

Diagnostic barriers to using PM_{2.5} concentrations as metrics of indoor air quality

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ABSTRACT

Particulate matter with a diameter of $\leq 2.5\mu\text{m}$ (PM_{2.5}) has been shown to be present in many buildings at concentrations that are harmful to human health. Accordingly, they should be used as metrics of indoor air quality (IAQ) and included in standards or norms. This paper uses measurements of PM_{2.5} concentrations made in three different environments using three different devices to show that there are barriers that must be before they can be considered viable diagnostics. Optical particle counters (OPCs) are a common device used to measure temporal changes in PM_{2.5} concentration. The PM_{2.5} concentration is inferred from the light scattering properties of the particles sampled, properties which vary by source and composition. Accordingly, OPC measurements must be scaled by a calibration factor, a simple multiplier derived from concurrent gravimetric sampling. A gravimetric sampler uses an impactor to remove large particles and measures the total mass of the collected on a filter during a sampling period allowing the calculation of an average PM_{2.5} concentration. OPCs are often most sensitive to PM_{2.5} with a particular size, and so OPCs capable of disaggregating PM_{2.5} concentrations by their diameter can be used to select an appropriate OPC for a particular source. Knowledge of these factors is important if a measurement of PM_{2.5} concentration is to be used to describe the quality of indoor air using a metric. In order to be useful, a standard or norm should provide clear guidance on device selection, device calibration, and calibration factors (for optical particle counters). For in-situ measurements, it must specify an appropriate measurement location, a sampling frequency, and time-averaging period. After measurements have been made, appropriate statistics and an error analyses should be prescribed to quantitatively describe the variance in the data.

Ambient PM_{2.5} concentrations have been measured in many countries for decades and so there is a mature library of guidance and standards. The indoor community should consider how it might adopt the methods it uses for regulatory monitoring by gravimetric sampling, and how it demonstrates the equivalence of real time OPCs to the gravimetric method so that they too can be used for regulatory purposes.

KEYWORDS

Measurement, gravimetric, optical particle counter, calibration factor, emission rate

1 INTRODUCTION

As policy makers strive to reduce the energy demand of buildings by sealing them or limiting ventilation rates, an unintended consequence could be a reduction in the quality of indoor air with corresponding negative health effects for individuals and increased burdens on public health care systems. Current standards specify a minimum ventilation rate that is fundamentally set for odour control and that is also assumed to minimize contaminant exposures and, therefore, protect occupant health. Accordingly, there is a need for

performance-based health-centred indoor air quality (IAQ) metrics supported by our best knowledge of health effects. They must be measurable, achievable, and have a positive impact on the physical health of occupants of buildings.

The AIVC estimates (Borsboom *et al.*, 2016) that the most dangerous pollutant is particulate matter with a diameter of $\leq 2.5\mu\text{m}$ (PM_{2.5}). They are small enough to bypass biological defences and are linked to chronic respiratory and cardiovascular diseases, and cancer. There are many indoor sources, such as combustion, cooking, vacuum cleaning, and the re-suspension from the pumping of doors and the action of sitting on soft furniture.

These are compelling arguments for using an indoor PM_{2.5} concentration as an IAQ metric, but there are also significant barriers. These include a lack of agreement on appropriate norms or standards; for example, the WHO recommends that mean PM_{2.5} concentrations in air breathed by a person is less than 25 $\mu\text{g}/\text{m}^3$ per day and 10 $\mu\text{g}/\text{m}^3$ per year but the U.S. National Ambient Air Quality Standards require 35 $\mu\text{g}/\text{m}^3$ and 12 $\mu\text{g}/\text{m}^3$, respectively. The WELL Buildings Standard sets a threshold of 15 $\mu\text{g}/\text{m}^3$ (IWBI, 2016) measured at least once per hour at a resolution of 10 $\mu\text{g}/\text{m}^3$ or finer.

