



Research article

The impacts of cooking and indoor air quality assessment in the southwestern region of Bangladesh

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ABSTRACT

The main objective of this study is to assess the impacts of cooking and indoor air quality (IAQ) in the southwestern region of Bangladesh. Here we report and compare the IAQ in considering a total of eight kitchens and living rooms of four selected households (HHs) in Jashore city and suburb area, the southwest district of Bangladesh. Air quality parameters, such as particulate matter (PM_{2.5}) and volatile organic compounds (VOC), were assessed continuously for 24 h. In addition, Carbon dioxide (CO₂) was evaluated in different phases during the study period. PM_{2.5}, VOC, and CO₂ levels were ranged from 18.52 to 207 µg/m³, 7.95–35.66 ppm, and 1061–2459 mg/m³, respectively, in the indoor cooking HHs. Conversely, while the average concentration was found between 20.63 and 23.72 µg/m³ PM_{2.5}, 11.18–12.36 ppm VOC, and 1097–1747 mg/m³ CO₂ in the outdoor cooking HHs. A significant increase in CO₂ due to kitchen activities (cooking, frying, boiling) was observed that ranged between 5 and 77% compared to the background level. The calculated range of toxicity potential (TP) values was between 0.8 and 8.3 for PM_{2.5} in the HHs. In most of the observations, PM_{2.5}, VOC, and CO₂ exceed the standard values. The study reports that well ventilation systems and clean fuel use significantly reduce the indoor air contaminants level. Our study offers new insights about the IAQ of the southwest region of Bangladesh, particularly for suburbs and urban setups, and provides a background for further study, and decision-making. It will serve as a reference for the formulation and implementation of policies to improve air quality.

1. Introduction

In recent years, Indoor air quality (IAQ) has become one of the most crucial issues that is directly related to the occupant's health and well-being. The IAQ is influenced by the air quality (AQ) within and around a building or structure [1]. Another significant concern regarding the issue is that IAQ is mostly damaged by indoor cooking (IC), which generates airborne contaminants that have

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significant health-related issues [2]. Numerous studies indicate that IC activities can significantly increase particulate matters (PMs), volatile organic compounds (VOCs), carbon dioxide (CO₂) and carbon monoxide (CO) [3,4], as well as produce intense oily-fumes and a large number of gaseous constituents [5]. Mabonga et al. [6] reported severe health issues (e.g., coughing, eye-irritation, respiratory problems) from PM_{2.5} in houses that use solid biomass fuel during cooking. Carcinogenic health risk potential belongs to different types of fries that produce VOC during the IC event. Zhang et al. [7] reported different cooking oils' VOC increment capacity, along with the risk of carcinogenic health impairment. Moreover, CO₂ and CO emissions are directly associated with the IC and biomass fuel burning reported in many studies [8-13], which have potential health risks in the indoor environment. Cheung et al. [13] reported that CO₂ and CO concentrations get higher than the before cooking (BC) (background) on the during cooking (DC) periods due to IC activity. Using fossil fuels in industrial and agricultural sectors causes CO₂ emissions, ultimately negatively impacting humans, crops, livestock, and the climate [14-16]. Therefore, adopting renewable energy sources is necessary to reduce the consequences [14]. Moreover, fine and ultra-fine particulate matter have a negative impact on the cardiovascular system and have been reported by epidemiological researchers [17-20]. More than 5.5 million people die each year as a result of respiratory diseases caused by breathing polluted air [21]. Peng et al. [20] demonstrated the potential risk of morbidity problems such as asthma, lung cancer, birth defects, and premature death from frequent exposures. The main concern arises from the view that humans spend 60–90% of their total life span staying in the indoor environment [3,22,23], and thus indoor pollution may continuously impair their health for over a lifetime.

In Bangladesh, the impact of IC on IAQ was reported in several studies: PMs determination from the cooking room and living room in rural areas depending on the fuel use, kitchen configuration, and ventilation [24,25]. Studies mentioned the PM_{2.5} increment due to biomass burning [26], which is associated with potential cardiovascular mortality and respiratory problems [27]. Furthermore, almost every study suggested women and children get more impaired due to indoor air pollution (IAP) since they spend a significant amount of time in contact with cooking areas [24,28–30]. Therefore, alternative cooking practices should be enhanced. As an alternative to solid biomass fuel-based cooking, various types of cleaner fuel (electricity, biogas, natural gas, etc.) could be used [31]. It is to be mentioned that most of the studies were conducted in the capital city of Bangladesh. However, the impact of IC in the southwestern part of Bangladesh is not well reported. Dasgupta et al. [28] reported only the PM₁₀ level for poor families in the Jashore region based on a variety of cooking locations, construction materials, ventilation practices, and the fuel used, but no information regarding the PM_{2.5}, VOC, CO₂, CO or other dominant indoor contaminants in the urban or suburban communities. In a separate study, Dey et al. [32] suggested the probable sources and health impacts of IAP in some village areas of the Jashore district by conducting a questionnaire survey. Still, they never mentioned the contamination level or IC impact on IAQ in the urban or suburban areas of the region.

Therefore, our research has been designed to assess the IAQ of the urban and suburban areas of Jashore to observe the cooking impact depending on the configuration of the kitchen, fuel use, building materials, and location of the houses in considering both types of indoor and outdoor cooking households (HHs).

