

# A Method to Measure Emission Rates of PM<sub>2.5</sub>s from Cooking

Catherine O'Leary<sup>1</sup> and Benjamin Jones<sup>\*1</sup>

*1 University of Nottingham  
B12 Paton House,  
Department of Architecture and Built Environment,  
University of Nottingham, NG7 2RD, UK*

*\*Corresponding author:  
Benjamin.Jones@nottingham.ac.uk*

## ABSTRACT

Exposures to airborne fine particulate matter with a diameter of  $<2.5\mu\text{m}$  (PM<sub>2.5</sub>) are linked to multiple negative health effects, including cardiovascular and respiratory disease. Existing investigations of PM<sub>2.5</sub> primarily focus on external sources and exposures, because outdoor air is easier to observe, and therefore, more widely monitored. However, as people spend up to 70% of their time in their own homes, exposures to indoor pollutants could have a greater impact on health. One method of investigating indoor exposures in a stock of houses is by modelling them. However, this process requires an understanding in the uncertainty of emission rates from internal sources.

Cooking has been identified as key source of PM<sub>2.5</sub> in non-smoking households. Existing studies of emission rates use a range of techniques from small scale test chambers, which give control over all parameters in unrealistic conditions, to personal monitoring studies in real conditions where there is no control over influencing parameters. Reported emissions rates for single sources vary significantly indicating poor repeatability, and are generally presented without an indication of the uncertainty in them; for example, as a probability density function (PDF). Therefore, existing emission rates have limited use for stochastic indoor air quality modelling.

This paper seeks to develop a methodology to measure emissions rates of PM<sub>2.5</sub>s from the cooking of foods. A two-phase investigation measured the variation in emission rates when toasting bread in an electric toaster, a process that is simple and repeatable with fewer variables than many other cooking processes.

Phase one was conducted in a domestic kitchen. A TSI SidePak<sup>TM</sup> AM510 optical monitor measured temporal concentrations during and after toasting (n=40). A number of problems with the procedure were identified: (i) the recording time-step was too long, which lead to insufficient data points during the emission period; (ii) the ventilation rate and mixing conditions were unknown, although steps were taken to control them, which increased measurement uncertainty; (iii) the relative humidity was not monitored, which can affect the performance of the SidePak<sup>TM</sup> at high levels, and (iv) when the toaster and bread were not isolated at the end of the toasting period, increasing uncertainty in the total emission period.

The second phase used a test chamber, which offered greater control over the indoor conditions. An experimental procedure was followed which was similar to that use during the first phase, but with a one-second time-step, the toaster and toasted bread were sealed after the emission period, and the chamber was flushed with outside air between tests. Relative humidity and temperature were monitored during tests. Emission rates are estimated using an established model and reported as a histogram.

## KEYWORDS

Food, Toast, Emission Rates, Fine Particulate Matter, Indoor Air Quality

## 1 INTRODUCTION

Fine particulate matter (PM<sub>2.5</sub>) is composed of airborne solid or liquid particles of diameter less than  $2.5\mu\text{m}$  (Pope and Dockery, 2006). Due to their small size, when breathed in, these particles can bypass the body's defences (Pope and Dockery, 2006) and exposure to elevated concentrations has been associated with chronic and acute, respiratory and cardiovascular diseases (Lewtas, 2007), as well as other possible health implications (Rückerl *et al.*, 2011).

The health impacts of exposure to air pollutants, including PM<sub>2.5</sub>s, can affect people at all stages of life, from before birth to death (RCP, 2016).

In the UK, energy use in residential buildings is estimated to account for around 26% of the country's total carbon dioxide (CO<sub>2</sub>) emissions (Wilkinson *et al.*, 2009), which has influenced the move to reduce the energy demand of the existing building stock (Wilkinson *et al.*, 2009). Retrofit and intervention strategies can involve reducing infiltration (Wilkinson *et al.*, 2009) but do not necessarily include any additional purpose-provided ventilation, potentially lowering overall airflow rates. Furthermore, whilst all new dwellings are required to have localised mechanical extract ventilation in the kitchen, when refurbishing existing dwellings, they are not required where they are not already present (HM Government, 2010).

