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Real-time indoor measurement of health and climate-relevant air pollution concentrations during a carbon-finance-approved cookstove intervention in rural India

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ABSTRACT

Biomass combustion in residential cookstoves is a major source of air pollution and a large contributor to the global burden of disease. Carbon financing offers a potential funding source for health-relevant energy technologies in low-income countries. We conducted a randomized intervention study to evaluate air pollution impacts of a carbon-finance-approved cookstove in rural South India. Prior research on this topic often has used time-integrated measures of indoor air quality. Here, we employed real-time monitors (~24 h measurement at ~ minute temporal resolution), thereby allowing investigation of minutely and hourly temporal patterns. We measured indoor concentrations of fine particulate matter ($PM_{2.5}$), black carbon (BC) and carbon monoxide (CO) in intervention households (used newer, rocket-type stoves) and control households ("nonintervention"; continued using traditional open fire stoves). Some intervention households elected not to use only the new, intervention stoves (i.e., elected not to follow the study-design protocol); we therefore conducted analysis for "per protocol" versus "intent to treat." We compared 24 h averages of air pollutants versus cooking hours only averages. Implementation of the per protocol intervention cookstove decreased median concentrations of CO (by 1.5 ppm (2.8 − 1.3; control − per protocol), $p = 0.28$) and $PM_{2.5}$ (by 148 $\mu g/m^3$ (365 − 217), $p = 0.46$) but increased BC concentration (by 39 $\mu g/m^3$ (26 − −12), $p < 0.05$) and the ratio of $BC/PM_{2.5}$ (by 0.25 (−0.28 − −0.03), $p < 0.05$) during cooking-relevant hours-of-day relative to controls. Calculated median effective air exchange rates based on decay in CO concentrations were stable between seasons (season 1: 2.5 h^{-1}, season 2: 2.8 h^{-1}). Finally, we discuss an analytical framework for evaluating real-time indoor datasets with limited sample sizes. For the present study, use of real-time (versus time-averaged) equipment substantially reduced the number of households we were able to monitor.

1. Introduction

Combustion of solid fuel (e.g., wood, animal manure, crop residue, or coal) in open fires and in traditional stoves affects human health and the environment (Venkataraman et al., 2005). The resulting household air pollution (HAP) includes CO, $PM_{2.5}$, and BC, and is associated with adverse health impacts in adults and children (Dherani et al., 2008; Smith et al., 2004) and affects regional and global climate (Bond et al., 2013; Janssen et al., 2012; Solomon et al., 2013). HAP from biomass and coal stoves was responsible for ~2.9 million premature deaths worldwide in 2015 (Forouzanfar et al., 2016), with low-income and industrializing countries most impacted.

Recently, there have been national and international efforts aimed to scale up stove and fuel interventions in India (Ministry of Petroleum and Natural Gas, 2016; Singh et al., 2017; Venkataraman et al., 2010). These efforts include cookstoves approved by the Clean Development Mechanism (CDM), established under the UN Framework Convention on Climate Change. Laboratory tests showed that, for example, "Chulika"

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rocket stoves had ~3-fold greater thermal efficiency than traditional open fire stoves (31% vs. 10%) and a two-fold wood savings; those stoves were subsequently approved for carbon financing (Central Power Research Institute, 2010; CDM Executive Board, 2006; Gold Standard Local Stakeholder Consultation Report, 2009).

Given the large impacts on health and the environment from solid-fuel combustion, a natural assumption would be that introducing a less polluting stove into a household would provide a net benefit for both. However, empirical evidence from intervention and observational studies has yielded mixed results (Chen et al., 2016; Khandelwal et al., 2017; Leavey et al., 2015; Pope et al., 2017; Wangchuk et al., 2017). For example, a meta-analysis of stove interventions in low- and middle-income countries conducted by Pope et al. (2017) reported improvements in HAP concentrations for intervention stoves over traditional stoves, though the 'improved' cookstoves often failed to achieve $PM_{2.5}$ concentrations close to the 24-h air quality guideline limit values. In addition, Khandelwal et al. (2017) highlight that adoption of intervention cookstoves over traditional stoves is limited despite promotion for decades; they highlight that stoves satisfy cultural and household needs beyond just cooking. Consequently, there is a need for better implementation and exposure assessment of intervention cookstoves. With few exceptions (Hankey et al., 2015; Carter et al., 2016; Chen et al., 2016), previous evaluations of cookstoves in rural areas have typically measured daily-average concentrations and have not analyzed data from real-time instrumentation, thereby preventing investigation of temporal patterns that may otherwise elucidate the effectiveness of the intervention stoves in reducing cooking pollution.

The present work is part of a larger energy intervention evaluation study of carbon-finance-approved cookstoves in the Koppal District of Karnataka State, India. Here, we discuss indoor concentrations of CO, $PM_{2.5}$, and BC concentrations during baseline and follow-up measurements (after the intervention) in houses using either traditional open fire stoves or carbon-finance-approved cookstoves. Seasonal and diurnal trends in the levels of indoor pollutants were analyzed using varying definitions of cooking time, to evaluate the effectiveness of the intervention stoves in reducing air pollution concentrations. Additionally, indoor air-exchange rates (AERs) were calculated based on CO decay patterns.