There are technical disagreements on appropriate diagnostic approaches; for example, a gravimetric sampler gives a mean concentration over a period of time, but in areas of low concentrations the sampling period is long and the standards for processing filters is onerous and expensive. Light scattering devices instantaneously show temporal variations in concentrations but require an appropriate calibration factor to correct for the disparity in the light scattering properties of particles from different sources. There is also a lack of consensus in many other areas, such as tester competence or test methods, appropriate regulation of mitigation measures, and the assumption that one standard or norm is appropriate for every building in every country.

Measuring PM_{2.5} is clearly problematic and so this paper considers measurements of PM_{2.5} concentrations made by the authors in three environments for four different purposes to highlight diagnostic barriers that must be overcome before PM_{2.5} can be included in IAQ standards and norms. Section 2 summarizes four types of health based IAQ norms that require PM_{2.5} measurements and Section 3 describes the measurement methods. Section 4 presents the results and discusses diagnostic confounding factors that can affect the use of measurements of PM_{2.5} concentrations as an IAQ metric.

2 HEALTH BASED IAQ STANDARDS AND NORMS

An air quality standard or norm should identify when the quality of indoor air is unacceptable and be based on its effects on human health and comfort, acknowledging that they may not be immediate. Section 1 identified that PM_{2.5} is an important metric of IAQ. Accordingly, four norms are introduced that might contain a PM_{2.5} metric, following the review of Jones (2017), and are considered to identify appropriate diagnostic methods.

2.1 Ratings Systems

Rating systems (RS) identify properties of a building that are known to affect IAQ directly, for example using a tick-box approach. Each feature might be weighted according to their hazard and aggregated to produce a single metric. This method could be used to develop a third-party rating system, similar to any existing energy rating scheme, and could help those particularly sensitive to specific contaminants. An example system is the WELL Building Standard (IWBI, 2016), which awards Silver, Gold, and Platinum status to non-domestic buildings depending on the extent of their compliance with its requirements. An RS requires spot measurements of PM_{2.5} concentration or the identification of known emitters and mitigation methods, which can be determined by measuring their PM_{2.5} emission rates.

2.2 Sub-Indices

To obtain a comprehensive picture of the IAQ in a building it is necessary to measure PM_{2.5} and other contaminants whose individual concentrations may be incomparable; for example, radon (Bq.m⁻³) and PM_{2.5} (µg.m⁻³) concentrations. These individual contaminant concentrations can be converted into sub-indices (SI), which may be a function of their health risks, before they are aggregated into a single index. An SI requires temporal measurements of PM_{2.5} concentration made *in-situ*.

2.3 Exposure Limit Values

Exposure limit values (ELV) are used in occupational environments to prevent or reduce risks to health from hazards, such as vibrations, by setting a maximum quantity experienced per person per day. This principle could be applied when measuring the concentrations of PM_{2.5} and other contaminants in a building. Here, the ratios of their maximum concentrations to their respective ELV concentrations give a quick indication of risk, where a ratio $\ll 1$ might be acceptable but one approaching or exceeding unity may be problematic. Here, an indication of the relationship between exposure and health consequences is required. An ELV requires temporal measurements of PM_{2.5} concentration made *in-situ*.

2.4 Health Adjusted Life Years

Health adjusted life years (HALY) are measures of health over time and give weighted years a person of cohort lives with a disease or disability. Disability is weighted by its effect on person's life in general, and so can account for mental illness. There are two key HALY metrics. The first is the disability adjusted life year (DALY), which is a measure the disease burden in a population, expressed as the sum of the number of years lost due to morbidity and mortality, where a value of 0 represents no loss. In the case of IAQ, the disease burden is a measurement of the difference between the current health status of a population of building occupants and an ideal situation where they all live into old age, free of disease and disability (WHO, 2009). The second is the quality-adjusted life year (QALY), which reflects the quality of life of a person or cohort but is the approximate inverse of a DALY because it considers the health gained from an intervention where a value of 1 represents a year lived in perfect health and 0 is death. Both the QALY and DALY can be used to assess the financial values of exposures to poor IAQ and interventions designed to minimize it.