2. Methodology

2.1. Study area

The study was conducted in the urban and suburban parts of Jashore District, Bangladesh (Figure S1). The district encompasses an area of 2606.94 sq. km. of which 23.39 sq. km is riverine [33]. The district is bordered by Magura and Jhenaidah districts to the north, Khulna and Satkhira districts to the south, Narail district to the east, and West Bengal of India to the west. Climatologically, the Jashore district is one of the most extreme districts in Bangladesh. The mean annual rainfall varies between 1305 and 2178 mm, the temperature ranges between 25.5 and 26.5 °C, and the humidity fluctuates between 76% and 77% [34]. The Jashore district hosts the largest land port between India and Bangladesh and thus has become a commercial hotspot. Diverse economic activities, busy traffic (due to the land port activity), and climatological extremes make the district a potential air pollution hotspot. The district is inhabited by over 2.7 million people across eight administrative Upazilas. Among them, Jashore Sadar (435.22 sq. km) is the most densely populated area (1707 population per sq. km.) and has 169,164 households [33]. Doratana is the central part of the Jashore Sadar municipality. The majority of the commercial, traffic, and administrative offices of the district are located in that area. Considering the location of the HHs, we carefully chose a total of four HHs; the living room (L) and kitchen (K) of each HH were selected for the 24-h observation. Two HHs were chosen from the urban area and two from the suburbs of the Jashore Sadar. HH1 and HH2 were selected from the Jhumjhumpur suburb. Between them, HH2 was located adjacent to the edge of the Bhairab river. HH3 and HH4 were taken from the city's commercial areas, Doratana and Monihar, respectively. Among the HHs, HH3 was taken from the low-income groups of urban residents. The HH locations are shown in Figure S1. Sampling was conducted in each HH's living room and kitchen continuously and simultaneously.

2.2. Building selection

The study was conducted in two different types of one-storey buildings (building and kitchen information is available in Figure S2 and Table S1). In the suburbs, HH1 and HH2 were pucca houses (completely brick and concrete built). In the city area, HH3 was semi-pucca (brick-built and tin-shaded), and HH4 was completely pucca. The complete floor plans of all selected HHs are presented in Figure S2, which depicts the kitchen and living room configuration, ventilation system, and sampling points. The HH1 cooking place (K1) was completely open and located within a 10-m distance from the living room (L1). In HH2, the kitchen (K2) did not have any door panels, and the living room (L2) was 8 m away from the cooking place. The HH3 kitchen (K3) and living room (L3) were tin-shaded, and there were no openable windows in the K3. The distance between L3 and K3 was comparatively close (within 1-m). A relatively well-

ventilated system has been observed at the HH4 kitchen (K4) with an openable window and an exhausting fan (1350 RPM and 8-inch diameter) installed in that kitchen. The HH4 living room (L4) was 6-m away from the K4, and the ventilation systems were also comparatively better. The HHs' living rooms were all naturally ventilated, and there were no closed-ventilated buildings selected for the study.

2.3. Information about the cooking events

A traditional mud stove was used at the K1 for cooking, whereas gas stoves were installed at the K2 and K4. Two types of single-pot stoves (mud stove and gas stove) were installed at the K3 and operated for cooking simultaneously. During the field study, biomass fuels were used in the mud stoves and LPG in the gas stoves in all the selected HHs (as mentioned in Table S1). Generally, a cooking event took around 45 min to 1 h in every kitchen, and the exhaust fan was kept running during the cooking time at the K4.

2.4. Data collection

Portable air quality monitors and exchangeable laser particle counter sensor heads (Aeroqual-500 series, New Zealand) with data logger facilities (Table S4) were used to monitor PM_{2.5} from the main sleeping area and the cooking spaces. For the detection of VOC, gas sensitive semiconductor sensor head was adapted in the Aeroqual device. Those two parameters (PM_{2.5} and VOC) were continuously measured from the L1, L2, L3, L4, and K1, K2, K3, and K4 (n = 8) for 24 h. The instrumental setup in the living room and kitchen area was installed at the height of normal breathing level (1.2 m) with an adjustable tripod stand [35]. Additionally, to assess the IC impact on CO₂ enhancement, two phases of observation were conducted: BC (background) and DC. All these phases were taken for 30 min individually in each kitchen (n = 4). During all sampling periods, data were collected at 1 min intervals for each parameter. In addition, it is to be mentioned that no tobacco smoking occurred during the whole assessment period.

2.5. Statistical analysis

Descriptive statistical analysis and graphical visualization were performed using different graphing tools such as ggplot2, corrplot, and gridExtra packages on R Studio (Desktop, V-1.4.1717). Spatial differences in pollutant concentrations were investigated using one-way analysis of variance (one-way ANOVA) in different sampling sites (urban and suburban) L1, L2, L3, L4, K1, K2, K3, and K4 at a statistical significance level of P < 0.001. Pearson product-moment correlation (PPMC) determined the relationship between the IC contamination level increment and the impact on the IAQ at a 95% confidence interval ("CI"; P-value < 0.05). The spatial distribution of sampling locations was pointed out on a map using ArcGIS Desktop-10.5.

3. Results

3.1. Fine particulate matters (PM_{2.5})

The study observed a significant variation between and among the kitchen and living room pollutant concentrations (p < 0.001). The overview of the levels of contaminants is represented in Table 1. At the K1, the cooking started at 10:30 on a mud stove with wood powder. With the introduction of cooking, the PM level started to increase and then got maximum at 11:00 a.m. However, the L1 PM level did not enhance with cooking. Since the location of the OC fireplace is far away from the L1, no walls or roofs were comprised, and the concentration of PM_{2.5} found was comparatively lower than the other HHs. A maximum peak was observed (Fig. 2a) in the L1 the next morning (6:00 a.m.), showing the highest PM_{2.5} concentrations of 220 µg/m³. This sharp increase in PM_{2.5} occurred due to sweeping the floor early morning. The living room activity (such as furniture cleaning and sweeping) during the evening (20:00–23:00) resulted in a small to moderate peak of PM_{2.5}. Otherwise, a more or less stable PM_{2.5} level was observed during the day and night. Moreover, the L1 PM_{2.5} concentration is also similarly lower (Fig. 1a) than in the other living areas. Hence, the cooking impact might not be solely responsible for impairing the PM_{2.5} level of that residence.