The three contributions of this study are: (1) evaluation of the effectiveness of a carbon-financed-approved cookstove intervention in the field, via a randomized control trial, (2) use of real-time rather than time-integrated measures of air pollution, thereby shedding light on impacts during times of cooking, and (3) calculation of air-exchange rates for a context where few AER measurements exist. This work can be referenced to, for example, create more detailed and sensitive emissions inventories, energy-use patterns, and health analyses in regions impacted by air pollution.

2. Material and methods

2.1. Study setting and research design

In this intervention study, households were randomly assigned to receive ("intervention") or not receive ("controls") the CDM-approved intervention. "Intervention" consisted of replacing the traditional open fire stoves with a new hearth and two chulika stoves. The chulika stoves are single-pot "rocket stoves", a type of natural-draft biomass stove. The study was conducted in two seasons: a pre-intervention baseline (September 2, 2011–December 10, 2011; "Season 1"/"S1") and a post-intervention follow-up (March 11, 2012–August 1, 2012; "Season 2"/"S2").

Our study was conducted in a rural village in the Koppal district of northern Karnataka, a state with a population of ~1.2 million people that covers 7190 square kilometers. Approximately 35% of Koppal residents are day-wage laborers earning less than one dollar per day and an estimated 99% of households use traditional stoves (indoor, open fires) to cook food and heat bath water (Fair Climate Network, 2012).

We partnered with a local nongovernmental organization that was the first in India to obtain CDM approval for a cookstove intervention program. The overall goal of the larger study was to evaluate climate and health impacts of a CDM-approved intervention. Additional details on the study design, the setting, and the CDM intervention are provided in Aung et al. (2016) and Grieshop et al. (2017).

Fig. 1 displays the study design of the field campaign, which used a parallel assignment structure. Of the 300 eligible households in the study village, 187 households met the inclusion criteria and were eligible to enroll into the study. Households were excluded if the family did not primarily burn biomass for cooking, if more than seven people lived in the household, or if the family planned to seasonally migrate during the next year. Of enrolled households, 96 households were randomly selected to receive the intervention CDM-approved cookstoves following baseline assessment, while the remaining 91 households served as controls and received the stoves following the completion of the study. We randomly selected 32 households (16 interventions, 16 controls) for 24 h, real-time monitoring of CO, $PM_{2.5}$, and BC in both seasons. Adherence to protocol was determined through a questionnaire that asked occupants about stove use practices at each visit and through visible inspection of the kitchen.

2.2. Indoor air pollution monitoring

Three instruments, sampling from a common inlet installed approximately 1 m from combustion zone and 0.6 m above the floor were used for continuous monitoring of household air pollution concentrations: a DustTrak Aerosol Monitor (Model #8520, TSI, Inc., Shoreview, MN) measured $PM_{2.5}$, an IAQ-Calc (Model #7545, TSI, Inc., Shoreview, MN) measured CO, and a MicroAethelometer (Model #AE51; wavelength: 880 nm, AethLabs, San Francisco, CA) measured BC. We selected the location of measurement to be consistent among households and to approximate the breathing location of people in this community when

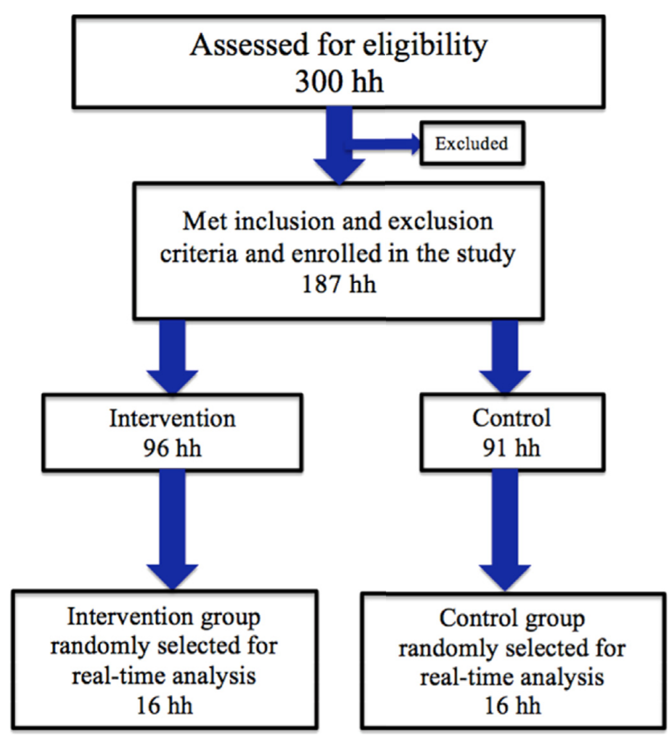

Fig. 1. Baseline enrollment and follow-up after a CDM-approved cookstove intervention in Koppal, India. Household (hh) eligibility assessment and inclusion and exclusion criterion were applied to establish randomized control and intervention treatment groups.

they cook. The temporal resolution of the instruments was set to 30 s for the Dustrak and 60 s for the IAQ and MicroAeth. A more detailed treatment of sampler location can be found in Aung et al. (2016).

