
<td>-0.76 (-7.70%)</td>
<td>-0.45 (5.85 %)</td>
<td>-0.30 (-5.49%)</td>
</tr>
<tr>
<td>Room Dimensions</td>
<td>0.80 (7.72%)</td>
<td>0.73 (7.40%)</td>
<td>0.41 (5.33%)</td>
<td>-0.38 (5.22%)</td>
</tr>
<tr>
<td>Smoking</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Deposition Velocity</td>
<td>-1.74 (-0.69%)</td>
<td>0.87 (5.22%)</td>
<td>-1.57 (-0.92%)</td>
<td>0.35 (2.80 %)</td>
</tr>
<tr>
<td>Interzonal Airflow</td>
<td>18.39 (7.32%)</td>
<td>0.46 (2.76%)</td>
<td>10.76 (6.32%)</td>
<td>0.28 (2.24 %)</td>
</tr>
<tr>
<td>Outdoor Concentrations</td>
<td>-5.39 (-2.14%)</td>
<td>-1.85 (-11.09%)</td>
<td>-4.12 (-2.42%)</td>
<td>-1.37 (-10.97%)</td>
</tr>
<tr>
<td>External Airflow</td>
<td>-3.70 (-1.47%)</td>
<td>-0.06 (-0.36 %)</td>
<td>-2.80 (-1.65%)</td>
<td>0.20 (1.60 %)</td>
</tr>
<tr>
<td>Emission Strength</td>
<td>-49.57 (-19.72%)</td>
<td>-2.80 (-1.65%)</td>
<td>-2.80 (-1.65%)</td>
<td>0.20 (1.60 %)</td>
</tr>
<tr>
<td>Room Dimensions</td>
<td>36.59 (14.59%)</td>
<td>-1.06 (6.47%)</td>
<td>24.64 (14.59%)</td>
<td>0.57 (4.64 %)</td>
</tr>
<tr>
<td>24 Hour Profile</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Deposition Velocity</td>
<td>6.49 (1.78%)</td>
<td>3.14 (6.15 %)</td>
<td>2.42 (0.85 %)</td>
<td>1.03 (2.70 %)</td>
</tr>
<tr>
<td>Interzonal Airflow</td>
<td>11.99 (3.28%)</td>
<td>-0.83 (-1.63%)</td>
<td>11.00 (3.87%)</td>
<td>-0.30 (-0.79%)</td>
</tr>
<tr>
<td>Outdoor Concentrations</td>
<td>-3.30 (-0.90 %)</td>
<td>-1.89 (-3.70%)</td>
<td>0.07 (0.02 %)</td>
<td>-1.33 (-3.48%)</td>
</tr>
<tr>
<td>External Airflow</td>
<td>1.86 (0.51%)</td>
<td>4.63 (9.07 %)</td>
<td>4.68 (1.65 %)</td>
<td>4.07 (10.65%)</td>
</tr>
<tr>
<td>Emission Strength</td>
<td>-71.41 (-19.54%)</td>
<td>-8.16 (-15.99%)</td>
<td>-57.79 (-20.32%)</td>
<td>-6.18 (-16.17%)</td>
</tr>
<tr>
<td>Room Dimensions</td>
<td>56.71 (15.52%)</td>
<td>5.10 (9.99%)</td>
<td>44.08 (15.50%)</td>
<td>2.84 (7.43%)</td>
</tr>
</tbody>
</table>

Table 2.10: The actual and percentage differences in modelled PM\(_{10}\) and PM\(_{2.5}\) concentrations following a 20% independent decrease in each input parameter. Peak and mean concentrations represent values in the kitchen while the doors are open, under three different emission scenario.

In the absence of indoor emission sources, outdoor PM concentrations have the greatest influence on indoor PM concentrations; without large percentage dif-
ference being observed between peak and mean concentrations. In the presence of indoor emission sources, peak concentrations are dominated by emission rates and room dimensions; and their influence on mean concentrations increases as indoor emissions increase. The influence of outdoor concentrations decreases as indoor emission rates increases; however, the influence of external air exchange rate still remains an important factor, varying from a positive to a negative influence as emission rates increase. Without indoor emission sources, indoor concentrations reduce as outdoor air exchange rates are reduced. However, in the presence of indoor emission sources, a reduction in outdoor air exchange rates increase indoor concentrations as emission rates increase.

In the absence of indoor emission sources, internal air exchange rates do not appear to have a major influence on PM concentrations that are already uniform throughout the residence. In the smoking scenario (Section 2.2.3 (i)), lower internal air exchange rates increase PM concentrations in the kitchen, however in the 24 hour profile, the internal air exchange rate appears to be the least important factor; this is due to the similar concentrations in the hall arising from emission sources in the living room.
2.6 Discussion

In this paper, the main features of a multi-zone multi-source compartmental model for estimating indoor PM concentrations have been demonstrated. The importance of modelling at an appropriate time resolution was illustrated in Section 2.3.1, where it was seen that estimating PM concentrations from a smoking source at intervals greater than one minute led to an over-estimation of PM decay, and consequently an under-estimation of the peak concentration, and this effect worsened for greater modelled time intervals; there was a 20% difference in estimated PM concentration for a smoking event modelled over a 15 minute interval, compared with a one minute interval.

The adapted PM$_{10}$ deposition method demonstrates the benefit of separating PM$_{2.5}$ from PM$_{10}$, for the purposes of calculating PM$_{10}$ deposition. Further separating PM$_{10}$ or PM$_{2.5}$ into smaller size fraction bins (e.g. PM$_{1}$) would allow a greater level of accuracy in simulating PM deposition. However, at present, insufficient information is available in the literature on PM$_{1}$ emission rates and deposition velocities.

In Section 2.3.3, overall PM concentrations from different indoor emission sources were compared, and it was observed, when comparing the PM concentrations associated with the solid fuel fire and the incense stick, that the source giving rise to the largest peak PM concentration does not necessarily give rise to the greatest mean PM concentration. Further, it was observed that the source with the largest PM$_{10}$ contribution does not necessarily have the largest PM$_{2.5}$ contribution. In addition, the contribution of various emission sources was seen to be strongly influenced by the internal door configuration; for the series of emission
sources modelled, the solid fuel fire was seen to result in the second highest PM$_{10}$ peak concentration when it was in a closed room, but when the doors were open, it resulted in the second lowest peak concentration. The order of the mean PM concentrations was not however affected by door opening.

When analysing the decay of particles (Section 2.3.4) it was found that PM$_{10}$ and PM$_{2.5}$ concentrations remained elevated by 60.4% and 54.2% one hour after emissions ceased. This confirms earlier findings (Ott et al., 2003)) which showed that Environmental Tobacco Smoke, remained elevated by 40-70% of background concentrations for an hour after one cigarette has been smoked.

In Section 2.3.5 it was demonstrated that source location influences PM concentrations. While doors remain closed, adjoining rooms experienced increased PM concentrations, although rooms separated by two or more closed doors experienced no noticeable effects. This finding demonstrates that the closing of internal doors can be an effective PM exposure control strategy in a multi-zone environment. While the doors remained open, it was seen that source location also influenced PM concentrations, the additional rooms allowing for greater PM dilution and deposition. A similar finding is reported in an experimental study by Ferro et al. (2009), who observed the influence of both internal room layout and internal door configuration on measured PM concentration.

As stated earlier, sample data for PM outdoor concentrations and air exchange rates were used for model parameterisation in this study, so that demonstration simulations could be carried out. It is nonetheless useful to compare the calculated PM concentrations with literature values associated with comparable dwellings and emission scenarios, to provide an indication of the representativeness of the current simulations. Semple et al. (2012b) reported PM$_{2.5}$ concentrations from
Chapter 2: Development of a Multi-zone Multi-source Computational Model to Evaluate Passive Exposure to Particulate Matter Indoors

sources such as solid fuel fires, gas cooking and environmental tobacco smoke in 100 Irish and Scottish living rooms, which had a mean volume of 57 m$^{-3}$. 24-hour mean concentrations for wood burning were found to be 7.7 µg m$^{-3}$ (ranging from 2-23 µg m$^{-3}$) and for peat burning to be 15.6 µg m$^{-3}$ (ranging from 2-24 µg m$^{-3}$). These are lower than the concentrations reported in the current work, and there may be a number of reasons for this, including a lower outdoor PM concentration in the work of Semple et al. (2012b) (a mean value of 8.2 µg m$^{-3}$ was reported).

Coggins et al. (2013) observed 13 PM$_{2.5}$ smoking peaks in a sample household; 11 of the 13 peaks fell in the PM concentration range of 100 - 250 µg m$^{-3}$, with seven in the range of 100 - 150 µg m$^{-3}$, similar to the concentration of 167 ± 43 µg m$^{-3}$ estimated for the “door open” scenario in the current work (Table 2.4).

2.7 Conclusion

This paper has demonstrated IAPPEM, an advanced probabilistic model capable of simulating a realistic representation of the residential environment, with a one minute time resolution and numerous indoor emission sources. This model has afforded an analysis of the influence of internal layout, emission source location, and internal door configuration on indoor PM concentrations, and would allow an investigation of how such parameters may be manipulated to optimise PM decay and dilution and hence minimise inhalation exposure.

One application of IAPPEM is it can be used to determine the minimum feasible external air exchange rates in dwellings, which when minimised to reduce heat loss and increase energy efficiency, does not negatively impact on human health. One potential strategy could examine increasing external air exchange rates in rooms that experience higher air pollutant concentrations and decreasing exter-
nal air exchange rates in rooms that experience lower air pollutant concentrations.

This paper has only demonstrated the model’s adaptations in relation to the home microenvironment. However, IAPPAM includes these adaptations in the simulation of a full range of additional microenvironments that include offices, transport and recreational areas, with the aim of simulating an individual’s exposure over a 24 hour period as they travel throughout a range of microenvironments.

2.8 Acknowledgements

This work was funded by Irish Environmental Protection Agency (EPA) under the STRIVE Programme.
Chapter 3

Simulated PM$_{2.5}$ Concentrations
due to Air Exchange Rate
Variations caused by Internal
Door Opening Patterns
Simulated PM$_{2.5}$ Concentrations due to Interzonal Airflow Variations caused by Internal Door Opening Patterns

J. A. McGrath$^a$, M. A. Byrne$^a$, M.R. Ashmore$^b$, A. C. Terry$^b$, C. Dimitroulopoulou$^c$

$^a$School of Physics, National University of Ireland Galway, Galway, Ireland
$^b$Environment Department, University of York, York
$^c$Department of Mechanical Engineering, University of West Macedonia, Sialvera and Bakola, Kozani, Greece

Abstract

Short-term variations in interzonal airflows can cause significant fluctuations in gaseous or particulate matter concentrations in indoor environments and therefore, interzonal airflow variations need careful consideration when modelling indoor pollutant concentrations. This study investigates the potential accuracy of modelling interzonal airflow variations and assesses the effect of interzonal airflow variations on indoor pollutant concentrations. A variable interzonal airflow is compared with a time-weighted average interzonal airflow, and the differences in the resulting estimates of indoor pollutant concentrations are analysed. Interzonal airflow variations were simulated by the opening/closing of internal doors for periods of 1, 2, 5, 10, 15 and 30 minutes. Based on experimental comparison, it can be concluded that the modelling approach used, accurately predicts PM$_{2.5}$ concentrations for interzonal airflow variations for durations of 10 minutes or greater, with increasing accuracy for longer durations. The simulations demonstrate that both the time of occurrence and duration of the interzonal airflow variations are critical in determining indoor concentrations, and indicating that a time-weighted average interzonal airflow is not a suitable substitute for modelling interzonal airflow variations, as it under-predicts mean PM$_{2.5}$ concentrations by up to 28%.

Keywords: Experimental Validation, Modelling, Interzonal Airflow, PM$_{2.5}$
Chapter 3: Simulated PM$_{2.5}$ Concentrations due to Air Exchange Rate Variations caused by Internal Door Opening Patterns

3.1 Introduction

The linkage of Particulate Matter (PM) exposure to significant health problems has been well documented (Dockery et al., 1992; Fairley, 1999; European Commission, 2005), and since people spend up to 89% of their time indoors (Klepeis et al., 2001), exposure received in the indoor environment deserves particular attention. In addition to infiltrated outdoor air, indoor air contains PM arises from combustion events such as cooking, smoking and heating fuel usage (Morawska and Zhang, 2002; He et al., 2004; Hussein et al., 2006) and resuspension events such as vacuuming and indoor walking (Ferro et al., 2004; Ozkaynak et al., 1996). The importance of the contribution to human health of short-lived, high air pollution concentration events such as these has been noted, relative to a longer term, lower concentration exposure (Michaels and Kleinman, 2000).

Airflow represents a key factor in understanding indoor PM concentrations. External airflow is the primary mechanism for transfer of outdoor air into a building by natural ventilation, air leakages or mechanical ventilation (Niachou et al., 2008). Interzonal airflow is the transfer of air from one room to another within a building through doorways, air ducts and heating, ventilation, and air conditioning (HVAC) systems (Ferro et al., 2009; Du et al., 2012). The dilution of indoor PM concentrations through external airflow variation and the re-distribution of PM through interzonal airflow variation (i.e. opening and closing of internal doors) are important factors in accurate exposure assessment.

Ott et al. (2003) showed how interzonal airflows vary considerably depending on internal layout and door opening patterns, and many studies have modelled or measured air pollution concentrations in the room of a building other than that
which contains the air pollution source (Dimitroulopoulo et al., 2001a; Ferro et al., 2009). The focus of the majority of these studies has been on second hand smoke, and in most cases, the approach has been to consider time-averaged concentrations over periods of hours (an exception is the work of Klepeis and Nazaroff (2006) which simulated both second hand smoke and carbon monoxide variations on a minute-by-minute basis in two buildings with one and four rooms, respectively).

The Energy Performance Buildings Directive (EPBD, 2002/91/EC; 2010/31/EU) sets minimum requirements regarding the energy performance of new and existing buildings (European Commission, 2010). This drive towards energy conservation through tighter buildings and lower external airflow implies a greater significance for interzonal airflow variations as a modifier of PM concentration. However, to date, no model or measurement study has considered this factor in isolation; the study of Klepeis and Nazaroff (2006) assumed that internal doors were fully closed or fully open.

In the present work, the development of a modelling approach is described which examines PM concentration variations in rooms containing a source, and in a number of adjacent rooms in a dwelling, and compares the results obtained when interzonal airflows are varied over short timescales (associated with short periods of door opening/closing) and when a time-weighted average is used for the interzonal airflow.
3.2 Methods

3.2.1 Model Structure

The basis of the model approach used in this work is the INDAIR model (Dimitroulopoulou et al., 2006), which was used to predict air pollutant concentrations in home microenvironments (ME) in the UK, using probability functions for four different pollutants. These pollutants were NO$_2$, CO, PM$_{10}$ and PM$_{2.5}$, under three different emission scenarios: a scenario where there were no sources present, a scenario where there was cooking taking place and thirdly where smoking took place. Various adaptations made to the model to allow specific simulations are described elsewhere by McGrath et al. (not yet published).

The model calculates the change in pollutant concentration by solving the differential equation

$$\frac{dC_k}{dt} = \frac{(\lambda_{0k})}{V_k} (f_k * C_0 - C_k) - \nu_g \left( \frac{A_k}{V_k} \right) C_k + \frac{Q_k}{V_k} + \sum_{i=1}^{15} \frac{\lambda_{ik}}{V_k} (C_i - C_k) \quad (3.1)$$

Equation 3.1 is solved for each $k$, where $k$ represents one of the compartments for each individual time step. $k$ can vary between 1 - 15, depending on how many rooms are being modelled. $C_k$ represent the concentration of the pollutant in that compartment ($\mu g m^{-3}$), and $C_0$ represents the outdoor concentration ($\mu g m^{-3}$). $f_k$ represents the building filtration factor between outdoors and that compartment. $v_g$ is the deposition velocity of the pollutant ($m hr^{-1}$); $\lambda_{(ik)}$ is the interzonal airflows representing the transport of pollutants between internal compartments, e.g. $\lambda_{(12)}$ represents the transport of pollutants from room 1 into room 2, or $\lambda_{(0k)}$
from outside into room \( k \) (m\(^3\) hr\(^{-1}\)); \( A_k \) is the surface area of the compartment \( k \) (m\(^2\)); \( V_k \) is the volume of the compartment \( k \) (m\(^3\)); \( Q_k \) is the indoor emission rates of the pollutant in compartment \( k \) (\( \mu g \) hr\(^{-1}\)).

### 3.2.1.1 Variable airflow

Although variable airflows were a possibility within the framework of earlier INDAIR code versions, this feature has not been fully examined to date, as the emphasis has been on investigating the effect of emission source presence on indoor air pollution concentrations. In previous cases where airflow variations have been considered, the technique employed was to use a time-weighted average airflow, but this did not reveal very short-term variations in interzonal airflow due to door opening events.

The improved time resolution in the current approach, allows the simulation of minute-by-minute variation and therefore an analysis of their effect on indoor pollutant concentrations. In the current work, an airflow value was set at each individual time step in the model which allows both external and interzonal airflow events to be considered.
Chapter 3: Simulated PM$_{2.5}$ Concentrations due to Air Exchange Rate Variations caused by Internal Door Opening Patterns

Figure 3.1: Diagrammatic representation of the apartment layout. The floor area for each room is displayed in the figure; each room has a height of 2.54m. The X represents the location of the emission source and the shaded black rectangular represents the interior structure of the building.

3.2.2 Simulations

Two sets of simulations are described. The objective of the first set was to provide the basis for an experimental validation, the success of which confirmed the feasibility of modelling short term variations in airflow. The second set focussed on modelling test case scenarios to highlight the significance of incorporating variable interzonal airflows in the model. Both sets of simulations were based on an apartment in Galway City, located approximately 100 metres from a main road; the layout, and the dimensions of the individual rooms, can be seen in Figure 3.1. The details of the simulations are presented in Section 3.2.5.
3.2.3 Experimental Data Generation for Model Validation

To provide data for experimental validation, external and interzonal airflows were measured, using the tracer gas decay method (Sherman, 1990), with CO$_2$ as the tracer gas. A GrayWolf probe (GrayWolf Sensing Solutions; Shelton, CT, USA) was used for gas detection. Internal temperature remained constant ($\pm 0.5^\circ C$) across single sampling events with only minor fluctuations occurring between overall samples, ranging from 19.2 - 21.7 $^\circ C$. Experimental data was collected in the period 10th - 18th September 2012. A total of 45 CO$_2$ dispersion measurements were made. Interzonal air flow between the sitting room and hall was measurement on each day that data were collected.

The values in Table 3.1 are listed in terms of volumetric airflow rate between the hall and other zones, measured in $m^3 h^{-1}$. The front door into the apartment remained closed for the entire duration of the measurements, and all the rooms were naturally ventilated. As shown in Figure 3.1, neither the hall, bathroom 1 or en suite bathroom are directly connected to outdoors. The en suite bathroom is connected to Bedroom 1, but the connecting door remained closed during all of the experiments. In Table 1, the heading “Doors Open” refers to the situation when the doors connected to the hall were opened at an angle of 12$^\circ$.