The PM_{2.5} concentration in HH2 was unstable throughout the 24-h monitoring period (Fig. 2b). At the L2, a gradual increase of

Table 1

Overview of the contamination level in indoor.

Indoor Area	PM _{2.5} (µg/m ³)			VOC (ppm)		
	Mean ± Std.	Median [IQR]	Range	Mean ± Std.	Median [IQR]	Range
L1*	20.63 ± 5.72	21 [16–24]	10–36	11.18 ± 3.05	12 [8–14]	6–16
L2	145.2 ± 51.91	145 [122–161]	63–226	29.54 ± 7.41	28 [24–34]	17–49
L3	207.3 ± 121.07	194 [109–271]	70–532	7.95 ± 1.46	7 [7–9]	6–12
L4**	18.52 ± 6.62	17 [14–22]	9–40	11.63 ± 3.11	12 [10–14]	5–20
K1*	23.72 ± 7	24 [18–29]	12–44	12.36 ± 2.80	12 [11–14]	7–18
K2	182 ± 40.43	153 [132–237]	70–386	35.66 ± 13	32 [25–42]	19–66
K3	106.7 ± 28.21	109 [80–126]	54–195	9.90 ± 2.53	9 [8–12]	6–18
K4**	51.62 ± 20.74	49 [41–61]	14–98	11.66 ± 3.18	12 [9–15]	7–19

L = living room, n = 4; K = Kitchen, n = 4; IQR = Interquartile Range, * Outdoor cooking HH, ** Improved ventilation + clean fuel used.

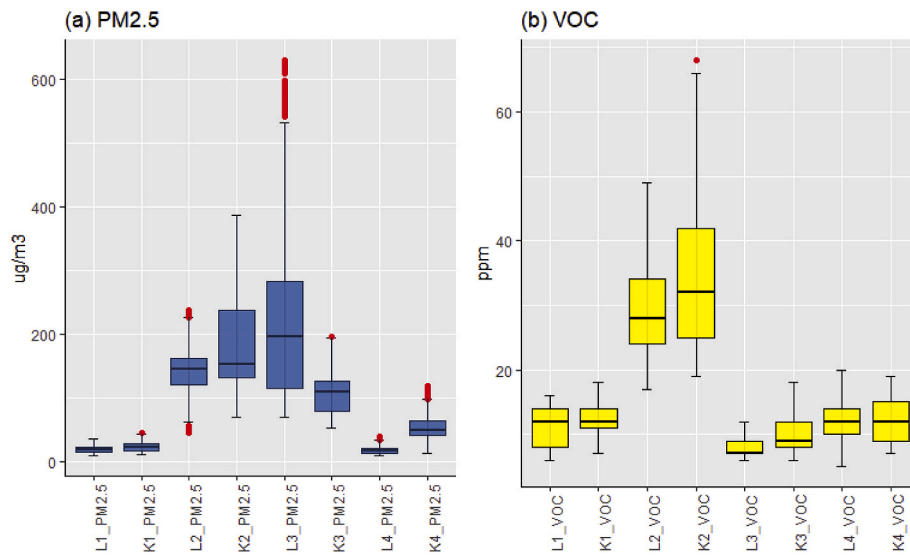


Fig. 1. (a) PM_{2.5} and (b) VOC of the living room (L) and Kitchen area (K). The graphs were generated with the exclusion of possible outliers of the dataset. (The box edges are quartiles, so the difference between the third and first quartile is regarded as the interquartile range (IQR). The median is marked by a line within the box).

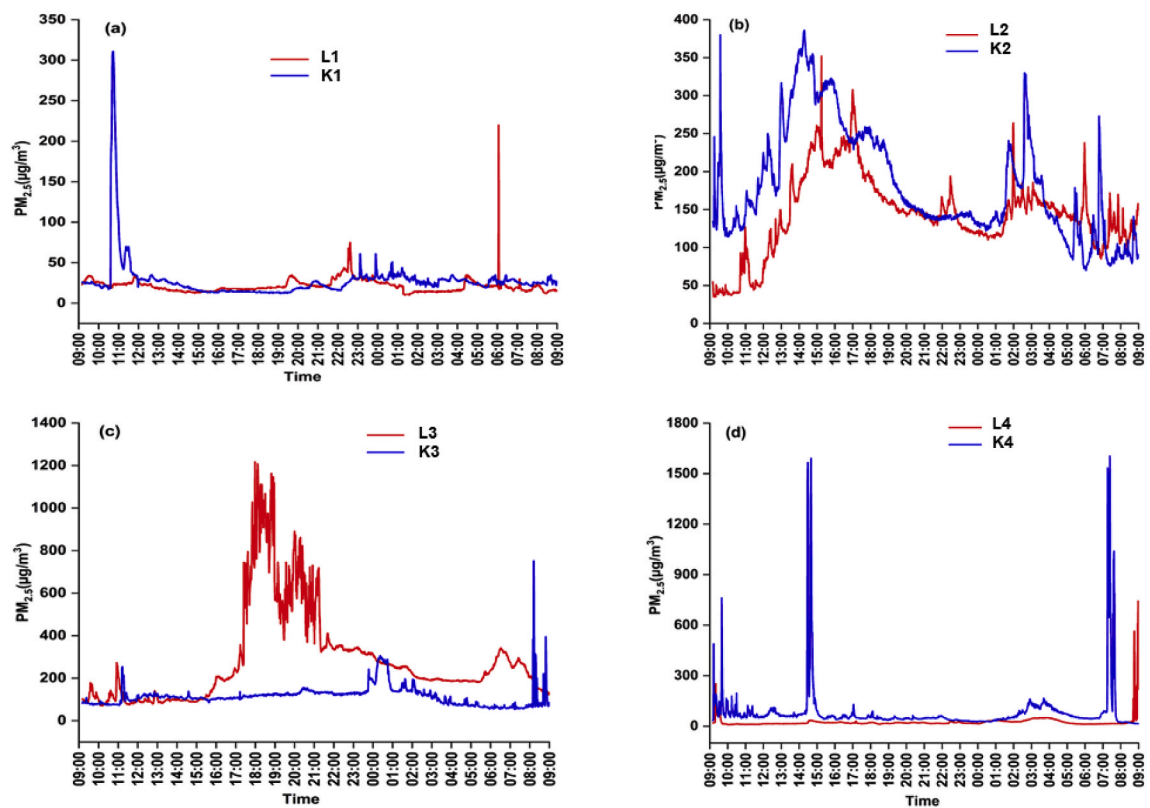


Fig. 2. 24-hours PM_{2.5} fluctuations in the (a) HH1, (b) HH2, (c) HH3, and (d) HH4 living room (L) and kitchen (K).