The DustTrak uses aerosol optical scattering to calculate concentrations. To correct for instrument bias and flow rate inconsistencies, $PM_{2.5}$ values were corrected against time-integrated indoor filter measurements collected from the same inlet (Aung et al., 2016). Additionally, RH was independently measured and used to correct for $PM_{2.5}$ levels; correction factors obtained in this study were specific to this location.

The MicroAeth determines raw BC concentrations (BC_0, ng m^{-3}) from attenuation values. Following corrections from Kirchstetter and Novakov (2007), corrected BC mass loading (BC_{corr}) was determined using Eq. (1):

$$BC_{corr} = BC_0/(0.88 \times Tr + 0.12). \tag{1}$$

Here, Tr is transmissions calculated from attenuation values, according to Eq. (2):

$$Tr = e^{-ATN/100}. \tag{2}$$

The MicroAeth measures attenuation values using filter tickets, which need to be changed when they become overloaded. To limit the filter loading rate and reduce the frequency with which filter changes were required, a diluter loop was employed upstream of the MicroAeth inlet. Briefly, the diluter consists of a flow-split controlled by a needle valve: most air travels through a mass flow sensor (Honeywell AWM 3300) and through a filter that removes all particles; the remaining air, which contains the environmental concentrations of particles, encounters a flow orifice as a resistance to flow. This method for dilution is regularly applied for other particle instruments in high concentration settings (Knibbs et al., 2007; Apte et al., 2011). The two streams (particle-free and containing particles) then combine and enter the MicroAeth. Typical dilution ratios are ~3:1 to 8:1. In order to correct for varying flow rates inside the diluter, an additional dilution ratio correction factor (CF; see Eq. (3)) was applied to the BC data for each household sampling day per season:

$$CF = \frac{[Aethalometer\ Flow]}{[Aethalometer\ Flow] - [Diluter\ Flow]}. \tag{3}$$

Household-specific correction factors can be found in Tables S3 and S4 in the Supplementary Information.

2.3. Data analysis

All real-time pollutant concentrations were processed into 10-min averages. Multiple time windows were defined to analyze the effects of cooking events. These time windows included: 1) a 24 h time-weighted average (24 h average), 2) fixed windows in which cooking events were defined based on observations of typical cooking periods in households (5:00–9:00 and 18:30–21:00 "cooking"; all other times "noncooking"), and 3) variable-cooking windows in which pollutant concentrations were at least 150% of a household's background baseline (established from midnight to 2:00am; time windows with concentrations less than 150% of this background were considered non-cooking).

Households were categorized into three stove use groups: "Control", "Followed Protocol" (intervention households that exclusively used the intervention stove), and "Mixed" (intervention households that used both the Chulika and traditional stoves, i.e., did not follow protocol). The label "intent to treat" refers to a comparison between the control households and all intervention households (whether following protocol or not); the label "per protocol" refers to a comparison between the control households and only the "followed protocol" households.

Nominally and statistically significant changes in pollutant concentration between stove types were established using a one-tailed Wilcoxon signed-rank test with p-values of 0.10 and 0.05, respectively.

To further evaluate the effectiveness of the intervention, we conducted statistical tests (difference-in-differences) and generated visual displays of the raw data. Given the small number of data-points per group, violin plots were used to display data trends. Violin plots visually display probability density distributions (Hintze and Nelson, 1998) and are useful for comparing multiple categories within a small dataset (<30 observations). Households that were missing more than 20% of their time series data (i.e., >5 h) were excluded from the matched-pairs analysis. Results of this study were compared to those discussed in companion papers (Aung et al., 2016; Grieshop et al., 2017).

AERs were calculated based on real-time indoor CO because it is a nonreactive tracer gas that is not affected by deposition or secondary chemistry associated with particles (Johnson et al., 2011; Samfield, 1995; Soneja et al., 2015). AERs for CO were separately calculated for both seasons. The calculation involved identifying the time window following cooking, when the CO concentration decayed from its peak concentration to a cutoff of 0.2 ppm. During those times, AER was calculated assuming well-mixed conditions (Sherman, 1990), using Eq. (4):

$$AER = \frac{Q}{V} = \frac{\ln\left(\frac{C_t}{C_0}\right)}{(t_2 - t_1)}, \tag{4}$$

where Q is ventilation rate (m^3 min^{-1}), V is room volume (m^3), and C_0 and C_t are the CO concentrations at times t = 0 and t, respectively.

The AER (Q V^{-1}) was determined by the regression of $Ln(C_2/C_1)$ against Δ(time) for each household. The simplified linear regression used to obtain this value was:

$$Ln(C_t) = \beta_0 + \beta_1 t + \varepsilon$$

where

$$\varepsilon \sim N(0, \sigma^2)$$

In this case, t represents time in minutes, C_t is the concentration of CO at time t, β_1 represents the -AER, and β_0 represents $Ln(C_0)$, where C_0 is the concentration of CO at time 0. Additional details regarding the method for calculating AER are in Fig. S1.

3. Results

Five of the 16 households (31%) did not follow intervention protocol; these five households comprise the "mixed-use" group; they are included in "intent-to-treat" analyses but not "per protocol" analyses. Additionally, data from two households in the assigned intervention group were corrupted and unusable for this analysis.

