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Indoor Particulate Matters Measured in Residential Homes in the Southeastern United States: Effects of Pandemic Lockdown and Holiday Cooking

Sierra Laltrello[1], Azita Amiri[2], Shan-Hu Lee[1*]

[1] Department of Atmospheric and Earth Science, The University of Alabama in Huntsville, Huntsville, AL 35805, USA
[2] College of Nursing, The University of Alabama in Huntsville, Huntsville, AL 35805, USA

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ABSTRACT

Although humans spend a majority of their lives in indoor environments, indoor air quality is immensely understudied, compared to ambient air. Here, we show the first long-term measurements of household indoor PM concentrations in the southeastern United States, for one year (May 2019 through April 2020) covering the COVID-19 hard-lockdown period (March and April 2020). Particle size distributions between 0.25–35 μm were measured with a low-cost sensor, which does not utilize hazardous chemicals and radiation sources and is ideal for indoor air monitoring in real households without disruption of residents' living conditions. Our observations show that while cooking and cleaning are two major emissions sources for the residential indoor PM, consistent with the literature knowledge, but we also show that human occupancy affects the indoor PM level substantially. During the hard lockdown during the COVID-19 pandemic, the background level of indoor PM increased by ~200%, while the ambient PM decreased by ~50% during the same period. Before the pandemic, the indoor PM level was lower than the outdoor, but it became similar or higher than the outdoor level during the pandemic. Thanksgiving holiday cooking (prior to COVID-19) produced high concentrations of PM for an extended period (e.g., over 6 hours) even with active kitchen ventilation. PM concentrations during a cooking and cleaning event usually increased linearly to a maximum value and then decayed exponentially. The decay time of indoor PM ranged from several minutes up to ~100 minutes and increased with the particle size, indicating that particle deposition to the interior surfaces is the main sink process of the indoor PM.

Keywords: Pandemic, Indoor air, Indoor PM, Holiday cooking, Cleaning

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* **Corresponding Author:**
shanhu.lee@uah.edu

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1 INTRODUCTION

Humans spend between 80% to 90% of their lives in indoor environments (Diffey, 2011; Klepeis *et al.*, 2001). Globally, air pollutions cause several millions of premature deaths annually (WHO, 2016). Inhalation of PM, especially $PM_{2.5}$ (PM with a diameter smaller than 2.5 μm), causes long-term respiratory issues such as asthma and chronic bronchitis (Jedrychowski *et al.*, 2013; Li *et al.*, 2020; Nel, 2005). PM inhalation can also lead to heart attack and heart failure, and the $PM_{2.5}$-related cardiovascular issues are the fifth leading risk factor for global deaths (Cohen *et al.*, 2017; Nel, 2005). Exposure to PM has been linked to central nervous system diseases including Alzheimer's, Parkinson's, and strokes (Block and Calderón-Garcidueñas, 2009). Fetuses and children are more susceptible to environmental pollutants than adults, and prenatal or childhood exposure to high levels of $PM_{2.5}$ may lead to premature birth, postnatal respiratory problems, and morbidity during early childhood (Isiugo *et al.*, 2019; Jedrychowski *et al.*, 2013). The World Health Organization (WHO) has defined the recommended indoor PM limits (e.g., the daily limit of 45 and 15 μg m^{-3}, and the annual limit of 15 and 5 μg m^{-3} for PM_{10} and $PM_{2.5}$, respectively), applicable to both

indoor and outdoor air (WHO, 2021). This guidelines However, currently, the United States Environmental Protection Agency (U.S. EPA) does not have any quantitative regulations exclusively applied to *indoor* air pollutants (including PM).

Most of the instruments that are used for the detection of PM size distributions use hazardous chemicals and radiation sources. For example, Scanning Mobility Particle Sizers (SMPS), a particle sizing instrument most commonly used for urban air measurements (Lee and Allen, 2012), utilizes radiation sources, such as ^{210}Po or ^{85}Kr, typically at the activity level of 10 mCi (e.g., https://tsi.com/products/aerosol-neutralizers/aerosol-neutralizer-3077a/). This is many orders of magnitude higher than the exempted radiation source activity levels in the United States. And the measurements with these instruments require a radiation source license from the respective state agencies (U.S. NRC, 2011). Due to these regulations, special shipping is required for transportation of these sources within United States. And in some states (e.g., Alabama), it is not feasible or practical to make residential measurements with these instruments. Thus, these invasive instruments severally limit the scope, location, and length of the measurements.

Recently, various low-cost, portable air monitoring sensors have been used for indoor air quality measurements (Bi *et al.*, 2021; Borrego *et al.*, 2016, 2018; Jodeh *et al.*, 2018; Krause *et al.*, 2019; Kumar *et al.*, 2015; Massey *et al.*, 2012; Rai *et al.*, 2017; Ścibor *et al.*, 2019; Wang *et al.*, 2020). These miniature-sized sensors can be used under unattained conditions for an extended period, and they produce little noises; thus, they are ideal for indoor measurements at the extensive temporal and spatial scales. Singer and Delp (2018) conducted inter-comparison of different portable indoor air quality sensors case studies of indoor activities in laboratory environments.

Indoor PM is emitted via a multitude of complex sources, such as infiltration from ambient air and various human activities (e.g., cleaning, cooking, smoking, occupancy, and the use of electrical appliances) (Abdullahi *et al.*, 2013; Morawska *et al.*, 2013; Zhang *et al.*, 2021). Indoor air studies have been conducted mostly either in a laboratory setting or in a test-home to simulate a real household environment. But there has been a very limited number of long-term, continuous measurements of household air pollutants in real living conditions (Geng *et al.*, 2019; He *et al.*, 2004; Jodeh *et al.*, 2018; Krause *et al.*, 2019; Li *et al.*, 2016; Wallace, 2006). For example, a study reported particle size distribution measured continuously over 3 years in a sub-urban single home in Virginia, and showed that cooking is the major source of indoor ultrafine particles (Wallace, 2006).