Measured and modelled PM_{2.5} concentrations can be used to estimate chronic health impacts in HALYs lost or gained. Here, temporal measurements made *in-situ* and source emission rates are required.

3 METHODS OF MEASUREMENT

The diagnostic techniques used to quantify PM_{2.5} must be understood and clearly stated in a norm. PM_{2.5} concentrations have been measured by the authors in three unique environments using three different devices for a range of purposes. Each of the locations and devices has benefits and confounding factors that affect measurements in different ways.

3.1 Locations

Location 1 is an enclosed **indoor chamber** with controlled ventilation and volume 26m³ located at TNO offices, in Delft, The Netherlands. Air is supplied by an HVAC unit equipped with an AFPRO F7 filter via a single ceiling diffuser. Air is extracted from a single location in the middle of the chamber 94cm below the ceiling at 21l/s. To minimize uncontrolled

infiltration and exfiltration between other zones, the supply air flowrate is adjusted so that the pressure difference between the test chamber and its surroundings was less than 0.5Pa.

Location 2 is an **outdoor chamber** (all sides exposed) with volume 21.5m³ located in Nottingham, UK; see O'Leary & Jones (2017) for full details. A door and window are located in the smallest facades and directly opposite one another. The floor is lined with plastic sheeting and the chamber is sparsely furnished to avoid particle re-suspension and excessive deposition. Custom boards are 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 only window, designed to provide an exhaust flow rate of 85m³/h (AER=3.96h⁻¹). A desk fan was available to aid mixing and an electric oil-filled radiator was used to change the relative humidity. The chamber is manufactured to Passivhaus standards and so infiltration is considered negligible.

Location 3 is an open plan **domestic kitchen** and living room exposed on two facades, with volume 77m³, located in Nottingham, UK. It is connected to a hallway via a door. Purpose provided ventilation is via a window, backdoor, and an extractor fan with damaged flue. Food is cooked on four gas hobs and in a standard electric oven. The house is considered to be *leaky* and so infiltration is a prominent source of ambient air when the purpose provided openings are closed and off.

3.2 Devices

A TSI **SidePak**TM AM510 measures PM_{2.5} concentrations at intervals of 1 second, which may be time-averaged when used over long periods of time. To filter particles >2.5µm, it uses an impactor that draws a sample of air through a convoluted path so that the inertial and aerodynamic drag forces cause the larger particles to collide with, and stick to, a greased plate. The remaining particles are detected using the *light scattering* method, which detects the amount of laser light scattered by them at 90°. However, the degree of scattering varies according to particle density, size distribution, shape, and refractive index, and so particles of equal diameter and different origins have divergent degrees of scattering. Optimal performance occurs when both the airflow rate through the device and the scattering are calibrated. The SidePak is calibrated by the manufacturer using A1 (Arizona) test dust defined by ISO 12103-1 (BSI, 2016). Accordingly, non-A1 particle types require a calibration factor (CF), a multiplier that scales a reading; see Section 3.3.3. For example, if the SidePak reads 10µg/m³ but the particles are from a known source with CF=0.5, then the true reading is 0.5×10=5µg/m³.

A **GRIMM** Model 11-R Mini Laser Aerosol Spectrometer measures particles with diameters between 0.25-32µm and classifies them into 31 size bins using the light scattering method of detection. It detects concentrations between 0.1µg/m³ and 100mg/m³, and has a sampling frequency of 6 seconds. It is calibrated using solid, spherical, and non-absorptive polystyrene latex particles with a defined distribution of diameters. The GRIMM also requires a CF when the source of the measured particles differs from those used to calibrate it.