People spend 70% of their time in their houses (Lader *et al.*, 2005) and so a reduction in the supply of fresh air could offer protection from outdoor pollutants, but could also trap pollutants emitted indoors (Shrubsole *et al.*, 2012). Key indoor sources of PM<sub>2.5</sub>s include cooking, smoking, spray aerosols, and various forms of combustion, including burning candles and incense (Afshari *et al.*, 2005). Cooking is interesting because it is a common activity where total source removal is not possible.

One method to investigate indoor exposure to PM<sub>2.5</sub>s at stock level is to use models. However, their predictions are highly sensitive to the emission rates used (Das *et al.*, 2014). When representing cooking sources, a constant value of 1.6 mg/min is used by Shrubsole *et al.* (2012), despite emission rates having been shown to vary with fuel type (Olson and Burke, 2006), food type and cooking method (Fortmann *et al.*, 2001), and oil type (Torkmahalleh *et al.*, 2012). Additionally, reported emission rates are highly varied, from 0.025 mg/min for pork roast (Fortmann *et al.*, 2001) to 1496 mg/min (Olson and Burke, 2006). They are also frequently based on a small number of measurements from which is it difficult to determine uncertainty and the probability density functions (PDFs) required by stochastic stock modelling. This paper evaluates existing measurements of PM<sub>2.5</sub> emission rates from the cooking of food in Section 2. It then applies these to develop methods for deriving PDFs of PM<sub>2.5</sub> emission rates in Sections 3 and 4.

## 2 EXISTING MEASUREMENT METHODOLOGIES

Existing measurement methodologies of emission rates from foods cooked in houses can be divided into five categories, each of which has its own benefits and frequency in the literature. The first method is personal monitoring, which is generally rare. Olson and Burke (2006) used it to determine emission rates using measurements of PM<sub>2.5</sub> concentration made during four, seven-day monitoring periods by 37 participants. Concentration peaks from cooking were identified using diaries, and the emission rate calculated using a mass balance approach that considered a house to be a single well-mixed space. A total of 411 cooking events found that emission rates ranged between 0.6 to 1496 mg/min ( $\mu=36$  mg/min). However, the approach was found to overestimate the true emission rates because of the combined effect of the close proximity of the sensor to the source, and the assumption of the whole house being considered well-mixed. The cooking methods were recorded but details of the food were not.

The second approach uses monitors located in a place of interest, most frequently in dwellings. He *et al.* (2004) measured PM<sub>2.5</sub> mass concentrations for 48 hours in 15 houses in Brisbane, Australia. Elevated concentrations were linked to events using occupant diary entries and catalogued into 21 activity types. In each house an air exchange rate (AER) test and controlled cooking test were conducted. The latter consisted of frying half an onion in vegetable oil on "high" heat under both minimum and normal ventilation conditions. This provided an opportunity for direct comparison between houses. Emission rates were determined using a mass balance relationship for a total of 106 cooking events where emission rates ranged from 0.03 to 2.78 mg/min ( $\mu=0.11$  mg/min). The lack of control over

ventilation rates and emission periods led to large errors. A monitoring study is a simple way of monitoring typical concentrations found in dwellings and the emission rates associated with them. But, the lack of information on the food prepared and the cooking methods limits the ability of its findings to be extrapolated to the wider population.

The third method uses managed tests conducted in semi-controlled conditions in the field, typically in a residential setting; see Dacunto *et al.* (2013) and Fortman *et al.* (2001). Both studies investigate emission rates from varying sources and in a residential setting, for some of the tests, using gas and electric stoves. They use different mass balance relationships to estimate emission rates. The AERs were measured using tracer gas decay, and the test environments flushed with outside air between tests. Dacunto *et al.* (2013) conducted 66 measurements in 3 locations, including both cooking and non-cooking sources, with duplicates of most, but not all, tests. The lowest emission rate of 0.1 mg/min was from oven cooking frozen pizza. The highest cooking emission rate of 15.2 mg/min was from fried chicken breast. Fortmann *et al.* (2001) conducted 32 cooking tests using a variety of foods and cooking methods based on the US diet, under multiple ventilation conditions, in a single location. 7 tests were repeated but comparing changes using gas and electric heat sources. However, only 4 tests were duplicated exactly to investigate variability, suggesting high variability that exceeds 100% and a need for more repetitions. Estimated emission rates ranged from 0.025 mg/min, for pork roast with exhaust ventilation, to 10.3 mg/min, for frying tortillas on an electric range. The range of emission rates from both of these tests highlights the need to understand the population diet when choosing sources for investigation, whilst the low repeatability – indicated in the repeatability tests (Fortmann *et al.*, 2001) and large standard deviations (Dacunto *et al.*, 2013) – calls for more repetitions.