<table>
<thead>
<tr>
<th>Location</th>
<th>Outdoor</th>
<th>Doors Closed</th>
<th>Doors Open</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sitting Room</td>
<td>21.9 (0.24)</td>
<td>5.4 (0.31)</td>
<td>125 (7.19)</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>7.5 (0.29)</td>
<td>9.5 (0.54)</td>
<td>76.7 (4.41)</td>
</tr>
<tr>
<td>Bedroom 2</td>
<td>3.4 (0.16)</td>
<td>4.5 (0.26)</td>
<td>38.4 (2.20)</td>
</tr>
<tr>
<td>Bathroom</td>
<td>-</td>
<td>5.1 (0.29)</td>
<td>-</td>
</tr>
<tr>
<td>Ensuite</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

Table 3.1: External and interzonal airflow rates measured in $m^3 h^{-1}$ with respect to the hall. The air change is included in brackets is measured in hr$^{-1}$.

No Irish data exist to compare with the airflows reported in this study (Ta-
Chapter 3: Simulated PM$_{2.5}$ Concentrations due to Air Exchange Rate Variations caused by Internal Door Opening Patterns

Table 3.1. However, previous studies in United States reported volumetric airflows between 0.4-5.1 m$^3$ h$^{-1}$ and 60-245 m$^3$ h$^{-1}$ with the doors closed and the doors opened respectively (Ott et al., 2003; Miller and Nazaroff, 2001; Ferro et al., 2009).

For the purposes of experimental validation, indoor PM concentrations were monitored using a SidePak Personal Aerosol Monitor Model AM510 (TSI Incorporated, Shoreview, MN, USA) adjusted to measure PM$_{2.5}$. The SidePak was set to log data at 1 minute intervals and calibrated to the recommended flow rate of 1.7 l/min using a primary airflow meter (DryCal DC Lite; BIOS International, NJ, USA.). To greater emphasise the variation in PM concentration in each room, an emission source, an incense stick, was lit in the sitting room, and burned for 40-50 minutes. A calibration factor of 0.34 was used to correct the measured PM concentrations, as recommended by Jiang et al. (2011).

3.2.4 Model Parameterization

3.2.4.1 Outdoor and Indoor PM Concentration

For the first set of simulations, the SidePaks were used to simultaneously monitor both indoor and outdoor PM$_{2.5}$ concentrations, and these concentrations were then used as a direct input parameter into the model. For the second set of simulations, it was decided to leave the outdoor concentrations constant throughout, so as to emphasise the effect of a varying interzonal airflow, rather than a varying outdoor concentration, on the indoor concentration. The outdoor concentration was set to 10 ± 2 µg m$^{-3}$, outdoor values were generated using a log normal distribution based upon the geometric mean and geometric standard deviation from experimental observations from a nearby monitoring station (EPA, 2013). For the first set of simulations, the initial PM concentration in each room was
extracted from the experimental data. For the second set of simulations, the initial concentration in each room was set to $4 \pm 1 \mu g \ m^{-3}$, this value is comparable with concentrations measured for the no source scenario (Section 3.3.2).

### 3.2.4.2 Deposition Velocities and Penetration Factors

The PTEAM study (Ozkaynak et al., 1996) found a deposition loss rate for PM$_{2.5}$ of $0.39 \pm 0.16 \ h^{-1}$, and Dimitroulopoulou et al. (2006) interpreted this to correspond to a deposition velocity of $(1.8 \pm 1.00) \times 10^{-4} \ m \ s^{-1}$ for PM$_{2.5}$. This value was considered to be relevant in the present context since it considers a particle size distribution, rather than a discrete particle sizes. However, it is limited by only considering particles that deposit on the floor; Byrne et al. (1995) found that approximately 60 - 65% of the particles between 0.7 to 2.5 $\mu m$ deposited on the floor of a test chamber with the remainder depositing on the walls. In that study, particles with a diameter of 0.7 and 2.5 microns had deposition velocities of $4.1 \times 10^{-5}$ and $6.2 \times 10^{-5} \ m \ s^{-1}$ respectively, and based on this, a value of $5.15 \times 10^{-5} \ m \ s^{-1}$ is employed in the present simulations.

A building fabric filtration factor of 1.0 was used in these simulations, as this was consistent with the observations from other studies (Ozkaynak et al., 1996; Thatcher et al., 2003). Although, Chen and Zhao (2011) summarised PM$_{2.5}$ filtration factors to range from 0.32-0.82, a sensitivity analysis concluded that vary the filtration factor from unity had a negligible effect on modelling PM concentrations in this study. This is because the contribution from the indoor emission source outweighs the contribution from outdoor concentrations.
3.2.4.3 Indoor Source Emission Rate

The incense stick used had a mean emission rate of 51.6 ± 7.1 mg h\(^{-1}\) (obtained by experimental observations) which is consistent with other studies (See and Balasubramanian, 2011; Lee and Wang, 2004; Jiang et al., 2011). In the present work, the incense stick was positioned on the kitchen table at a height of 80cm above the floor.

3.2.5 Scenarios Tested

For the first set of simulations, the time intervals selected for the door to be opened were 1 minute, 2 minutes, 5 minutes, 10 minutes, 15 minutes and 30 minutes. For the second set of simulations, a time period of six hours, from 18:00 until 24:00, was selected. An emission source was present from 18:00 until 18:45 in the sitting room. Six scenarios were chosen, as follows (in the descriptions below, the sitting room door refers to the door between the sitting room and the hall, and the bedroom door refers to the door between Bedroom 1 and the hall).

The scenarios chosen were as follows:

i ) All internal doors remained closed and no emission source was present.

ii ) All internal doors remained closed and an emission source was present in the sitting room.

iii ) The sitting room door was opened for the entire period but all other doors remained closed.

iv ) The sitting room door and Bedroom 1’s door was open for the entire period.

v ) The sitting room door was opened for several different 15 minute intervals (each interval was simulated separately), but Bedroom 1’s door remained closed.
vi) The sitting room door was opened for a fixed 15 minute interval and Bedroom 1’s door was opened for several different 15 minute intervals (separate simulations).

### 3.3 Results

#### 3.3.1 Experimental Validation

It was observed that the incense sticks burned for periods between 40 and 50 minutes and clearly had varying emission rates. For this reason, simulations commenced after the emission period, and the initial PM concentration for the model was derived from the experimentally-observed indoor PM concentration at that time. This approach allows for a more direct comparison of the interzonal airflows variations between simulated and experimental concentrations, as emission variations (both strength and duration) will have no influence on results. Once the emission source had ceased, PM$_{2.5}$ concentrations in the room were allowed ten minutes to become uniform before each simulation commenced. Each scenario started by modelling five minutes with the doors closed and then modelling the door opening event.

Figure 3.2 shows the experimental and modelled PM concentrations for the sitting room when the doors remain closed. In all simulations, the linear regression value of experimental vs. modelled PM concentrations ranged from 0.97 - 1.07 for the sitting room. However, for the hall, the linear regression values varied far more widely, from 0.72 to 1.72, as seen in Table 3.2. This apparent discrepancy is due to the large difference in room sizes between the sitting room and hall, in terms of volumetric airflow, the hall has much less impact on the sitting room that the sitting room has on the hall.
Chapter 3: Simulated PM$_{2.5}$ Concentrations due to Air Exchange Rate Variations caused by Internal Door Opening Patterns

Figure 3.2: Observed comparison between model concentrations and experimental concentration in the sitting room when the door is closed.

Table 3.2 represents a summary of three data sets comparing experimental and predicted PM concentrations in the hall, when the door between the sitting room (which contains an emission source) and the hall is opened for a variety of intervals. It can be seen that for door opening periods of the order of minutes, experimental vs. model agreement is relatively poor, and this is attributed to two major factors. Firstly, the opening and closing of a door could create a suction effect and consequently a brief increase in the interzonal airflow which the model, relying on an experimental determined average interzonal airflow value, does not consider. Secondly, this suction effect could have an impact on PM deposition velocity; this was also observed by Hussein et al. (2009) where the operation of a fan being turned on increased the deposition velocity of particles with diameters less than 4$\mu$m. It can be seen from the data in Table 3.2 that the longer the duration of the door opening event, the more accurately the model performs. This is most likely due to the fact that the above mentioned causes for fluctuations have time to average out.
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Table 3.2: This table shows the comparison between experimental versus predicted concentration in the hall for the linear regression and $R^2$ along with the peak concentrations.

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Linear Regression</th>
<th>$R^2$</th>
<th>Experimental Peak ($\mu g \ m^{-3}$)</th>
<th>Predicted Peak ($\mu g \ m^{-3}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 Minute</td>
<td>1.72</td>
<td>0.65</td>
<td>262</td>
<td>149</td>
</tr>
<tr>
<td>2 Minute</td>
<td>0.72</td>
<td>0.41</td>
<td>166</td>
<td>183</td>
</tr>
<tr>
<td>5 Minute</td>
<td>1.11</td>
<td>0.75</td>
<td>159</td>
<td>214</td>
</tr>
<tr>
<td>10 Minute</td>
<td>1.07</td>
<td>0.95</td>
<td>317</td>
<td>280</td>
</tr>
<tr>
<td>15 Minute</td>
<td>1.05</td>
<td>0.64</td>
<td>240</td>
<td>230</td>
</tr>
<tr>
<td>30 Minute</td>
<td>1.02</td>
<td>0.94</td>
<td>489</td>
<td>473</td>
</tr>
</tbody>
</table>

### 3.3.2 Test Case Scenarios

In section 3.3.1 it was demonstrated that the model could reasonably simulate 10, 15 and 30 minute interzonal airflow variations due to door openings. In this section, the aim is to highlight the importance of the time periods that are chosen in the context of evaluating personal exposure to PM.

The first scenario simulated was where no source was present. Mean concentrations ranged from 2.89-5.81 $\mu g \ m^{-3}$ between various rooms. These concentrations reflect the background concentration in the apartment (simulation (i)), with variations attributed to interzonal airflows and room dimensions. The highest concentrations are observed in the sitting room and bedrooms, which were directly connected to the outside.

Table 3.3 shows peak PM concentrations in each of the rooms for a simulated scenario when the doors remained closed (simulation (ii)) and an emission source is present in the sitting room. Following this simulation, the highest concentrations are observed in the sitting room, as expected, it being the location of the source. The hall, being the only room directly connected to the sitting room, has the sec-
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The second highest concentrations. The variation in concentrations in the two bedrooms and the bathroom are accounted for by the various interzonal airflows. Although Bedroom 1 has a larger interzonal airflow than the bathroom, the smaller volume of the bathroom results in higher PM concentrations in that room.

<table>
<thead>
<tr>
<th>Location</th>
<th>Time of Peak Concentration (Hours)</th>
<th>Peak Concentration ($\mu g m^{-3}$)</th>
<th>Mean Concentration ($\mu g m^{-3}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sitting Room</td>
<td>18:46</td>
<td>376 ± 9</td>
<td>168 ± 9</td>
</tr>
<tr>
<td>Hall</td>
<td>19:49</td>
<td>60 ± 3</td>
<td>43 ± 3</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>21:23</td>
<td>26 ± 2</td>
<td>19 ± 1</td>
</tr>
<tr>
<td>Bedroom 2</td>
<td>22:03</td>
<td>20 ± 2</td>
<td>15 ± 1</td>
</tr>
<tr>
<td>Bathroom</td>
<td>21:33</td>
<td>35 ± 3</td>
<td>26 ± 2</td>
</tr>
<tr>
<td>En Suite</td>
<td>23:00</td>
<td>15 ± 2</td>
<td>11 ± 1</td>
</tr>
</tbody>
</table>

Table 3.3: Predicted peak and mean PM$_{2.5}$ concentrations in each of the rooms, with an emission source present in the sitting room. All internal doors remained closed for the entire simulation.

Table 3.4 shows the peak PM concentrations in each room when the sitting room, which contains a source, has its door open for the entire period (simulation (iii)). It can be seen that all rooms have higher PM concentrations than those shown in Table 3.3 except for the sitting room. With the door open, it takes approximately 10 minutes for the PM concentrations in the hall to reach its maximum after the source has ceased emission. PM concentrations also increase in the remaining rooms, but at a slower pace; it is a further 90 minutes before the peak concentrations are reached in the bedrooms and bathroom in this scenario.

Table 3.5 shows the concentrations when both the sitting room and Bedroom 1’s door are open (simulation (iv)). A similar result is observed as is shown in Table 3.4, namely a dilution effect in the sitting room and the hall due to Bedroom 1’s door being opened.
Table 3.4: Predicted peak and mean PM$_{2.5}$ concentrations in each of the rooms when an emission source is present in the sitting room. The sitting room door is open for the entire simulation.

<table>
<thead>
<tr>
<th>Location</th>
<th>Time of Peak</th>
<th>Peak Concentration</th>
<th>Mean Concentration</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(Hours)</td>
<td>(µg m$^{-3}$)</td>
<td>(µg m$^{-3}$)</td>
</tr>
<tr>
<td>Sitting Room</td>
<td>18:46</td>
<td>323 ± 7</td>
<td>135 ± 6</td>
</tr>
<tr>
<td>Hall</td>
<td>18:54</td>
<td>258 ± 7</td>
<td>120 ± 6</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>20:10</td>
<td>74 ± 4</td>
<td>51 ± 3</td>
</tr>
<tr>
<td>Bedroom 2</td>
<td>20:45</td>
<td>54 ± 4</td>
<td>41 ± 3</td>
</tr>
<tr>
<td>Bathroom</td>
<td>20:21</td>
<td>109 ± 7</td>
<td>76 ± 6</td>
</tr>
<tr>
<td>En Suite</td>
<td>21:45</td>
<td>40 ± 4</td>
<td>28 ± 3</td>
</tr>
</tbody>
</table>

Table 3.5: Predicted peak and mean PM$_{2.5}$ concentrations in each of the rooms when an emission source is present in the sitting room. The sitting room and Bedroom 1’s door are open for the entire simulation.

<table>
<thead>
<tr>
<th>Location</th>
<th>Time of Peak</th>
<th>Peak Concentration</th>
<th>Mean Concentration</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(Hours)</td>
<td>(µg m$^{-3}$)</td>
<td>(µg m$^{-3}$)</td>
</tr>
<tr>
<td>Sitting Room</td>
<td>18:46</td>
<td>308 ± 7</td>
<td>123 ± 6</td>
</tr>
<tr>
<td>Hall</td>
<td>18:54</td>
<td>211 ± 6</td>
<td>105 ± 5</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>19:21</td>
<td>162 ± 7</td>
<td>89 ± 5</td>
</tr>
<tr>
<td>Bedroom 2</td>
<td>20:49</td>
<td>48 ± 3</td>
<td>36 ± 3</td>
</tr>
<tr>
<td>Bathroom</td>
<td>20:27</td>
<td>94 ± 6</td>
<td>66 ± 5</td>
</tr>
<tr>
<td>En Suite</td>
<td>20:48</td>
<td>74 ± 6</td>
<td>51 ± 4</td>
</tr>
</tbody>
</table>

The final set of scenarios examine PM concentrations in each room during periodic door opening events. Although concentrations in all rooms were simulated, Table 3.6 shows the peak concentrations in the hall only, when the sitting room (which contained the emission source) door was open for various 15 minute periods (simulation (v)). Table 3.7 shows the peak concentrations in the Bedroom 1 under the same conditions of sitting room door opening.
Chapter 3: Simulated PM$_{2.5}$ Concentrations due to Air Exchange Rate Variations caused by Internal Door Opening Patterns

<table>
<thead>
<tr>
<th>Door Opening Times</th>
<th>Time of Peak Concentration (Hours)</th>
<th>Peak Concentration ($\mu g m^{-3}$)</th>
<th>Mean Concentration ($\mu g m^{-3}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>18:00-18:15</td>
<td>19:22</td>
<td>69 ± 3</td>
<td>51 ± 3</td>
</tr>
<tr>
<td>18:15-18:30</td>
<td>18:31</td>
<td>152 ± 5</td>
<td>64 ± 4</td>
</tr>
<tr>
<td>18:30-18:45</td>
<td>18:46</td>
<td>230 ± 6</td>
<td>75 ± 4</td>
</tr>
<tr>
<td>18:38-18:53</td>
<td>18:53</td>
<td>246 ± 7</td>
<td>77 ± 4</td>
</tr>
<tr>
<td>18:45-19:00</td>
<td>17:00</td>
<td>241 ± 7</td>
<td>75 ± 4</td>
</tr>
<tr>
<td>19:00-19:15</td>
<td>19:15</td>
<td>222 ± 7</td>
<td>71 ± 4</td>
</tr>
<tr>
<td>19:15-19:30</td>
<td>19:30</td>
<td>204 ± 8</td>
<td>67 ± 4</td>
</tr>
<tr>
<td>Mean Airflow</td>
<td>19:36</td>
<td>92 ± 4</td>
<td>60 ± 4</td>
</tr>
</tbody>
</table>

Table 3.6: Predicted peak and mean PM$_{2.5}$ concentrations in the hall, when an emission source is present in the sitting room. The sitting room door is opened for 15 minutes at independent intervals. Note: each row of the table represents a separate simulation.

In Table 3.6, it can be seen that for the initial period of door opening, 18:00-18:15, the PM concentration peak occurs nearly two hours later. In all other cases, the peak concentration corresponds to the end of the door opening period, as after this period, the decay rate of particles in the hall is greater than the infiltration rate from the sitting room. The highest concentration reached in the hall occurs in the interval 18:38-18:53, 7 minutes prior and 8 minutes after the emission has ceased in the sitting room. This highlights the observation that the time at which the highest concentrations is reached in adjoining rooms does not simply correspond to the end of the emission period.

The term “mean airflow” in Table 3.6 refers to the time-weighted average airflow of the door being opened and closed. Figure 3.3 shows the predicted PM concentration variations associated with two different door opening patterns compared with those simulated using a mean airflow. The results clearly show that using a mean airflow is not an accurate substitute for simulating indoor PM concentrations in multiple rooms.
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Figure 3.3: PM$_{2.5}$ Concentrations in the hall for three different simulations, two with variation in interzonal airflow at different time intervals and the third using a time-weighted average airflow.