3.1. Diurnal trends

Fig. 2 displays the overall trends in diurnal concentrations of $PM_{2.5}$ by season. Analogous data for CO and BC are in Figs. S2 and S3. Concentrations during stove combustion events were generally higher in S2 compared with S1 for $PM_{2.5}$.

As expected, indoor concentrations for all three pollutants generally peaked in the morning and evening around cooking events. Shaded regions in Fig. 2, Figs. S2 and S3 indicate the cooking periods used for the "fixed-time windows" analyses below. The plots suggest that the fixed-cooking windows generally captured indoor biomass burning events, although imperfectly (e.g., start time of evening peaks in S1 for $PM_{2.5}$). Concentrations during non-cooking hours for both seasons (S1: 58 μg/m^3 [$PM_{2.5}$], 0.07 ppm [CO], 1.65 μg/m^3 [BC] and S2: 18 μg/m^3 [$PM_{2.5}$], 0.11 ppm [CO], 0.10 μg/m^3 [BC]) were relatively constant within that season (pooled coefficient of variability for S1 and S2: 47% [$PM_{2.5}$], 59% [CO], 62% [BC]).

Temporal variation within groups was similar, with the exception of the mixed-use group. The concentrations in the mixed-use group households were more variable during cooking periods, possibly

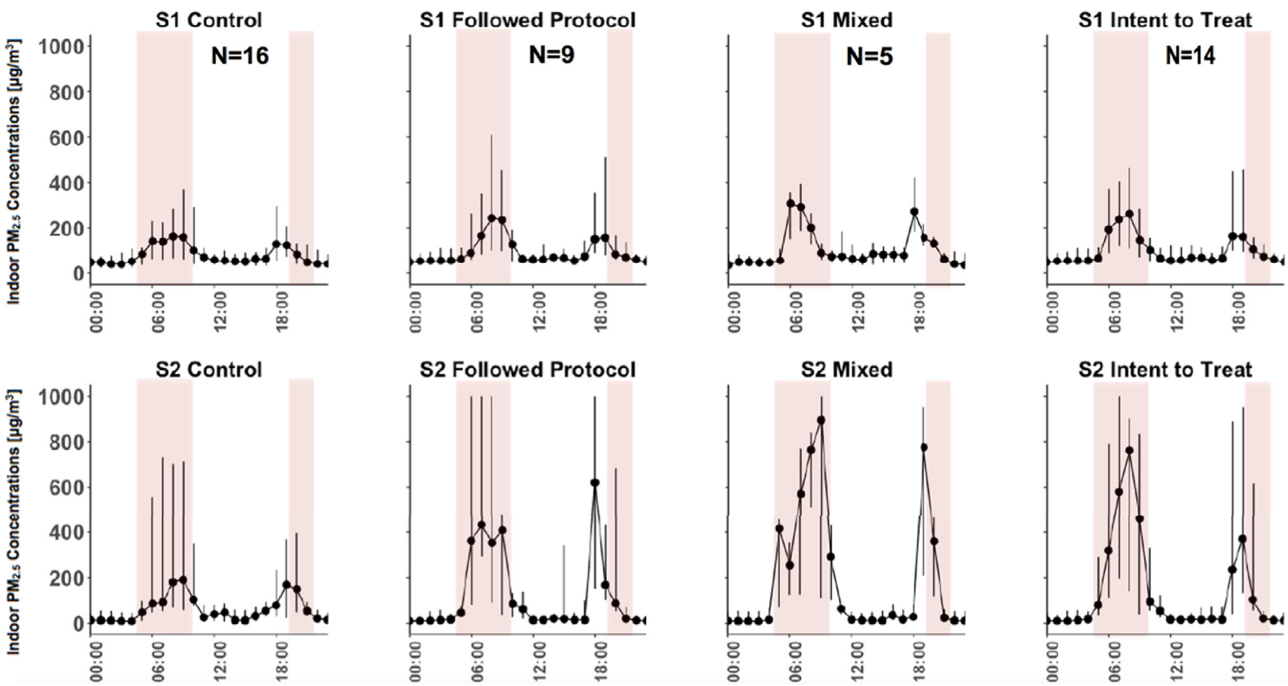

Fig. 2. Average hourly indoor $PM_{2.5}$ concentrations by season and stove use category. Shaded regions indicate fixed-window cooking periods. Error bars represent the 25th and 75th quartiles. Sample size for each season and stove category are indicated at the top of each panel. "S1" (top row) is pre-intervention; "S2" (bottom row) is post-intervention.

reflecting that the mixed-use group had the smallest available sample size (N = 5) or that conditions were more variable for this group. During cooking, the mixed-use group yielded concentration readings that were similar neither to the control nor the followed protocol groups.

3.2. Seasonal and diurnal distributions of $PM_{2.5}$

Following the methods of Carter et al. (2016), Fig. 3 shows diurnal variation in the distributions of PM concentrations across households and by season. These distributions suggest that peak pollution events occur at 8:00 a.m. and 7:00 p.m. for both seasons. During midday, $PM_{2.5}$ concentrations generally decreased to the overnight background concentrations.