PM_{2.5} from 50 to 350 $\mu\text{g}/\text{m}^3$ (PM_{2.5}) was obtained from 10:30 am to 16:00, after which it started to decline until 22:00 and then fluctuated for the remaining hours. Because of the riverside location of HH2, wind velocity was much higher. Furthermore, some working practices, such as cement bag recycling and sewing, might significantly contribute to the increase in PM_{2.5} in the HH2. The maximum level of PM_{2.5} was 386 $\mu\text{g}/\text{m}^3$ and was recorded from the K2 (Table 1). The level started to readily upsurge with the

beginning of cooking activity with LPG burning at 9:00 am. The magnitude soon decreased with the stopping of the cooking. However, PM_{2.5} concentration steadily increased from 11:00 am. to 14:30 before and during a second cooking event in mid-day. Afterward, PM_{2.5} levels started to decrease for the rest of the day. Despite using clean fuel, the highest PM_{2.5} vol was detected at the K2. Moreover, several peaks were observed at midnight from 1:00 am to 3:00 am indicating the possibility of other indoor and outdoor sources (except IC) being responsible for PM_{2.5} enhancement in the K2 and L2.

The maximum PM_{2.5} level was recorded at the L3 located in the urban commercial area (HH3); the peak values of PM_{2.5} were 1217 µg/m³ between 17:00 and 21:00 (Fig. 2c). After the beginning of the PM_{2.5} assessment, two mild peak values were obtained at 9:30 and 11:00 am. After that, concentration increased gradually until 17:00. We found a dramatic increase from 18:00 to 19:00. After all, it steadily decreased from 21:00 to 6:00 am, although some fluctuations were observed in the remaining timeframe. Among the selected HHs, the HH3 was in the noisiest, dustiest, and fumiest atmospheric conditions, which occurred from moderate to large-scale manufacturing industries, the extra number of vehicles on the road, and day-to-day outdoor commercial activities. During the 24-h monitoring of K3, we obtained a moderate peak value of PM_{2.5} at 11:00 am during a cooking event. However, it started to decrease sharply after completing the cooking period, and at noon, it increased again at the second cooking event. At the K3, 106.7 ± 28.1 µg/m³ PM_{2.5} were observed using biomass fuel and LPG at the separate cooking stoves simultaneously. However, the living room yielded much more PM_{2.5} volume than the cooking area among all the findings. Since the HH3 was selected from a low-income urban area, the building structure was not so well improved, and that was tin-shaded. It could be possible to infiltrate the outdoor contamination through the open space under the tin shade and the open windows of L3.

Among the selected HHs, HH4 was equipped with an improved mechanical ventilation system at K4 compared to the others. Furthermore, LPG was used for cooking purposes. However, the IC impact is clearly visible in Fig. 2d, which indicates three peak values from 9:00 to 10:00 am, 14:00 to 15:00, and the following day, 7:00 to 8:00 am. During these periods, three cooking events occurred. At the same time, the L4 contamination level increased in the morning, then decreased, and remained stable over the day. However, it observed another peak value at 9:00 am the next day. The average PM_{2.5} was comparatively higher in the kitchen in the HH4 over the detection period.

Among the study areas, the commercial urban and riverside suburb areas had high concentrations of PM_{2.5}. Nevertheless, the open-fire burning contributed to a lower amount of PM_{2.5}. Another significant finding of the study is that adequate ventilation and exhausting air systems could potentially reduce the volume in residential urban areas HH.

3.2. Volatile organic compounds (VOC)

The VOC of the living room and kitchen were assessed from all the selected HHs. The overall mean concentration of living room VOC was significantly different from that in the OC spaces ($p < 0.001$). The VOC observation from the living rooms ranged between 7.95 and 29.54 ppm and was slightly lower than that of the Kitchens (9.90–35.66 ppm) (Fig. 1b). In that sense, it could be assumed that the kitchen activities might impact the volume of VOC in the living room. The average VOC was 11.18 ± 3.05 and 12.36 ± 2.80 ppm in the L1 and K1, respectively. In Fig. 3a, the peak value at L1 was 16 ppm and 18 ppm at K1. From the beginning of the assessment, the concentration was higher due to kitchen activities that started at 8:00 am. After the first cooking has been finished, the VOC level starts to decrease gradually. It peaked in the second cooking period from 18:00 to 19:00. Two additional peak values were detected between 20:00 and 2:00 am due to other sources of VOC. From morning to evening, L1 VOC was found to decrease during the daytime. However, VOC in L1 increased gradually from late-night 21:00 to early morning 5:00 am.

The maximum concentration was obtained in the living room and kitchen (29.54 ± 7.41 and 35.66 ± 13 ppm) of the HH2, respectively (Fig. 3b). The concentration of K2 was significantly higher ($p < 0.001$) than the L2. That might indicate the possibility of IC as a source for living areas' contamination level increment. However, one unusual fluctuation was observed at 5:00, while there was no IC occurred, which may potentially indicate other sources except the IC. With the normal ventilation system, it is likely to enter the outside air into the indoors. Fig. 3b shows the unorderly fluctuation of VOC indoors over the study period.