Table 3.7 represents the concentrations in Bedroom 1, when the sitting room door is opened in the interval 18:38-18:53 and variations occur in the opening of Bedroom 1’s door (simulation (vi)). A similar trend arises, as was observed in Table 3.6, whereby the highest concentrations reached in the Bedroom are a combination of the door being opened prior to and during the hall reaching its peak concentrations.
Chapter 3: Simulated PM$_{2.5}$ Concentrations due to Air Exchange Rate Variations caused by Internal Door Opening Patterns

<table>
<thead>
<tr>
<th>Door Opening Times</th>
<th>Time of Peak Opening (Hours)</th>
<th>Peak Concentration ($\mu g\ m^{-3}$)</th>
<th>Mean Concentration ($\mu g\ m^{-3}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>18:00-18:15</td>
<td>19:57</td>
<td>50 ± 3</td>
<td>34 ± 2</td>
</tr>
<tr>
<td>18:15-18:30</td>
<td>19:58</td>
<td>50 ± 3</td>
<td>34 ± 2</td>
</tr>
<tr>
<td>18:30-18:45</td>
<td>19:37</td>
<td>59 ± 4</td>
<td>39 ± 3</td>
</tr>
<tr>
<td>18:44-18:59</td>
<td>19:00</td>
<td>94 ± 10</td>
<td>46 ± 4</td>
</tr>
<tr>
<td>18:45-19:00</td>
<td>19:01</td>
<td>93 ± 10</td>
<td>45 ± 4</td>
</tr>
<tr>
<td>19:00-19:15</td>
<td>19:16</td>
<td>82 ± 8</td>
<td>40 ± 3</td>
</tr>
<tr>
<td>19:15-19:30</td>
<td>19:31</td>
<td>76 ± 6</td>
<td>39 ± 3</td>
</tr>
<tr>
<td>Mean Airflow</td>
<td>19:58</td>
<td>51 ± 3</td>
<td>35 ± 2</td>
</tr>
</tbody>
</table>

Table 3.7: Predicted peak and mean PM$_{2.5}$ concentrations in the Bedroom 1, when an emission source is present in the sitting room. The sitting room door is opened for a fixed 15 minute interval (18:38-18:53) in each simulation, while Bedroom 1’s door is opened for independent 15 minute intervals. Note: each row of the table represents a separate simulation.

3.4 Discussion

Most indoor air pollution models have, to date, used time-weighted average airflow values. This study has highlighted major differences in predicted PM concentrations indoors when a time-weighted average airflow approach is compared with one that considers short-term airflow variations.

In the simulations conducted to predict PM$_{2.5}$ concentrations in the hall and Bedroom 1 during source emission in the sitting room, it has been demonstrated that the time-weighted airflow approach under-predicts the peak concentrations by approximately a factor of two and under-predicts the mean concentration by approximately 28%.

The analysing of a variable airflow approach to modelling has indicated that accuracy improves according to the time duration for which an airflow value remains constant. When modelling one-minute airflow variations, predicted PM
concentrations were underestimated (and overestimated) by up to 80% from experimental data. Possible inaccuracies at lower airflows can be explained by the door movements creating additional suction effects that would have been averaged out, and therefore not obvious, when using tracer gas methods to measure airflow. Model/measurement comparisons demonstrate that the model can accurately simulate airflow variations for events lasting ten minutes or more. If the interzonal airflow associated with the suction effect could be determined, the model could incorporate this additional airflow, potentially improving the accuracy at lower time intervals. However, it is anticipated that a great deal of variability in the “suction effect” airflow would exist between individual opening/closing events, which would place a large uncertainty on model outputs following the incorporation of this effect.

The comparison between the no source scenario and the emission scenarios demonstrates the occurrence of inter-room air pollution transport through closed doors, and the enhancement of this process when doors are open, while having the effect of reducing PM concentration in the room containing the source.

A mean PM concentration reduction of 19% is observed in the sitting room if the door is left open for the entire period, and it is estimated that much greater PM concentration reductions in the sitting room would be observed, if the volume ratio (currently 5:1) of the sitting room to hall were lower.

When the sitting room door is opened for any period, higher concentrations are observed in all adjoining rooms. The highest mean concentration in Bedroom 1 was doubled compared with the “doors closed” scenario and was approximately half the mean concentrations if both doors remained open for the entire period. This is important when it is considered that it represents a total of only 30 minutes
of airflow variations (approximately 8% of the entire time period) throughout the entire apartment, with the doors being only opened 12°.

This finding has relevance when modelling individual’s personal exposure. Peak concentrations were approximately doubled compared to a time-weighted average airflow in the additional rooms. Individuals moving through a series of rooms are exposed to a combination of multiple peak concentrations. In these simulations only a single emission event was present, however in most households multiple emission events will take place over a 24-hour period with individuals present in different rooms. For an accurate exposure assessment within a household, proper consideration must be given to potential airflow variations.

3.5 Conclusion

It has been well established that emission source strengths and durations are key factors in determining indoor PM concentrations. Most emission sources (e.g. cooking, smoking, incense) are dependent on individuals being present, and it is highly plausible that human presence will alter the interzonal airflows, through door opening during or shortly after an emission period. This paper has clearly demonstrated that changes in interzonal airflow during periods of high PM concentrations can substantially alter internal PM concentrations. Without proper consideration of interzonal airflow variations, simulated concentrations could be dramatically under-predicted and, as a result, individuals’ personal exposure. Although all the simulations referred to interzonal airflow variations resulting from internal door opening patterns, the same principles apply for variation in external airflow. Variations in interzonal airflows could also arise from operation of central heating systems, window opening, and use of extractor fans and Heating,
Ventilation and Air Conditioning (HVAC) units.

While all scenarios simulated above referred to PM$_{2.5}$, the same implications of airflow variation (both time and duration) also apply to the modelling of any other gaseous or particulate indoor air pollutant, as the transfer of pollutant between adjoining rooms is governed by interzonal airflow.

### 3.6 Acknowledgements

This work was funded by Irish Environmental Protection Agency (EPA) under the STRIVE Project. The authors would like to acknowledge the guidance given by Dr. Jerome Sheahan in statistical analysis.
Chapter 4

Simulated Variations in PM Exposure due to Different Time Activity Profiles Within the Same Dwelling
Chapter 4: Simulated Variations in PM Exposure due to Different Time Activity Profiles Within the Same Dwelling

Simulated Variations in PM Exposure due to Different Time Activity Profiles Within the Same Dwelling

J. A. McGrath\textsuperscript{a}, M. A. Byrne\textsuperscript{a}, M. R. Ashmore\textsuperscript{b}, A. Terry\textsuperscript{b}, C. Dimitroulopoulou\textsuperscript{c}

\textsuperscript{a}School of Physics, National University of Ireland Galway, Galway, Ireland
\textsuperscript{b}Environment Department, University of York, York
\textsuperscript{c}University of West Macedonia, Department of Mechanical Engineering, Sialvera and Bakola, Kozani, Greece

Abstract

This paper describes the combination of a physical pollutant model with a time activity model, to create an air pollutant exposure model. A previous study has already demonstrated the capability of IAPPEM in predicting indoor PM concentrations in a residential environment. The present work investigates exposure variability between individuals, by examining the movements of four individuals through the same dwelling. The results of the simulations highlight the distinction between exposure and indoor concentrations, providing computational validation to well-documented experimental differences that exist between indoor concentrations and exposure. Using IAPPEM, exposure according to the percentage contribution from the time spent in each room is calculated, illustrating that the periods of presence, in relation to the times of indoor source emission, are as important as the durations spent in each room. This study provides a comparison between two approaches for evaluating exposure, a time-weighting profile and a time activity profile. The results showed that a time averaged profile is a poor substitute when compared with exposure calculated based on a time activity profile, as, in each simulated scenario, exposures were under-predicted using the time-weighted approach, in some cases by up to 74.56%.

Keywords: Modelling Exposure, PM\textsubscript{10}, PM\textsubscript{2.5}, Time-Activity Profile, Indoor Environment
4.1 Introduction

In recent years, air pollution studies have found evidence linking exposure to Particulate Matter (PM) with adverse health outcomes. WHO (2006) reported on long-term exposure to particulate matter (PM) in the European Union, reducing life expectancy on average by 8.6 months and accounting for more than 280,000 premature deaths every year. Less severe health effects were also reported ranging from chronic bronchitis, increased hospitals admissions, respiratory and cardiovascular disease in children and adults (Clancy et al., 2002; Tsai et al., 2003; Host et al., 2008).

Kleinman et al. (2003) reported significant differences in chemical composition between coarse and fine particle fractions. Schins et al. (2002) analysed macrophages and cells and reported finding significant toxicity in coarse PM but not in fine PM. Pope et al. (2011) reported that cardiovascular deaths were accounted for at low exposure levels, while at high PM$_{10}$ exposure, lung cancer becomes of greater concern. Ebelt et al. (2005) highlighted the need to separate total personal particle exposures into their ambient and non ambient components. In order to assess the health implications, accurate data is needed on short-term and long-term exposures for both PM$_{10}$ and PM$_{2.5}$.

Since most people in North America or Europe spent 89% of their time indoors, with 69% spent in the residential indoor environment (Klepeis et al., 2001), the home environment deserves particular attention. Indoor PM concentrations are influenced by infiltration of outdoor PM concentrations, indoor emissions sources and removal of PM by deposition and internal and external air exchanges (Ferro et al., 2009; Singer et al., 2002; Ott, 1999). Indoor emission sources range from
smoking, cooking, and candle usage to solid fuel burning (Semple et al., 2012b; Buonanno et al., 2009; Hussein et al., 2006).

Human exposure is defined as follows: “exposure of person (i) to pollutant concentration (c) is viewed as two events occurring jointly: person (i) is present at a particular location, and concentration (c) is present at the same location” (Ott, 1982). This definition illustrates that both the concentration level and the time the individual spends in contact with the pollutant are exposure determinants. Variability in personal exposures arises from variations in individual personal activity profiles (Van Ryswyk et al., 2013). Numerous studies have highlighted variations in exposure as individuals move through a range of different microenvironments e.g. home, office, car, bus, outdoor etc. (Gulliver and Briggs, 2004; Nasir and Colbeck, 2009; Steinle et al., 2013). The consideration of a large variety of individual profiles is justifiable and necessary as different population groups have greater health risks associated with them; Pope et al. (2009) reported that a daily $10 \mu g m^{-3}$ increase in outdoor PM$_{2.5}$, increases heart disease mortality by 18% in men over the age of 30 and 24% in women between the ages of 50 - 75.

It is often expensive or impractical to obtain direct indoor measurements (or personal exposure measurements) for large population groups in epidemiological studies and computational models are a recognised substitute. Models are effective methods of separating out the contribution from indoor and outdoor air pollution sources, allowing effective exposure reduction strategies to be devised. A number of different models exist to simulate the home microenvironment, predicting concentrations for different pollutants arising from indoor emission events, ranging ranging from NO$_2$, CO, PM$_{10}$, PM$_{2.5}$ to ETS, in single rooms and in multiple scenarios (up to four) (Dimitroulopoulou et al., 2006; Miller and Nazaroff,
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2001; Ott, 1999; Fabian et al., 2012). However, these models have focussed on the prediction of the PM concentrations in households, and they did not investigate an individual’s exposure.

Two methods for calculating exposure have already been widely employed. Firstly, a mean exposure can be calculated by multiplying the mean microenvironmental concentrations by the percentage time spent in each microenvironment (Kousa et al., 2001; Shrubsole et al., 2012). Alternatively, mean exposure can be calculated by setting the exposure at each time interval equal to the concentrations in that microenvironment at that time interval, and then dividing by the total time (Klepeis and Nazaroff, 2006; Ashmore and Dimitroulopoulou, 2009; Dimitroulopoulou et al., 2001b). The second approach requires a more detailed time activity profile, requiring details on both period and duration in each microenvironment.

McGrath et al. (2013a) describe the the development of IAPPEM, which includes the capability of estimating individual PM exposure through a range of different microenvironments. In the current work, the variability in PM exposure for a number of individuals with different activity profiles will be examined, using the indoor emission scenarios described in McGrath et al. (2013a). In addition, a comparison will be made between the two methods, described above, for calculating exposure.
4.2 Methodology

4.2.1 Model

McGrath et al. (2013a) demonstrated IAPPEM, an advanced probabilistic model capable of predicting air pollutant concentrations in realistic representation of the residential environment, with a one minute time resolution and numerous indoor emission sources. McGrath et al. (2013a) only analysed the physical pollutant concentrations throughout a dwelling. In order to accurately access the impact on human health, a model must have the capability to predict exposure to air pollutants.

A physical pollutant model combined with a time activity model predicts exposure to air pollutants (Figure 4.1). IAPPEM incorporates both the physical pollutant model and the time activity model. The physical pollutant aspect of IAPPEM was previously examined in the Chapter 2. This chapter examines the
pollutant exposure model, investigating exposure variability solely due to differences in personal activity profiles.

Equation (4.1) adapted from Klepeis (2006) shows the method used to calculate exposure,

\[ E_i = \sum_{k=1}^{m} \left( \int_{t_{k1}}^{t_{k2}} C_{ik}(t) \, dt \right) \]  (4.1)

where \( E \) represents exposure (\( \mu g \, m^{-3} \, \text{min} \)) of individual \( i \), \( C_{ik} \) are the concentrations (\( \mu g \, m^{-3} \)) experienced by individual \( i \) in microenvironment \( k \) between time \( t_1 \) to \( t_2 \), where \( t \) is in minutes (representing the resolution of IAPPEM) and \( m \) represents the number of different microenvironments.

The physical pollutant model assumes uniform distribution of the air pollutant throughout each room. Therefore, the exposure model must assume that the pollutant concentration in each person’s breathing zone is the same as the concentration in the room.

4.2.2 Simulations

4.2.2.1 Physical Pollutant Model

Tables 4.1 show a sample 24-hour activity profile which was simulated in McGrath et al. (2013a). This simulation incorporating a number of emission events in an 11 room dwelling, commencing at 8 a.m. Tables 4.2 and 4.3 shows the PM\(_{10}\) and PM\(_{2.5}\) peak and mean concentrations for the key rooms in the dwelling. Bedrooms 2 and 3 were not included, as very little variation existed between the rooms. Full details on the physical pollutant simulation are given in McGrath et al. (2013a).
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<table>
<thead>
<tr>
<th>Time</th>
<th>Location</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>10:30-10:39</td>
<td>Living Room</td>
<td>Single Smoking Event</td>
</tr>
<tr>
<td>12:00-12:09</td>
<td>Living Room</td>
<td>Single Smoking Event</td>
</tr>
<tr>
<td>13:30-13:38</td>
<td>Kitchen</td>
<td>Single Frying Event</td>
</tr>
<tr>
<td>17:00-17:09</td>
<td>Living Room</td>
<td>Single Smoking Event</td>
</tr>
<tr>
<td>18:00-21:30</td>
<td>Living Room</td>
<td>Solid Fuel Fire</td>
</tr>
<tr>
<td>18:00-18:08</td>
<td>Kitchen</td>
<td>Single Frying Event</td>
</tr>
<tr>
<td>19:30-19:39</td>
<td>Living Room</td>
<td>Single Smoking Event</td>
</tr>
<tr>
<td>22:00-22:09</td>
<td>Living Room</td>
<td>Single Smoking Event</td>
</tr>
</tbody>
</table>

Table 4.1: A 24-hour profile showing a number of different emission sources present at different times and locations throughout the dwelling.

<table>
<thead>
<tr>
<th>Source</th>
<th>Time of Peaks (Hours:Minutes)</th>
<th>Peak PM$_{10}$ Concentration ($\mu g \ m^{-3}$)</th>
<th>Mean PM$_{10}$ Concentration ($\mu g \ m^{-3}$)</th>
<th>Time exceeding WHO guidelines (Hours:Minutes)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Doors Closed</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>18:08</td>
<td>556 ± 234</td>
<td>68 ± 6</td>
<td>7:29</td>
</tr>
<tr>
<td>Dining Room</td>
<td>19:11</td>
<td>30 ± 4</td>
<td>16 ± 1</td>
<td>0:00</td>
</tr>
<tr>
<td>Hallway</td>
<td>20:35</td>
<td>84 ± 10</td>
<td>33 ± 2</td>
<td>5:13</td>
</tr>
<tr>
<td>Living Room</td>
<td>19:39</td>
<td>1104 ± 172</td>
<td>260 ± 20</td>
<td>16:07</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>21:48</td>
<td>17 ± 1</td>
<td>13 ± 1</td>
<td>0:00</td>
</tr>
<tr>
<td>Doors Open</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>13:38</td>
<td>363 ± 100</td>
<td>54 ± 3</td>
<td>8:00</td>
</tr>
<tr>
<td>Dining Room</td>
<td>18:20</td>
<td>163 ± 30</td>
<td>47 ± 2</td>
<td>7:54</td>
</tr>
<tr>
<td>Hallway</td>
<td>19:47</td>
<td>205 ± 20</td>
<td>60 ± 2</td>
<td>10:34</td>
</tr>
<tr>
<td>Living Room</td>
<td>19:39</td>
<td>642 ± 83</td>
<td>105 ± 4</td>
<td>11:14</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>20:08</td>
<td>165 ± 12</td>
<td>52 ± 2</td>
<td>9:45</td>
</tr>
</tbody>
</table>

Table 4.2: The peak and mean PM$_{10}$ concentrations in different rooms after simulating the 24-hour profile scenario with doors opened and doors closed. Additional information is provided on the times of peak concentrations and duration in each room for which the WHO 24-hour guideline values are exceeded.
Chapter 4: Simulated Variations in PM Exposure due to Different Time Activity Profiles Within the Same Dwelling

<table>
<thead>
<tr>
<th>Source</th>
<th>Time of Peak PM$_{2.5}$ Concentration (Hours:Minutes)</th>
<th>Peak PM$_{2.5}$ Concentration ($\mu g \ m^{-3}$)</th>
<th>Mean PM$_{2.5}$ Concentration ($\mu g \ m^{-3}$)</th>
<th>Time exceeding WHO guidelines (Hours:Minutes)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Doors Closed</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>18:08</td>
<td>437 ± 182</td>
<td>58 ± 6</td>
<td>10:01</td>
</tr>
<tr>
<td>Dining Room</td>
<td>19:19</td>
<td>26 ± 4</td>
<td>13 ± 1</td>
<td>0:35</td>
</tr>
<tr>
<td>Hallway</td>
<td>20:34</td>
<td>56 ± 6</td>
<td>25 ± 2</td>
<td>10:55</td>
</tr>
<tr>
<td>Living Room</td>
<td>19:39</td>
<td>595 ± 100</td>
<td>167 ± 11</td>
<td>18:24</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>22:48</td>
<td>12 ± 1</td>
<td>10 ± 1</td>
<td>0:00</td>
</tr>
<tr>
<td><strong>Doors Open</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>18:08</td>
<td>282 ± 76</td>
<td>41 ± 2</td>
<td>13:20</td>
</tr>
<tr>
<td>Dining Room</td>
<td>18:21</td>
<td>130 ± 24</td>
<td>37 ± 2</td>
<td>12:26</td>
</tr>
<tr>
<td>Hallway</td>
<td>19:47</td>
<td>123 ± 12</td>
<td>42 ± 2</td>
<td>14:26</td>
</tr>
<tr>
<td>Living Room</td>
<td>19:39</td>
<td>361 ± 53</td>
<td>64 ± 2</td>
<td>14:34</td>
</tr>
<tr>
<td>Bedroom 1</td>
<td>20:08</td>
<td>104 ± 8</td>
<td>38 ± 2</td>
<td>14:11</td>
</tr>
</tbody>
</table>

Table 4.3: The peak and mean PM$_{2.5}$ concentrations in different rooms after simulating the 24-hour profile scenario with doors opened and doors closed. Additional information is provided on the times of peak concentrations and duration in each room for which the WHO 24-hour guideline values are exceeded.