Midday concentrations were generally higher during S1 (typically, 100–250 $\mu g/m^3$) than during S2 (35–100 $\mu g/m^3$). In contrast, concentrations during cooking were generally lower during S1 (typically, 250–500 $\mu g/m^3$) than during S2 (500–1000 $\mu g/m^3$). Also, during cooking, the percent of events with concentrations higher than 1000 $\mu g/m^3$ was lower during S1 than during S2 (7% vs. 13%). That finding is consistent with the mean concentrations being lower during S1 than during S2 (Fig. 2). Although midday and nighttime concentrations of $PM_{2.5}$ were higher in S1 than S2, cooking events dominate the overall average concentrations; as a result, average $PM_{2.5}$ was lower during S1 than during S2. The seasonal diurnal distributions for CO and BC were similar to $PM_{2.5,}$ but household variation in concentration ranges was smaller. Importantly, this finding is only uncovered because of the real-time measurements; time-integrated measurements would be unable to identify differences between cooking and noncooking time windows.

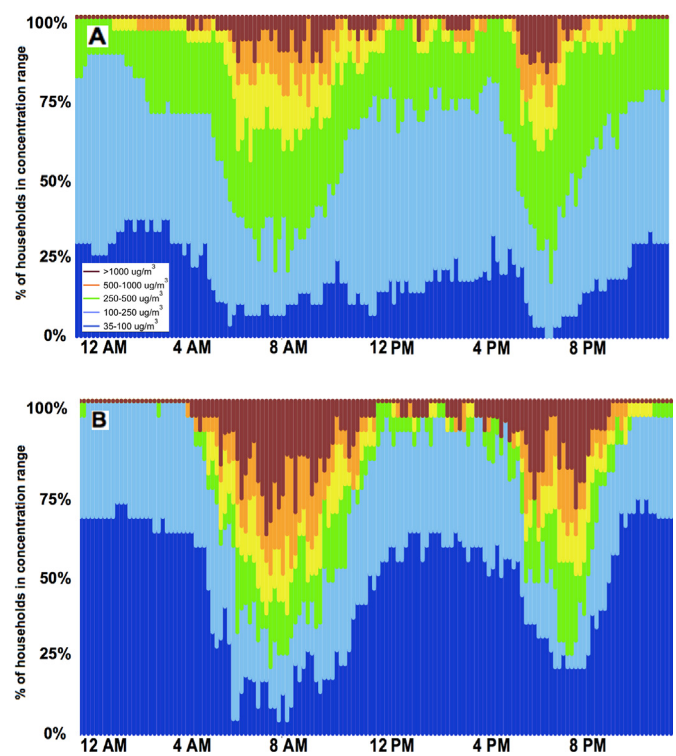

Fig. 3. Diurnal distribution of real-time $PM_{2.5}$ concentrations with 10 min resolution. "S1" (A) is the pre-intervention baseline and "S2" (B) is the post-intervention follow-up.

3.3. Effectiveness evaluation

The violin plots in Fig. 4 compare matched-pair differences (S2—S1) among stove use categories and definitions for cooking time-windows. Here, medians are employed for comparisons. In these plots, a positive median value indicates that for the median household, concentrations were higher in S2 than in S1; a median value of 0 would suggest a median of no change in concentration from S1 to S2. For the intervention group, if the median value is lower than the control group, this indicates that intervention households fared better than control households (e.g., concentrations improved more or worsened less for interventions than for controls). Conversely, if the median change is higher for the intervention than for the control, this suggests that intervention households

