ASSESSMENT OF PERSONAL EXPOSURES TO HOUSEHOLD AIR POLLUTION AMONG PREGNANT AND NON-PREGNANT ADULT WOMEN IN GUATEMALA, INDIA, PERU, AND RWANDA

by

DEVAN ANTHONY CAMPBELL

(Under the Direction of Luke Peter Naeher)

ABSTRACT

Objectives: 1) To evaluate the effectiveness of a liquefied petroleum gas (LPG) intervention in reducing HAP exposures among middle- and older-aged women, 2) characterize the major factors associated with black carbon (BC) exposures among pregnant women, 3) assess the role these factors play in modifying the BC exposure reduction via an LPG stove intervention, and 4) compare BC measures from the same optical instrument using two slight variations of the same optical absorption technique.

Methods: Up to six repeated exposure measures were conducted for approximately 3,200 households enrolled in the Household Air Pollution Intervention Network (HAPIN) trial. The HAPIN study population included 3,195 pregnant women, their children, and 418 nonpregnant adult women. Personal particulate matter ($PM_{2.5}$), carbon monoxide (CO), and BC were collected. Light absorbing carbon (LAC) was measured as a surrogate for BC. At each measurement visit, questionnaires were administered to collect individual- and household-level information. After baseline exposure visit, households were randomly allotted to either control (continued use of traditional biomass stove) or intervention (use of LPG stove) arm.

Results: The relationship between BC measures using pre-sampled filter scans versus laboratory blank filters as a reference was strong (R^2: 0.97); however, this relationship ($R^2 = 0.25 – 0.26$)

was weak in low exposure settings (1.3 – 2.9 μg/m^3). We identified stove type, kerosene use, study site, whether cooking activity, primary lighting source, other sources of smoke, kitchen location, roof material, participant occupation, hours of stove use, season, temperature, and relative humidity to be significant predictors of personal BC exposures among pregnant women. The contrast in prenatal BC exposures between study arms, post-randomization, differed according to study site, adherence to the assigned study stove, and cooking activity. We observed significant reductions in household air pollution exposure among middle- and older-aged women from the HAPIN LPG intervention.

Conclusion: Our results suggests that an LPG stove intervention can substantially and consistently reduce HAP exposures, including BC, for individuals relying on solid fuels for cooking. Our data also highlights several important factors that can potentially attenuate the effectiveness of an LPG intervention in reducing BC as well as measurement procedures that can provide more accurate estimates of BC exposure.

INDEX WORDS: Exposure Assessment, Household Air Pollution, Randomized-Controlled Cookstove Intervention, Black Carbon (BC), Fine Particulate Matter ($PM_{2.5}$), Carbon Monoxide (CO)

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DEDICATION

This dissertation is dedicated to the Most High. Everything that I have and will accomplish was and is only possible through the blessings that You have bestowed upon me. I am, and will remain, forever grateful.

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TABLE OF CONTENTS

Page

ACKNOWLEDGEMENTS v

LIST OF TABLES x

LIST OF FIGURES xii

CHAPTER

1 INTRODUCTION 1

PROBLEM STATEMENT 1

PURPOSE OF RESEARCH 2

OUTLINE OF DISSERTATION 3

2 LITERATURE REVIEW 5

OVERVIEW 5

BLACK CARBON (BC) 7

MEASUREMENT TECHNIQUES FOR BC 8

KEY SOURCES AND SPATIAL VARIATION OF BC EMISSIONS 12

HEALTH EFFECTS FROM RESIDENTIAL BURNING 15

EFFECTIVENESS OF COOKSTOVE INTERVENTION TRIALS IN REDUCING HAP EXPOSURES 21

FACTORS ASSOCIATED WITH BC EXPOSURE IN LMICs 25

SUMMARY 30

3 COMPARISON OF BLACK CARBON MEASUREMENTS USING FILTER SPECIFIC REFERENCE TRANSMITTANCE TO THOSE USE LAB BLANKS OR AN AVERAGE OF UNLOADED FILTERS 31
ABSTRACT....................32
INTRODUCTION 34
METHODS 36
RESULTS 39
DISCUSSION....................41
CONCLUSION....................43
4 FACTORS ASSOCIATED WITH PERSONAL EXPOSURE TO BLACK CARBON AMONG RURAL PREGNANT WOMEN IN FOUR COUNTRIES....................52
ABSTRACT....................53
INTRODUCTION 55
METHODS 56
RESULTS 61
DISCUSSION....................65
CONCLUSION....................69
5 EXPOSURE CONTRASTS OF ADULT WOMEN DURING THE HOUSEHOLD AIR POLLUTION INTERVENTION NETWORK RANDOMIZED CONTROLLED TRIAL....................81
ABSTRACT....................82
INTRODUCTION 84
METHODS 86