### 4.2.2.2 Time Activity Profiles

Tables 4.4 and 4.5 show profiles for individuals moving through the home environment, defining the periods for which durations they are present in each room. The time it takes individuals to move between rooms is approximated at less than one minute and considered negligible in these simulations. It should be noted that the profiles, although having some basis in literature (the percentage of time spent in each microenvironment), are for demonstrative purposes, and may not be truly representative of the population behaviour. In these simulations, individuals spend the entire 24-hour period in the home environment, removing any external environments as a cause of variability in individual exposure.
Chapter 4: Simulated Variations in PM Exposure due to Different Time Activity Profiles Within the Same Dwelling

<table>
<thead>
<tr>
<th>Time</th>
<th>Locations</th>
<th>Person 1</th>
<th>Person 2</th>
<th>Person 3</th>
<th>Person 4</th>
</tr>
</thead>
<tbody>
<tr>
<td>00:00-09:30</td>
<td>Bedroom 1</td>
<td>Bedroom 1</td>
<td>Bedroom 2</td>
<td>Bedroom 3</td>
<td>Bedroom 3</td>
</tr>
<tr>
<td>09:30-10:00</td>
<td>Kitchen</td>
<td>Bedroom 1</td>
<td>Bedroom 2</td>
<td>Bedroom 3</td>
<td>Bedroom 3</td>
</tr>
<tr>
<td>10:00-10:30</td>
<td>Dining Room</td>
<td>Dining Room</td>
<td>Dining Room</td>
<td>Dining Room</td>
<td>Dining Room</td>
</tr>
<tr>
<td>10:30-12:00</td>
<td>Dining Room</td>
<td>Living Room</td>
<td>Living Room</td>
<td>Dining Room</td>
<td>Dining Room</td>
</tr>
<tr>
<td>12:00-13:00</td>
<td>Kitchen</td>
<td>Living Room</td>
<td>Living Room</td>
<td>Bedroom 3</td>
<td>Bedroom 3</td>
</tr>
<tr>
<td>13:00-14:00</td>
<td>Kitchen</td>
<td>Living Room</td>
<td>Bedroom 2</td>
<td>Bedroom 3</td>
<td>Bedroom 3</td>
</tr>
<tr>
<td>14:00-15:00</td>
<td>Dining Room</td>
<td>Dining Room</td>
<td>Dining Room</td>
<td>Dining Room</td>
<td>Dining Room</td>
</tr>
<tr>
<td>15:00-17:00</td>
<td>Kitchen</td>
<td>Living Room</td>
<td>Living Room</td>
<td>Bedroom 3</td>
<td>Bedroom 3</td>
</tr>
<tr>
<td>17:00-18:00</td>
<td>Kitchen</td>
<td>Living Room</td>
<td>Bedroom 2</td>
<td>Bedroom 3</td>
<td>Bedroom 3</td>
</tr>
<tr>
<td>18:00-18:30</td>
<td>Bedroom</td>
<td>Living Room</td>
<td>Living room</td>
<td>Bedroom 3</td>
<td>Bedroom 3</td>
</tr>
<tr>
<td>18:30-19:30</td>
<td>Dining Room</td>
<td>Dining Room</td>
<td>Dining Room</td>
<td>Dining Room</td>
<td>Dining Room</td>
</tr>
<tr>
<td>19:30-21:00</td>
<td>Kitchen</td>
<td>Living Room</td>
<td>Bedroom 2</td>
<td>Bedroom 3</td>
<td>Bedroom 3</td>
</tr>
<tr>
<td>21:00-22:00</td>
<td>Kitchen</td>
<td>Living Room</td>
<td>Living Room</td>
<td>Bedroom 3</td>
<td>Bedroom 3</td>
</tr>
<tr>
<td>22:00-23:00</td>
<td>Living room</td>
<td>Living Room</td>
<td>Bedroom 2</td>
<td>Bedroom 3</td>
<td>Bedroom 3</td>
</tr>
<tr>
<td>23:00-24:00</td>
<td>Bedroom 1</td>
<td>Bedroom 1</td>
<td>Bedroom 2</td>
<td>Bedroom 3</td>
<td>Bedroom 3</td>
</tr>
</tbody>
</table>

Table 4.4: The time activity profiles for Persons 1-4, indicating their location in each room over the 24 hour period.

<table>
<thead>
<tr>
<th>Time</th>
<th>Locations</th>
<th>Person 5</th>
</tr>
</thead>
<tbody>
<tr>
<td>00:00-09:30</td>
<td>Bedroom 1</td>
<td></td>
</tr>
<tr>
<td>09:30-10:00</td>
<td>Kitchen</td>
<td></td>
</tr>
<tr>
<td>10:00-10:30</td>
<td>Dining Room</td>
<td></td>
</tr>
<tr>
<td>10:30-13:30</td>
<td>living Room</td>
<td></td>
</tr>
<tr>
<td>13:30-15:00</td>
<td>Kitchen</td>
<td></td>
</tr>
<tr>
<td>15:00-18:00</td>
<td>living Room</td>
<td></td>
</tr>
<tr>
<td>18:00-19:30</td>
<td>Kitchen</td>
<td></td>
</tr>
<tr>
<td>19:30-23:00</td>
<td>living Room</td>
<td></td>
</tr>
<tr>
<td>23:00-24:00</td>
<td>Bedroom 1</td>
<td></td>
</tr>
</tbody>
</table>

Table 4.5: The time activity profiles for person 5 indicating their location in each room over the 24 hour period.
i ) Person 1 could represent the cook or potentially, the homemaker described in other studies. They spent 35%, 17%, 4% and 44% of time in the kitchen, dining room, living room and bedroom respectively. Averaging assumptions by Dimitroulopoulou et al. (2001b) over the time spend in the home environments gives 26%, 31%, 43% in the kitchen, dining room, living room and bedroom respectively.

ii ) Person 2 could represent an additional adult in the home environment, spending 10%, 44% and 46% of time in the dining room, living room and bedroom, respectively. If the kitchen and living room was treated as one room, this would be roughly consistent with the findings of Wilkinson et al. (2009) who assumed a time distribution of 45%, 45% and 10% in the living room, bedroom, and kitchen respectively.

iii ) Person 3 could represents a young adult in the home environment, spending 10%, 25% and 65% of time in the dining room, living room and bedroom, respectively. Livingstone and Bovill (2013) reported that majority of young people spend at least half of their waking time in their bedrooms when at home.

iv ) Person 4 could represent an infant, who spends 17% and 83% of the time in the dining room and bedroom. This distribution is based on the knowledge that newborn babies can sleep up to 16 to 20 hours a day. (Johnston et al., 2003).

v ) Person 5, for whom the time activity profile is shown separately in Table 4.5, represents an individual who moves back and forth between the kitchen and living room and resides in a particular room during the time that an emission
event takes place in that room.

4.3 Results

4.3.1 24-hour Mean Exposure

4.3.1.1 Personal Profiles

Figure 4.2 shows PM$_{10}$ exposure for Persons 1 and 2 moving throughout the dwelling, when the doors are open. This demonstrates the combination of a physical pollutant model with a personal activity model, to create a pollutant exposure model, and highlights how individuals are exposed to different levels of PM concentrations as they move throughout the dwelling.
Chapter 4: Simulated Variations in PM Exposure due to Different Time Activity Profiles Within the Same Dwelling

Figure 4.2: The difference in PM$_{10}$ exposure for Person 1 and Person 2 as they move through different rooms in the dwelling when the doors are open.

An example of this, occurred between 19:30 and 22:00 when both individuals are exposed to different PM concentrations levels. Person 1 is in the kitchen and Person 2 in the living room, and there is a major difference between their exposure during that period. At 22:00, when Person 1 entered the living room, both individuals are exposed to the concentrations level. Figure 4.2 shows PM$_{10}$ exposure profiles for the same two individuals.

4.3.1.2 Difference in Mean Exposures

As is shown in Tables 4.6 and 4.7, Person 2 has the highest mean exposure, due to the longer duration spent in the living room especially during the emission periods.

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<table>
<thead>
<tr>
<th>Individual</th>
<th>Time</th>
<th>Location</th>
<th>Peak PM$_{10}$ Exposure</th>
<th>24-hour Mean PM$_{10}$ Exposure</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(hh:mm)</td>
<td></td>
<td>(µg m$^{-3}$)</td>
<td>(µg m$^{-3}$)</td>
</tr>
<tr>
<td><strong>Doors Open</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Person 1</td>
<td>22:09</td>
<td>Living Room</td>
<td>448 ± 55</td>
<td>60 ± 3</td>
</tr>
<tr>
<td>Person 2</td>
<td>19:39</td>
<td>Living Room</td>
<td>642 ± 83</td>
<td>98 ± 4</td>
</tr>
<tr>
<td>Person 3</td>
<td>12:09</td>
<td>Living Room</td>
<td>358 ± 70</td>
<td>72 ± 3</td>
</tr>
<tr>
<td>Person 4</td>
<td>20:04</td>
<td>Dining Room</td>
<td>169 ± 13</td>
<td>53 ± 2</td>
</tr>
<tr>
<td><strong>Doors Closed</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Person 1</td>
<td>22:09</td>
<td>Living Room</td>
<td>932 ± 85</td>
<td>75 ± 5</td>
</tr>
<tr>
<td>Person 2</td>
<td>19:39</td>
<td>Living Room</td>
<td>1104 ± 172</td>
<td>201 ± 16</td>
</tr>
<tr>
<td>Person 3</td>
<td>21:00</td>
<td>Living Room</td>
<td>939 ± 90</td>
<td>90 ± 7</td>
</tr>
<tr>
<td>Person 4</td>
<td>19:09</td>
<td>Dining Room</td>
<td>30 ± 4</td>
<td>14 ± 1</td>
</tr>
</tbody>
</table>

Table 4.6: Predicted peak and mean PM$_{10}$ exposures with variations in personal activity profiles for each individual, under the same physical conditions when the door were open and closed.

When mean exposure for Person 1 and Person 3 is compared, it can be seen that Person 3 has the highest PM$_{10}$ and PM$_{2.5}$ mean exposure, both with the doors open and closed, although the difference between PM$_{2.5}$ mean exposure is considerably reduced. As will be suggested in Section 4.3.2 if frying took place for a longer duration or additional emission events took place in the kitchen, Person 1 could have experienced a higher PM$_{2.5}$ mean exposure than Person 3, but a lower PM$_{10}$ exposure.
Table 4.7: Predicted peak and mean PM$_{2.5}$ exposures with variations in personal activity profiles for each individual, under the same physical conditions when the door were open and closed.

When the doors were closed, Person 4’s exposure marginally increases compared with the no emission source scenario. In the no source scenario, mean PM$_{10}$ and PM$_{2.5}$ concentrations are $11 \pm 1 \, \mu g \, m^{-3}$ and $8 \pm 0 \, \mu g \, m^{-3}$ respectively (Table 3 and 4 in (McGrath et al., 2013a)). Person 4, who spent the majority of their time in a room that did not contain a source, is the only individual whose exposure increased when the doors were open.

When the doors were open, Person 2’s exposure seems to reflect mean PM concentrations in the living room, where the mean PM$_{10}$ concentration was $101 \pm 4 \, \mu g \, m^{-3}$ while mean PM$_{10}$ exposure was $92 \pm 4 \, \mu g \, m^{-3}$. However, when the doors were closed, mean PM$_{10}$ concentrations was $242 \pm 20 \, \mu g \, m^{-3}$, while mean PM$_{10}$ exposure was $201 \pm 17 \, \mu g \, m^{-3}$. 
4.3.1.3 Percentage Contribution to Exposure from Time Spent in Each Room

Table 4.8 analyses each individual’s mean exposure in terms of the percentage contribution from each room; it does not reflect the contribution from emission sources, but only the contribution from rooms in which an individual was present. As each person only spends time in one of the bedrooms, Bedrooms 1, 2 and 3 are all combined into a single row in the table.

Person 1, 2 and 3 all spent 44% or greater, of their time, in the bedroom, but the majority of their percentage exposure contribution is attributed to time spent in the living room, kitchen and/or dining room. For these persons, the bedroom contributes less than 11% to exposure when the doors remained closed. Although Person 3 only spends 25% of their time in the living room, when the doors were closed, the living room contributes up to 88.11% and 86.07% respectively of their to PM$_{10}$ and PM$_{2.5}$ mean exposure.

When the doors were open, it can be seen from Table 4.8 that all four individuals experience a reduced percentage exposure contribution from time spent in the living room, and a correspondingly increased percentage contribution from time spent in the bedrooms due to a transfer of PM throughout the dwelling.
Chapter 4: Simulated Variations in PM Exposure due to Different Time Activity Profiles Within the Same Dwelling

<table>
<thead>
<tr>
<th>Time</th>
<th>Locations</th>
<th>Locations</th>
<th>Locations</th>
<th>Locations</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Person 1</td>
<td>Person 2</td>
<td>Person 3</td>
<td>Person 4</td>
</tr>
<tr>
<td>PM\textsubscript{10} Doors closed</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>50% (35%)</td>
<td>0% (0%)</td>
<td>0% (0%)</td>
<td>0% (0%)</td>
</tr>
<tr>
<td>Dining Room</td>
<td>4% (17%)</td>
<td>1% (10%)</td>
<td>3% (10%)</td>
<td>22% (17%)</td>
</tr>
<tr>
<td>Living Room</td>
<td>39% (4%)</td>
<td>96% (44%)</td>
<td>88% (25%)</td>
<td>0% (0%)</td>
</tr>
<tr>
<td>Bedroom 1,2,3</td>
<td>7% (44%)</td>
<td>3% (46%)</td>
<td>9% (65%)</td>
<td>78% (83%)</td>
</tr>
<tr>
<td>PM\textsubscript{10} Doors open</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>48% (35%)</td>
<td>0% (0%)</td>
<td>0% (0%)</td>
<td>0% (0%)</td>
</tr>
<tr>
<td>Dining Room</td>
<td>18% (17%)</td>
<td>12% (10%)</td>
<td>15% (10%)</td>
<td>21% (17%)</td>
</tr>
<tr>
<td>Living Room</td>
<td>15% (4%)</td>
<td>75% (44%)</td>
<td>41% (25%)</td>
<td>0% (0%)</td>
</tr>
<tr>
<td>Bedroom 1,2,3</td>
<td>18% (44%)</td>
<td>13% (46%)</td>
<td>44% (65%)</td>
<td>79% (83%)</td>
</tr>
<tr>
<td>PM\textsubscript{2.5} Doors closed</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>55% (35%)</td>
<td>0% (0%)</td>
<td>0% (0%)</td>
<td>0% (0%)</td>
</tr>
<tr>
<td>Dining Room</td>
<td>4% (17%)</td>
<td>2% (10%)</td>
<td>3% (10%)</td>
<td>23% (17%)</td>
</tr>
<tr>
<td>Living Room</td>
<td>34% (4%)</td>
<td>95% (44%)</td>
<td>86% (25%)</td>
<td>0% (0%)</td>
</tr>
<tr>
<td>Bedroom 1,2,3</td>
<td>7% (44%)</td>
<td>3% (46%)</td>
<td>11% (65%)</td>
<td>77% (83%)</td>
</tr>
<tr>
<td>PM\textsubscript{2.5} Doors open</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kitchen</td>
<td>49% (35%)</td>
<td>0% (0%)</td>
<td>0% (0%)</td>
<td>0% (0%)</td>
</tr>
<tr>
<td>Dining Room</td>
<td>18% (17%)</td>
<td>9% (10%)</td>
<td>13% (10%)</td>
<td>20% (17%)</td>
</tr>
<tr>
<td>Living Room</td>
<td>16% (4%)</td>
<td>80% (44%)</td>
<td>46% (25%)</td>
<td>0% (0%)</td>
</tr>
<tr>
<td>Bedroom 1,2,3</td>
<td>17% (44%)</td>
<td>11% (46%)</td>
<td>41% (65%)</td>
<td>80% (83%)</td>
</tr>
</tbody>
</table>

Table 4.8: The percentage each room contributes to PM\textsubscript{10} and PM\textsubscript{2.5} exposure for person 1-4 for the doors open and closed scenarios. The corresponding percentage of time spent in each room is included in brackets.

4.3.1.4 A Comparison between the Time-Weighting and Step-by-Step Approach for Calculating 24 hour Exposure

A comparison is made between two techniques for calculating air pollutant exposure. The first technique, as, employed in this paper, calculates air pollutant exposure based on a step-by-step approach as seen in Table 4.4. The second technique involves using time-weighted values, and these values are summarised in Section 4.2.2.2. The time-weighted technique calculates exposure based on time-weighted factors and multiplies these by the mean concentrations in the room in
which time is spent. A 24 hour mean exposure can then be calculated from a summation of the exposure in each room. Table 4.9 presents the calculated 24 hour mean exposures for each individual using the time-weighted technique; corresponding peak exposure values cannot be calculated using this approach.

<table>
<thead>
<tr>
<th></th>
<th>PM$_{10}$ Doors Closed (µg m$^{-3}$)</th>
<th>PM$_{10}$ Doors Open (µg m$^{-3}$)</th>
<th>PM$_{2.5}$ Doors Closed (µg m$^{-3}$)</th>
<th>PM$_{2.5}$ Doors Open (µg m$^{-3}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Person 1</td>
<td>43 (75)</td>
<td>54 (60)</td>
<td>33 (58)</td>
<td>40 (45)</td>
</tr>
<tr>
<td>Person 2</td>
<td>123 (201)</td>
<td>75 (98)</td>
<td>80 (127)</td>
<td>49 (64)</td>
</tr>
<tr>
<td>Person 3</td>
<td>75 (90)</td>
<td>65 (72)</td>
<td>50 (60)</td>
<td>45 (50)</td>
</tr>
<tr>
<td>Person 4</td>
<td>13 (14)</td>
<td>52 (53)</td>
<td>10 (11)</td>
<td>38 (39)</td>
</tr>
</tbody>
</table>

Table 4.9: Predicted 24-hour mean PM$_{10}$ and PM$_{2.5}$ exposures using the time averaged approach for the doors open and closed scenarios. The 24-hour mean exposure calculated based on a personal activity profile are reported in the brackets.

By comparing the step-by-step exposure values shown in Tables 4.6 to 4.7 with the time-weighted values shown in Tables 4.9, it can be seen that, in every case, the time-weighted approach underpredicts exposure. Percentage differences in exposures calculated by the two methods range from 3.0-74.56%, with the step-by-step values being higher in each case. This is due to the time-weighted approach including extended periods when no emission sources were present, and hence reducing the 24-hour mean concentrations calculated.

### 4.3.2 Mean Exposure versus Mean Concentrations

In this section, a comparison of calculated peak and mean exposures and peak and mean concentrations are made for PM$_{10}$ and PM$_{2.5}$, for Person 5 only.

When the doors were open, peak PM$_{10}$ and PM$_{2.5}$ exposure was $651 \pm 79 \ \mu g \ m^{-3}$ and $369 \pm 52 \ \mu g \ m^{-3}$ respectively, which equaled the peak concentrations
recorded in the living room (Table 4.2 and 4.3). Mean PM$_{10}$ exposure was $106 \pm 4 \, \mu g \, m^{-3}$, while mean PM$_{10}$ concentrations in the living room were $105 \pm 4 \, \mu g \, m^{-3}$. Mean PM$_{2.5}$ exposure was $69 \pm 3 \, \mu g \, m^{-3}$ while mean PM$_{2.5}$ concentrations in the living room were $64 \pm 2 \, \mu g \, m^{-3}$. Although the individual’s peak exposure can not exceed peak concentrations in the dwelling, the 24-hour mean exposure can exceed the 24-hour mean concentrations in the living room (the room with the highest mean concentrations).