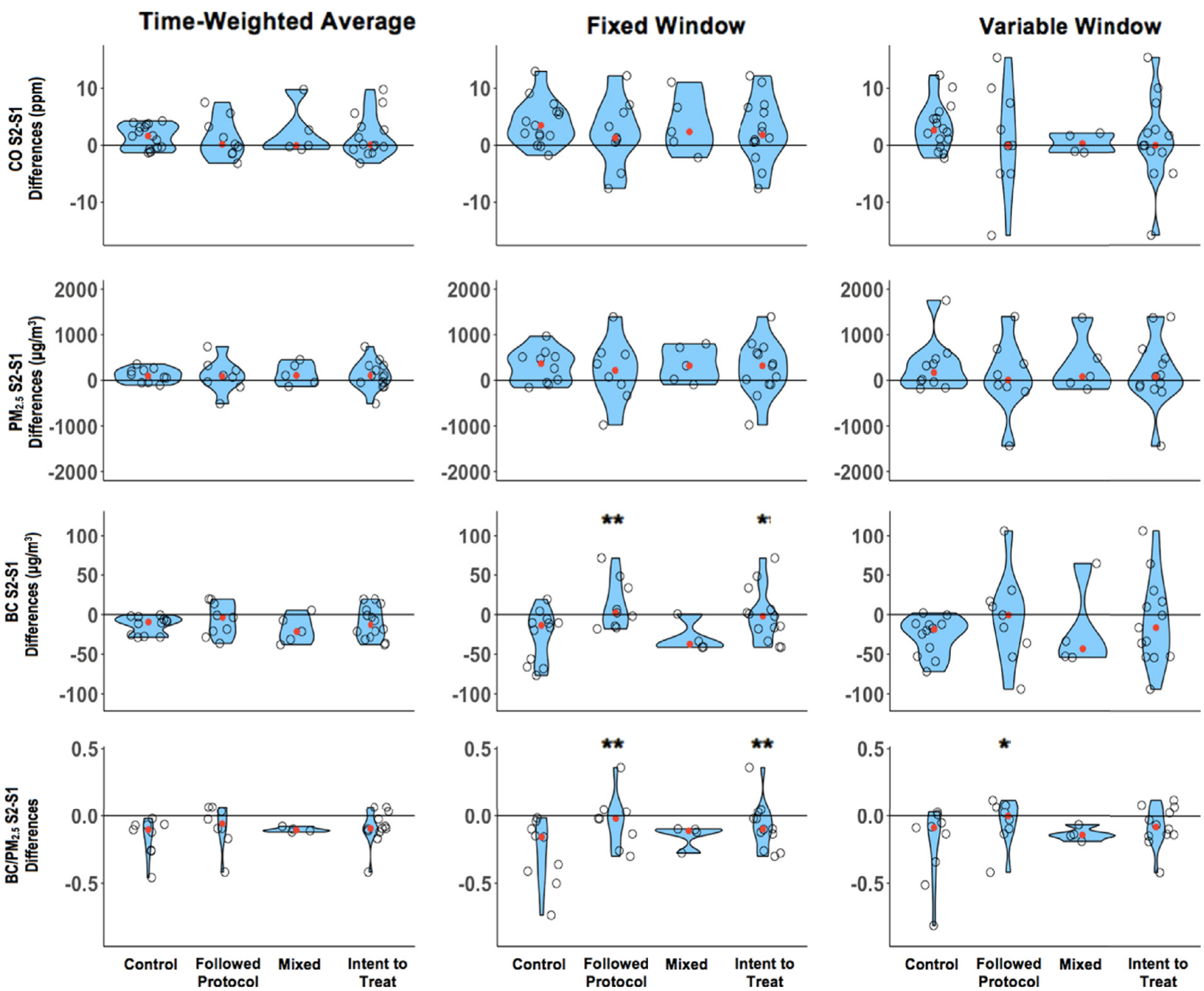

Fig. 4. Violin plots of indoor CO (ppm), $PM_{2.5}$ (μg/m^3), BC (μg/m^3) concentration and BC/$PM_{2.5}$ ratio differences between seasons (S2−S1) for time-weighted average, fixed time-window, and variable time-window cooking-only concentrations, by treatment type. Filled circles represent median values. The shapes of the distributions reflect a given treatment's probability density. Unfilled circles within a density plot represent individual matched-paired household observations for a given treatment type. A slight stochastic jitter in the x-axis was added to reduce overlap among icons. The horizontal line at 0 denotes the case of no change in concentration from S1 to S2. Statistical significance between the control and intervention groups were determined by a one-tailed Wilcoxon signed-rank test indicated by * ($p < 0.10$; "nominally significant") or ** ($p < 0.05$; "significant").

fared worse than control households (e.g., concentrations improved less or worsened more for intervention households than for controls).

The CO median seasonal difference (S2−S1) values of the followed protocol group in all time window definitions (24 h average: 0.14, Fixed: 1.3, Variable: −0.06; ppm) were lower than the control group median values (24 h average: 1.6, Fixed: 2.8, Variable: 2.9; ppm), although these differences were not statistically significant (p = 0.32, 0.28, 0.26). The difference in median CO concentration was 1.5 ppm lower in the followed protocol stove group (1.3 ppm) compared with the control group (2.8 ppm) during fixed-window cooking events. The lower relative median values for the intervention stove suggests improvement in indoor CO concentrations, though this difference is not statistically significant (p = 0.28). The followed protocol groups had greater variability in pollutant concentrations in all time window definitions. In addition, the median and spread of the mixed group was often similar to that of the control group in all time windows, which may indicate that those households reverted back to their traditional stoves frequently.

In the "followed protocol" category, median differences for $PM_{2.5}$ (24 h average: 88, Fixed: 217, Variable: 10; μg/m^3) were lower than the control group medians (24 h average: 97, Fixed: 365, Variable: 169; μg/m^3), although these differences were also not statistically significant (p = 0.45, 0.46, 0.39, respectively). The difference in median $PM_{2.5}$ concentration was 148 μg/m^3 lower in the followed protocol stove group (217 μg/m^3) compared with the control group (365 μg/m^3) during fixed-window cooking events. As with indoor CO concentrations, median and spread of $PM_{2.5}$ were similar for the mixed and control groups.

Fig. 4 indicates that BC median concentration difference was higher in the followed protocol group (24 h average: −6, Fixed: 12, Variable: −4; μg/m^3) than in the control group (24 h average: −13, Fixed: −26, Variable: −27; μg/m^3) for all time window definitions. Furthermore, the BC increase was statistically significant for the followed protocol group and nominally significant for the intent to treat group in the fixed-window analysis (followed protocol: p = 0.02, intent to treat: p = 0.08). The difference in median BC concentration was 38 μg/m^3 higher in the followed protocol group (12 μg/m^3) than in the control group (−26 μg/m^3) during fixed-window cooking events, suggesting a worsening of indoor BC pollution. In addition, the spread in BC of the control and mixed groups were smaller than in the followed protocol group.