RESULTS 93
DISCUSSION 101
CONCLUSION 108
6 SUMMARY AND CONCLUSIONS 146
SUMMARY 146
CONCLUSIONS 147
FUTURE RESEARCH 148
REFERENCES 149

LIST OF TABLES

Page

Table 3.1: Summary of I_0 values by study site and filter type 48

Table 3.2: Limit of Detections (LoDs) for each method and study site combination 49

Table 3.3: Summary of mean (SD) eBC exposure measures ($\mu g/m^3$) 50

Table 3.4: Model performance statistics for the linear regression between methods 51

Table 4.1: Summary of baseline participant and household characteristics 71

Table 4.2: Summary statistics for personal BC measurements ($\mu g/m^3$) 72

Table 4.3: HAPIN-wide and IRC-specific association between personal BC and household, behavioral, and environmental characteristics 73

Table 4.S1: Pooled post-randomization median (IQR) BC exposures 74

Table 4.S2: Exposure summary and model performance with imputed data 75

Table 4.S3: Sample size (N measures) of comparison groups (Control vs Intervention) after randomization 76

Table 4.S4: Number (%) of baseline and post-randomization measures by treatment arm and kerosene use in India 77

Table 5.1: Household and other adult women participants characteristics at baseline, by IRC and study arm 110

Table 5.2: Summary of personal exposure to $PM_{2.5}$ of other adult women participants by IRC and study group 111

Table 5.3: Summary of personal exposure to BC of other adult women participants by IRC and study group 112

Table 5.4: Summary of personal exposure to CO of other adult women participants by IRC and study group.....113

Table 5.5: Comparison of personal exposures to $PM_{2.5}$, BC, and CO between pregnant and non-pregnant adult women from the same households by study period.....114

Table 5.6: Percent reductions in $PM_{2.5}$, BC, and CO associated with LPG intervention.....115

Table 5.S1: Wearing compliance.....116

Table 5.S2: Overall exposure data completeness of other adult women participants in HAPIN Trial.....117

Table 5.S3: Comparison of personal exposures to $PM_{2.5}$ between pregnant and non-pregnant adult women in the same households by arm, IRC, and study period.....118

Table 5.S4: Comparison of personal exposures to BC between pregnant and non-pregnant adult women in the same households by arm and study period.....119

Table 5.S5: Comparison of personal exposures to CO between pregnant and non-pregnant adult women in the same households by arm and study period.....120

Table 5.S6: Detailed pairwise testing results between non-pregnant and pregnant adult women by IRC, study period, arm, and pollutant.....121

Table 5.S7: Correlation (Spearman's ρ) of non-pregnant and pregnant adult women by pollutant, stove-type, IRC, and study period.....122

Table 5.S8: Correlations (Spearman's) of personal exposure to $PM_{2.5}$, BC, and CO between measurement rounds.....123

Table 5.S9: Correlations (Spearman's) between pollutants by stove type and IRC.....124

Table 5.S10: Percent decreases in $PM_{2.5}$, BC, and CO exposure associated with LPG intervention in Guatemala.....125

Table 5.S11: Percent decreases in $PM_{2.5}$, BC, and CO exposure associated with LPG intervention in India 126

Table 5.S12: Percent decreases in $PM_{2.5}$, BC, and CO exposure associated with LPG intervention in Peru.... 127

Table 5.S13: Percent decreases in $PM_{2.5}$, BC, and CO exposure associated with LPG intervention in Rwanda 128

LIST OF FIGURES

Page

Figure 3.1: Magee Scientific Sootscan model OT21 transmissometer..45

Figure 3.2: Light intensities (IR) for all lab blank and pre-sample filter scans 46

Figure 3.3: Bland-Altman plots 47

Figure 4.1: HAPIN-wide multivariable mixed-effects regression coefficients 78

Figure 4.S1: Percent of missing data for each covariate...79

Figure 4.S2: IRC-specific multivariable linear regression coefficients..80

Figure 5.1: HAPIN other adult women participants personal exposure to $PM_{2.5}$, BC, and CO by study group...129

Figure 5.2: Trends in personal $PM_{2.5}$ exposure 130

Figure 5.3: Estimated effects of the HAPIN LPG stove and fuel intervention on $PM_{2.5}$ exposure… 131