The recent HOMEChem (House Observations of Microbial and Environmental Chemistry) study (Farmer *et al.*, 2019; Patel *et al.*, 2020), which represents one of the most comprehensive indoor experiments so far, was carried out for a 4-week period in June 2018 in a test-house in Texas, where volunteers performed simulated residential activities (e.g., cooking and house cleaning). And indoor PM concentrations in a large size range in the size range from 1 nm to 20 μm were measured with a series of state-of-the-art particle instruments, including SMPS, Aerodynamic Particle Sizers, and Nano Condensation Nucleus Counter (Patel *et al.*, 2020).

To understand the effects of human activities on the indoor PM level, we have conducted long-term measurements of PM in a real residential home with living residents for approximately a year using Portable the Aerosol Spectrometer Dust Decoder (PASDD). PASDD measures PM size distributions without any chemicals, pumps, or radiation sources, and thus does not disrupt the daily life of the residences. This miniature PM sensor has been used in outdoor and indoor PM monitoring by other studies (Jodeh *et al.*, 2018; Massey *et al.*, 2012). To our knowledge, our study represents the first long-term measurements of indoor PM in a residential home in the southeastern US, especially including the strictest lockdown period during the COVID-19 pandemic. Additionally, we have conducted Thanksgiving holiday cooking experiments in another residential home within the same region, and compare with the HOMEChem experiments (Patel *et al.*, 2020), as holiday gatherings can provide a unique opportunity to study indoor air under conditions with enhanced indoor activities with the increased number of occupants and extended cooking.

2 METHODS

Two residential homes in Northeastern Alabama were used in the present study, and we denote them as Residences A and B, respectively. The exterior photos and the main floor plans of the two homes are shown in Figs. S1 and S2. Both homes had only non-smoking residents.

Table 1. Summary of the average PM mass concentrations with one standard deviation during different daily activities. The size range of each PM mass concentration shown here also applies to all tables and figures showing the results from this study.

Residence Activity	Number of Events	Average Concentration (μg m^{-3})				Peak Concentration (μg m^{-3})			
PM Size Range		$PM_{1.0}$ (0.25–1 μm)	$PM_{2.5}$ (0.25–2.5 μm)	PM_{10} (0.25–10 μm)	TSP (0.25–35 μm)	$PM_{1.0}$ (0.25–1 μm)	$PM_{2.5}$ (0.25–2.5 μm)	PM_{10} (0.25–10 μm)	TSP (0.25–35 μm)
Overall Cooking	193	6.5 ± 7.2	8.0 ± 8.2	14.6 ± 13.2	63.1 ± 1481.9	7.1 ± 9.6	9.7 ± 11.9	33.5 ± 29.6	1223.1 ± 12627.8
Pan Searing/Frying	32	7.0 ± 4.3	8.7 ± 5.4	15.3 ± 10.6	49.2 ± 78.9	8.5 ± 7.0	11.6 ± 8.7	36.0 ± 20.2	380.9 ± 291.9
Simmering	9	5.5 ± 3.4	6.7 ± 4.1	12.7 ± 9.8	44.0 ± 69.1	5.7 ± 4.1	8.8 ± 6.6	36.9 ± 13.4	356.8 ± 171.3
Toasting	40	4.9 ± 2.2	5.7 ± 2.4	10.1 ± 11.2	237.7 ± 5473.5	5.3 ± 2.7	6.6 ± 3.9	26.4 ± 42.5	4584.5 ± 27603.4
Baking	11	4.1 ± 2.3	5.4 ± 2.9	12.6 ± 10.1	50.4 ± 68.4	4.5 ± 2.5	6.4 ± 3.3	29.3 ± 21.3	226.0 ± 133.0
Boiling	15	3.6 ± 1.8	4.4 ± 2.0	10.2 ± 9.1	43.1 ± 80.4	3.9 ± 1.7	5.3 ± 1.9	25.5 ± 20.3	200.5 ± 183.5
Reheating	10	4.0 ± 1.9	5.7 ± 3.0	17.6 ± 14.4	57.3 ± 69.7	3.4 ± 1.5	5.4 ± 2.9	33.0 ± 22.1	269.5 ± 137.6
Burning/Over-cooking	3	18.9 ± 21.9	21.3 ± 24.6	26.2 ± 27.2	55.3 ± 74.7	48.0 ± 58.5	56.1 ± 67.0	86.1 ± 66.2	586.2 ± 121.8
Multiple-Method Cooking	68	7.4 ± 9.7	9.0 ± 11.0	15.3 ± 15.2	47.5 ± 75.8	7.6 ± 7.2	10.6 ± 10.0	34.7 ± 25.7	339.8 ± 280.2
Cleaning	5	4.2 ± 1.7	9.9 ± 8.2	61.3 ± 95.1	405.0 ± 1145.8	8.9 ± 2.9	60.7 ± 27.2	807.48 ± 180.08	11311.7 ± 4509.6

Continuous measurements were made in Residence A from May 2019 to May 2020 except for December 2019 to January 2020 (305 days in total, including the first two months of the hard lockdown during the COVID-19 pandemic). Residence A was a ten-year-old, three-story home located in North Alabama with five occupants. This residence was surrounded by trees and about 8.0 km away from the nearest major highway. This residence had a gas range stove, multiple ovens, and a toaster oven accompanied by an efficient ventilation system that pulls cooking fumes out of the house. The primary kitchen was cordoned off from the rest of the floor, and the PASDD instrument was placed approximately 3 m away from the stove. As different rooms may have different PM concentrations depending on the proximity to the emission sources, we placed the sensor in the kitchen to be close to the most frequently occurring source (cooking) in residential homes. The HVAC system had an age of less than a year and the filters were changed monthly. The HVAC fan was continuously on during the time of the study, and the room temperature was kept between 21–24°C. The house cleaning supplies included a vacuum cleaner and household cleaning products, all commercially available from common warehouses in the United States. Table 1 shows a summary of the main home activities, which will be discussed in detail in the following sections.