The fourth method uses large scale chambers to obtain a further level of control over ventilation rates and mixing conditions, pollutant concentrations in the supply air, and internal materials to minimise sink effects. By creating a test chamber within a building, the effect of outside air and natural ventilation processes is reduced, and over-pressurising the test chamber controls the direction of background ventilation (Afshari *et al.*, 2005). Afshari *et al.* (2005) investigated concentrations of fine and ultrafine particles from 10 cooking and non-cooking sources, under identical conditions, in a full-scale test chamber. Lee and Wang (2006) found PM<sub>2.5</sub> emission rates of 0.72–1.83 mg/min, for 5 mosquito coils tested in a large environmental chamber. Temperature, relative humidity, AER, mixing conditions, and leakage were all controlled during these tests, which reduces error in the estimated emission rates. Finally, Pagels *et al.* (2009) investigated chemical composition and mass emission of candle smoke particles. Although these are not cooking sources and the emission rates are not useful for direct comparison, they also used a positively pressurized stainless-steel chamber, with controlled relative humidity, temperature, ventilation rates and mixing conditions, and filtered supply air.

The fifth and final method uses small-scale chambers. Géhin *et al.* (2008) utilised a hexagonal test chamber, volume  $2.36 \pm 0.05 \text{ m}^3$ , first designed for vacuum cleaner efficiency characterisation, to investigate fine and ultrafine particle emission rates for cooking and non-cooking sources. The control measures used in large chamber tests are also used, and the internal surfaces were treated with an antistatic coating, and an upward current of filtered air were employed to reduce sink effects. Torkmahalleh *et al.* (2012), measured emissions from heating cooking oils in a laboratory fume hood operating at  $65 \text{ m}^3/\text{h}$  ( $80 \text{ h}^{-1}$ ), with 5 repetitions of each test. Mixing conditions were tested using SF<sub>6</sub> and a grid equivalent to the full size of the hood with grid points located 10cm apart. The small size of the chamber means the air is very well mixed, but the method of heating the cooking oils in a beaker on a hot plate is far abstracted from real cooking methods, and so the resulting emissions may not reflect those found in a domestic kitchen. This may also be true for the emissions from Géhin *et al.* (2008) because the airflow conditions may have impacted the emissions from the combustion source.

### 3 EXPERIMENT PHASE 1

#### 3.1 Method

The first phase of tests follows the third method described in Section 2. They were conducted in a residential kitchen, volume 35.4 m<sup>3</sup>, in Nottingham, UK, in July 2015. Although it was not possible to control the ventilation conditions precisely, all windows, and internal and external doors were closed during tests, and a wall mounted kitchen extractor fan was used to stabilise the ventilation rate. Mixing conditions were not tested, and no mixing fan used, additionally, the ventilation rate was not measured.

One SidePak™ AM510 Personal Aerosol Monitor (TSI Inc., Shoreview, MN, USA) was used to monitor PM<sub>2.5</sub> concentrations, at height 1.1 m, 1 m from the source; see Figure 1. Concentrations are time-averaged and reported at 1 minute intervals following Ott *et al.* (2006), with the calibration factor – required for the conversion of optical readings to mass concentrations – set to the default 1.0, as no concurrent gravimetric sampling was available.