The VOC levels of HH3's living room and kitchen were recorded at 7.95 ± 1.46 and 9.90 ± 2.53 ppm, respectively. Fig. 3c shows the peak value found at the first cooking in the K3 was 47 ppm, which was at the starting time of our monitoring (9:00). On the overall assessment, the concentration was comparatively higher than that of the L3. However, before the end of our 24-h monitoring period, the VOC level in the L3 (38 ppm) sharply exceeded the K3 at 8:30. It is mentioned that the HH3 was located in a low-income urban area with comparatively older building types, with unimproved ventilation and kitchen orientation. Besides the IC impact, other potential indoor and outdoor sources may be responsible for the VOC enhancement in the L3 and K3.

In the HH4, the VOC distribution in both the living room and kitchen followed a similar trend (11.63 ± 3.11 and 11.66 ± 3.18 ppm, respectively) during the 24 h of monitoring. However, at 10:00, there was a significant increase in VOC levels (32 ppm) at the L4 (Fig. 3d). This might have occurred due to the use of perfume and body spray by the occupants before leaving for the office. Besides, VOC concentrations in both rooms were found to be decreased throughout the night and started to increase in the early morning. This suggests that indoor activities during the daytime might influence VOC levels in HH4.

Among the four locations' HHs, the maximum VOC concentration could have been obtained from the riverside suburb, while the least amount was found in the commercial urban area. However, the volume could observe to be similar for the residential urban area and the normal suburb area (HH1). At the same time, the volume was detected higher in each kitchen comparatively than in their living rooms.

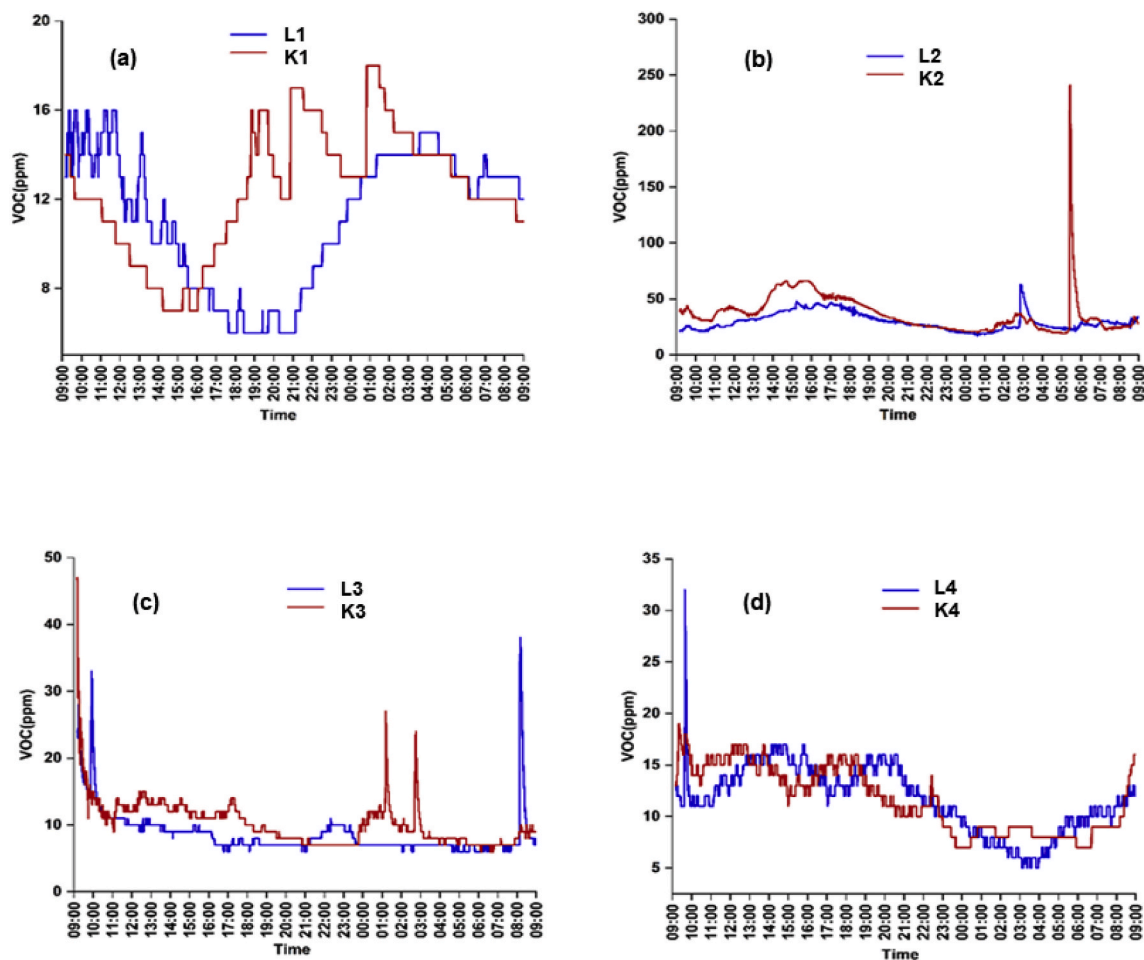


Fig. 3. 24-hour VOC fluctuation in the (a) HH1, (b) HH2, (c) HH3, and (d) HH4 living room (L) and kitchen (K) area.

3.3. Carbon dioxide (CO_2) fluctuations in the cooking place

Our study found a significant increase in CO_2 (75–77%) in the DC period compared to the BC. The most recorded change at the K3 on the DC period was 77% greater than in the background. During the DC period, it was discovered that K1, K2, and K4 had consecutively 59%, 51%, and 5% higher values (Table 2). The highest concentration was measured at K3 (2459 ± 37.51 mg/m³),

Table 2
Percent increase of CO_2 than the background level.