Additionally, we also conducted Thanksgiving cooking experiments in Residence B (Fig. S1), where continuous measurements of PM were made for 78 days (from the end of November 2019 through January 2020). Table S1 specifies the activities that took place during the Thanksgiving cooking. Residence B was a one-story, one-bedroom apartment home located on the first level of a three-story apartment building with two occupants and a dog. This home was located at an urban site in North Alabama, approximately 0.8 km from the nearest major highway. There was an electric range stove, a microwave oven, and a recirculating ventilation system recirculating the air to the ceiling. The HVAC system was original to the building with an age of 6 years, and the filters were routinely changed every 6 months. The HVAC fan was continuously on during the time of the study, and the room temperature was maintained at 21°C. The instrument was placed approximately 1 m from the stove.

In both residences, cooking took place

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always with overhead kitchen ventilation (used in an active mode). The residents in both homes, acting as citizen scientists, recorded their daily activities (e.g., cooking, cleaning, changing the indoor temperature, and exiting the residence). The log information also included the activity duration, how many people were in the room during the activity, if any windows were open during the activity, and the list of appliances that were running at the time of the activity. No human subjects or substances were collected or used during the study; this study was approved for exemption by the Institutional Review Boards (IRB).

A PASDD (Model 11-D, Grimm Technologies Inc., Netherland) was used to measure particle size distributions between 0.25–35.15 μm within 31 size bins. This sensor is based on light-scattering of single particles with diode laser. The time resolution of the instrument is 1 s, which allows us to monitor the rapid evolution of PM in an indoor environment. For particle mass concentrations, we assumed particles to be spherical with a density of 1.0 g cm^{-3}. We present the PM mass concentrations in four different ranges: $PM_{1.0}$ (0.25–1.0 μm), $PM_{2.5}$ (0.25–2.5 μm), PM_{10} (0.25–10 μm), and Total Suspended Particles (TSP, 0.25–35.15 μm). The instrument was calibrated at the manufacturer right before the measurements began. Previous studies have shown that the performance of PM sensors can be affected by high humidity conditions (Jayaratne *et al.*, 2018); but as stated earlier, the two homes both used HVAC system for 24/7, so we expect the effects of humidity and temperature fluctuations were minimal.

The PASDD sensor was calibrated at the manufacture prior to our study. Calibration was made using polydisperse dolomite dust to adjust the attenuation thresholds on each size channel of the instrument for mass concentrations. However, we have not done any parallel measurements together with a gravimetric instrument at residential homes. This is the main limitation of measurements, along with limited size range (starting from 0.24 micron) in the PASDD instrument. However, this sensor has a significant advantage over filter-based gravimetric instruments. The PASDD measures particle concentrations at real-time with fast time response (1 Hz) and thus is free from sampling artifacts, whereas in gravimetric measurements, particles undergo chemical evolution (e.g., evaporation, condensation and aging) on the filter during the long sampling period (often over hours).

3 RESULTS AND DISCUSSION

3.1 Background PM Level

The *background* condition was defined as a period when the residence was absence of occupants for longer than 6 hours (without any occurrence of the elevated PM level). The average background $PM_{2.5}$ mass concentrations of Residents A and B were 4.7 and 1.7 μg m^{-3}, respectively (Table 2). The lower background level in Residence B was likely because it was a newer house and had only two occupancies. In comparison, the background environment during HOMEChem's study was 2.3 μg m^{-3} (Patel *et al.*, 2020), perhaps because no residents lived in the HOMEChem *test-home*.

Table 2. Background PM Mass concentrations in the two studied households in μg m^{-3}. In comparison, HOMEChem (Patel *et al.*, 2020) are also shown. Note the range of particles sizes in different studies. Note different size ranges between this study and HOMEChem. As discussed in the text, due to various factors (house volume, nature of the house, and usage of kitchen ventilation), this table here only illustrates a general trend of high concentration of indoor PM during a holiday cooking, and does not provide a "quantitative" comparison.

PM Size Range This Study	Residence A This Study	Residence B This Study	PM Size Range HOMEChem	HOMEChem
$PM_{1.0}$ 0.25–1 μm	4.2 ± 1.2	1.7 ± 0.6	$PM_{1.0}$ 4 nm to 1 μm	1.5 ± 0.3
$PM_{2.5}$ 0.25–2.5 μm	4.7 ± 1.6	1.7 ± 0.6	$PM_{2.5}$ 4 nm to 2.5 μm	2.3 ± 0.3
PM_{10} 0.25–10 μm	5.6 ± 4.6	1.8 ± 0.8	PM_{10} 4 nm to 10 μm	2.5 ± 0.4
TSP 0.25 to 35.15 μm	9.3 ± 33.3	2.1 ± 2.6	PM_{20} 4 nm to 20 μm	2.5 ± 0.4

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The background concentrations shown here for Residences A and B also indicate that the majority of the particle mass was in the size range of $PM_{1.0}$ in the unattended home environment.

3.2 COVID-19 Pandemic Lockdown

Fig. 1(a) shows the monthly average PM concentrations measured in Residence A. There was a large variability in indoor PM, as expected in real homes, where there are always a large number of random variables that cannot be controlled (much more than in a laboratory or test-home setting). Before the COVID-19 pandemic, the average $PM_{2.5}$ concentrations were 4.3 μg m^{-3} and 3.5 μg m^{-3} in September and October (2019), respectively (Fig. 2(a)). During these months, the school was in session, which allowed the number of occupancy to decrease, and thus fewer activities were performed in the home. During the summer months (June–August 2019), the average $PM_{2.5}$ concentrations were higher on average (4.9 μg m^{-3}), due to the increasing occupancy and activities. In particular, May 2019 had one of the highest TSP values of all of the months (29.5 μg m^{-3}). Ambient air is known to be a main contributor to indoor air (Hänninen *et al.*, 2011), and since Residence A was surrounded by trees, pollen could be a contributor to more coarse-mode particles during this period.