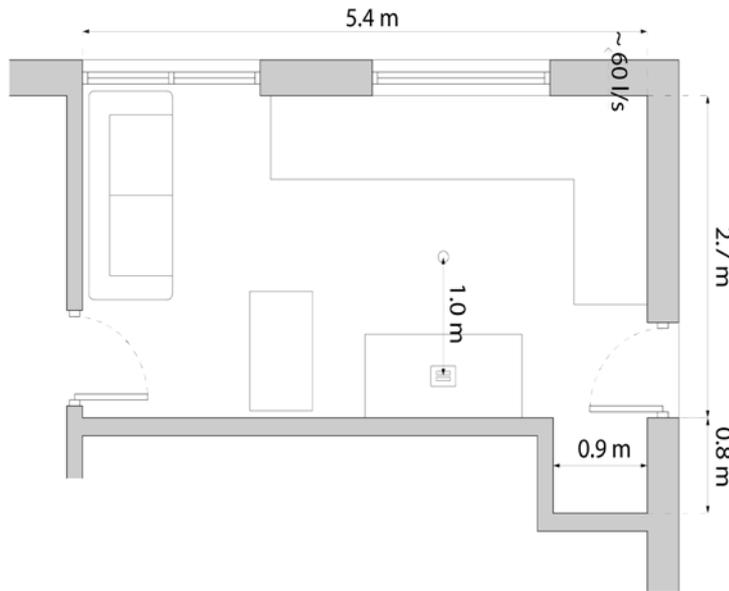


Figure 1 - Kitchen Plan

The test consisted of toasting 2 slices of bread (medium sliced white or wholemeal Hovis 800g loaf) in a supermarket branded toaster on setting 6, the highest setting. The test is simple and repeatable, with a predefined cooking time, and slices of bread with consistent weight and geometry. The start and end of the toasting time were recorded, and concentrations were allowed to return to background levels between tests, or left for a minimum 25 minutes. The test was repeated a total 40 times with 50% wholemeal. The toaster was new at the start of testing, and was not cleaned between tests.

In addition, 6 plume tests were conducted in the kitchen in October 2015, to observe the concentration profile during the emission period. A tube attached to the SidePak™ inlet was positioned in the plume, 20 cm above the toaster, with concentrations logged at 1 second intervals. The decay was not monitored and the room was flushed with outside air between tests.

#### 3.2 Emission Rate Calculation

The method used to determine emission rates is based on the widely used mass balance model; see Dacunto *et al.* (2013), Olson and Burke (2006) and Ott *et al.* (2006).

$$C(t) = C_b + \frac{G}{(\lambda + k)V} + \left( C(0) + C_b + \frac{G}{(\lambda + k)V} \right) e^{-(\lambda+k)t} \quad (1)$$

Here,  $C(t)$  is the concentration at time  $t$ ,  $C_b$  the background concentration,  $C(0)$  the initial concentration,  $G$  the emission rate,  $V$  the mixing volume, and  $(\lambda+k)$  the total decay rate due to ventilation, deposition, and coagulation.

Ott established three phases in during emissions: an  $\alpha$ - or emission period when air is not well-mixed, a subsequent  $\beta$ -period when emissions cease and mixing is incomplete, and a  $\gamma$ -period of well mixed decay (Ott *et al.*, 2006). During the  $\beta$ - and  $\gamma$ -periods, equation (1) is simplified to give

$$C(t) = C_b + (C_p - C_b)e^{-(\lambda+k)(t-t_p)} \quad (2)$$

$C_p$  is the peak concentration at time  $t_p$ , and  $(\lambda+k)$  is determined by log-linear regression. For a rectangular source function, the peak concentration should occur at the end of the  $\alpha$ -period, but it is often observed later indicating that full-mixing is not instantaneous. Ott *et al.* (2006) proposes the *theoretical peak estimation* method where the decay rate is determined from the  $\gamma$ -period and extrapolated back to the start of the  $\beta$ -period. This theoretical peak,  $C_p$ , is then used to estimate the emission rate over an emission period of time  $T$ , where

$$G = (\lambda + k)V \left[ \frac{(C_p - C_b) - (C(0) - C_b)e^{-(\lambda+k)T}}{1 - e^{-(\lambda+k)T}} \right] \quad (3)$$

### 3.3 Calibration Factors

Concurrent gravimetric sampling to determine custom calibration factors (CFs) was unavailable. TSI calibrate SidePak<sup>TM</sup>s using Arizona Test Dust, and so if the aerosol has a different density, size distribution, shape, or refractive index, the CF is expected to vary (Jiang *et al.*, 2011). Two CFs are available for toasting bread; Dacunto *et al.* (2013) found CF=0.47 for burned toast, whereas Jiang *et al.* (2011) found CF=0.79. Dacunto attributes the difference to levels of charring. In absence of further information, the data are processed twice, applying each CF.