Consistent with the BC findings, the median BC/$PM_{2.5}$ ratio difference was higher for the followed protocol intervention households (24 h average: −0.08, Fixed: −0.03, Variable: −0.05) than the control (24 h average: −0.16, Fixed: −0.28, Variable: −0.22). The BC/$PM_{2.5}$ ratio was greater in the followed protocol than for the control group in the fixed-window analysis (followed protocol: p = 0.02, intent to treat: p = 0.04) and also greater in the variable-window analysis (followed protocol: p = 0.08). Furthermore, during fixed-window cooking events followed protocol BC/$PM_{2.5}$ ratio (−0.03) was greater than for the control group (−0.28). The spread of the control group ratios were similar to the spread of the followed protocol group ratios.

3.4. Air-exchange rate results

The overall median AER was 2.5 h^{-1} (mean: 3.4 h^{-1}, IQR: 1.5–3.9 h^{-1}). Median AER was 52% higher in the evenings than the mornings (3.2 vs. 2.1 h^{-1}, $p < 0.003$; IQR: 1.5–6.0 h^{-1} vs. 1.4–3.3 h^{-1}) (Fig. 5). Differences between seasons 1 and 2 were small and not statistically significant (see Supplementary Information Tables S1, S2, and Fig. S4). Diurnal variation in AERs may help explain why CO concentrations (Supplementary Information Fig. S2) were typically lower in evenings compared with mornings. Higher AERs will result in greater dilution of emissions, thereby (if all else is equal) reducing indoor concentrations.

4. Discussion

In this study of real-time HAP concentrations in 32 households in the Koppal district of Karnataka state in India, the CDM-approved intervention stove improved some, but not all aspects of air quality. In households that exclusively used the intervention stove, the median household concentration exhibited a smaller increase in CO and $PM_{2.5}$ between seasons than the control group, but higher BC concentrations and thus higher BC/$PM_{2.5}$ ratios. Aung et al. (2016) also reported that $PM_{2.5}$ median concentration increase was lower in the followed protocol group (65 μg/m^3) compared with the control group (162 μg/m^3) using 24 h integrated gravimetric measurements, BC median concentration increase was higher in the followed protocol group (23 μg/m^3) compared with the control group (16 μg/m^3) using 24 h absorbance measurements, and BC/$PM_{2.5}$ ratio median difference was higher in the followed protocol group (0.08) than in the control group (0.04).

For $PM_{2.5}$, the direction of reported seasonal differences in concentration between the followed protocol and control households were consistent with exposure measurements reported in Aung et al. (2016) and emission factors reported in Grieshop et al. (2017). For BC, seasonal differences are slightly different in this work than for emission factors reported in Grieshop et al. (2017) and time-integrated indoor filter measurements in Aung et al. (2016): here, average reported BC values are slightly higher in S1 than S2 (but with some variability among groups; see Fig. 4); Grieshop et al. reported the reverse (though also with variability among groups). We suspect these differences found in one pollutant are attributable to variability and uncertainty potentially due to dilution correction. Measurements reported in the three publications (the

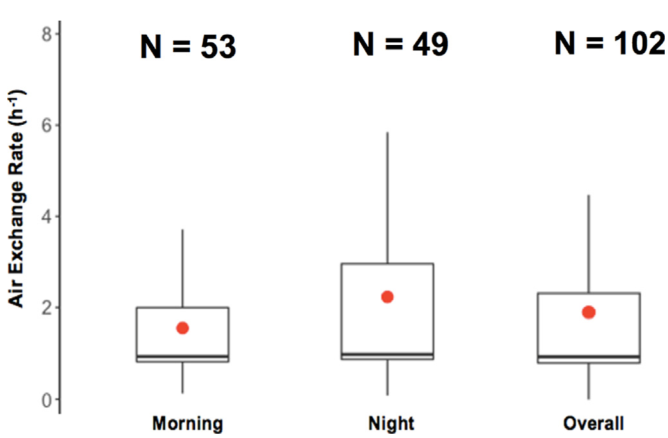

Fig. 5. Distribution of morning, night, and overall CO AERs for all households during both monitoring seasons in stove intervention evaluation study in Koppal, India. The boxplot indicates 10^{th}, 25^{th}, 75^{th}, and 90^{th} percentiles; the first black line in the box marks the median, and a dot marks the mean. Sample sizes (number of decay events) are indicated at the top of each boxplot.

present article; Aung et al. (2016); Grieshop et al. (2017)) also were conducted using different instruments.

The existing literature suggests mixed results for cookstove interventions. A systematic literature review and meta-analysis of stove intervention studies by Pope et al. (2017) found post-intervention mean reductions of 1.6 ppm for CO (eight studies, ten estimates) and 100 μg/m^3 for $PM_{2.5}$ (four studies, six estimates), respectively. These reductions are consistent with our findings. Notably, Pope et al. cautions that $PM_{2.5}$ measured during pre-intervention seasons are highly variable across all studies, which is also consistent with this work; substantial seasonal effects on control household concentrations were noted.