The pandemic measurements demonstrate the effects of human occupancy on the indoor PM pollutant level. In the state of Alabama, the COVID-19 Pandemic stay-at-home order took place in the mid-march in 2020 and the *hard lockdown* period continued until the end of April. During this

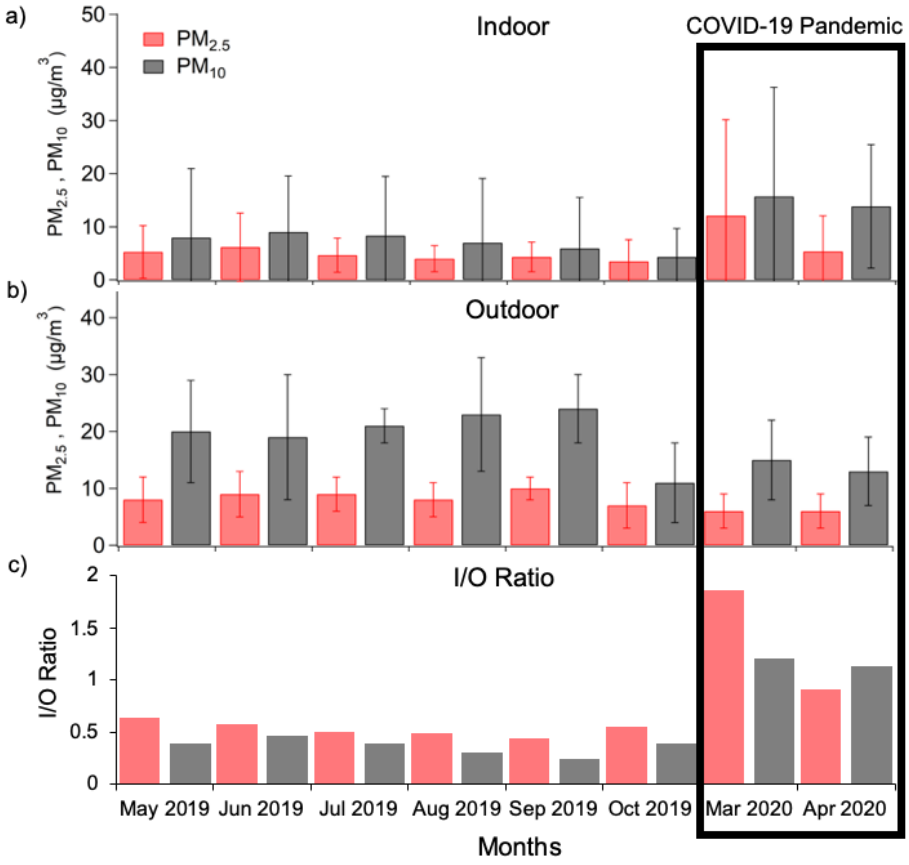

Fig. 1. Monthly average mass concentration of PM in Residence A (a) followed by the corresponding outdoor ambient PM mass concentrations of $PM_{2.5}$ and PM_{10} (b). We included here only the months where indoor PM measurements were made for the entire month. Outdoor measurement data was gathered at the EPA ground station (https://www.epa.gov/outdoor-air-quality-data). (c) Calculated ratios of the indoor PM versus the EPA ambient PM data. Note that since two different measurement methods were used for the indoor (light scattering) and EPA (filter), and as the EPA site was 15 km away from Residence A, this ratio shown here is NOT the same as the "I/O ratio" typically used in indoor air studies.

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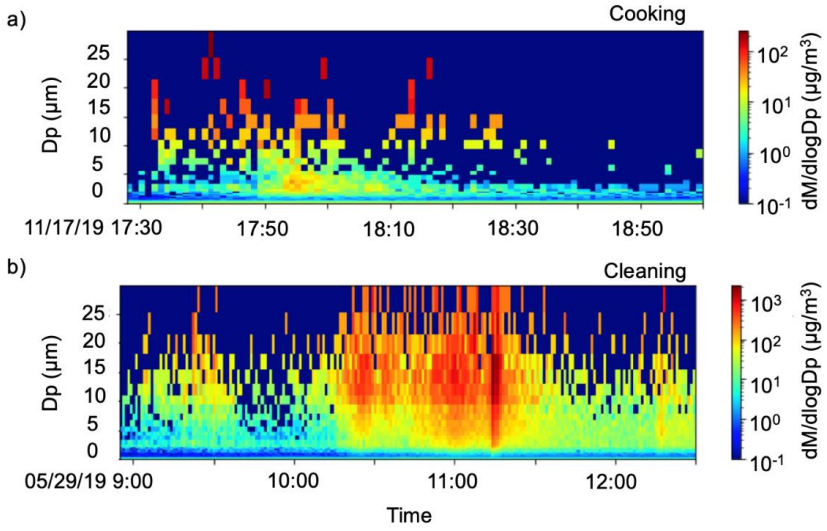

Fig. 2. Particle mass size distribution during (a) a typical cooking event on 17 November 2019 from 5:30 PM until 7:00 PM and (b) cleaning event on 29 May 2019 from 9:00 AM until 12:30 PM. Note, shown here are the measured particle mass (as opposed to number) size distributions.