### 3.4 Results

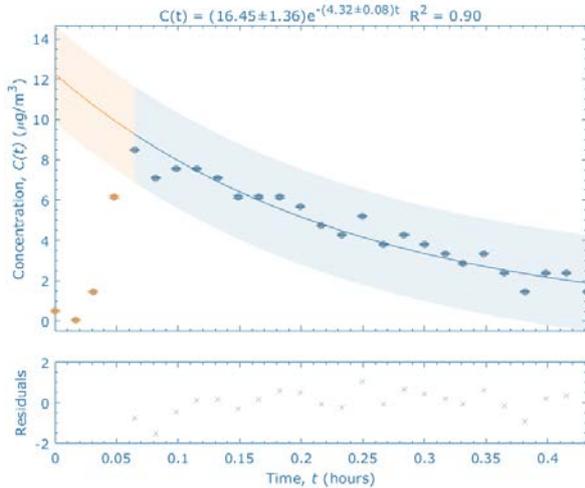


Figure 2 - Example Decay and Emission, CF = 0.47

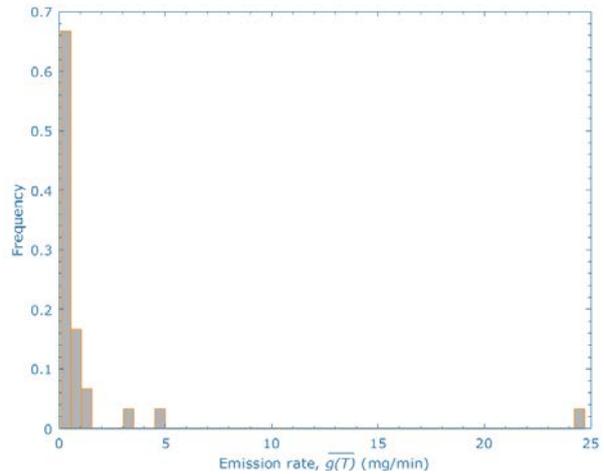


Figure 3 – Phase 1 Emission Rates Histogram, CF = 0.47

Figure 2 shows the concentrations recorded in a sample test, highlighting the  $\alpha$ - and  $\beta$ -periods (orange), the  $\gamma$ -period (blue), and the fit of the curve predicted using the log-linear regression during the  $\gamma$ -period. All tests were processed using the same method and were eliminated from further analyses if the  $R^2$  value for the log-linear regression of the decay was  $<0.7$ , and when emission rates were calculated to be negative (indicating a problem with the test) or could not be estimated. One further outlier was eliminated using *Chauvenet's Criteria*, leaving 29 of 40 tests to determine an average emission rate.

Figure 3 shows the distribution of emission rates calculated for CF=0.47, although the shape of the distribution is identical for both CFs. The data is significantly positively skewed and the median is much less than the mean. This is attributed to a few tests having a high emission rate. Table 1 presents summary the statistics for both CFs, and shows that its value has a significant impact.

Table 1: Phase 1 Emission Rates (mg/min)

CF	N	Mean	SD	2 <sup>nd</sup>	25 <sup>th</sup>	Percentiles		
						50 <sup>th</sup> (Median)	75 <sup>th</sup>	98 <sup>th</sup>
0.47	29	0.662	1.009	0.097	0.179	0.309	0.626	0.097
0.79	29	1.113	1.697	0.163	0.301	0.519	1.052	0.163

The peak concentration occurred after the end of the toasting period. The reasons are unclear and investigations are hampered by the poor resolution of the 1 minute logging interval. The calculation method assumes the delay is due to the time taken for the air to become well-mixed, but emissions may have continued after the toasting period (the toast and toaster were unsealed), or the composition and chemistry of the particles may have evolved. The plume tests in Figure 4 show that the PM<sub>2.5</sub>s are emitted during toasting, but their emission rate is variable, so toasting is a non-rectangular source function.