		CO_2 (mg/m ³)	
K1	Phase	Mean \pm Std.	Range
	BC	1097 \pm 10.02	1087–1105
	DC	1747 \pm 9.74	1726–1756
	Increase (%) *	59	
K2	BC	1061 \pm 58.40	1000–1298
	DC	1599 \pm 144.13	1376–1816
	Increase (%) *	51	
K3	BC	1392 \pm 4.61	1041–1473
	DC	2459 \pm 37.51	1080–2172
	Increase (%) *	77	
K4	BC	1280 \pm 113.88	1382–1400
	DC	1345 \pm 278.03	2276–2494
	Increase (%) *	5	

*Percent changes between BC and DC period.

which could have resulted from intense cooking and burning for a significantly more significant number of family members and the simultaneous use of biomass fuel and gas stoves. However, the CO₂ level fluctuated at DC, indicating a higher contamination emission potential of IC compared to the study reported by Cheung et al. [4], which showed less than a 5% change in CO₂ compared to the background at the DC period. However, our study found that the level (>50%) at the K1, K2, and K3 on the DC was significantly higher than the background level, which indicates that the IC is responsible mainly for emitting CO₂ in the indoor environment.

In addition, the minor increment could have been observed (5%) at the K4 than the background, most probably due to the improved mechanical ventilation and cleaner fuel practice.

4. Discussion

4.1. Cooking impact on indoor air quality

Our study suggests that the IC has the potential to damage the IAQ significantly in almost every observation. However, the contamination was found to be comparatively lower in the open cooking place. Sharma et al. [36] reported a relatively lower level of PM_{2.5} in open cooking places than the other places in a particular study, which is congruent with the finding of our research. Tan et al. [37] reported that PM_{2.5} increased with the start of food preparation and was between the 300–700 µg/m³ range. In the current study, the range was detected at 23.72 ± 7–182 ± 40.43 µg/m³ in the kitchen areas, which is congruent to the findings from the previous studies. However, no significant correlation could be found between the association between the open cooking and living room contamination enhancement (Fig. 4a and e) in the open kitchen area in our study. Therefore, open cooking might not be responsible for the living room air quality impairment. In addition, the spikes of VOC from the L1 during the night might be attributed to the burning of mosquito coils in the house [38,39], old painting of the walls, and outdoor infiltration through the open window [29]. In the open area, natural wind flow dilutes the contaminants' volume constantly, although these cooking configurations cannot be recommended because they pose a risk to the neighboring exposures. Salje et al. [29] reported increased IAP due to the use of biomass fuels by neighbors, despite using cleaner fuels in the HHs. In our study, However, we found a moderate positive correlation between the cooking place PM_{2.5} and VOC concentration and the pollution enhancement in the living space of the riverside suburb (HH2) (Fig. 4b and f) (PPMC, cor = 0.55 p = 0.000) and (PPMC, cor = 0.51 p = 0.000), respectively. This may indicate that the IC is primarily responsible for PM_{2.5} emissions [37]. Furthermore, the IC can potentially damage the living spaces of the selected HHs. It has been reported in previous studies that cooking and frying have more potential to increase IAP compared to boiling [40–42]. Despite the use of LPG in the kitchen, the highest amount of PM_{2.5} and VOC could have resulted from the adjacent HHs solid biomass fuel burning [29]. Moreover, apart from the cooking activity, different indoor and outdoor activities could impact the indoor contamination level. In the