period (March–April, 2020), $PM_{2.5}$ and PM_{10} increased by 188% and 209%, respectively, compared to the all months prior to the pandemic (Fig. 2(a)). The sustained high level of indoor PM during the lockdown was probably due to increased occupancy, home-cooking and extensive cleaning activities (at the unprecedented level). The ambient $PM_{2.5}$ and PM_{10} concentrations, retrieved from a nearby EPA monitoring station (https://www.epa.gov/outdoor-air-quality-data) are included in Fig. 2 for a comparison. The EPA AQS Air Quality System (AQS) ambient data were obtained with the filter-based PM mass detector at Huntsville International Airport (about 15 km away from Residence A). As ambient PM concentrations may vary across the urban topography and due to various factors such emission sources, in this study, we did not aim to analyze effects of outdoor PM on indoor PM concentrations. Rather, by comparing our indoor measurements with the EPA ambient data, we wanted to look at the *general* trends of ambient and indoor PM within the same area, specifically, in the context of the COVID-19 Pandemic lockdown. In contrast to the indoor PM monthly trends, ambient PM concentrations were decreased by 50% during the pandemic, as traffic and industrial activities were significantly reduced. The overall effect is changes in the "proxy" ratio of the indoor to outdoor PM concentrations (I/O) (Fig. 1(c)). Before the pandemic, the "proxy" I/O ratio was between 0.3–0.5, but during the COVID-19 Pandemic, it became close to or even much higher than 1 (0.9–1.8).

3.3 Cleaning vs. Cooking

There is a large volume of the literature that shows cooking and cleaning are the two everyday household activities that mostly affect indoor air quality (e.g., Balasubramanian and Lee, 2007; Wallace, 2006; Wallace *et al.*, 2008; Wan *et al.*, 2011), and our observations support this universal conclusion. Overall, cleaning dominated larger particles, with an average PM_{10} and TSP value of 61.3 and 405.0 μg m^{-3}, respectively, compared to cooking, which had an average PM_{10} and TSP value of 14.6 and 63.1 μg m^{-3} (Table 1). Cooking, however, had higher amounts of smaller particles (0.2 μm to 5 μm), consistent with previous studies (e.g., Farmer *et al.*, 2019; Kang *et al.*, 2019; Patel *et al.*, 2020; Wallace *et al.*, 2008). This feature is also illustrated in the measured aerosol size distributions (Fig. 3). Very importantly, these aerosol mass size distributions show that even with its limited size range (0.25–35 μm), this inexpensive portable sensor still can capture the majority of *mass* concentrations from these events, thus useful for indoor PM monitoring.

3.4 Cooking Methods

Numerous studies have shown that cooking methods directly affect the indoor PM level (e.g., Abdullahi *et al.*, 2013; Kang *et al.*, 2019; Wallace, 2006; Zhang *et al.*, 2010). In this study, we

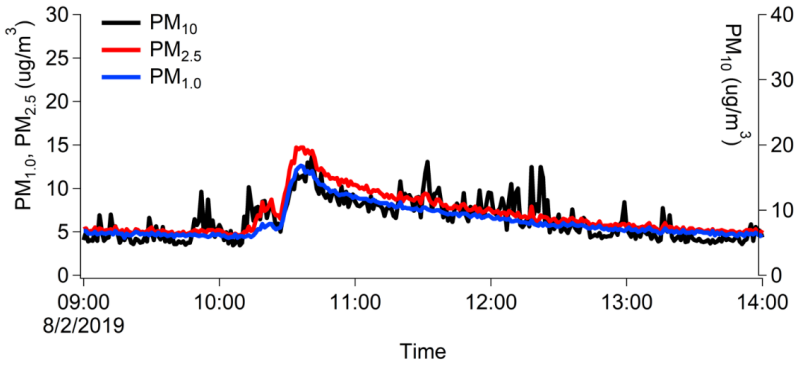

Fig. 3. Time series of PM concentrations during a typical cooking event in Residence A on August 2, 2019.

categorized cooking methods into eight different groups: Pan Searing/Frying, Simmering, Toasting, Baking, Boiling, Reheating, Burning/Overcooking, and Multiple Methods (where food was made using different methods simultaneously). Table 1 summarizes the average PM concentrations in different cooking methods. Residence A reheated their food on their gas range stove, which explains the increased levels of $PM_{2.5}$ and PM_{10} associated with Reheating cooking activities at 5.7 and 17.6 μg m^{-3}, respectively. Multiple Methods and Burning/Overcooking were the methods that had high emissions of $PM_{2.5}$ (9.0 μg m^{-3} for Multiple Methods and 21.3 μg m^{-3} for Overcooking). These methods also resulted in larger particles, with average PM_{10} values of 15.2 μg m^{-3} for Multiple Methods and 26.2 μg m^{-3} for Burning/Overcooking. Simmering also had a considerable amount of $PM_{2.5}$ (6.7 μg m^{-3}) in comparison to Boiling (4.4 μg m^{-3}). This could be the result of longer cooking times, since an individual will simmer a dish longer than boiling water. The cooking duration was an important variable when considering the emissions by cooking indoors, as cooking time increases, the emitted PM increases. The occupants typically spent a longer amount of time simmering a dish (142 minutes in average) compared to frying a meal (28 minutes on average). Frying emitted a considerable amount of indoor $PM_{2.5}$ with an average of 8.7 μg m^{-3} over the events studied, which is comparable to past studies (Abdullahi *et al.*, 2013; Lee *et al.*, 2001; Massey *et al.*, 2012; Zhang *et al.*, 2010). Methods that are cleaner to perform were Boiling and Baking, with a combined average $PM_{2.5}$ mass concentrations 2 and 1.63 times lower than Multiple Methods and Pan Searing/Frying, respectively. Boiling has been mentioned as a clean method of cooking (Abdullahi *et al.*, 2013; Lee *et al.*, 2001; Zhang *et al.*, 2010); however, in this study, baking produced the indoor $PM_{2.5}$ 38.1% less than frying, likely due to the closed system of the oven.