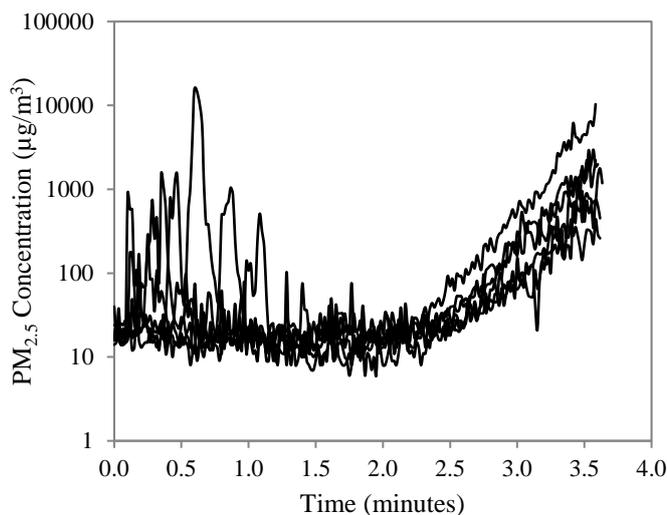


Figure 4 - Plume Test Concentrations during the  $\alpha$ -period

## 4 EXPERIMENT PHASE 2

The second test phase was developed in response to the problems encountered in phase 1. It combines conditions similar to field tests (method 3 in Section 2) by following Dacunto *et al.* (2013), and large scale chamber tests (method 4 in Section 2), following Afshari *et al.* (2005).

### 4.1 Method

Tests were conducted in a chamber with dimensions 2.62×3.49×2.35m, in Nottingham, UK, during May 2017; see Figure 5. The test chamber was repurposed for air quality measurements using plastic sheeting on its floors and by removing unnecessary furniture and fixings. Custom boards were installed into the open window and door to control the ventilation conditions. A low level 0.1×0.1m opening is located in the doorway, and an extractor fan fitted into the window, designed to provide an exhaust flow rate of 85m<sup>3</sup>/h (AER=3.96h<sup>-1</sup>) (Screwfix, 2017). A desk fan was used to aid mixing.

Two SidePak™ AM510 personal aerosol monitors logged PM<sub>2.5</sub> mass concentrations at 1 second intervals, to improve resolution. Both SidePak™s

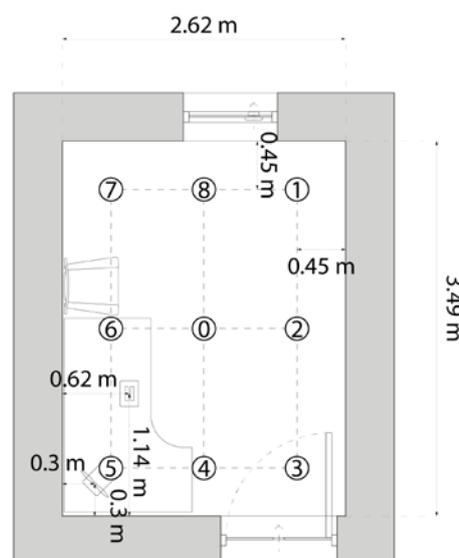


Figure 5 - Test Chamber Plan

were mounted on tripods at height 1.1 m. The first was centrally located (see position 0 in Figure 5) whereas the second was moved through all 9 positions between tests to evaluate the room mixing conditions, the results of which are not discussed here. Gravimetric sampling was conducted, and so a default CF=1.0 was used.

In addition, 2 IAQ-Calc Indoor Air Quality Meters (Model 7545, TSI Inc., Shoreview, MN, USA) monitored indoor and outdoor temperatures, CO<sub>2</sub> concentration, and relative humidity. Chamber relative humidity was maintained below 70% for all tests, to minimise its influence on SidePak™ performance. When required, an electric oil-filled radiator was used to maintain the relative humidity.

At the start of each test day, PM<sub>2.5</sub> background concentrations were monitored for 5 minutes periods. The phase 1 toasting tests were repeated exactly using wholemeal bread, as there was no noticeable difference in emissions from white and wholemeal bread in Phase 1. The toaster was located on a table; see Figure 5. The start and finish of the  $\alpha$ -period were recorded when the toaster was switched on, and when the toasting ended, respectively. Thereafter, the toast and toaster were sealed in a box and the  $\beta$  and  $\gamma$ -periods were observed for 20 minutes. The room was then flushed with outside air to restore concentrations to background levels.