Most available measurements of AERs are from office buildings and residences in developed countries; in contrast, there are few AER measurements in low-income countries (McCracken and Smith, 1998; Smith, 1987; Park and Lee, 2003; Sundell et al., 2011; Williams and Unice, 2013). Of the limited studies conducted in rural villages, Soneja et al. (2015) measured a mean AER of 11.9 (h^{-1}) and a range of 2.3–41.8 (h^{-1}) in Nepal, Williams and Unice (2013) measured an AER range of 2–20 (h^{-1}) in the northern highlands of Peru, and Carter et al. (2016) measured AERs of 18 ± 9 (h^{-1}) in the summer and 15 ± 7 (h^{-1}) in the winter in the Eastern Tibetan Plateau. Bhangar 2006 reported a range of 7–27 (h^{-1}) in Maharashtra, India and Pillarisetti et al. (2015) reported a mean of 33 (h^{-1}) in Pokhara, Nepal. This work measured a mean AER of 3.4 (h^{-1}) and a range of 0.08–12.7 (h^{-1}), which is consistent with the mentioned literature, but also offers a substantially lower range of values. Lower AERs tended to occur more frequently during mornings than nights for both seasons (Supplementary Information Fig. S4). Values presented here are consistent with our expectations, based on informal observation of the buildings studied (i.e., households in this community are typically enclosed structures with small wall and ceiling openings, flat and low ceilings and no active ventilation, resulting in AER values that are smaller than, but comparable to, those typical of values for houses in low-income countries). We have no evidence that residents increase circulation of air during cooking or that AER would be systematically different during cooking versus other times, though are mindful that AER may vary by hour-of-day. Our results contribute to a knowledge gap regarding field-based measurements of AER in rural low-income settings.

Our study has several limitations to consider for future studies of real-time household air pollution. The foremost two difficulties for the monitoring campaign and analysis were instrument breakage at very high concentrations and the small sample size. As a result of high concentrations, there were often inconsistencies in the dilution ratio measured upstream of the MicroAeth and DustTrak flow errors. A series of correction factors were used to adjust for these errors (described in Supplementary Information Tables S3 and S4). Moreover, if a continuously monitoring instrument did not function for several hours in a household, that loss of data was critical because of the small number of homes evaluated per treatment group (fewer than 16 households). Although obtaining real-time concentrations is useful, especially in rural and under-monitored areas, the smaller sample size of the study may prevent establishing statistical significance that may otherwise be viable with a larger dataset of lower time resolution measurements.

As noted above, the intervention stove was not fully adopted in this study. Of 14 households in the intervention group (excluding the two households with corrupted measurements), only nine (64%) households followed the protocol. The seasonal trends in the five mixed households (36% of the intervention group) displayed medians and spreads more similar to households with traditional stoves than to the intervention group, thus complicating the results and for some comparisons (namely, the "per protocol" analyses) reducing statistical power of the study. The 36% of households with nonadherence to protocol is consistent with the findings in Aung et al. (2016) in which 40% of intervention-assigned homes continued to use their traditional cookstove during the field campaign.

One important aspect of this study is the use of multiple methods to analyze a small sample (fewer than 30 households) of real-time, indoor monitoring data. This work not only used typical approaches for investigating HAP trends, such as time series and diurnal analyses, but also applied counterfactual analysis using double-sided probability density distribution plots (violin plots), real-time diurnal distributions segregated into different concentration ranges, fixed and variable definitions of timeframes, and calculation of air-exchange rates from decay of CO. Despite a limited sample size, this work provides an analytical framework for future analyses and visualization of real-time household air pollution data. An important finding is that concentration differences by season were different for cooking times than for other times. Analysis of real-time data enables us to calculate time-dependent metrics such as AER, to understand at what points pollution is largest, and to evaluate differences between homes using similar stoves.

Investment of resources into rural energy intervention programs has great potential to improve household air quality in developing countries and thus increase quality of life, improve public health, and address climate change. Laboratory studies have clearly demonstrated the potential benefits of cookstove interventions (Grieshop et al., 2011), but demonstrating these same benefits in real households has proven to be more challenging and complex. This work and other recent publications suggest that reduction in indoor pollution from intervention cookstoves might not be occurring in practice to the same extent as is expected from laboratory evaluation, and that benefits from such interventions should not be assumed. As mentioned above, the reduction in CO and $PM_{2.5}$ concentrations were substantial (1.5 ppm for CO, 148 $\mu g/m^3$ for $PM_{2.5}$) but not statistically significant. Given our small sample size, these results may not be representative of the potential of Chulika cookstoves; further careful evaluation is needed. Additional research should focus on how to design and implement future stove interventions such that the focus is meeting user need in order to encourage and enable the full adoption of the new technology (and – perhaps more importantly – dis-adoption of the old technology). Furthermore, this work and that of Aung et al. (2016) and Grieshop et al. (2017) suggest that intervention cookstoves need further refinement and improved emission performance in order to more dramatically reduce all household air pollutants produced during indoor cooking.

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Appendix A. Supplementary data

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