3.5 Evolution of Indoor PM

Fig. 3 represents a typical time variation of the indoor PM concentrations during a cooking event. When the PM was emitted from a single cooking (or cleaning) event, there was a rapid linear increase, then followed by a near-exponential decay, as also previously shown by (Xia *et al.*, 2009). To characterize such time evolution of the indoor PM, we use the rising rate (R), pseudo-first order decay rate constant (k_D), and the decay time (τ), at which PM concentrations become 1/e of the initial peak concentrations. The rising rate was estimated from the approximation of a linear function (1):

$$R = \frac{V_{max} - V_0}{t_{max} - t_0} \quad (1)$$

where V_{max} and V_0 is the peak and initial value of PM during the activity, respectively; t_{max} is the time where the peak value occurred during the activity; and t_0 is the time when the experiment started according to the activity logbook. The pseudo-first order decay rate constant was derived from the following equation:

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$$k_D = \frac{\ln\left(\frac{V_{max}}{V_0}\right)}{t_{max} - t_{final}} \quad (2)$$

where t_{final} is the time at which the PM then decreased back to the background level (which is approximately the same as V_0). And the decay time of PM was calculated as the following:

$$\tau = \frac{1}{k_D} \quad (3)$$

Overall, the average R and k_D did not vary much throughout the day (Fig. 4). The R ranged about tens of μg m^{-3} hr^{-1} for $PM_{1.0}$ and $PM_{2.5}$, hundreds of μg m^{-3} hr^{-1} for PM_{10}, and thousands of μg m^{-3} hr^{-1} for TSP, regardless of the time of the day. On the other hand, the k_D values showed strong size dependencies, with larger k_D for larger particles, indicating the main decay process of the indoor PM is deposition on home interior surfaces. Likewise, there were the longer decay times for $PM_{1.0}$ and $PM_{2.5}$, in comparison to those for PM_{10} and TSP for both cooking and cleaning events (Table 3). During cleaning, $PM_{1.0}$ and $PM_{2.5}$ had an average τ of 68 and 23 minutes, respectively, in comparison to the PM_{10} and TSP's τ of 16 and 9 minutes, respectively. In terms of different cooking methods, during Pan searing/Frying, the average τ of $PM_{1.0}$ and $PM_{2.5}$ were 79.2 and 59.8 minutes, and the τ of PM_{10} and TSP were 24.5 and 11.2 minutes, respectively, again strongly dependent on the particle size.

3.6 Thanksgiving Holiday Cooking

Thanksgiving is one of the most popular (non-religious) American holidays, where large gatherings of people come together and cook copious amounts of food, utilizing multiple cooking methods over long hours. The holiday cooking experiment was conducted (on 24 November 2019) in Residence B; there were 8 individuals. Human occupancy is known to affect indoor air quality (Avery *et al.*, 2019; Hospodsky *et al.*, 2012). The guests started to arrive at approximately 9:00 AM, and cooking started around 11:00 AM (Fig. 5). Before the cooking began, the occupancy increased by

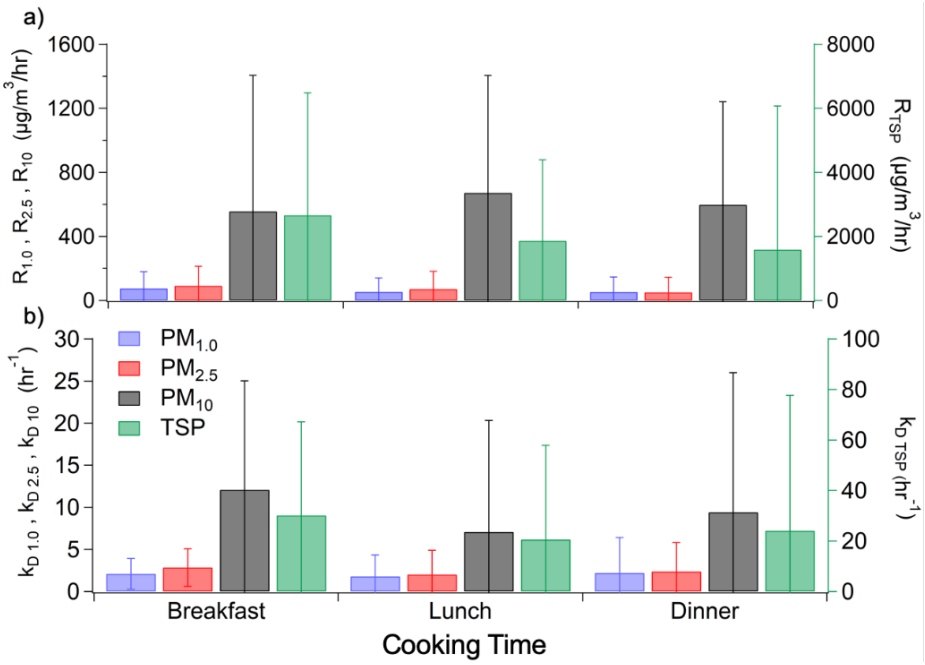

Fig. 4. (a) Average rise rate (R) and (b) decay rate (k_D) for breakfast (N = 53), lunch (N = 44), and dinner (N = 86). Note that the right axis is for $PM_{1.0}$ ($R_{1.0}$, $k_{D\,1.0}$), $PM_{2.5}$ ($R_{2.5}$, $k_{D\,2.5}$), and PM_{10} (R_{10}, $k_{D\,10}$), whereas the right axis is for TSP rise and decay rates (R_{TSP}, $k_{D\,TSP}$).

Table 3. Average decay times of indoor PM during cooking with different methods in Residence A.