Figure 5.S1: Monitoring wearing compliance 132

Figure 5.S2: Boxplot of personal exposure to $PM_{2.5}$ among non-pregnant adult women participants by IRC, study group and visit...133

Figure 5.S3: Boxplot of personal exposure to BC among non-pregnant adult women participants by IRC, study group and visit 134

Figure 5.S4: Boxplot of personal exposure to CO among non-pregnant adult women participants by IRC, study group and visit 135

Figure 5.S5: Trends in personal $PM_{2.5}$ exposure among the HAPIN non-pregnant adult women participants in each IRC...137

Figure 5.S6: Trial-wide and IRC-specific comparison of personal exposures to $PM_{2.5}$ between pregnant and non-pregnant adult women in the same households by arm and study period… 138

Figure 5.S7: Trial-wide and IRC-specific comparison of personal exposures to BC between pregnant and non-pregnant adult women in the same households by arm and study period… 139

Figure 5.S8: Trial-wide and IRC-specific comparison of personal exposures to CO between pregnant and non-pregnant adult women in the same households by arm and study period… 140

Figure 5.S9: Correlation between log_{10}-transformed $PM_{2.5}$ and BC exposure by stove type and IRC…....141

Figure 5.S10: Correlation between log_{10}-transformed $PM_{2.5}$ and CO exposure by stove type and IRC…....142

Figure 5.S11: Correlation between log_{10}-transformed BC and CO exposure by stove type and IRC……....143

Figure 5.S12: Estimated effects of the HAPIN LPG stove and fuel intervention on BC exposure........ 144

Figure 5.S13: Estimated effects of the HAPIN LPG stove and fuel intervention on CO exposure…. 145

CHAPTER 1

INTRODUCTION

PROBLEM STATEMENT

Household air pollution (HAP) results from the residential combustion of solid fuels (i.e., biomass in the form of wood, dung, charcoal, etc.) for cooking and heating purposes. The proportion of solid fuel users, worldwide, has decreased by 11% since 2010, but marked increases in solid fuel users have been reported in Sub-Saharan Africa (SSA) where an estimated 900 million people still rely on traditional fuel sources for cooking.[1] An emphasis on increasing access to cleaner energy sources have yielded reductions in the percentage of solid fuel users in Asian countries like India (73% to 61%) and China (54% to 36%), although these proportions remain much higher than relative proportions in more developed nations like the United States of America or Japan (both < 1%).[2] The disparity in solid fuel use is primarily driven by both limited access to and infrastructure for clean energy fuels in low- and middle-income countries (LMICs).

Exposure to HAP accounted for 2.3 million deaths and greater than 90 million disability-adjusted life years (DALYs) globally in 2019 with disproportionate adverse health effects in LMICs.[3,4] HAP is associated with adverse respiratory, cardiovascular, pediatric, and maternal health outcomes.[4–9] However, a causal link between HAP and these adverse health outcomes has yet to be established via randomized cookstove interventions largely due to insufficient reductions in HAP exposures.[10,11]

Comprehensive exposure assessments are needed to better understand the determinants of HAP-related exposures in LMICs. Such studies would provide the information needed to implement more impactful cookstove interventions of cleaner burning fuel alternatives.

PURPOSE OF RESEARCH

Our understanding of HAP as a risk factor is hindered due to our limited understanding of the levels and determinants of exposures to HAP and its constituents in LMICs. HAP is primarily characterized via measurements of fine particulate matter with an aerodynamic diameter of ≤2.5 µm ($PM_{2.5}$) and/or carbon monoxide (CO).[12] Yet, repeated measures of personal exposures to these pollutants, as well as black carbon (BC), a constituent of $PM_{2.5}$, are currently lacking in LMICs. BC is a specific marker for combustion related emissions and has been suggested as a stronger predictor for certain health outcomes than $PM_{2.5}$.[13] Herein lies a need for HAP exposure studies that (1) characterizes HAP exposure reductions from cookstove interventions for susceptible populations, (2) elucidate the major determinants of BC exposure in LMIC settings, and (3) assess the potential for increased accuracy of BC measures. **Therefore, my dissertation sought to compare BC measures from the same instrument using two slight variations of the same optical absorption technique; characterize the major factors associated with BC exposure with the intent to understand the role these factors play in modifying the exposure reduction via a liquefied petroleum gas (LPG) stove intervention; and assess the effectiveness of an LPG intervention in reducing HAP exposures among middle- and older-aged women.** My dissertation research has **four** overarching objectives, of which results are presented in **three** separate manuscripts:

1. Compare BC measures from two slight variations of the same optical absorption technique, (Manuscript 1)

2. Assess the association between select factors and BC exposure among pregnant women, (Manuscript 2)
3. Characterize the role these factors play as effect modifiers for the reduction of BC exposures among pregnant women via a liquefied petroleum gasoline stove intervention, (Manuscript 2)
4. Characterize HAP exposures among middle- and older-aged adult women and assess the effectiveness of an LPG intervention in reducing said exposures. (Manuscript 3)

OUTLINE OF DISSERTATION

This dissertation consists of **three** manuscripts, all of which are products from the Household Air Pollution (HAPIN) trial, a randomized controlled LPG stove and fuel intervention conducted in Guatemala, India, Peru, and Rwanda.[14] Chapter 1 states the problem and overall purpose of the research while also providing a general summary of each chapter presented herein. Chapter 2 consists of a background and literature review pertinent to the research. Chapter 3 consists of a third manuscript that will be submitted for publication to the ***Atmospheric Environment***. This paper focuses on comparing BC measurements from slightly modified methods using the same optical absorption technique (Main objective: 1). Chapter 4 comprises of a manuscript that will be submitted for publication to the ***Environmental Science & Technology***. This paper reports BC exposures among pregnant women, factors associated with said BC exposure, and the role these factors play in reducing BC exposures via an LPG intervention (Main objective: 2 & 3). Chapter 5 presents a manuscript that will be submitted for publication to the ***Science of the Total Environment***. This paper characterizes HAP exposures among older adult women enrolled in HAPIN (Main objective: 4). Chapter 6 provides an overall summary and conclusion of this dissertation and gives remarks on possible future directions.

CHAPTER 2

LITERATURE REVIEW

OVERVIEW

The global reduction in the percentage of populations exposed to HAP is driven by aggressive fuel switching campaigns in India and China; however, the percentage of solid fuel users, in these countries, remains substantially higher than that in more developed nations.[2] Even with the reported reductions in solid fuel users, approximately half of the world's population still relies on solid fuel for household energy needs.[2] HAP is a leading risk factor in the global burden disease with disproportionate impacts among children and the elderly in low- and middle-income countries (LMICs).[15] Specific health outcomes associated with HAP include adverse respiratory (asthma, acute respiratory infection in adults and children, chronic obstructive pulmonary disease, lung cancer), cardiovascular (cerebrovascular disease, ischemic heart disease, and cardiovascular mortality), and birth outcomes.[4–9]

Intention-to-treat analyses, in the form of cookstove interventions, have estimated the differences in HAP exposures and related adverse health outcomes between treatment groups to establish a causal link between the two. To date, cookstove interventions of both improved biomass cookstoves and cleaner burning fuel alternatives have largely failed to produce sufficient improvements to health or significant reductions in HAP exposure to levels expected to produce meaningful health benefits.[10,11]

The major health risk-related components of HAP are fine particulate matter with an aerodynamic diameter of ≤2.5 μm ($PM_{2.5}$) and carbon monoxide (CO). Black carbon (BC), the most light-absorbing portion of $PM_{2.5}$, is a by-product of the incomplete combustion of carbonaceous material and is therefore a more specific marker for combustion related sources than $PM_{2.5}$ alone. There are limited epidemiology studies that have assessed adverse health outcomes from HAP-related BC exposures in LMICs. Instead, the studies associating BC to health outcomes such as lung cancer and cardiopulmonary, cardiovascular, and all-cause mortality have been conducted primarily on ambient air in developed nations.[16–18] Measurements of BC exposure are integral for stakeholders that aim to fully characterize the global burden of disease attributable to HAP, especially when considering the potential for BC to be a more robust indicator for adverse health than $PM_{2.5}$.[13,18] Therein lies a need for more exposure assessments in LMICs that accurately characterize the magnitude of as well as the factors influencing exposures to HAP and its constituents. Such studies will provide reliable primary data that underpin global HAP-attributable risk estimates.[16]

BC is a catch all term describing a wide range of carbonaceous particles from incomplete combustion processes. BC is measured in various ways that leverage the unique chemical and physical properties of these particles. Optical methods for BC estimation leverage the light-absorbing properties of BC particles collected on filter media. These methods are known to be more cost-effective alternatives to thermal optical techniques and as such are more practical for exposure studies conducted in resource scarce areas. A uniform method of BC estimation that maximizes the accuracy of filter-based optical measurement techniques is warranted to improve the characterization of exposure response relationships in health risk studies.