## 4.2 Results

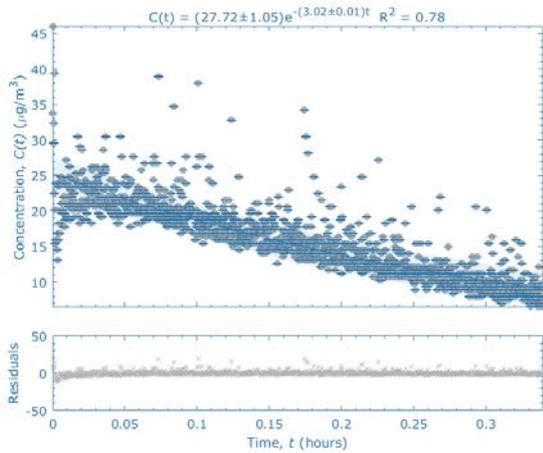


Figure 6 - Example Decay, CF = 0.47

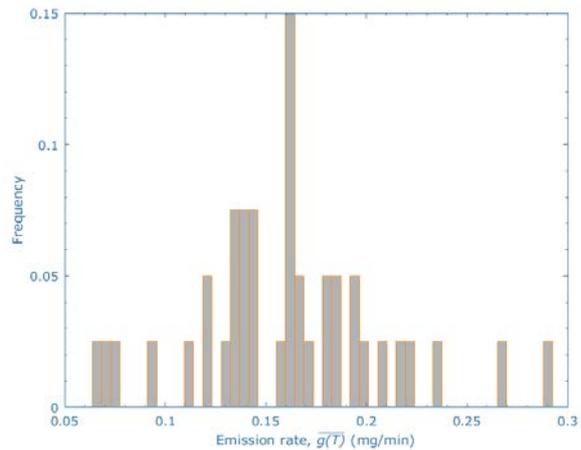


Figure 7 - Phase 2 Emission Rates Histogram, CF = 0.47

The data were processed using the method described in Sections 3.2 and 3.3, except that the decay was determined using the  $\beta$ - and  $\gamma$ -periods together, due to the difficulty in identifying the peak concentration over the background noise attributable to the 1-second logging interval (see Figure 6). Tests were not eliminated according to a minimum threshold R<sup>2</sup>-value, because visual inspections indicate a good fit even when R<sup>2</sup><0.7. Here, noisy data from the 1-second time-step makes the R<sup>2</sup> a poor indicator of goodness-of-fit. A single outlier test was eliminated according to *Chauvenet's Criteria*.

Mean and median decay rates were 3.97 h<sup>-1</sup> and 3.33 h<sup>-1</sup>, respectively, compared to an expected air change rate 3.96 h<sup>-1</sup> from the fan. This suggests minimal decay due to other factors, such as deposition. It is also possible that the extract fan operated at a lower rate than specified, although this was confirmed independently. Figure 6 shows that the  $\beta$ -period is short, indicating good mixing.

Table 2: Phase 2 Emission Rates (mg/min)

CF	N	Mean	SD	Percentiles				
				2 <sup>nd</sup>	25 <sup>th</sup>	50 <sup>th</sup> (Median)	75 <sup>th</sup>	98 <sup>th</sup>
0.47	39	0.157	0.043	0.071	0.135	0.161	0.182	0.071
0.79	39	0.265	0.073	0.119	0.226	0.270	0.306	0.119

The resulting mean and median emission rates are approximately 25% and 50% of the values measured during phase 1. It is possible the different conditions and variations in the bread used may have modified the emissions. Additionally, as the  $\beta$ -periods were included when determining the decay rates, both they and the theoretical peaks may have been underestimated leading to lower emission rates. Further developments of the post-processing method are required to identify the peak concentration for high frequency data to help reduce this source of error.

The distribution of emission rates is given in Figure 7, and is smaller than the distribution obtained during phase 1 (see Figure 3). It has no outliers that increase the mean and the distribution is less skewed, but still appears to be non-Gaussian. The median is slightly greater than the mean (see Table 2) so there may be a slight negative skew, but this difference is small.

## 5 DISCUSSION

The mean emission rates from both phases, calculated using both CFs, are an order of magnitude lower than those reported by Dacunto and Jiang (see Table 3). There are two main unknowns which could have caused this difference: firstly the lack of gravimetric sampling, so the true calibration factor required may be very different, and secondly, the degree of charring. Whilst Dacunto and Jiang use a standard char-index to report the degree of charring in their tests, the authors have been unable to obtain this index for comparison. The estimated mean emission rates from Phase 2, for both CFs, are similar to the median 0.11 mg/min reported by He (see Table 3).

Table 3: Emission Rates Comparison

Source	N	Mean Emission Rate (mg/min)	SD	RSD
Phase 1 – CF 0.47	29	0.66	1.01	1.53
Phase 1 – CF 0.79	29	1.11	1.70	1.53
Phase 2 – CF 0.47	26	0.16	0.04	0.24
Phase 2 – CF 0.79	26	0.27	0.07	0.24
Toast, 90-95% char (Dacunto <i>et al.</i> , 2013)	3	9.5	10.8	1.14
Toast, 70-80% char (Jiang <i>et al.</i> , 2011)		4.2		
Toasting (He <i>et al.</i> , 2004)	18	0.11*	0.37	3.36

\* median; RSD, relative standard deviation.