The individual is present during the periods of highest concentrations in the living room and kitchen. While PM concentrations were decaying in the living room after each smoking event, higher concentrations were present during that period in the kitchen during/following the frying event. This results in higher exposure than mean concentrations for any individual room.

Although, there is no significant differences between mean exposure and mean concentrations reported here, this is only a reflection on the scenario modelled in this paper. The significantly higher concentrations in the living room dominate over the kitchen, which has a lower impact on mean exposure. The mean concentrations in the kitchen could be increased by a factor of three and still equal concentrations in the living room, resulting in higher mean exposure.

If frying was extended to $3 \times 20$ minute frying events when the doors were open, mean PM$_{10}$ and PM$_{2.5}$ concentrations in the living room become $126 \pm 5 \, \mu g \, m^{-3}$ and $84 \pm 3 \, \mu g \, m^{-3}$, mean PM$_{10}$ and PM$_{2.5}$ concentrations in the kitchen become $103 \pm 4 \, \mu g \, m^{-3}$ and $82 \pm 4$, while mean PM$_{10}$ and PM$_{2.5}$ exposure increases to $152 \pm 6 \mu g \, m^{-3}$ and $107 \pm 5 \, \mu g \, m^{-3}$. In this case, mean exposure is 20% and 28% higher than the highest mean concentrations.
4.3.3 Peak Exposures

4.3.3.1 Location of Peaks

Person 1, 2 and 3 experienced peak exposures in the living room, while Person 4’s experience of peak exposure depends on the doors being open or closed.

When the doors were closed, Person 4 experienced peak PM$_{10}$ and PM$_{2.5}$ exposure in the dining room, due to infiltration from the kitchen, which was the only room where person 4 is present that is adjoining a room which contained an emission source; it was shown in previous work that rooms separated from an emission source by two or more closed doors experience only marginal increases in concentrations (McGrath et al., 2013b). However, when the doors were open, Person 4 experienced peak PM$_{10}$ exposure in the bedroom, while peak PM$_{2.5}$ exposure occurred in the dining room. Peak PM$_{10}$ exposure is mostly accounted for by the solid fuel fire and smoking emissions infiltrating into the bedroom. These emissions have a lower PM$_{2.5}$ to PM$_{10}$ ratio, when compared with frying. The result caused the peak PM$_{10}$ exposure in the bedroom without the corresponding PM$_{2.5}$ peak exposure. Peak PM$_{2.5}$ exposure occurred in the dining room, having a higher PM$_{2.5}$ but a lower PM$_{10}$ concentration than the bedroom, due to the higher PM$_{2.5}$ to PM$_{10}$ ratio of the frying sources.

4.3.3.2 Times of Occurrence of Peak Exposure

As seen in Tables 4.6 and 4.7, Person 2 encountered peak exposure at a time when peak concentrations occurred in the living room, this time remains the same regardless of the doors been open or closed. However, Person 1, who only enters the living room at 22:00, experiences peak exposure at the end of the last
cigarette-burning event (22:09), as PM concentrations were already elevated in the living room due to the previous emission events, and exceeded those associated with earlier frying events in the kitchen.

Tables 4.6 and 4.7 indicates that the time at which Person 3 encountered peak exposure varied substantially, depending on the doors being open or closed. When the doors were open, peak exposure occurred at 12:09 at the end of a cigarette emission period. When the doors were closed, peak exposure occurred at 21:00, when Person 3 re-entered the living room; at this time PM concentration were decaying from a cigarette emission event which ceased at 19:39, although the concentrations are decaying at a slower rate, due to the emissions from the solid fuel fire and the doors being closed.

As seen in Tables 4.6 and 4.7, Person 4, who resided primarily in the bedroom, had their peak PM$_{10}$ and PM$_{2.5}$ exposure at different times, depending on whether the doors were open or closed. When the doors were open, peak PM$_{10}$ exposure occurred at 20:04, 25 minutes after peak concentrations occurred in the living room. The time lag is a result of PM transport to the bedroom via the hallway. When the doors were closed, peak exposure occurred in the dining room, but the PM$_{2.5}$ exposure peak occurred five minutes later than the PM$_{10}$ exposure peak. This is due to the impact of air exchange and decay on peak PM concentration, as discussed by McGrath et al. (2013b). The lower PM$_{2.5}$ deposition velocity, compared with PM$_{10}$, means that PM$_{2.5}$ concentration remains elevated for a longer period, allowing greater infiltration into the dining room.
4.3.3.3 Magnitude of Peak Exposures for the Various House Occupants.

Person 1, 2 and 3 all experience peak exposure in the living room (Section 4.3.3.1), although there is difference between the times each experiences peak exposure (Section 4.3.3.2). As seen in Tables 4.6 and 4.7, when the doors were open, there is a clear difference in magnitude between the exposures experienced by the four individuals, but in each case, peak PM$_{10}$ exposures and peak PM$_{2.5}$ exposures are similar.

However, when the doors were closed, Person 1 and 2 experience almost identical peak PM$_{2.5}$ exposure levels, but there is a notable difference between their peak PM$_{10}$ exposure levels. Similarly, Person 1 and 3 have nearly identical PM$_{10}$ exposure but there is a larger difference between their peak PM$_{2.5}$ exposure. When comparing Person 1 and 3, it is observed that Person 3 had a higher PM$_{2.5}$ peak exposure, while Person 1 had the higher PM$_{2.5}$ peak exposure.

When the doors were closed, Person 4 who resided primarily in the bedroom, only experienced a fraction of the peak exposure of the other three individuals, Person 3 experienced a PM$_{2.5}$ exposure that was 19 times higher than Person 4. In contrast, when the doors were open, peak exposure is still lower compared with the other three individuals, but the percentage difference is substantially reduced.
4.4 Discussion

It is evident that an individual’s exposure is dependent on the time that they spend in a location and the concentrations in that location during that period. In the present work, the simulations of a range of different personal profiles demonstrates how different individuals, all present in the same dwelling, under the same physical conditions can experience a large variation in exposure. Individuals can spend shorter durations in a room, but if higher concentrations are present during this period, this can result in a greater mean exposure than a long period spent in an area where there are lower concentrations.

The time-weighted approach to exposure estimation is found to be a poor substitute for a time-activity profile. In all cases examined in this paper, the time-weighted approach underpredicts an individual’s exposure, in some cases by up to 74.56%, also it fails to yield information on peak exposures. Gulliver and Briggs (2005) reported similar findings when assessing the journey time through the outdoor environment.

As can be seen from the results shown, peak exposure can vary by location, time and concentration, depending on the individual’s time activity profile, and individuals experiencing the highest peak exposure do not necessarily experience the highest mean exposure. Individuals experiencing the highest PM$_{10}$ peak exposure can differ from those experiencing the highest PM$_{2.5}$ peak exposure; a similar effect is observed when mean exposures are considered. These findings are important as they have implications for health risk assessment associated with PM exposure; it is well known that PM-related health effects vary depending on whether exposure is to PM$_{10}$, or PM$_{2.5}$, and whether mean or peak exposure is
Chapter 4: Simulated Variations in PM Exposure due to Different Time Activity Profiles Within the Same Dwelling

considered.

The results of the present work demonstrate that an individual’s mean PM exposure can exceed the maximum mean concentrations throughout the dwelling. This has important implications for analysing individual rooms within a household. Experimental observations or computational models that examine background concentrations provide information that is limited to conclusions regarding 24-hour mean exposure. For accurate exposure approximations, the results of the current work imply that concentrations in every room the individual is present must be examined in combination with a time-activity profile. This could help account for some, but not all, of the previously reported differences between personal exposure monitoring and indoor sampling (Monn et al., 1997; Janssen et al., 2000; Wheeler et al., 2011).

An interesting scenario occurs when comparing four individual’s exposure, for the two different cases of the doors being open and closed. When the doors were open, exposure reduced by approximately 20% for Persons 1 and 3, and 51% for Person 2, compared with the case when the doors were closed. Contrarily, when the doors were open, exposure increased by 280% for Person 4. With the doors open, Person 1 and 3 experienced a combined reduction in exposure of 34 $\mu g m^{-3}$ and 23 $\mu g m^{-3}$ for PM$_{10}$ and PM$_{2.5}$ respectively, while exposure for Person 4 increases by 39 $\mu g m^{-3}$ and 29 $\mu g m^{-3}$, respectively. If Person 2 is included, the combination reduction in exposure for Persons 1, 2, and 3 increases to 136 $\mu g m^{-3}$ and 86 $\mu g m^{-3}$ for PM$_{10}$ and PM$_{2.5}$ respectively. The overall reduction for Person 1, 2 and 3 is three times greater than the corresponding increase experienced by Person 4.

In attempting to analyse the health implications of the above findings, it is
necessary to review the relevant literature. Klemm et al. (2000) carried out a reanalysis of results from the Six U.S Cities Study (Schwartz et al., 1996), and reported that a 10% increase in a two day mean PM$_{2.5}$ exposure was associated with a 1.3% increase in daily mortality. If this finding is applied to the present data, and if Persons 1, 2 and 3 are each considered to represent a population group, their individual daily mortalities increase by 1.6%, 8.2% and 1.3% respectively, when the “closed door” case is compared the “open door” case for the simulated dwelling. However, if Person 4 is considered to represent a population group, the percentage mortality increases for this group alone when the doors are opened, by 3.8% compared with the doors closed. The overall population daily mortality increases by 3.7% in the doors closed case compared with the doors open.

Person 4, who spent most of their time in the bedroom in this simplified scenario, could be representative of an infant population group, or a population of seriously ill people. This finding of location-dependent variation in exposure with door configuration could have important implications for the design of simple behavioural strategies that would lead to exposure reduction for susceptible individuals who are confined to certain parts of a dwelling. Pattenden et al. (2006) reported on a postnatal parent smoking on the respiratory health of children in the range of 6 - 12 years of age; the study confirmed a range of respiratory symptoms associated with postnatal exposure. For susceptible population groups, additional weighting factors must be applied in exposure estimation. Woodruff et al. (2006) reported that, in post-neonatal infants who were examined from 2 weeks to one year old, a 10 $\mu g\ m^{-3}$ increase in overall exposure resulted in a 7% increase for overall post-neonatal mortality and a 213% increase for respiratory-related post-neonatal mortality. If this mortality estimation was applied to Person 4’s group,
daily mortality for that group would increase by 20.1% for the “door open” case compared to the “door closed” case. The overall population’s daily mortality increase by 7.7% in the **doors open** case compared with the **doors closed**.

Although this approximation is only for demonstrative purposes, and the behavioural profiles of the individuals in the simulations carried out are highly simplified, the mortality estimates provided do illustrate the importance of combining studies of health risk with population exposures. This highlights the need for careful consideration of exposure reduction strategies, as an overall lower population exposure might not necessarily correspond to the lowest daily mortality rates.

It should be noted that the above analysis does not distinguish between differences in chemical composition of the particles. Studies such as that of Franklin et al. (2008) have shown that chemical species modify the association between PM$_{10}$ and mortality when evaluating health effects of PM exposure. The composition of smoking particles are known to contain at least 67 carcinogens and 172 toxic substance in both gas and particulate phases (Repac, 2007). Miller and Hurley (2006) reported that for a single birth cohort a 10 µg m$^{-3}$ reduction in airborne PM$_{2.5}$ is predicted to add seven months to life expectancy, compared with 2 - 3 months for the elimination of passive smoking. However, the consideration of chemical composition is beyond the scope of this work.

### 4.5 Conclusion

This paper has demonstrated that a physical pollutant model must be combined with time activity profiles to create an accurate air pollutant exposure model. The results of the simulations carried out show that PM exposure varies considerably from mean concentrations predicted within a dwelling. In addition, it was seen...
that calculating exposure based on time averaged profiles is a poor substitute for calculating exposure based on time activity profiles.

This paper has demonstrated the important result that an individual’s exposure to PM$_{10}$ and PM$_{2.5}$ varies with their movements as well as the physical conditions within the dwelling. Within the same household, 24-hour mean exposures were seen to differ by up to a factor of 10. This can have a major impact in epidemiology studies where it is assumed exposure is constant within the same household. This paper has also emphasised the importance of combining such studies as this one, with investigations into chemical composition and health impacts with PM sources.

The model described in this work has the potential to simulate additional microenvironments and determine exposure for individuals as they travel throughout a range of microenvironments. The model can aid in the development of strategies to reduce population exposure through changes in physical environment (e.g. air exchange rate and emission events) but also lifestyle choices (amount of time spent in certain location).

4.6 Acknowledgements

This work was funded by Irish Environmental Protection Agency (EPA) under the STRIVE Programme.
Chapter 5

IAPPEM Validations

This chapter is divided into three sections; each section compares experimental to predicted concentrations under a variety of different physical conditions. Experimental measurements were gathered from a range of different households; each section summarises the input parameters used for the simulations.

Section 5.1 examines additional airflow validations conducted in a separate location to that described in Chapter 3. The airflow variations in this section were caused by the opening and closing of external windows and a sunroof.

Section 5.2 focuses on simulating a solid-fuel-fire burning event over a period of four hours. Simulations were run based on mean emission and airflow rates, a mean emission rate and a variable airflow rate, and finally variable emission and airflow rates.

Section 5.3 examines validations for different indoor emission source; no indoor emission source, multiple smoking events, an incense burning event with the subsequent six hour decay, and a combination of frying and kettle emission event.
5.1 Additional Airflow Validations

Two additional validations of the adapted airflow have been carried out by monitoring simultaneous indoor and outdoor concentrations of PM$_{10}$ and PM$_{2.5}$, using a SidePak Personal Aerosol Monitor Model AM510 (TSI Incorporated, Shoreview, MN, USA). The SidePak was set to log data at 1 minute intervals and calibrated to the recommended flow rate of 1.7 l/min using a primary air flow meter (DryCal DC Lite; BIOS International, NJ, USA.). The outdoor concentrations are needed as an input parameter for the model. The measurement of the indoor concentration can then be compared to the model’s output to test the accuracy of the model. To greater stress the variation that changes in airflow have, an emission source was used. The emission source selected was a tealight candle, with an approximate burning period of four hours, the emissions remains relatively constant over the four hour period. Twelve of these candles were lit to increase the concentrations.

The airflow values for a room of dimensions 2.2 m x 2.6m x 3 m (height) were determined (by CO$_2$ tracer gas decay measurement) under two different scenarios; one caused by opening and closing of windows, the second by the opening and closing of a sunroof. The CO$_2$ tracer gas decay measurements involved releasing CO$_2$, from a sealed cylinder, into the room, until concentrations exceed 2500ppm. A GrayWolf probe (GrayWolf Sensing Solutions; Shelton, CT, USA) was used for gas detection at one minute intervals. The natural log of the CO$_2$ concentrations was used to extrapolate the airflow rates.

The location chosen, as seen in Figure 5.1, was a conservatory, and this was chosen for the following reasons.
Chapter 5: IAPPEM Validations

- There was a direct airflow between the conservatory and the outdoor environment and the air does not pass through any additional rooms prior to entering the conservatory.

- The room is only connected to one additional room, in which there was a very low airflow. The adjoining room was a sealed room itself (not connected to the rest of the house by any internal doors) and as a result, would not have been affected by any activity taking place in the house.

- The floor in the conservatory was a tiled floor which greatly reduced the contribution that resuspension would have had if this was conducted on a carpet surface.

Figure 5.1: The conservatory in which the monitoring took place.
5.1.1 Airflow Variations caused by Windows Opening Patterns

The first scenario demonstrated airflow variations generated by the opening and closing of window patterns. The windows remained closed for a period, allowing PM$_{10}$ and PM$_{2.5}$ concentrations to rise sufficiently above background concentrations. All four windows (only two opened windows are shown in Figure 5.2) were opened to 30° for approximately 30 minutes.

![Figure 5.2: A picture showing the windows opened in the conservatory.](image)

Table 5.1 indicates the times when the candles were lit and when changes in the window patterns occurred. Comparing Figures 5.3 and 5.5 shows how indoor concentrations increase when the candles are lit and the indoor concentrations continue to increase when the windows remain closed. The opening of the windows results in increasing the AER, reducing the indoor concentrations in line with the outdoor concentrations. The same sequence of events can be observed when the windows are closed again. There is no sizeable increase in the outdoor concentrations during the monitoring period, with the exception of two spikes in the outdoor PM$_{2.5}$ values caused by an external interference.
Table 5.1: The different variations in airflow rates for different events, each measured using the tracer gas method. The time-weighted averaged airflow (Mean airflow) is also included.

Figures 5.4 and 5.6 demonstrate the enhancement of the model by using a variable airflow, showing better performance than the simulation using a mean airflow over the time period. In IAPPEM, simulated PM concentrations follow more accurately the approximate profile of the experimental concentrations than the INDAIR model. This is most evident when the windows are closed, with the reduced air flow, as with the reduced air flow the emissions result in higher PM concentrations in the room which the traditional model would under-predict.
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Figure 5.4: Measurement of the indoor PM$_{10}$ concentrations, along with mean airflow and variable airflow predicted concentrations.

Figure 5.5: Measurement of the indoor and outdoor PM$_{2.5}$ concentrations during the sampling period.
Figure 5.6: Measurement of the indoor PM$_{2.5}$ concentrations compared with the predicted concentrations using the mean airflow and the variable airflow concentrations.

5.1.2 Further Airflow variations

The second scenario considered airflow variations generated by the multiple variations in the sunroof, as seen in Table 5.2. Table 5.2 indicates the times at which candles were lit and when the changes occurred in the sunroof opening patterns. Figure 5.9 shows how indoor concentrations increase when the candles are lit and the indoor concentrations continue to increase when the sunroof remains closed. Figure 5.8 show that concentrations reach a steady state when the sunroof is partially opened; the emission from the candles equals the overall decay rate, so no increase or decrease is observed. When the sunroof is fully opened, the increased airflow increases the overall decay rate resulting in a decrease in PM$_{2.5}$ concentrations. Once the sunroof is closed, PM$_{2.5}$ concentrations begin to increase again.
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<table>
<thead>
<tr>
<th>Time</th>
<th>Event</th>
<th>Airflow Rate</th>
</tr>
</thead>
<tbody>
<tr>
<td>21:10</td>
<td>Sunroof Closed</td>
<td>3.5 m³ h⁻¹</td>
</tr>
<tr>
<td>21:20</td>
<td>Candles Lit</td>
<td>N\A</td>
</tr>
<tr>
<td>22:15</td>
<td>Sunroof 1/3 Opened</td>
<td>5.1 m³ h⁻¹</td>
</tr>
<tr>
<td>22:45</td>
<td>Sunroof Fully Opened</td>
<td>13.7 m³ h⁻¹</td>
</tr>
<tr>
<td>23:35</td>
<td>Sunroof Closed</td>
<td>3.5 m³ h⁻¹</td>
</tr>
</tbody>
</table>

Mean airflow: 6.3 m³ h⁻¹

Table 5.2: Sunroof scenarios with three different airflow rates.