An Airmetrics **MiniVol** Tactical Air Sampler is used for gravimetric sampling. A pump draws air through a PM_{2.5} impactor and a 47mm Millipore fluoroporeTM PTFE membrane filter at 5l/min for a defined period of time. Filters are weighed before and after sampling under controlled conditions described by EN 12341 (BSI, 2014). The average particle concentration over the sampling period is determined from the sampling time, the increase in filter mass, and the airflow rate.

Each device has its own advantages. The SidePak can be battery operated and can be worn to measure personal exposure and has the shortest sampling frequency. The GRIMM cannot be worn, but can identify particle size distributions at 6 second intervals, which the SidePak

cannot. The MiniVol gives the most accurate mean mass concentration of the three, but cannot identify changes over the short time periods.

3.3 Measurements

3.3.1 Gravimetric Sampling

The MiniVol was used in the outdoor chamber to calculate a CF for the toasting of bread, which was then used to calculate emission rates for the toasting process; see Section 3.3.4. The MiniVol and two SidePaks were located side by side in the centre of the chamber. Three measurements were made where the chamber was sealed, door and window closed, extractor fan off, and mixing fan switched on. Three tests were used and the chamber was purged before each test. Test 1 toasted two slices of bread at the same time for 225 seconds. The decay (γ) period was 1 hour 37 minutes. Tests 2 and 3 generated higher PM_{2.5} concentrations by toasting ten slices of bread, two at a time, for an average of 215 seconds and 219 seconds, respectively. There was a gap of 10 minutes between each cooking event and the decay period was 1 hour. The MiniVol and SidePaks took samples during each decay period and, after each period, the filter was removed for weighing using a Sartorius Analytical 5 decimal place balance. The CF for each SidePak is determined from the linear regression through the origin of the mean concentrations measured by the MiniVol and SidePak.

3.3.2 Temporal Changes

The measurements of changes in PM_{2.5} concentration over time were made in all three locations; see Section 3.1. The GRIMM was used in the indoor chamber to measure changes when cooking a meal comprising chicken breast fried in olive oil, pre-cooked sliced potatoes fried in olive oil, and boiled French beans. The cooking process was repeated 6 times to test repeatability.

The SidePak was used in the outdoor chamber and domestic kitchen. In the outdoor chamber it was used to measure the decay of particles emitted by a specific source so that the emission rate can be calculated; see Section 3.3.4. The sampling frequency was 1 second. It was also placed in a domestic kitchen for 7 days to record the changes in concentration occur as food is cooked by the occupants. Cooking times, methods, and ingredients were recorded by the occupants in a diary. The SidePak was located 2m away from the cooker at height 1.5m with a sampling frequency of 1 second but stored time-averaged over 1 minute intervals.

All measurements made by a SidePak or GRIMM require a CF to accurately report temporal changes. The CF for the GRIMM was obtained by a 3rd party and is not discussed here. The method of obtaining a CF for the SidePak is described in Section 3.3.1.

3.3.3 Size Distributions

The GRIMM was used in the indoor chamber to measure size fractions when a meal was cooked; see Section 3.3.2.

3.3.4 Source Emission Rates

The outdoor chamber was used to measure the emission rates from (i) the toasting of 2 slices of bread (medium sliced white or wholemeal Hovis 800g loaf) in a supermarket branded toaster on the highest setting; and (ii) Dove Extra Fresh aerosol deodorant. The method used to determine emission rates is based on the widely used mass balance model following Ott *et al.* (2006) and O'Leary & Jones (2017). Three phases during emissions: an α or emission period when air is not well-mixed, a subsequent β -period when emissions cease and mixing is incomplete, and a γ -period of well mixed decay. For a rectangular source function, the peak concentration should occur at the end of the α -period, but it is frequently observed afterward indicating that complete-mixing is not instantaneous. The *theoretical peak estimation* method

is used where the decay rate, ϕ , is determined from the log-linear regression of the $PM_{2.5}$ concentration during the γ -period and extrapolated back to the start of the β -period. This theoretical peak, C_p , is then used to estimate the emission rate over an emission time period T , where

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

Here, G is the emission rate, V the mixing volume, C_b the background concentration, and $C(0)$ the initial concentration.