Despite the larger number of repetitions in Phase 1, the variance of the emission rates remains large, indicated by the large relative standard deviation (RSD). Therefore, the test methodology from this phase is not highly repeatable, although it is significantly more consistent than the methodology from He. The improved method of Phase 2 used a smaller room volume with improved mixing to give a better defined  $\alpha$ -period and a more reproducible test. This is indicated by its RSD, which is smaller than the Phase 1 RSD by an order of magnitude.

The calibration factor is the largest source of uncertainty, because gravimetric sampling was not used. This problem is not unique to the SidePak™, but is also applicable to other optical monitors, which are commonly used to measure temporal concentrations of particulate matter. It is recommended to simultaneously conduct gravimetric sampling to determine custom calibration factors, although this may not always be possible.

The *peak estimation* calculation method may also be flawed because it assumes a constant emission rate over the emission period, which is not observed for this source, and may not be the case for other cooking sources. Pagels *et al.* (2009) used an iterative process to calculate emission rates by fitting a curve to the observed data and adjust the emission rate to obtain the best possible fit. There are two potential problems with this method. It shares the first with the *peak estimation* method, because they both assume a constant emission rate during the  $\alpha$ -

period, which was not observed here; see Figure 6. Therefore, the predicted curve would never be a good fit. Secondly, the curve predicted using the mass balance equation assumes instantaneous mixing, which is not observed in reality (Ott *et al.*, 2006). Ott suggests an alternative calculation method, known as the *area-under-the-curve* method. It uses the total mass emitted to estimate the average emission rate (Ott *et al.*, 2006). This method is not well established in literature, where preference is usually given to the *peak estimation* approach. The *area-under-the-curve* method is theoretically exact, but it assumes instantaneous full-mixing, which is not observed in practice.

Indoor environment modelling at stock level often use stochastic methods to explore uncertainties in indoor pollutant concentrations (Das *et al.*, 2014). These require probability density functions (PDFs) of emission rates, which could be obtained from the Phase 2 data by methods such as fitting a curve to Figure 9, or using bootstrapping techniques to account for uncertainty in all of the measurements. Future work will investigate this further.

## 6 CONCLUSIONS

There is growing evidence linking outdoor PM<sub>2.5</sub> exposure to a range of health effects, yet the impacts of exposure indoors is relatively unknown. Indoor environment modelling at stock level can be used to investigate these impacts, but they are highly sensitive to the emission rates used. Therefore, a better understanding of the emissions rates and their uncertainties from key indoor sources, such as cooking, is required. They should be presented as probability density functions so that probabilistic sampling methods can be employed to estimate uncertainty in indoor concentrations.

Five common methodologies are used to measure temporal variations in PM<sub>2.5</sub> concentrations during the cooking of foods. Most use the *peak estimation method*, derived from a mass-balance equation, to calculate emission rates from these measurements. Controlled field tests and large scale chamber tests are preferential methods because they offer a high level environmental control without abstracting conditions so far from reality that they render the calculated emission rates implausible. Most studies typically have a low number of repetitions, which is insufficient to produce PDFs or demonstrate emission rate distributions.

Two phases of tests use undertaken in different indoor environments to develop a methodology appropriate for measuring PM<sub>2.5</sub> emission rates from the cooking of foods. The toasting of bread is used as a simple and repeatable source. The first phase identified key flaws in the *field test* method and was used to inform a second phase. These proved to be better, but still problematic. Key problems identified are the need for calibration factors used by the optical measurement devices frequently used to measure temporal PM<sub>2.5</sub> concentrations, the need for full-mixing and a means of identify mixing conditions, and the need for a method to identify the  $\alpha$ -,  $\beta$ - and  $\gamma$ -periods used to determine emission and decay rates. The method does not require a known ventilation rate, though the model does depend on a constant decay rate, and so it must be steady. Additionally, further investigations into the measuring and modelling of non-constant emission rates is needed, as the *peak estimation method* used here may under- or over-estimate the total mass emitted depending on the emission profile.

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