Figure 5.7: A picture showing the sunroof in two different opening positions, demonstrating different airflow conditions.
Figure 5.8: Measurement of the indoor PM$_{2.5}$ concentrations, and predicted concentrations using the mean airflow and variable airflow.

Figure 5.9: Measurement of the indoor and outdoor PM$_{2.5}$ concentrations during the sampling period.
5.2 Simulation of Solid Fuel Burning

An important aspect of modelling air pollution levels in any indoor environment has to be the source of heating that is used within the environment. Most homes in Ireland contain a primary heating system operating on either oil or gas as the fuel source. However, in Ireland, many homes have an additional source of heating, consisting of an open or closed solid fuel fire. Guo et al. (2008) investigated the effect of domestic fireplaces as a supplementary heating source and where peat or coal was used in combination with smoking, that 7 out of 9 houses had PM$_{10}$ concentrations that exceed the WHO threshold of 50 µg m$^{-3}$ between 20 - 30% of the time. Open combustion sources have not been given major consideration in previous versions of INDAIR.

Figure 5.10: An example of a solid fuel fire burning event.

A measurement of the emissions from a solid fuel fire, dealing specifically with the emission of PM$_{2.5}$ from coal burning was carried out. PM$_{2.5}$ concentrations were recorded throughout the burning events (at a distance of 1.5 m from the fire), using a SidePak Personal Aerosol Monitor Model AM510 (TSI Incorporated,
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Shoreview, MN, USA). The SidePak was set to log data at 1 minute intervals and calibrated to the recommended flow rate of 1.7 l/min using a primary air flow meter (DryCal DC Lite; BIOS International, NJ, USA.). Simultaneous outdoor PM$_{2.5}$ concentrations were also recorded. Indoor sampling was conducted in a room of dimensions 3.58 m x 4.36 m x 2.67 m (height), and varying airflow rates were achieved through a range of door opening patterns and door sealing exercises. Airflow rates were measured using the tracer gas decay method, with CO$_2$ as the tracer, following the same procedure as described in Section 5.1

Following 30 minutes of background concentration measurements the fires were lit. Figure 5.11 demonstrates how indoor and outdoor PM$_{2.5}$ concentrations remain relatively similar until the fire is lit, at which stage indoor PM$_{2.5}$ concentrations start to exceed outdoor PM$_{2.5}$ concentrations.

![Figure 5.11](image)

Figure 5.11: Comparison of indoor and outdoor concentrations for PM$_{2.5}$, during the solid fuel burning event.

All fires were started with a compressed sawdust log (Figure 5.12a), which
was allowed to burn for approximately 30 minutes. After this period, 3 briquettes of compressed peat (Figure 5.12b) were added to the fire, as it was found that this reduced the smouldering effect that resulted from the direct addition of coal (Figure 5.12c). After a further 5-10 minutes, coal was added and this was the only fuel that was burnt for the remainder of the sampling period.

Figure 5.12: Examples of the fire log, briquette and coal that are burnt during the solid fuel burning event.

Figure 5.13 shows the comparison between experimental and predicted indoor PM$_{2.5}$ concentrations when the simulations were based on a mean airflow and a mean emission rate.

On further investigation, it was found that the solid fuel burning altered the airflow within the room. The heat emitted by the fire created an additional suction effect up the chimney, increasing the airflows for the room. Figure 5.14 shows
Figure 5.13: Comparison of indoor PM$_{2.5}$ concentrations to predicted PM$_{2.5}$ concentrations simulated using a mean airflow and a mean emission rate.

the comparison between the experimental and predicted concentrations when the airflows change at different stage, as seen in Table 5.3.

<table>
<thead>
<tr>
<th>Event</th>
<th>Airflow Rates</th>
</tr>
</thead>
<tbody>
<tr>
<td>Before Lighting Fire</td>
<td>8.3 m$^3$ h$^{-1}$</td>
</tr>
<tr>
<td>Added Briquettes</td>
<td>14.6 m$^3$ h$^{-1}$</td>
</tr>
<tr>
<td>Added Coal</td>
<td>19.17 m$^3$ h$^{-1}$</td>
</tr>
<tr>
<td>An hour after added Coal</td>
<td>13.3 m$^3$ h$^{-1}$</td>
</tr>
</tbody>
</table>

Table 5.3: The different airflows rates at different stages during the solid fuel burning event.

In order to determine the most efficient method of modelling solid fuel burning emissions in Irish fireplaces, it was observed that both variable airflow (Values in Table 5.3) and emission rates (Values in Table 5.4) were needed, as shown in Figure 5.15. It was observed that the fire had three primary stages of emission, the initial intensity at the start of the fuel burning event, it reaches steady state and the eventual decrease as the fire begins to cool down. Due to the nature
Figure 5.14: Comparison of indoor PM$_{2.5}$ concentrations to predicted PM$_{2.5}$ concentrations simulated using a variable airflow and mean emission rate.

Figure 5.15: Comparison of indoor PM$_{2.5}$ concentrations to predicted PM$_{2.5}$ concentrations simulated using a variable airflow and a variable emission rate.
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<table>
<thead>
<tr>
<th>Event</th>
<th>Emission Rate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Initial Stage</td>
<td>372 $\mu g \ h^{-1}$</td>
</tr>
<tr>
<td>Log Lighting</td>
<td>120 $\mu g \ h^{-1}$</td>
</tr>
<tr>
<td>Added Briquettes</td>
<td>124 $\mu g \ h^{-1}$</td>
</tr>
<tr>
<td>Added Coal</td>
<td>127 $\mu g \ h^{-1}$</td>
</tr>
<tr>
<td>An hour after added Coal</td>
<td>127 $\mu g \ h^{-1}$</td>
</tr>
</tbody>
</table>

Table 5.4: The different emission rates at different stages during the solid fuel burning event. Emission rates were calculated from experimental observations in previous datasets using Equation (3.1).

of the fire, the increase in fire’s intensity has the secondary effect of creating an additional suction up the chimney; so as the fire intensity increases, so too will the airflow values and vice versa. Figure 5.15. shows good agreement between modelled and experimental concentrations, the two peaks at 14:44 and 15:47 which are under-estimated by the model, are believed to be caused due to the addition of fuel to the fire creating a brief upsurge of particles as fuel is placed on top of the existing fire.
5.3 Additional Experimental Validations

A number of indoor emission source validations were also performed. The same experimental procedure that was described in Sections 5.1 and 5.2 for the comparison was also employed in this section, although in different locations. The emissions rates and deposition velocities used in these simulations are found in Table 2.2 and section 2.2.4.4. The simulations in Figures 5.16 to 5.18 all occurred in the sitting room in Figure 3.1.

Figure 5.16: A no source scenario examining the indoor-outdoor relationship, comparing predicted PM$_{2.5}$ concentrations with indoor PM$_{2.5}$ concentrations.
Figure 5.17: Simulation of burning an incense stick simulating for six hours, to examine the decay period. Predicted PM$_{2.5}$ concentrations are compared with indoor PM$_{2.5}$ concentrations.

Figure 5.18: A simulation examining multiple smoking events. A cigarette was smoked every 30 minutes. Predicted PM$_{2.5}$ concentrations are compared with indoor PM$_{2.5}$ concentrations.
Chapter 5: IAPPEM Validations

Figure 5.19: Measured PM concentrations, compared with the output of the IAPPEM model.

Figure 5.19 shows the IAPPEM model output when run under the same conditions as those under which the sampling was conducted. From 12:00 midnight until 09:00am, the individual carrying the sampler was in the home environment, and after this time, he left this environment. In the period 12:00 to 12:15, the kettle was boiled, after which time the concentrations of PM$_{10}$ are seen to reduce. From 08:00 to 08:05, the kettle was boiled again and from 08:24 to 08:28 the individual was making toast. Experimental Data for this simulation was provided by Andrew McCreddin TCD (McCreddin, 2013).
Figure 5.20: Measured PM concentrations, compared with the output of the IAPPEM model.

Figure 5.20 shows an individual who returns to their home at 20:25 for the remainder of the night. He started to fry at 20:57 until approximately 21:03. For the rest of the time there was no other activity in the house. Experimental Data for this simulation was provided by Andrew McCredden TCD (McCredden, 2013).
Chapter 6

General Discussion, Final

Conclusions and Future Work

The overall aim of this work was to develop a computational model to evaluate exposure to Particulate Matter (PM) as individuals move through a series of microenvironments. The project focused on adapting an existing model, the INDAIR model, further developing the physical pollutant component and incorporating a time activity model to create an overall exposure model, IAPPEM. IAPPEM simulates 24-hour activity profiles, predicting exposures as individuals move through different microenvironments.

6.1 General Discussion

IAPPEM can determine the PM concentrations in a number of rooms throughout a dwelling, identifying the contribution from both indoor and outdoor sources, analysing indoor emission sources in terms of emission strength and duration, determine the necessary decay rates for different sources and the influence internal
configuration has on indoor concentrations.

Separating indoor and outdoor concentrations is essential for an accurate estimation of health effects associated with PM exposure. Emission sources vary in chemical composition and toxicology and in turn, different health risks are associated with each emission source. Identifying the contribution that each source has to exposure is vital in understanding the health implications. Such techniques become extremely difficult to accomplish through experimental observations. In experimental observations, photometric devices are one of the most widely employed techniques in gathering experimental measurements. These devices operate on the principle of light scattering using the volume of particles to determine particle mass. Applying a calibration factor adjusts measurements to account for variations in particle densities arising from different emission sources. Unfortunately, photometric measurements do not distinguish between particles from multiple sources. Calibration factors can only be applied to entire data sets, leading to uncertainties in estimating mass concentrations. Computational models overcome this issue, either by determining concentrations based on the input parameters, or by separating the contribution from each source, allowing calibration factors to be applied in each case.

An important outcome of the current study has demonstrated on a number of occasions, the differences between exposure and concentrations; this was as discussed in Chapter 4. While concentrations must be known in order to determine exposure, without a detailed consideration of a time activity profile and the full range of microenvironments, conclusions about exposure become limited. Short term fluctuations in concentrations can have considerably more impact on exposure than on mean concentrations.
Although, all simulations reported in this work have focussed on PM$_{10}$ and PM$_{2.5}$, the model is capable of simultaneously simulating up to four pollutants. The remaining two unparameterised pollutants could be used to simulate a number of different gaseous or particulate pollutants, e.g. NO$_2$, CO (as in previous versions of the model). Provided that the same modelling principles apply to the additional pollutants, the assumptions regarding uniform distribution and the mass balance equation within a zone. Parameterising IAPPEM with the appropriate outdoor concentrations, deposition velocity and emission rates would predict concentrations for the remaining pollutants.

6.2 Summary of Model Adaptations

The adaptations made to the INDAIR model in developing IAPPEM have been discussed in Chapters 2 and 3 but only in terms of the impact on PM concentrations. Appendix A provides the details regarding the necessary modifications to the model’s source code that are required to achieve the adaptations. The benefits of each adaptations in terms of predicting individual exposure are summarised below:

a) In contrast to INDAIR, which involved a simplistic three room layout of the home environment, the current work developed the model into fifteen interconnecting rooms which can be combined to represent different household layouts. This allows the examination of source location and household layout on indoor concentrations examined in Section 2.3.5, while predicting exposure for multiple individuals present in the same dwelling. The model determines the rooms containing the lowest concentrations and this information can be used to develop strategies to help reduce exposure.
b) The resolution of the model was improved from fifteen minutes to one minute. The increased resolution improves predictions of peak PM concentrations, discussed in Section 2.3.1. While no sizeable difference occurs in the 24-hour mean concentrations, the same is not true for the 24-hour mean exposure. Short-term peaks in concentrations can have a considerably greater impact on mean exposure than on mean concentrations. An individual moving through a series of microenvironments can be exposed to multiple peaks in different microenvironments, although each peak might only occur once in each microenvironment.

c) The adapted PM$_{10}$ deposition is an important feature, especially with the inclusion of multiple indoor emission sources. PM$_{2.5}$ to PM$_{10}$ ratios vary for both outdoor PM concentrations and indoor PM emission sources. Failure to separate out the PM$_{2.5}$ contribution from PM$_{10}$ results in the PM$_{2.5}$ contribution assumed to decay at a rate calculated for the PM$_{10}$ deposition velocity, hence over-estimating PM$_{10}$ decay. However, separating PM$_{2.5}$ from PM$_{10}$, results in the prediction of higher PM$_{10}$ concentrations for longer durations, as discussed in Section 2.3.2. This results in greater PM$_{10}$ transfer throughout a dwelling and higher exposure.

d) Simulations using a variable airflow compared with a time-weighted average airflow shows potential under-estimations in both peak and mean concentrations in adjoining rooms, especially when emission sources are present, as discussed in Chapter 3. Additional validations are shown in Section 5.1. The greatest discrepancies occur in peak concentrations, having implications when calculating individual’s exposure. This is a significant finding, as short-term fluctuations in concentrations can account for a large exposure variation
between population groups.

e) The inclusion of multiple emission sources increases the ability to simulate real life scenarios; in most households multiple emissions sources are present. Peak and mean PM indoor concentrations vary depending upon the emission source, emission duration and the source’s location as discussed in Sections 2.3.3 and 2.3.5.

f) The INDAIR model had only three additional microenvironments; outdoors, shop, restaurants and transport. Within these microenvironments PM concentrations were calculated based on indoor/outdoor ratios. IAPPEM include the capability to conduct simulations in a total of 10 microenvironments (in addition to the home microenvironment), with the full functionality of the home environment. The microenvironments included are the classroom, office, supermarket, gym, car, tram, train, bus, outdoors, restaurant and a pub. Each environment has names attached in the code, this does not limit alternative parameterisation being applied for the simulation of different microenvironments. The additional microenvironments, although not demonstrated in this work, have the potential to better evaluate individual exposure over a 24 hour profile.

6.3 Additional Applications

Although IAPPEM was designed primarily to simulate PM$_{10}$ and PM$_{2.5}$ exposure in the residential environment, the model can easily be adapted to allow exposure simulations for a range of alternative scenarios as described below.
6.3.1 Microenvironments

a) The model can analyse population exposure resulting from extreme outdoor events. Earlier work has involved the provision of PM$_{2.5}$ population exposure estimations following the collapse of the World Trade Center (Ng et al., 2005), which released a large amount of dust and inhalable toxic substances into the air. The INTAIR model (Dimitroulopoulou et al., 2001a) was used to provide indoor exposure estimations to the population of lower Manhattan.

Alternative events, such as the eruption of the volcano, Eyjafjallajökull, resulted in large-scale volcanic ash emission. Thorsteinsson et al. (2012) reported, on 7 May 2010 in Vik, a small town 38 km SE of the volcano, 10 minute average values exceeded 13,000 $\mu g$ $m^{-3}$ and the 24 hour mean PM$_{10}$ concentrations were 1230 $\mu g$ $m^{-3}$. The volcanic cloud also travelled across Europe, O’Dowd et al. (2012) reported PM$_{2.5}$ concentrations of 37.7 ± 0.6 $\mu g$ $m^{-3}$ and PM$_{10}$ concentrations of 46.9 ± 0.6 $\mu g$ $m^{-3}$ at Mace Head, Galway, Ireland. Other studies also reported increased PM concentrations across Europe (Chazette et al., 2012; Perrone et al., 2012). While outdoor models can predict outdoor concentrations of the ash at different heights, IAPPEM could be used to estimate potential increases in indoor PM concentrations resulting from the ash cloud.

b) Within the manufacturing sector, there exist a wide variety of emission sources in enclosed areas. Numerous experimental studies reported on the risk of welding fume exposure as a cause of concern (Hedmer et al., 2013; Woskie et al., 1994; Jenkins et al., 2005). Healy et al. (2013) reported peak personal exposure of respirable dust between 1-10 $mg$ $m^{-3}$ when cutting or grinding
sandstone with an angle grinder indoors. Baatjies et al. (2010) reported, in a study of 18 South African bakeries, that bread bakers were exposed to average flour dust particulate concentrations of 1.33 mg m\(^{-3}\), with an average of 20 workers per bakery. Paudyal et al. (2011) investigated personal exposure to inhalable dust during full-shifts for 114 workers in Nepal. The study reported geometric mean exposure of 3.36 mg m\(^{-3}\) in the recycling sector and 1.16 mg m\(^{-3}\) in the carpet sector.

In the majority of the studies mentioned above, the focus was based on assessing the worker’s exposure. However, any additional workers in the same or adjoining rooms could still be exposed to high concentrations. With appropriate parameterisation, IAPPEM could be used to investigate the exposure of individuals not directly involved in tasks that generate workplace PM.

c ) Restaurants have a wide range of cooking activities that can emit large amounts of particulate (Buonanno et al., 2010; To et al., 2007; Yeung and To, 2008). While the cooking takes place in a separate kitchen area, PM may transfer into the dining area where customers are present (Pan et al., 2008; Huboyo et al., 2011). The IAPPEM model could be used to provide a clearer understanding of the concentrations in the dining area, and customers’ exposure.

d ) A number of studies have highlighted the effect of biomass fuel use on indoor air quality in developing countries (Fullerton et al., 2009; Akintan, 2012). In recent years, a number of projects (Global Alliance for Clean Cookstoves, 2011; Clark et al., 2009; Dutta et al., 2007) have focused on replacing cooking on open fires with wood stoves, to reduce PM exposure. With appropriate parameterisation, IAPPEM could be used in this context to provide exposure
estimations, highlighting current high risk practises and providing exposure estimations after strategies have been implemented and the associated health benefits.

6.3.2 Potential PM Reduction Strategies

IAPPEM predicts 24-hour PM exposure, identifying periods and locations that contribute most to exposure and hence allowing the design of exposure reducing strategies. Two different types of strategies can be devised. One strategy examines individuals’ behavioural patterns and highlights the benefits of reducing or eliminating the time spent in certain microenvironments. However, this strategy is limited to specific scenarios, e.g. when an individual is cooking food, they are forced to spent time in that microenvironment. The second strategy examines the physical environment, investigating mechanisms to reduce concentrations by eliminating emission sources or altering the physical environment to increase the removal of particles.

It is important to consider that an overall lower mean exposure does not always correspond with an overall lower concentration, as was demonstrated in Section 4.3.2. Forcing concentrations to remain high in certain microenvironments, for example by keeping doors closed, could lower concentrations in other microenvironments. This could reduce overall exposure, depending on the durations spent in each microenvironment.

6.3.2.1 Behavioural Strategies

To date, IAPPEM has been largely employed for domestic simulation. The inclusion of additional microenvironments would allow the simulation of the profiles
of individuals as they move through a range of different locations, permitting the following investigations:

a) Investigations into the exposure impact of different modes of transport used by commuters could be carried out. Although certain modes of transport are potentially used for a longer duration than others, lower PM concentrations could be present, e.g. trains interiors could have lower pollutant concentrations than car interiors, as they are separated from traffic congestion. Dirks et al. (2012) examined carbon monoxide exposure while commuting by different modes on separate routes; in two cases individuals travelling by bus had a lower exposure, and in the third case individuals who travelled by car had a lower exposure. Other studies reported similar results (Nasir and Colbeck, 2009; Gulliver and Briggs, 2004) which highlights that the key factors are the time travelled and the route taken, rather the actual mode of transport.