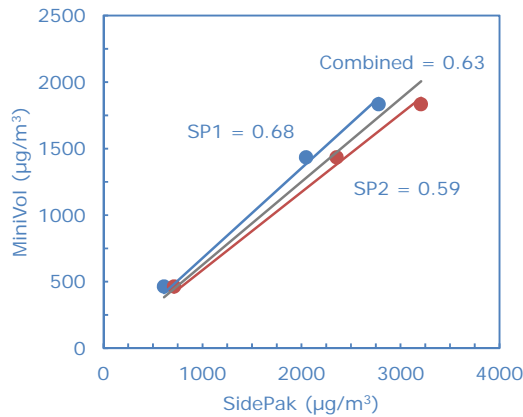


Figure 1: SidePak Calibration Factors. blue, SidePak 1 (SP1); orange, SidePak 2 (SP2); grey, combined.

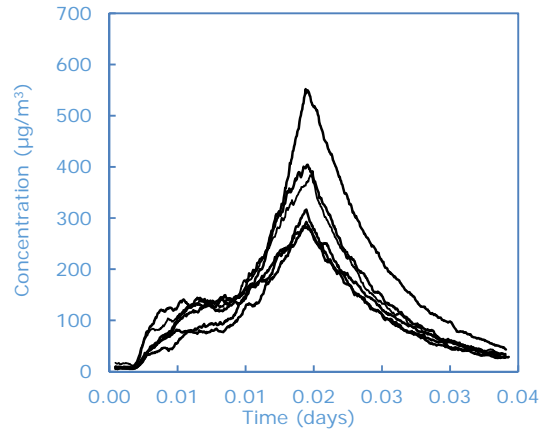


Figure 2: Indoor chamber. Temporal changes during cooking repeatability test.

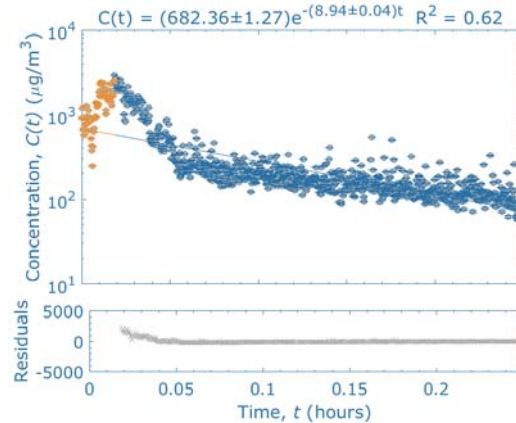
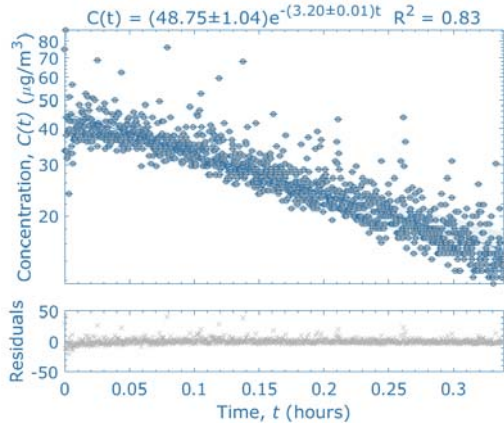


Figure 3: Left, decay of toast; Right, decay of aerosol deodorant. Orange indicates α and β periods.

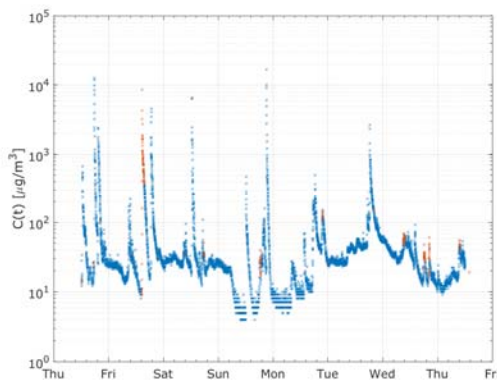


Figure 4: Temporal changes over 1 week in a domestic kitchen during the heating season. Orange indicates humidity > 70%.