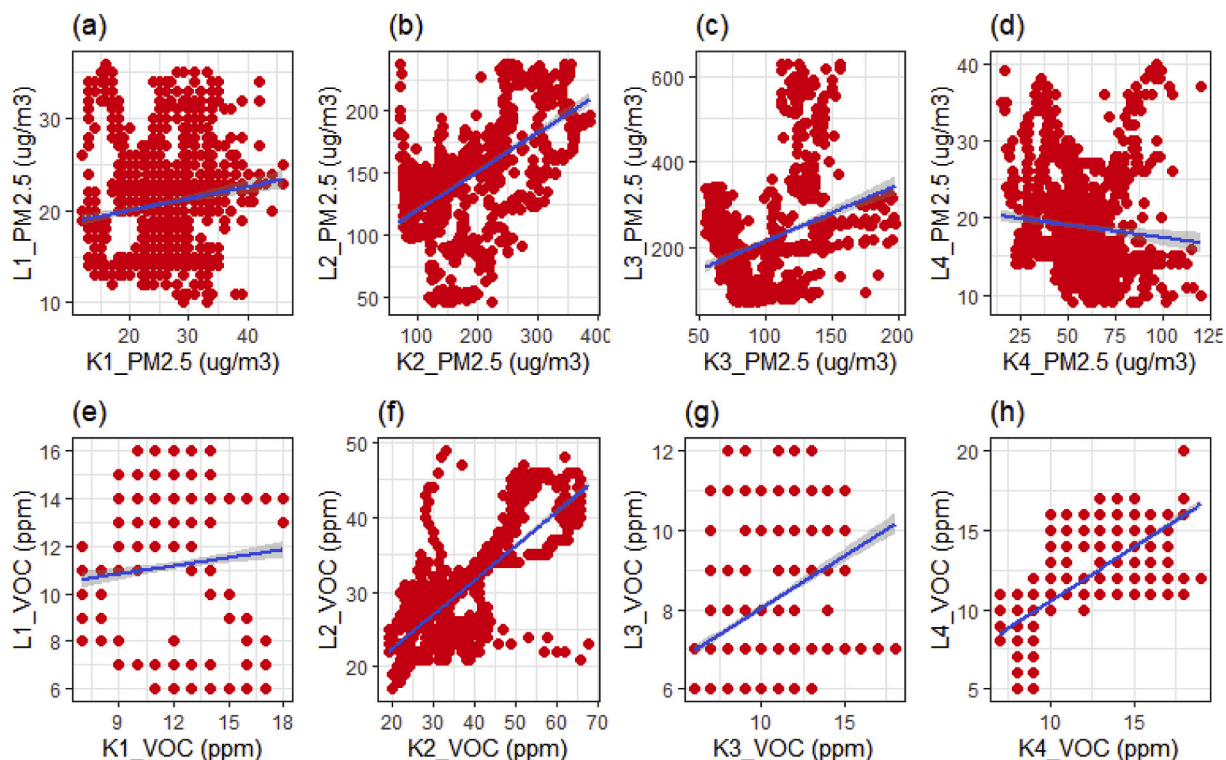


Fig. 4. Indoor cooking impact on the living room contamination enhancement. Pearson product-moment correlation (PPMC) shows the association between the kitchen (K) activity and living room (L) PM_{2.5} and VOC level.

commercial urban area (HH3), for example, no significant correlation could be found between cooking activity and bedroom contamination increase (Fig. 4c and g). Therefore, static (adjacent temporary food markets, industrial solvent use, production, and combustion processes) and dynamic sources (vehicular transport) may be considered as the potential outdoor sources of that urban environment [43–46]. On the other hand, old building materials, paints, cleaning products, solvents, and clothes washing might be considered indoor sources of VOC emissions [46–48]. In the current study, $51.62 \pm 20.74 \mu\text{g}/\text{m}^3$ $\text{PM}_{2.5}$ was detected in the urban residential area's kitchen, where LPG was burned for food preparation. This finding is congruent with the $40.6 \mu\text{g}/\text{m}^3$ $\text{PM}_{2.5}$ that was reported in a similar study in an urban area [37]. Nevertheless, no significant correlation was observed between the kitchen and living room $\text{PM}_{2.5}$ enhancement in the residential urban area (Fig. 4d). It is to be noted that, despite the well-improved ventilation system at the K4, the IC was significantly responsible for the L4 VOC level increment (PPMC, $\text{cor} = 0.68$, $p = 0.000$) (Fig. 4h). Therefore, the exhaust fan could not reduce the VOC in the cooking area significantly. Oil-based cooking practices, particularly frying, primarily cause VOC emissions from cooking. Moreover, water added to the oily cooking pan emits fumes and higher VOC contamination [17]. So, alternative cooking practices should be recommended. Furthermore, more improved technological adaptation is necessary, such as the well-placement of exhausting fans and fume hood installation. The exhausting fan continuously enhanced the air exchange capacity and directly removed the emissions from the cooking source [49].

4.2. Spatial variation of the contaminant's level

The study discovered a significant ($p < 0.001$) variation in IAP level based on the location of different HHs. The least volume of $\text{PM}_{2.5}$ originated in the suburban area's open kitchen (K1). Because of the separated outside location and open cooking place, there was a proper dilution of pollution instantly with the wind turbulence. Moreover, the wood powder was used as fuel, although there was no significant elevation of $\text{PM}_{2.5}$ or VOC in the HH. Therefore, the open cooking events could have a minor influence on IAQ in HH1. In contrast to HH1, the highest levels of $\text{PM}_{2.5}$ were found in lower-income urban commercial areas, with K2 recording the second-highest volume. In addition, the unusual fluctuations observed throughout the study could be attributed to floor sweeping [50], inadequate ventilation, the absence of an exhausting system, and the location of the kitchen, all of which serve as important determinants of $\text{PM}_{2.5}$ levels in the indoor environment [51]. In the riverside location, the maximum VOC observed at the L2 and K2 produced might have some influence from fine particles and water droplets carried by the strong wind. Dust particles, tiny water droplets, fragments, and debris of bio and non-biogenic substances could be carried by the strong wind flow [52]. Moreover, VOC emissions can be influenced by natural sources (e.g., vegetation, aquatic phytoplankton, and soil microbiota) [43,53–55]. In addition, the poor ventilation system could be another potential reason for the increased VOC in the HH2 [56]. On the other hand, with the improved ventilation, exhausting fan, and openable windows, HH4 VOC levels were significantly lower than HH2.

Spatial variation was reported in a study depending on the distance from the highway in urban areas of England [57]. In addition to the traffic impact, industrial activity, power generation, air recirculation, and soil resuspension were suggested in a study [58]. Moreover, they reported that the predominant wind at the particular location could potentially impact the contaminant's dispersion. Since all the HHs were naturally ventilated, it could be possible to have about 70% of outdoor infiltration happening [59]. That is from the consideration that $\text{PM}_{2.5}$ has a lower deposition velocity in comparison to coarse particulates [60]. Therefore, the unimproved ventilation of urban areas, largely impaired by outdoor pollution, poses a potential health risk. Based on different locations of the study, the risk potential could follow the order: $L3 > K2 > L2 > K3 > K4 > K1 > L1 > L4$ for $\text{PM}_{2.5}$, and $K2 > L2 > K1 > K4 > L4 > L1 > K3 > L3$ for VOC.

4.3. Health risk assessment of indoor air

The National Ambient Air Quality Standards (NAAQS), US EPA, and Bangladesh AQ standards [61] are shown in Table S2 after considering the WHO guidelines. But in practice, no pollution level is safe for human health [62]. For $\text{PM}_{2.5}$, it is $25 \mu\text{g}/\text{m}^3$ for a 24-h exposure period. In the present study, we found the L1 and L4 living room $\text{PM}_{2.5}$ concentrations were below the WHO and NAAQS guidelines of 24-h exposure time. Since HH1 is located in a suburban area and the cooking stove was outside the living room, the contamination from the outdoors and kitchen into the living room was limited. For HH4, which is located in an urban residential area where fewer indoor and outdoor influences resulted in $\text{PM}_{2.5}$ levels within the standard limits. In the L3, $\text{PM}_{2.5}$ was 728% higher than the WHO guideline, while 182% higher in the L2. Higher levels of $\text{PM}_{2.5}$ were also found in K2, K3, and K4. The $\text{PM}_{2.5}$ was 628% and 327% higher than the WHO guideline found in the HH2, and HH3 kitchens, respectively. In addition, the VOC levels exceeded the ASHRAE standard value of 3 ppm (mentioned in Table S2) in every location except the living room and kitchen. The CO_2 levels exceeded the standard limits in most cases during our various monitoring phases on the day. Overall, our results suggest an unhealthy condition of IAQ in studied HHs, particularly HH2 and HH3.