Depending on the bus route used, an individuals’ exposure can vary considerably and be affected by a number of factors; such as the duration at bus stops, outdoor concentrations and the number of people onboard (Song and Ashmore, 2007). Huang and Hsu (2009) attributed lower PM$_{2.5}$ concentrations in their study to non-stop driving and highway surroundings.

IAPPEM combined with a GIS model (to provide outdoor concentrations), would allow individuals to determine which routes and modes of transport would result in lower air pollution exposure.

b) Investigations into the effect on exposure of internal door opening patterns could be carried out. For example, if the bedroom doors remained open for the majority of the day, but were closed prior to individuals going to bed, this
would achieve a greater particle dilution throughout the day, while ensuring that bedroom concentrations decay before individuals are present, provided sufficient time is allowed for particles to decay. This results in lowered exposure during the day, without higher exposure at night. Additionally, leaving internal doors open in unoccupied rooms, allows for greater dilution and deposition of particles. Since these rooms are unoccupied, the only result could be decreased concentrations in the remaining rooms, resulting in decreased exposure. IAPPEM could be used to allow the definition of appropriate time for occupancy of various rooms in order to minimise exposure. Encouraging individuals to leave doors open in unoccupied rooms, would be a simple method of reducing overall population exposure, although such strategises would undoubtably create tension between measures to conserve heating and energy consumption.

c IAPPEM permits comparison of several potential air pollution strategies, in order to identify the strategy with the greatest benefit to human health. For example, studies have commented on the effect smoking in the car has on exposure (Semple et al., 2012a; Ott et al., 2008). Delaney and McGettigan (2013) reported on the potential PM$_{10}$ reduction from the introduction of the Port Tunnel in Dublin, reducing vehicle traffic flow through Dublin city centre. While this reduction lowers outdoor concentrations and effects the large population, the elimination of smoking in cars would have a major exposure reduction on the individuals involved. The model could provide policy makers with an insight into which strategies would yield the greatest overall health benefit.
6.3.2.2 Building Design Related Strategies

a) Examining different household layouts could aid in the design of strategies to passively reduce PM concentrations in a dwelling by maximising decay and minimising PM transfer, as demonstrated in Section 2.3.5. It was shown in Section 2.3.5 that particles travelling through an additional room to that containing the source reduces the PM transfer through the dwelling. Connecting the dining room directly to the hallway instead of the kitchen forces particles to travel through the additional room before reaching the hallway, resulting in lower concentrations in the remainder of the household. Additionally, if both the kitchen and dining room doors were closed, particles would be contained away from the remaining household and overall exposure would be reduced, especially if multiple individuals were present as was seen in Chapter 4.

b) The model could be used to investigate the potential difference between a separate kitchen and dining room layout or a combined layout. The separate layout effectively reduces exposure for anyone not present in the kitchen, particles decay in the kitchen prior to infiltrating into the dining room. Keeping the doors closed considerably reduces exposure for anyone present in the dining room. Alternatively, for the “cook”, the combined layout results in greater particle dilution reducing the cook’s exposure.

Chapter 4 demonstrated how population based exposure varies within a household depending on the number of individuals present and on the doors being open or closed. Architects, building engineers or planning authorities could aim to achieve the lowest population exposure in the residence, insisting, for example on self closing chains on kitchen doors.
c) Investigations into the effect of varying airflow in different rooms for different types of dwelling could be carried out. Chapter 3 demonstrated the effect airflow variations had on influencing indoor PM concentrations. Increasing external airflow in rooms containing emission sources increases the removal of particles; while decreasing airflow in rooms without emission sources reduces the infiltration of outdoor particles, providing these rooms do not experience high internal PM transfer.

The European directive (2010/31/EU, 2010) sets minimum energy performance requirements for new buildings. Irish legislation (S.I. No. 259/2008, 2008) refers to limiting the heat loss from pipes, ducts and vessels that are used for the transport of air, increasing the air tightness of buildings, decreasing airflow. The choice of appropriate airflow needs careful consideration in order to maintain energy efficiency in homes, while not impacting on human health. The above factors vary depending on room dimensions and internal layout, each dwelling would require separate analysis.

It should be noted that although increasing airflow decreases short-term peaks in concentrations due to emission sources, it also results in an increase in the infiltration of outdoor particles. Depending on the relationship between indoor sources and outdoor concentrations, increasing external airflow potentially increases or decreases 24-hour mean concentrations, depending on the outdoor concentrations.
6.3.3 Population Exposure Studies

The IAPPEM model predicts 24-hour exposure for individuals or population groups, and in order to accurately assess the health risk to population groups, exposure to air pollution must first be determined. Once exposure has been calculated, it needs to be combined with health risk analysis to provide a clearer picture of the health implications.

Simulating different population groups, identifying groups with a higher health risk is possible with the IAPPEM model. Once specific groups have been identified, public awareness campaigns could encourage these groups to become more health conscious, specifically in relation to high risk health effects. The medical sector could become more prepared for specific symptoms in certain regions, and potentially request additional resources for regions or periods of higher health risk. A number of suggested simulations using IAPPEM are listed below that could potentially identify different population groups with higher exposures.

a) Seasonal variability in PM exposure could be investigated i.e. due to elevated heating fuel usage in the winter, increases in traffic flow, or variation in airflow due to increased indoor-outdoor temperature differences.

b) Differences in exposure in various types of dwelling structures could be investigated. For example, the comparative exposure of individuals living in sub-urban housing estates versus city centre apartments could be simulated. Apartments typically do not have additional heating sources in contrast to houses, Irish Central Statistics Office (2012) report that almost half of all occupied apartments in 2011 were heated by electricity, while 75% of all detached house, semi-detached and terraced house were heated by oil or gas in
Ireland. City centre apartments are typically located closer to main roads (higher outdoor concentrations) but occupants may have lower commuting times as a result of living in the city.

c ) The air pollution in a family home compared with a house of young professionals could be investigated. In a family home, meals would be expected to be prepared for the entire household, whereas in a house of young professionals where each individual prepares their own meals. In the family home, children would be more physically active in the house, generating additional resuspension and the family are more likely to spend time eating together.

d ) The effect of smoking bans in pubs has been well documented (Semple et al., 2007; Cains et al., 2004; McNabola et al., 2006). Many pubs have created ill-defined outdoor smoking areas. Irish legalisation (06/2002, 2002) and (42/2004, 2004) refers to an outdoor smoking area as “an outdoor part of a place or premises covered by a fixed or movable roof, provided that not more than 50 per cent of the perimeter of that part is surrounded by one or more walls or similar structures (inclusive of windows, doors, gates or other means of access to or egress from that part)”. There is no legal requirement as to the minimum proximity from external doorways or windows. Mulcahy (2010) reported an 83% reduction in air nicotine concentrations in Irish bars following post smoking ban measurements. This study suggested that outdoor tobacco smoke may be infiltrating into the indoor environment. IAPPEM could be used to assess the indoor concentrations based on the proximity of the outdoor smoking area and determine appropriate distances that the smoking area should be located from the pub to minimise tobacco related indoor air pollution concentrations. For example in Canada, The Tobacco Control Act
(in force May 31, 2004) together with the Environmental Tobacco Smoke Work Site Regulations (Section 25 of the Safety Act, 31\textsuperscript{st} of May, 2004) prohibits smoking within 15 m of any entrance/exit to a school and within 3m of any entrance/exit to an enclosed workplace.

6.4 Limitations of the studies

The limitations of IAPPAM can be divided into two categories. The first set of limitations reflects the assumptions regarding how the model calculates exposure. Overcoming these limitations, would require further development of the model. The assigned input parameters from literature imposed the second set of limitations. However, as additional information becomes available in literature, these values need only be updated in a text file, requiring no additional modifications to the model.

6.4.1 Limitations of the model

The model assumes all particles distribute evenly throughout each room. The assumption of uniform distribution assumes no variation in concentrations at different horizontal or vertical positions, which implies that during decay, concentration closer to ground level should remain higher for longer durations compared with higher vertical positions. The overall mean concentration might remain constant, but height variations would have a considerable impact on exposure, especially for young children or babies who have breathing zones located closer to the ground.

Although the current model considers a realistic number of rooms in the dwelling, in a room with multiple adjoining rooms (e.g. a hallway), no emphasis has been placed on particles entering rooms in a given order. For example, par-
particles are assumed to enter rooms at the end of the hallway at the same time as rooms at the other end, although one room could be adjoining a room containing an emission source.

The use of airflows in the model assumes that particles are removed evenly from a room while in practice, particles are removed in stages with particles closest to the door being remove first. There is no account for local ventilation in the model, as airflows are averaged for an entire room.

The model assumes that deposition occurs evenly on all wall surfaces. In literature (Afshari et al., 2007) different deposition velocities are associated with different wall surfaces, but in IAPPEM deposition to a smooth uniform wall is assumed. Afshari et al. (2008) reported on the correlation between the amount of carpet on a floor and the increase in deposition rates and Thatcher et al. (2002) reported that furnished rooms had increased deposition velocities compared with unfurnished rooms. Afshari and Reinhold (2008) reported that carpet and curtain resulted in higher PM deposition rates compared with glass and gypsum board surfaces. In reality, homes contain a mixture of curtains, carpets, glass windows and wall surfaces. In offices, glass facades can encompass up to 50% of the wall surface (Pasquay, 2004). However, as the sensitivity analysis (as discussed in Section 2.5) showed that deposition velocity was one of the least influential factors affecting PM concentrations, this should not be viewed as a major limitation of the model.

6.4.2 Limitations of the Parameterisation

Following a review of current literature, a large number of values used to generate the modelled PM$_{2.5}$ emission rates for smoking, incense and candles; however only
a single source of values for frying and solid fuel burning were available, leading to large uncertainties in frying and solid fuel burning emission rates. Fewer PM$_{10}$ emission rates are available in literature, forcing values to be extrapolated based on PM$_{2.5}$ to PM$_{10}$ ratios from a number of different studies. This leads to uncertainty in predictions made using these emission source strength values.

A wide range of deposition velocity values are reported in the literature. However, these values are mainly based on studies that have typically analysed monodispersed sized particles (Byrne et al., 1995; Fogh et al., 1997) which do not truly reflect PM$_{10}$ or PM$_{2.5}$ deposition. Other studies (Ozkaynak et al., 1996; Olson and Burke, 2006) focused on particles decaying from a single emission source, but these do not apply to every emission source.

In the model, emission sources are assumed to remain uniform over the emission period. For short emission durations, e.g. smoking or frying, no major limitations are expected. However for emission sources with longer durations, variation in emission rates would have an impact on predicted concentrations; Section 5.2 shows how the use of a mean emission rate does not accurately model emission from a solid fuel fire.

In modelling terms, resuspension can be treated as an emission source. Although in practice, emissions rates for resuspension will vary depending on the actual surroundings. Rosati et al. (2008) reported that the amount of PM that can be resuspended from carpets is directly proportional to the amount of PM deposited on the carpet. Ozkaynak et al. (1996) reported on technicians gathering PM concentrations in Riverside who assigned a variable, the “dirty level”, in order to provide estimations for the levels of resuspension.

While information is available on the percentages of the Irish population that
live in different types of dwellings, and the typical number of rooms and occupants (Irish Central Statistics Office, 2012), on Irish commuting patterns (Irish Central Statistics Office, 2009) and distribution by sector of the Irish workforce (Irish Central Statistics Office, 2008), only limited information is available on recreational activities; times, durations and locations where individuals spend their time. There is no comprehensive information available on the behavioural patterns of individuals in the home in Ireland.

6.4.3 Future Model Parameterisation Needs

While emission rates have been widely reported based on experimental studies, the primary focus of these studies has often been examining 24-hour concentrations, and the actual emission rates have been extrapolated from a large data set. Detailed experimental studies where the primary focus is determining PM$_{10}$, PM$_{2.5}$ and PM$_{1}$ emission rates for the purposes of parameterising computational models, are rare. Separating emission sources into smaller size fractions, which would provide better estimations for PM deposition rates is desirable.

While some preliminary modelling validations have been carried out, as shown in Section 5.3, an extensive experimental comparison with predicted concentrations should be carried out. Such a validation should encompass a range of different dwellings and microenvironments. Validating a model against concentrations reported in literature has generally resulted in considerable uncertainties; Dimitroulopoulou et al. (2006) reported linear regression values of 0.525 to 0.597 for PM$_{10}$ and 0.647 to 0.706 for PM$_{2.5}$, when comparing the model with values from literature. The weak correlation was attributed to unknown variations in conditions (volume, airflow etc.) between studies. There is a need for a validation
study, where data on input parameter values is collected at the same time as the time-resolved exposure and location information.

6.5 Future Model Development

Further development of the IAPPEM model should investigate the assumptions regarding uniform PM distribution in a room. Subdividing each room in the model into additional zones (a sub-zonal model as discussed in Section 1.3) would allow investigations and resolution of some of the above limitations already described. Possible developments in this context include the following:

a) In a sub-zonal model, weighting factors could be applied to the transfer of particles between zones, to investigate any time lags in reaching uniform distributions, or to identify regions of higher concentrations. The weighting factors could relate to convection currents controlling particle distribution throughout the room. This would require a detailed house-layout specifying the locations of internal heating sources and windows/doors.

b) Sub-zonal modelling could analyse the vertical and horizontal locations of emission sources. This approach would allow the investigation of individual’s exposure with proximity to emission sources. Sub-zonal modelling could allow for accurate representation of the “personal cloud effect” (Ozkaynak et al., 1996), and the effects of resuspension on personal exposure.

c) Although, sub-zonal models are not necessary to simulate variations for a single wall or floor surface, the variation needs to be uniform over the whole surface. However, if half a wall had window or curtains, modelling sub-zones would account for these variations.
d) The sub-zones could be used to analyse PM differences during vertical decay, and to investigate PM concentrations at different heights, particularly around the breathing zone.

e) In rooms with multiple doors, e.g. hallways, a sub-zonal approach would allow particles to enter certain rooms in a certain order. This could impact on exposure as rooms nearer an emission source could have higher concentrations than rooms further away.

f) Air flow could be given greater importance in a sub-zonal model, allowing for cross draughts or effects of local ventilation due to open windows and extractor fans in rooms to be considered.

While the above development would overcome many of the limitations in the current model, the practicality of these enhancements needs further consideration. Splitting each room into two zones, doubles the amount of time and memory needed to complete the simulations. However with continuous technological improvements in computing speed, the complexity of such simulations is perhaps a real prospect. At present too much uncertainty exists regarding input parameters for current simulations, to justify such enhancements to the model.

6.5.1 Communication to the General Public

In order for the model to be of most benefit, the general population need to have access to it, to raise awareness of the harmful effects of particulate matter on health. By modifying the model to be incorporated into on a website, individuals could assess their own exposure comparing it to the corresponding health risk. This would raise awareness of benefits that could result form lowering their
exposures.

It would be advantageous to develop the model into a mobile app (mobile application) designed to run on smart-phones or tablet computers e.g. iPads. The app could include a wide range of scenarios that accurately reflects the population, allowing a user to select the scenario that most closely reflects their living conditions.

Combining IAPPEM with GIS modelling would enable up-to-date data collection on day-to-day exposures. Encompassing meteorology and traffic conditions in the model would facilitate choices of which mode of transport and route would result in the lowest exposure.

It is now mandatory that all homes in Ireland have a Building Energy Rating (BER) under European law (S.I. 243 of 2012, 2012) for the inclusion in the sale of property or with rental accommodation. Based on the results in this study, similar evaluations are recommended on indoor air quality. IAPPEM has the potential to provide an excellent tool to promote better indoor air quality in homes.
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Appendix A

Modifications to the Model

Source Code

Chapters 2 to 4 described the adaptations to the model focusing on the effects the adaptations have on predicting concentrations. No details were provided on how these modifications were implemented on the source code. This section provides additional information describing the general approaches and algorithms that were applied to the model.

One of the major changes to the code was converting the model from C++ to Java. C++ is not designed as cross-platform software, meaning if the model is written in C++ on one computer does not mean it will run on another computer even if they support the C++ programming language. Java, on the other hand, is a cross-platform language running on the Microsoft Windows, Linux and Mac OS X operating systems.

During the model’s development, one of the main aims was to ensure that the model would be user friendly. The model was designed for the Irish EPA, with the hope of becoming available to multiple users, for example other research
Appendix A: Modifications to the Model Source Code

projects, government departments or individuals wishing to assess their own personal exposure. The model needed to be operable by external users who were perhaps unfamiliar with the model or lacked skills in computer programming. Some adaptations had multiple possible approaches to implement the adaptations, but careful consideration was needed to assess which approach best suited the user.

A.1 Principle of the Probabilistic Model

This section provides the general principles of how the probabilistic nature of the model operates. Although this has not changed from previous versions, the principles have greater implications for the adaptations of the model, in terms of memory allocation and run time, but this will be discussed further in Sections A.6 and A.7.

Figure A.1 shows the layout for storing solutions for one room with the probabilistic model. Each room is subdivided into the individual time steps in this case, one minute intervals. Each individual time step is further subdivided into separate iterations, each iteration representing a single solution to the differential equation. The model solves a large number of iterations for each time step, varying depending on the user’s requirements, typically 1,000 times or greater. For the purposes of this discussion, it is assumed that the number of iterations is set to 1,000.

Each solution is obtained by solving Equation (3.1). The solution for room 1 at time 12:30 on iteration 1, becomes the initial concentration for room 1 at 12:31 on iterations 1, and so on. If multiple rooms are being simulated, the solution for iteration 1 is used in the calculation for iterations 1 for the different rooms. This ensure that the model has a 1,000 independent solutions allowing mean and
Figure A.1: A layout demonstrating how each room is subdivided into different time steps, and each timestep is further subdivided into iterations.

standard deviations to be calculated.

A.2 Header File for Defining the Variables

The header file stores all the necessary parameters to perform the simulations, while removing them from the source code. This allows the user to change the values in the model without a needing to alter the code. These parameters perform various tasks; some parameters contain the values for solving differential equations, others control the simulations, while the remaining parameters specify the file path to the input files.

The assigned values in the model are the initial concentrations, room dimensions, emission rates and deposition velocities. Figure A.2 provides an example of how the emission values for smoking are enter in the header file.

The parameters that control the model, are number of rooms, the step size and the number of iterations.

A function not available in the INDAIR model, is to control which pollutants
Appendix A: Modifications to the Model Source Code

Figure A.2: A screenshot of the mean, standard deviation, minimum and maximum values for generating the PM$_{10}$ emission rates.

are simulated. IAPPEM allows the user to “switch off” the calculations of certain pollutants. Figure A.3 show the parameters (public static final boolean are a part of the java construct), when runPM$_{10}$ and runPM$_{2.5}$ are set to true, this will perform the calculations on their pollutants. While NO$_2$ and CO are set to false, this instructs the model to perform no calculations on these pollutants.