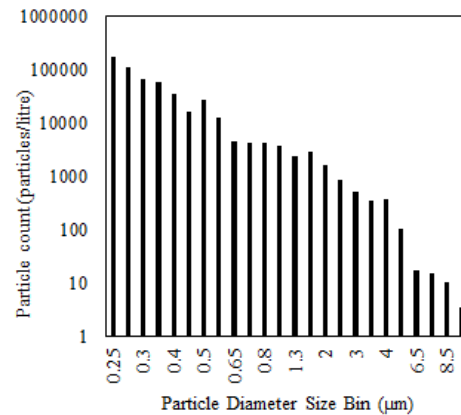


Figure 5: Mean particle size distribution for cooking in the indoor chamber.

The SidePak was located in the centre of the chamber and the mixing fan was switched on for the toast and off for the deodorant. A CF Calibration factor required to obtain an accurate G , although it is not required to obtain ϕ . Good mixing was demonstrated by using a second SidePak to obtaining similar mean measurements of concentration over a sampling period as the first at different locations in the chamber.

4 RESULTS AND DISCUSSION

The measurements described in Section 3 are used to results show how outcomes that affect the application of PM_{2.5} measurement in IAQ standards. It is our intention to discuss the process of taking measurements rather than specific outcomes, such as emission rates, and to show how these might affect the use of measurements in health based standards or norms.

4.1 Gravimetric Sampling and Calibration Factors

The gravimetric sampling described in Section 3.3.1 is used to derive relationships between the mean measurements made by the MiniVol and two SidePaks; see Figure 1. It shows that the CF is different for each SidePak and that a less accurate combined CF can also be obtained. More samples, and more SidePaks would reduce uncertainty in the CFs. PM_{2.5} from ambient sources were present in the chamber air but because there is no ventilation—supported by a very small decay in PM_{2.5} over the sampling period (not shown here)—and so they are not expected to have a significant effect on the CFs in this instance. An unbiased CF requires all ambient air to be filtered and a negligible background concentration.

4.2 Temporal Changes

Measurements of temporal changes are described in Section 3.3.2 and are used to show trends and a range of behaviours. Figure 2 gives PM_{2.5} concentrations as a meal is repeatedly cooked and shows repeatable temporal trends in times concentrations and gradients that occur as ingredients are added to the pan and as they undergo Maillard browning. There is variance in the magnitude of the concentrations between tests attributed to differences in pan temperature, ingredient constitution, chamber mixing, and less-quantifiable errors.

Figure 3 shows the decay of PM_{2.5} emitted by toast (left) and an aerosol deodorant (right). All toast measurements have been scaled using an appropriate CF (see Section 4.1) whereas those for the deodorant have not because a suitable value is unavailable. Accordingly, these measurements should only be used to indicate trends and behaviours. The data are plotted using a logarithmic y-axis and so the exponential element of the γ -period decay is shown linearly. The residuals show that the toast decay is largely exponential but that the deodorant's decay is not initially exponential but becomes so after around 2 minutes of decay. This is explored further in Section 4.4.

Figure 4 shows the changes in concentrations measured in a domestic kitchen over 7 days. The peaks are attributable to cooking and correlate with the activities recorded by the occupants in their diary. The peaks are substantial and are two orders of magnitude higher than the ELVs given by the WHO; see Section 1. These measurements have not been scaled using a CF because of the range of cooking methods and ingredients used. CFs for food are generally <1 (O'Leary & Jones, 2017) but Figure 1 still shows that the peak concentrations are substantial and interventions are required to minimize exposure risk. Here, it is possible for a low cost OPC to switch on a targeted ventilation device, such as a range hood, and this can be done when the local concentration is clearly above the ambient concentration. The absence of a CF should not affect the device's ability to act as a switch, and in well ventilated spaces, it should be possible for it to self-calibrate. An OPC can determine ϕ from smooth