PM Size	Pan Searing /Frying	Simmering	Toasting	Baking	Boiling	Reheating	Burned/ Overcooked	Multiple Methods	Cooking Overall	Cleaning
Number of Events	32	9	40	11	15	10	3	68	193	5
$PM_{1.0}$	103.3 ± 116.9	122.5 ± 121.7	63.1 ± 67.0	80.2 ± 28.4	94.9 ± 82.4	88.8 ± 99.9	143.5 ± 61.1	109.8 ± 108.8	99.8 ± 110.5	98.2 ± 35.8
$PM_{2.5}$	84.3 ± 117.3	95.5 ± 84.0	47.5 ± 54.3	38.6 ± 22.2	45.8 ± 42.9	63.3 ± 82.0	135.1 ± 50.6	62.8 ± 56.7	63.1 ± 71.7	33.7 ± 6.8
PM_{10}	35.0 ± 44.3	28.0 ± 35.0	11.9 ± 11.7	12.60 ± 12.9	9.7 ± 6.9	19.0 ± 19.1	55.9 ± 42.8	25.4 ± 27.9	22.2 ± 28.5	23.5 ± 15.9
TSP	15.9 ± 17.9	16.8 ± 17.7	4.9 ± 5.1	4.9 ± 5.1	4.9 ± 4.0	9.7 ± 11.4	40.9 ± 35.8	11.0 ± 13.5	10.3 ± 13.7	12.8 ± 4.2

5 individuals. During the time, the $PM_{2.5}$ increased from an average of 3.0 μg m^{-3} during 7:00–9:00 AM to 5.3 μg m^{-3} during 9:00-11:00 AM, and the PM_{10} increased from 12.2 to 19.8 μg m^{-3} the same time. There were also peaks of PM_{10} that occurred before the cooking began, when occupancy increased (Fig. 6(a)). It is possible that as the occupancy increased before the cooking began, dust resuspension and personal clouds may have increased the PM_{10} concentration. Thus, our holiday experiments also demonstrate that the increased PM due to occupancy. This feature is consistent with the increase in the indoor PM during the COVID-19 lockdown months (Fig. 2), as discussed above.

Once the cooking began, the TSP level increased from 1.6 μg m^{-3} to 1,283 μg m^{-3}, with the increasing number of appliances over 8 hours (Fig. 6(a)). During the peak cooking period (from 14:00 PM to around 15:00 PM), an oven, two slow-cookers, and a pressure cooker were operated simultaneously, and the $PM_{2.5}$ reached up to 38.8 μg m^{-3}. The overall cooking during this experiment ended at approximately 15:15 PM, and the elevated PM level lasted until about 22:00 PM (over ~6 hours). Fig. 6 also shows the comparison of our study with the HOMEChem Thanksgiving experiments. Here, the data within the same size ranges are shown for both studies: $PM_{1.0}$ (0.253–1 μm), $PM_{2.5}$ (0.253–2.5 μm), and PM_{10} (0.253–10 μm). The average concentrations of $PM_{1.0}$, $PM_{2.5}$, PM_{10} in our study were higher by a factor of 1.3, 1.4, and 2, respectively, compared to HOMEChem (Patel *et al.*, 2020). Note that the HOMEChem cooking experiments were made without kitchen ventilation. As smaller particles are diffused more easily than larger particles, the lack of ventilation may explain the high concentrations of nuclei-more particles reported from the HOMEChem study (Patel *et al.*, 2020).

To quantify the exposure of PM during the Thanksgiving cooking period, we estimated the Integrated Exposure (*IE*) based on (Morawska *et al.*, 2013):

$$IE(t_1,t_2)=\int_{t_1}^{t_2} c(t)dt \tag{4}$$

where t_1 and t_2 are start and end times of the exposure, respectively; $C(t)$ is the concentration function in hours and μg m^{-3}. The *IE* estimated during our Thanksgiving cooking was 118, 168, 312, and 1,010 μg m^{-3} hr^{-1} for $PM_{1.0}$, $PM_{2.5}$, PM_{10}, and TSP, respectively. During the peak cooking period, the 2-hour average rose to higher than 25 μg m^{-3}.

4 CONCLUSIONS AND IMPLICATIONS

This study represents the first long-term measurement of indoor PM in authentic households in the southeastern United States. While our study confirms that cooking and cleaning are two major emission sources of indoor PM, our study also shows human occupancy has a strong effect on the indoor PM level. During the COVID-19 pandemic with increased human occupancy, the average mass concentrations of indoor PM increased by 200%, while the ambient concentrations decreased by 50% due to decreased traffic and industrial activities. These results are consistent with the

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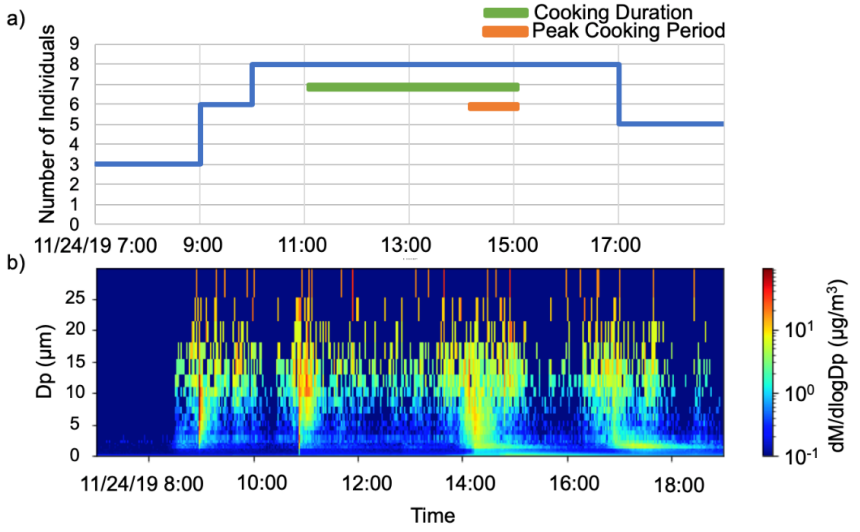

Fig. 5. (a) The number of occupancies and (b) the measured mass size distribution during the Thanksgiving experiment in Residence B on 24 November 2019.