Figure A.3: A screenshot demonstrating how the user controls which pollutants are simulated.

Two methods could be employed in separating the pollutants in the model; isolating each pollutant outside of the time loop or isolate the pollutants during each time loop. Isolating the pollutant outside of the time loop means the model only checks once whether the user requires the calculations for that pollutant. This method gives greater speed if the pollutant is turned off. However, if the pollutant is isolated inside in the time step, the model must check each time whether or not to perform the calculations on that pollutant, thereby slowing down the process. If multiple pollutants are required for simulations, this allows for greater speed in performing the calculations, as this is due to overlap in the parameters generated
for solving the differential equations. e.g. airflows are the same regardless of the pollutant. Overall it was decided that this approach was the better option, as the increased speed with multiple pollutants outweighs the decrease compared with single pollutants.

The user may not always require predictions for each microenvironment. Variables are declared in the header file, as shown in Figure A.4. If the user only requires simulations in certain microenvironments, the value is set to true. If they do not require simulations the values are set to false. This option saves considerable computing time if the user only requires specific simulations.

```java
public static final boolean runHome = true;
public static final boolean runOffice = false;
public static final boolean runGym = false;
public static final boolean runCar = false;
public static final boolean runLuas = false;
public static final boolean runBus = false;
public static final boolean runTrain = false;
public static final boolean runPub = false;
public static final boolean runSuperMarket = false;
public static final boolean runClassroom = false;
public static final boolean runRestaurant = false;
public static final boolean runOutdoor = false;
```

Figure A.4: A screenshot demonstrating how the user controls which microenvironments are simulated.

The model includes the function allowing the user to use mean values instead of generating the values with a distribution. Figure A.5 shows the parameters that control the distributions, for airflow, room dimensions, deposition velocity, microenvironments, emission rates, outdoor concentrations and initial concentrations respectively. Setting any of these parameters to true, those values are generated based on one of the probability functions. If set equal to false, the model
uses the mean value for those values. Setting these value to true or false, has no impact on the overall run time, but it gives the user the opportunity to focus on influence of certain parameters, or simply run the model deterministically.

```java
public static final boolean use_AER_Iterations = true;
public static final boolean use_RD_Iterations = true;
public static final boolean use_Vd_Iterations = true;
public static final boolean use_ME_Iterations = true;
public static final boolean use_Source_Iterations = true;
public static final boolean use_OutConc_Iterations = true;
public static final boolean use_initialCon_Iterations = true;
```

Figure A.5: A screenshot demonstrating how to control which values are generated using a distribution function.

### A.3 Outdoor Concentrations

The model was adapted to allow different environments to have different outdoor concentrations. This was a necessary addition as the office, the home and recreational environment can all be located in very different outdoor locations. It is especially important for the transport environment; where buses, cars or trains will be moving throughout different outdoor locations, with potentially varying outdoor concentrations.

### A.4 Airflow

IAPPEM changed the structure of how the model dealt with airflow. In the INDAIR model, a time-weighted average airflow is employed over a 24-hour period. The limitations surrounding a time-weighted average airflow have been discussed.
The use of a time-weighted average airflow meant that only four values needed to be specified in the header file for each airflow; the mean, standard deviation, minimum and maximum values. However, this is not possible for a variable airflow, and in order to assign multiple values for different time steps to a single airflow, a new approach was devised. The enhancements to IAPPEM allow for the possibility of these variations by creating individual airflow values for each individual time step in the model. This is achieved by creating a text file for every airflow. The user then enters the start and stop times along with the corresponding mean, standard deviation, minimum and maximum values for the desired airflow, as shown in Figure A.6. This approach allows the user to set the entire 24-hour period in a single block as in the INDAIR model, select individual blocks, or even apply changes on step-by-step basis based on the user requirements.

IAPPEM alters the differential equations to use air flow in terms of volumetric air flow \((m^3 h^{-1})\), as opposed to INDAIR, which used air changes per hour \((ACH^{-1})\). \(ACH^{-1}\) incorporates the dependance on the room’s volume. Using volumetric air flow, the model calculates the infiltration/exfiltration of room’s concentrations in terms of \(\mu g\), instead of \(\mu g m^{-3}\) (as with \(ACH^{-1}\)); the exfiltration from one room then equals the infiltration into the adjoining room. In INDIAR, it was assumed that exfiltration and the corresponding infiltration were the same using air changes per hour \((ACH^{-1})\), and this led to inaccuracies in predicting concentrations, the larger the difference in volume between the two rooms, the greater the inaccuracies.

The location of the airflow files are declared in the header file, allowing multiple airflows to correspond to the same file. For example, most rooms will have only one
Figure A.6: A view of the text airflow input file with a number of variations in airflow rates

or two doors, so a large portion of the airflows will be set to zero (no interaction between rooms). Instead of needing to change all the corresponding text files, simply changing the file path sets all airflows to the same value; in this case zero.

A.5 Sources

The INDAIR model only dealt with one emission source. It did, however, take into account two different scenarios, one with smoking and another with cooking, but it was only possible to model one of these at a time. While each scenario could be run for individual times, it was not possible to run the two scenarios in the same 24 hour period.

IAPPAM now incorporates up to 12 simultaneously-operating emission sources. In a similar respect to the airflow, the user specifies the conditions in a text file. The users inputs the start and stop times of the emission sources alongside corresponding sources. The user enters the word YES underneath the appropriate heading to indicate whether that emission event is taking place in that time frame. Otherwise, a NO indicates that the particular emission is not present, as seen in

<table>
<thead>
<tr>
<th>start time</th>
<th>stop time</th>
<th>Mean</th>
<th>StdDev</th>
<th>Minimum</th>
<th>Maximum</th>
</tr>
</thead>
<tbody>
<tr>
<td>06:00</td>
<td>05:30</td>
<td>0.590</td>
<td>0.134</td>
<td>0.25</td>
<td>1.0</td>
</tr>
<tr>
<td>05:30</td>
<td>13:00</td>
<td>1.350</td>
<td>0.5111</td>
<td>1.025</td>
<td>2.5</td>
</tr>
<tr>
<td>13:00</td>
<td>18:00</td>
<td>0.21320</td>
<td>0.025</td>
<td>0.0001</td>
<td>0.3</td>
</tr>
<tr>
<td>18:00</td>
<td>24:00</td>
<td>3.0230</td>
<td>0.15</td>
<td>2.5</td>
<td>3.5</td>
</tr>
</tbody>
</table>
A.6 Rooms

IAPPEM increases the number of rooms from three to fifteen. The complications that arose with this were the necessity to alter the differential equations to incorporate the additional airflow terms. All rooms are required to be connected to each other; otherwise this would force limitations on specific household layouts. For example, a hallway might have a varying number of rooms connected to it. However, the option to limit simulations to fewer rooms was also included to allow simplistic scenarios. Parameters declared in the header file, allow the user to specific the amount of rooms to be simulated and the calculations for each room have been encased in conditional statements, considerably reducing the time needed to run the simulations.

A second obstacle which occurs is that each additional room must have emission source terms, room dimensions, deposition terms, initial concentrations along with the airflows. For the total number of rooms, there will be that same number
of airflows for each room. e.g. for only one room there will only be AER0001 (between outdoors (00) and room 1 (01)), and for 15 rooms there would be 15 airflows, e.g. AER0015, AER0105, AER0205,...., AER1315, AER1415. For 15 rooms, there will be 60 emission rates, 60 deposition rates, 60 initial concentrations, 45 rooms dimensions and 120 airflows as well as with the outdoor concentrations. All the additional parameters needed have implications for memory allocation, as discussed in Section A.7.

A.7 Time

The INDAIR model had a time step of 15 minutes, and in IAPPEM, a modification is made to reduce the time step to one minute. This allows the capture of information relating to short term pollution concentration and air exchange variations, detailed in Section 2.3.1. Adjusting the time resolution of the model is relatively straightforward, but an issue arises with memory allocation when the time step is reduced. The problem which arises is due to the probabilistic nature of the model, as 1,000 iterations requires each parameter to generate a 1,000 independent values per time step. The model previously generated all of those values before continuing to solve the differential equations, but with the additional rooms, there are considerably more parameters, which requires more values to be generated.

Section A.6 discussed that with 15 rooms, there are over 244 parameters that vary over time. Calculating 1,000 iterations per time step with a total of 1,440 time steps (one minute resolution over a 24 hour period) corresponds to approximately 2.62 gigabytes of memory without consideration of the variables needed to perform the calculations and store the solutions. For high performance computing this
would not present any considerable problems. However, at present this would eliminate the ability to run the model on a common PC.

The problem described above required a restructuring of the code to accommodate the additional rooms and increased time resolution. The solution was as follows; instead of generating and storing all the values for each time step, the code was adjusted so the model generates the values for each time step individually. This results in the model only needing to store values for one time step. This method is implemented for all the input parameters. Once the calculations for the time step have been completed, the new values override the existing values, and do not require any additional memory.

Memory allocation is no longer a problem when it comes to each time step. Structuring the code in this way allows the user the ability to run simulations on up to 15 rooms with a time step of one minute, which allows up to 12,000 iterations for each time step.

From a computational point of view, an adverse effect resulting from this approach is, at each new time step the model is required to reread in the values from the text files, e.g. outdoor concentrations, airflows and emission files, before it can generate the values for the differential equation. This results in additional computational time needed to complete the simulations.
Appendix A: Modifications to the Model Source Code

A.8 Outputs

Once all the calculations have been completed, solutions exist for each iteration and for each time step as described in Section A.1. The model calculates the mean and standard deviations for each time step, printing the outputs to a file, as shown in Figure A.10. The values are printed into a simple file format, allowing the data to be easily imported into Microsoft Excel or another software package, for any further analysis.

<table>
<thead>
<tr>
<th>Time(h)</th>
<th>PM10</th>
<th>PM2.5</th>
<th>Range3</th>
<th>Range4</th>
<th>Range3</th>
<th>Range4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Room1</td>
<td>uq/m3</td>
<td>uq/m3</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>00:00</td>
<td>10.07</td>
<td>5.75</td>
<td>1</td>
<td>1.01</td>
<td></td>
<td></td>
</tr>
<tr>
<td>00:01</td>
<td>10.04</td>
<td>5.77</td>
<td>0.82</td>
<td>0.85</td>
<td></td>
<td></td>
</tr>
<tr>
<td>00:02</td>
<td>10.01</td>
<td>5.78</td>
<td>0.72</td>
<td>0.75</td>
<td></td>
<td></td>
</tr>
<tr>
<td>00:03</td>
<td>9.98</td>
<td>5.79</td>
<td>0.66</td>
<td>0.7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>00:04</td>
<td>9.95</td>
<td>5.81</td>
<td>0.62</td>
<td>0.66</td>
<td></td>
<td></td>
</tr>
<tr>
<td>00:05</td>
<td>9.93</td>
<td>5.82</td>
<td>0.59</td>
<td>0.63</td>
<td></td>
<td></td>
</tr>
<tr>
<td>00:06</td>
<td>9.91</td>
<td>5.83</td>
<td>0.57</td>
<td>0.61</td>
<td></td>
<td></td>
</tr>
<tr>
<td>00:07</td>
<td>9.89</td>
<td>5.85</td>
<td>0.55</td>
<td>0.6</td>
<td></td>
<td></td>
</tr>
<tr>
<td>00:08</td>
<td>9.87</td>
<td>5.86</td>
<td>0.53</td>
<td>0.58</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Figure A.8: A screenshot of the text output file, showing the PM$_{10}$ and PM$_{2.5}$ mean and standard deviations for room 1.

Some additional functionality has also been included in the model. The model searches through the mean concentrations for each time step, identifying the peak concentration and the corresponding time in each room. The model calculates the 24 hour mean and standard deviations for each room, by calculating the 24 hour mean concentrations for each iteration and then calculating the mean and standard deviations for each iteration. Figure A.9 shows this output file, which includes the room number with the corresponding peak concentration, the time of the peak concentrations and the 24 hour mean concentrations.
### Appendix A: Modifications to the Model Source Code

<table>
<thead>
<tr>
<th>Room Number</th>
<th>Peak Time</th>
<th>Peak Concentrations</th>
<th>24 Hour Mean Concentrations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Max PM10 values</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Room 1</td>
<td>11:15</td>
<td>90.56 +/- 18.19</td>
<td>30.00 +/- 2.79</td>
</tr>
<tr>
<td>Room 2</td>
<td>10:13</td>
<td>885.39 +/- 322.41</td>
<td>96.62 +/- 0.51</td>
</tr>
<tr>
<td>Room 3</td>
<td>11:13</td>
<td>40.21 +/- 5.06</td>
<td>18.72 +/- 1.31</td>
</tr>
<tr>
<td>Room 4</td>
<td>12:27</td>
<td>89.42 +/- 10.57</td>
<td>35.61 +/- 2.43</td>
</tr>
<tr>
<td>Room 5</td>
<td>11:40</td>
<td>1893.94 +/- 162.33</td>
<td>261.04 +/- 20.53</td>
</tr>
<tr>
<td>Room 6</td>
<td>13:40</td>
<td>22.40 +/- 2.10</td>
<td>14.38 +/- 1.05</td>
</tr>
<tr>
<td>Room 7</td>
<td>13:39</td>
<td>17.12 +/- 1.38</td>
<td>12.87 +/- 0.88</td>
</tr>
<tr>
<td>Room 8</td>
<td>13:41</td>
<td>17.37 +/- 1.39</td>
<td>12.84 +/- 0.86</td>
</tr>
<tr>
<td>Room 9</td>
<td>13:40</td>
<td>17.72 +/- 1.50</td>
<td>12.92 +/- 0.88</td>
</tr>
<tr>
<td>Room 10</td>
<td>13:42</td>
<td>17.77 +/- 1.54</td>
<td>12.98 +/- 0.90</td>
</tr>
<tr>
<td>Room 11</td>
<td>14:38</td>
<td>19.91 +/- 0.78</td>
<td>9.38 +/- 0.62</td>
</tr>
</tbody>
</table>

| Max PM2.5 values |           |                     |                             |
| Room 1          | 11:09     | 79.29 +/- 16.71     | 25.94 +/- 2.67              |
| Room 2          | 10:13     | 633.69 +/- 256.01   | 81.69 +/- 8.18              |
| Room 3          | 11:19     | 34.33 +/- 5.12      | 15.27 +/- 1.2               |
| Room 4          | 12:22     | 61.54 +/- 6.29      | 27.35 +/- 1.66              |
| Room 5          | 14:10     | 588.8 +/- 57.1      | 167.97 +/- 12              |
| Room 6          | 13:40     | 17.33 +/- 1.59      | 11.28 +/- 0.8               |
| Room 7          | 13:39     | 13.00 +/- 0.99      | 9.76 +/- 0.6                |
| Room 8          | 13:44     | 13.29 +/- 1.02      | 9.32 +/- 0.6                |
| Room 9          | 13:44     | 13.53 +/- 1.11      | 9.37 +/- 0.64              |
| Room 10         | 13:43     | 13.57 +/- 1.13      | 9.32 +/- 0.65              |
| Room 11         | 15:18     | 8.54 +/- 0.65       | 7.5 +/- 0.49               |

Figure A.9: A screenshot of the text output file showing the PM$_{10}$ and PM$_{2.5}$ peak and mean concentrations, along with the time of each peak for each room.
Appendix A: Modifications to the Model Source Code

A.9 Personal Profile

In order to calculate individual exposure as shown in Chapter 4, it is necessary to know the microenvironment where the individual is present and how long they spend in that microenvironment. This information is stored in a text input file, as shown in Figure A.10. The user enters the start time, stop time and the corresponding microenvironment where the individual is present.

<table>
<thead>
<tr>
<th>Personal Profile</th>
<th>Location</th>
</tr>
</thead>
<tbody>
<tr>
<td>Source</td>
<td></td>
</tr>
<tr>
<td>start time</td>
<td>stop time</td>
</tr>
<tr>
<td>hh:mm</td>
<td>hh:mm</td>
</tr>
<tr>
<td>00:00</td>
<td>08:00</td>
</tr>
<tr>
<td>08:00</td>
<td>09:00</td>
</tr>
<tr>
<td>09:00</td>
<td>17:00</td>
</tr>
<tr>
<td>17:00</td>
<td>18:00</td>
</tr>
<tr>
<td>18:00</td>
<td>20:00</td>
</tr>
<tr>
<td>20:00</td>
<td>20:30</td>
</tr>
<tr>
<td>20:30</td>
<td>22:00</td>
</tr>
<tr>
<td>22:00</td>
<td>22:15</td>
</tr>
<tr>
<td>22:15</td>
<td>24:00</td>
</tr>
</tbody>
</table>

Figure A.10: A screenshot demonstrating how the user selects the time and the microenvironments where an individual is present.

The model loops through each time step, identifying the microenvironment where the individual is present. This determines the corresponding mean and standard deviation concentrations for that microenvironment during that time step. The concentrations for each microenvironment have already been calculated. The individual’s exposure is printed to the personal output file as shown in Figure A.11.
Appendix A: Modifications to the Model Source Code

A.10 Activity Profile

The approach that is taken for the activity profile pattern is a style similar to that of the personal profile, but there is a difference in that instead of only one individual being modelled, a population group is modelled. In this case, the personal profiles are run for 100 individuals and an average population exposure is obtained.

For each individual iteration, the model generates a value for each microenvironment. The number of values generated for each microenvironment corresponds to the number in the activity profile file for that microenvironment during that time step. There can only ever be 100 values generated, as the total number across all the microenvironment will always add up to 100 for each individual time step. All the values generated are summed together and an average is obtained.

<table>
<thead>
<tr>
<th>Time(h)</th>
<th>Mean</th>
<th>Standard Deviation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Room1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Range3</td>
<td></td>
<td></td>
</tr>
<tr>
<td>PM10</td>
<td>10.12</td>
<td>1.02</td>
</tr>
<tr>
<td>PM2.5</td>
<td>5.8</td>
<td>1.01</td>
</tr>
</tbody>
</table>

Figure A.11: A screenshot of the text output file, showing the PM$_{10}$ and PM$_{2.5}$ mean and standard deviations including the location where the individual is present.
Appendix A: Modifications to the Model Source Code

Figure A.12: A screenshot demonstrating how the user selects the times and location of a population group. The names of location are provided for demonstrative purposes, and do not imply any limitations in changing the house layout.

(percentage or people in various hrs)

<table>
<thead>
<tr>
<th>Time (h)</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>6</th>
<th>7</th>
<th>8</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>utiltv</td>
<td>kitchen</td>
<td>dining room</td>
<td>hall</td>
<td>sitting room</td>
<td>bathroom</td>
<td></td>
<td></td>
</tr>
<tr>
<td>00:00</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>50</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>00:15</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>50</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>00:30</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>50</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>00:45</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
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<td>50</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>01:00</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>50</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>01:15</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>50</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>01:30</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>50</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>01:45</td>
<td>0</td>
<td>25</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>25</td>
<td>25</td>
<td>25</td>
</tr>
<tr>
<td>02:00</td>
<td>0</td>
<td>25</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>25</td>
<td>25</td>
<td>25</td>
</tr>
</tbody>
</table>