decays that follow a peaks that are significantly above the ambient concentration (see Figure 4) with the caveats that C_b is known and ϕ is not exclusively a ventilation rate. Most OPCs are confounded by high humidity. The SidePak has an operational range of 0-95% relative humidity (RH) (TSI, 2012), but there is concern about concentrations measured for RH>70% (Dacunto *et al.*, 2013). Figure 4 colours all measurements orange when RH>70%, and shows that they generally coincide with peaks and, therefore, with emissions. The data presented in Figures 2-3 was obtained with a high sampling frequency of at least 6 seconds and each sensor was set to sample as fast as it would allow obtaining maximum information. This gives clarity but can also show high levels of noise; see Figure 3-left. Time averaging is generally undesirable because it obscures behaviour but may be appropriate if storage space is limited (for the long-term measurements made in the domestic kitchen) or if the intention is to show compliance against a mean concentration, such as the WHO thresholds. Noise can be reduced by effective mixing, but this can also increase deposition and make measuring the decay from sources with a low emission rate difficult.

4.3 Size Distributions

Figure 5 gives the mean particle size distributions at the time of the peak concentrations shown in Figure 2. The distribution is right skewed and so most of the particles are very small and $<<2.5\mu\text{m}$. However, most of the mass will be in the larger size bins (not shown here). These factors should be considered when choosing a low-cost device to detect emissions from this meal.

4.4 Source Emission Rates

Section 4.2 identified an initial period of non-exponential decay in Figure 3 (right), which shows that the changes in concentration that are not captured by the first-order removal of particles represented by ϕ and that there are likely to be changes in the mass and size distribution of particles caused by agglomeration, evaporation, and other processes. Accordingly, it may be appropriate to estimate ϕ using the exponential element of the decay by assuming that it is the remaining particles that harm health.

When measured by a SidePak, the errors in the concentration are a constant rather than a proportion, which make the error analysis straight forward. However, when the error is closer to a percentage, a more robust approach, such as a Chi-squared goodness of fit analysis (Hughes & Hase, 2015), allows each data point to be individually weighted and a goodness-of-fit estimate is obtained. A further consideration is that a standard least-squares regression is

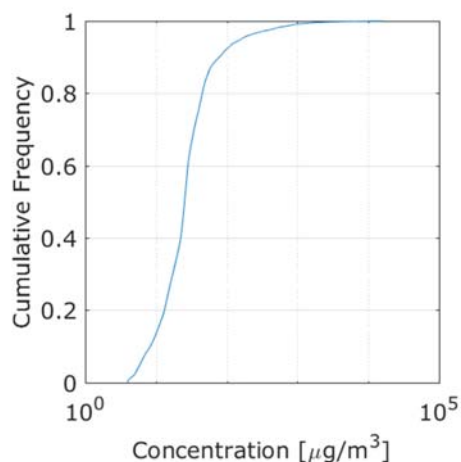


Figure 6: CDF of concentrations measure over a week in a domestic kitchen during the heating season.

biased by outliers of noisy data (see Figure 3 left) and so a robust regression method, such as least-absolute-deviations, may be more appropriate. It is important to take multiple measurements of emission rate for a single source to indicate uncertainty in them. They can be shared as a distribution and used in stochastic modelling.