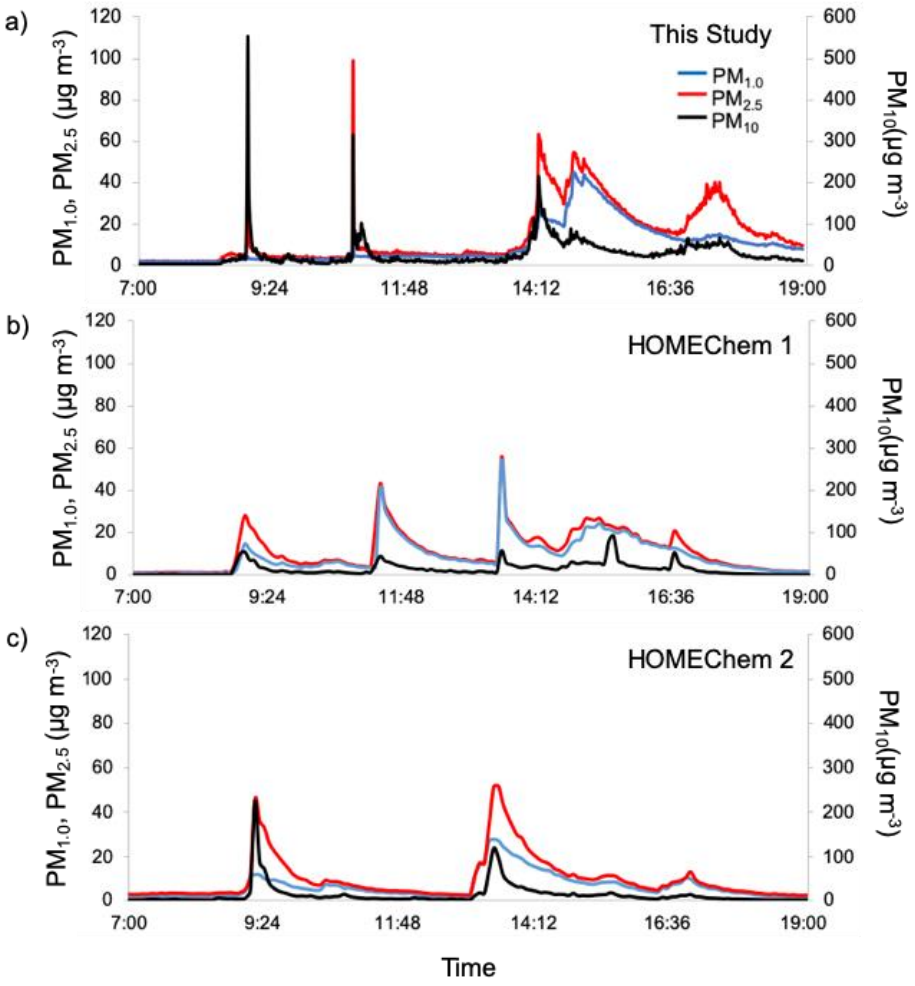

Fig. 6. (a) Comparison of mass concentrations measured during the Thanksgiving cooking experiments; in comparison, the results from the HOMEChem's two Thanksgiving experiments are including in (b) and (c). For a direct comparison, data with the same ranges are shown here in these three figures: $PM_{1.0}$ (0.253–1 μm), $PM_{2.5}$ (0.253–2.5 μm) and PM_{10} (0.253–10 μm). Note that HOMEChem cooking experiments were performed without kitchen ventilation in a test-home.

opposite trends between PM in indoor and outdoor environments observed during the pandemic lockdown at other locations (e.g., Domínguez-Amarillo *et al.*, 2020; Du *et al.*, 2021; Zhang *et al.*, 2022).

Thanksgiving holiday experiments further reinforced the effects of human occupancies on the indoor PM level. As occupancy increased, the $PM_{2.5}$ and PM_{10} increased from 3.0 to 5.3 μg m^{-3} and from 12.2 to 19.8 μg m^{-3}, respectively (before the cooking began). During the holiday cooking, even with kitchen ventilation. There was a high level of $PM_{2.5}$ of ~20 μg m^{-3} an extended period (e.g., over 6 hours). The comparison with HOMEChem study illustrates a general trend of the elevated PM emissions from intense cooking events and large gatherings, as Thanksgiving holiday provided a unique combination of these two factors. However, there are several other factors that hinder direct comparison between the two studies. Mostly, the volumes of the houses were different between Residence A in our study vs. the HOMEChem test-house; additionally, the HOMEChem did not use a kitchen ventilation whereas Residence A used the kitchen ventilation at all times during cooking. Even in the same residential home, with the same occupancies, depending on different activities, indoor PM concentrations may vary significantly due to many spontaneous factors, hourly and daily. These features thus represents an inherent challenge in indoor studies, where a direct "quantitative" comparison is difficult to make between different homes. While holiday gathering, in general, implies a substantial increase in home occupancies, at the same time, a quantitative interpretation will be dependent on the context, such as different cultures, countries, and geographical regions with vastly different living spaces.

Cooking emits particles ranging from 0.25–5 μm, while cleaning generated more particles ranging from 5–30 μm due to dust agitation. The PM level during the cooking events was mainly driven by the method (rather the time of the day). Cooking methods that were oil-heavy or involve multiple cooking styles emitted more indoor PM. Cleaner methods shown in this study were Boiling (due to water-based cooking) and Baking (due to the closed system of the oven). Regardless of the appliance type, cooking played a major role in the emission of PM within a household. The decay time of the PM decay from household activities (cooking and cleaning) ranged from < 10 to about 100 minutes, clearly dependent on the particle size, indicating the main sink process of the indoor air PM is deposition onto the interior surfaces.

Additionally, our study shows that the portable miniature PM sensor with its limited size range can still detect the majority of mass concentrations of residential indoor PM are measured, thus validating the usages of this type of portable sensors for indoor air quality measurements within larger temporal and spatial scales worldwide, cost-effectively.

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SUPPLEMENTARY MATERIAL

Supplementary material for this article can be found in the online version at https://doi.org/10.4209/aaqr.210302

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