Toxicity potentials (TP) refers to the potential capability of a detrimental effect on the health of the dwellers of a unit of pollutant released into the air [63,64], and was calculated according to the following formula [63–66]:

$$\text{Toxicity potential (TP)} = (\text{Cp}/\text{Sp}) \quad (\text{i})$$

Here, Cp is the measured concentration of $\text{PM}_{2.5}$ from the indoor area, and Sp is the WHO standard concentration value (mentioned in Table S3) for $\text{PM}_{2.5}$.

The TP value of $\text{PM}_{2.5}$ ranged between 0.82 and 8.3 found for the HHs, which is congruent to the range (1.40–10.3) reported by Zaman et al. [51] in the hospital environment, Dhaka, Bangladesh. Most of the observations of $\text{PM}_{2.5}$ exceeded the acceptable ratio of

1. Except for K1, L1 and L4, in the other three living rooms and all the kitchens, TP was calculated more than 1 (Table S3). The L3 had the highest PM_{2.5} TP (8.3) value, posing a significant health risk to its residents. The TP order is: L3 > K2 > L2 > K3 > K4 > K1 > L1 > L4. Because of the long time spent at home and direct exposure to indoor cooking, the significant proportion of TP in HHs indicates a high potential health risk for dwellers, particularly newborn babies and pregnant women [57]. The increasing risks from solid biomass fuel use for food preparation result in acute respiratory problems among children in many developing and underdeveloped countries [67]. The study [67] compared the exposure impacts on those who used biomass fuel and cleaner fuel and concluded there was less potential risk for the clean fuel user group or being otherwise less exposed. Therefore, proper intervention should be made to ensure the health and safety of the household dwellers. Proper building design, ventilation system, and clean fuel use in the kitchen, for example, should be ensured to maintain a habitable indoor environment.

4.4. Limitations of the study

This has some limitations regarding the small sample size, pollution source identification, quantification of the air exchange rate, and outdoor AQ assessment. Firstly, getting acceptance from the HH dwellers to install equipment for 24 h at their residences was problematic. Secondly, instrument installation outside 24 h was impossible due to device security concerns. All the problems arose because the study did not operate under any specific research grant. Moreover, the equipment shortage and unavailability of research volunteers to deploy in more than eight sampling locations could not make it possible. However, further study should find out the sources, conduct an outdoor impact assessment and collect more sampling locations to suggest a conclusive report on the IAQ of the region. In addition, we could not compare the CO₂ data with the other two continuously measured parameters due to the inadequacy of 24-h data. That occurred because of the equipment unavailability during the research conducted.

5. Conclusion

The study demonstrated that the enhanced PM_{2.5} led to higher TP for HH dwellers in the riverside and commercial urban areas. A significant proportion of contaminants originated from the IC and damaged the IAQ found in most of the observations. It was also found that the outside open kitchen impacted the least on IAQ. However, it cannot be suggested to be installed since the emitted pollutants severely affect the neighbors' IAQ. The study recommends installing an improved mechanical ventilation system in the kitchen since it was found effective in reducing the PM_{2.5} volume—it recorded $51.62 \pm 20.74 \mu\text{g}/\text{m}^3$ in the IC space and $18.52 \pm 6.62 \mu\text{g}/\text{m}^3$ in the living room. Moreover, only 5% CO₂ enhancement occurred due to the IC in the well-ventilated and clean fuel-used HH. In contrast, the unimproved and solid fuel used IC system yielded higher contaminants ($106.7\text{--}207.3 \mu\text{g}/\text{m}^3$ PM_{2.5}, $9.90\text{--}36.66$ ppm VOC, $1061\text{--}2459 \text{mg}/\text{m}^3$ CO₂) in the HHs air. Therefore, cleaner fuel use, alternative food cooking practices (e.g., relegating the water addition to oil), installation of fume hoods, and improved cooking stoves should be kept in practice to ensure a healthy indoor environment. Moreover, in future studies, more parameters (e.g., PM₁₀, formaldehyde, carbon monoxide, and metals) should be determined to assess the health risks from indoor contaminants. To overcome the limitations of the current study, future research should include a diurnal assessment from more sampling locations (both indoor and outdoor) and seasonal variation.

Authors contribution

Md. Akteruzzaman: Conceived and designed the experiments, Performed the experiments, Analyzed and interpreted the data; Md. Alinur Rahman: Performed the experiments, Analyzed and interpreted the data, and Wrote the paper; Fahim Muntasir Rabbi, Sohely Asharof, Mahi Muzammel Rofi, Md. Kamrul Hasan, Md. Abdul Muktaadir Islam: Performed the experiments; M. Azizur R. Khan: Analyzed and interpreted the data; Mohammad Mahfuzur Rahman: Contributed reagents, materials, analysis tools or data and analyzed and interpreted the data; Md. Hasibur Rahaman: Conceived and designed the experiments, analyzed and interpreted the data, and Contributed reagents, materials, analysis tools or data.

Submission declaration and verification

No parts of the manuscript have been published previously.

Data availability

We declare that the database that supports the findings of our research will be fully available, upon reasonable request.

Declaration of competing interest

The authors declare no conflict of interest.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at <https://doi.org/10.1016/j.heliyon.2023.e12852>.

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