4.5 Statistical Reporting

Figure 4 shows that concentrations measured in indoor environments can be varied and when all data points are represented as a distribution they are often found to be approximately log-normal (Salthammer, 2011). Figure 5 is a cumulative

distribution function (CDF) of the measurements made in the domestic kitchen. It shows that they are significantly right-skewed and mostly lie between 10-100 $\mu\text{g}/\text{m}^3$, that the peak concentrations shown in Figure 4 are rare events, and that the likelihood of receiving a dose in excess of the WHO thresholds is high. When reporting indoor concentrations, it is important that descriptive statistics (or graphic representations) are used that quantitatively describe the variance in the data. The skewed nature of indoor pollutant distributions suggests that a mean average is an unsuitable descriptive statistic and centiles are more useful. The shape of the distribution is important when comparing two samples, such as those taken before and after some intervention. Here, tests that determine the *effect size*, the size of the difference between two samples, must account for non-normality and for different sample sizes; for example, if data points measured when the RH is unacceptably high (see Section 4.2) are removed.

4.6 Diagnostic Techniques

The measurement of a PM_{2.5} concentration is challenging because the metric does not describe a physical or chemical component of the air but is defined by the measurement method itself (AQEG, 2012). These difficulties are reflected in a measurement uncertainty $\pm 25\%$ for PM_{2.5} required by the UK's Air Quality Directive. It is $\pm 15\%$ for most other pollutants (AQEG, 2012). The most accurate measurements of PM_{2.5} concentrations are achieved using gravimetric sampling because they directly measure the acquired mass of particles on a filter over a known period of time.

PM_{2.5} concentrations in ambient air have been systematically monitored in the UK since 1987. For a gravimetric measurement to be used to demonstrate compliance against an air quality standard or norm, the measurement procedure must follow the *Reference Method* described in EN 12341 (BSI, 2014). It controls the design of the gravimetric sampler (pipe work, filter holder, flow control system, leak tightness), the sampling period, the storage conditions of the filters, and the weighing facilities (balances and room) and procedures. In the US, these same methods are enforced by a Federal Regulation (EPA, 2006). Gravimetric measurements do have the granularity required to be a good diagnose tool (see Section 4.2) or to derive emission rates from known sources (see Section 4.4). The same is true in ambient air where real time devices must demonstrate *equivalence* with the Reference Method, defined in BS EN 16450 (BSI, 2017).

Without descriptions of the diagnostic procedures it is possible to manipulate both the conditions and the data to meet a benchmark. For example, WELL requires a threshold of 15 $\mu\text{g}/\text{m}^3$ measured at least once per hour at a resolution of 10 $\mu\text{g}/\text{m}^3$ or finer. Therefore, an imprecise device must measure <20 $\mu\text{g}/\text{m}^3$ to comply. Furthermore, an hourly spot measurement could lead to an elevated reading, perhaps attributable to poor mixing or the measurer's proximity to a source. Here, time averaging away from known sources and in areas of a room where the ventilation effectiveness is low, such as corners, are all advantageous and possible because no measurement location is specified.

To ensure that measurements are accurate, precise, and repeatable, it is necessary for a standard or norm to clearly define the device type, its calibration method, resolution, sampling frequency, a desired time averaging period, the measurement location, and adherence to the Reference method and the demonstration of equivalence.

Finally, Sections 3.3.2-3 show that the SidePak and Grimm were calibrated using different particle types. Although they cannot share calibration factors, it is desirable for the IAQ research community to agree on a calibration particle type so that CFs for a particular device and a range of sources can be reported in the literature and used widely.

5 CONCLUSIONS

This paper considers measurements of PM_{2.5} concentrations made in three different environments using three different devices to show that there are barriers that must be overcome before they can be viable diagnostics. In order to be useful, standards and norms should provide clear guidance on device selection, device calibration methods, and calibration factors (for optical particle counters). For *in-situ* measurements, it must specify an appropriate measurement location, a sampling frequency, and time-averaging period. After measurements have been made, appropriate statistics and an error analyses should be prescribed to quantitatively describe the variance in the data.

Ambient PM_{2.5} concentrations have been measured in many countries for several decades and so there is also a mature library of guidance and standards. The indoor community should consider how it might adopt the Reference Method for regulatory monitoring by gravimetric sampling, and demonstration the equivalence of real time devices to the gravimetric method so that they too can be used for regulatory purposes.

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