Our understanding of the levels of and factors influencing BC exposure in LMICs is limited. Also, there remains uncertainty regarding the level of air pollution reduction needed to reach health relevant targets via cookstove interventions. More comprehensive exposure assessments would provide the information needed to improve our understanding of HAP-related exposure-response relationships.[19] To date, the majority of studies assessing factors associated with BC exposure are cross-sectional in nature and have relatively small sample sizes often limited to a particular city or region. These limitations impact our ability to generalize study findings to other settings. A longitudinal analysis would elucidate our understanding of the spatiotemporal patterns of BC exposures among populations in LMICs, thereby providing the information needed for more impactful cookstove intervention programs.

The following literature review is intended to provide readers with an in-depth but succinct review of the available literature on 1) the recommended terminology for BC-containing aerosols, 2) measurement techniques for BC, 3) key BC emission sources, 4) health effects from residential biomass burning, 5), the effectiveness of cookstove intervention trials in reducing HAP exposures, and 6) factors associated with exposure to HAP and its constituents in LMICs.

BLACK CARBON (BC)

Black carbon (BC), or soot, is an all-encompassing term describing a wide range of carbonaceous particles emitted as a by-product of incomplete combustion. There is no universally accepted definition of BC, but it is typically defined by either its chemical and/or physical properties or operationally defined based on the measurement method used for estimating BC. Combining the technical terminology of both "black" and "carbon" we get a blanket definition of black carbon as the most light absorbing atmospheric carbonaceous particles.[20] However, this term does not represent a pure substance and instead refers to a collection of substances with light absorbing properties. In contrast, elemental carbon (EC) refers to materials that are virtually pure carbon and are not chemically bonded to other elements.[20] For this reason, it has been suggested to use the term "BC" as a qualitative description of light-absorbing carbonaceous (LAC) particles and to instead use the term equivalent black carbon (eBC) for quantitative applications.[21] Following the recommendations of Petzold et al.[21], BC is better suited to be distinguished according to the measurement technique used for estimation.

MEASUREMENT TECHNIQUES FOR BC

Refractory black carbon (rBC), EC, LAC, and soot are all terms that represent BC-containing atmospheric carbonaceous particle components. Each term characterizes different chemical or physical properties of BC-containing particles; therefore, these particle components can be distinguished by the techniques used to measure them. BC has four fundamental physical properties: (1) refractory with vaporization temperature near 4000K, (2) graphitic sp^2-bonded carbon with aggregate morphology, (3) strong visible wavelength-independent light absorption, and (4) insolubility in water and common organic solvents.[21,22] These properties can be leveraged to measure the BC mass content of aerosol particles. A brief summary of techniques used for estimating BC-containing atmospheric particles is provided below:

Refractory Black Carbon

Measures of refractory black carbon (rBC) utilizes the heat resistivity of BC particles and derives carbon mass via laser incandescence (LII) by directly measuring the thermal emission of carbon particles that absorbs the laser energy.[23] More specifically, the particles are introduced to intense radiation and heated to their boiling point (4300 K).[24] At this temperature, BC particles concurrently absorb and emit radiation in an approximately balanced manner.[23] The particle mass is then derived from the peak LII signal which decreases as the particle begins to evaporate and absorb less light.[23]

Elemental Carbon

EC can be measured via thermal optical analysis which employs a two-step process that separates OC and EC in filter samples. These filter based measurement techniques take advantage of the higher oxidation temperatures of EC (~650 - 1100 C) compared to that for OC and inorganic constituents that oxidize at lower temperatures (~350 – 550 C).[23,25] In the first phase, the sample is heated in an inert (O_2-free) atmosphere so that the OC can be converted to CO_2.[23] At this stage, certain OC components may combust to form light-absorbing "charred" materials that increases the absorptivity of the sample. To overcome this, the light reflected from (thermal optical reflectance or TOR) or transmitted through (thermal optical transmission or TOT) the sample is used to monitor the potential darkening of the particle deposits on the filter due to OC charring.[25] In the next phase, oxidizing gas is substituted for inert gas. In this oxidized environment, any leftover charred OC combusts and is considered to be removed when the reflected or transmitted light from the sample returns to its original intensity.[25] EC is directly measured as the remaining carbon on the filter.

Equivalent Black Carbon from Light Absorbing Carbon

LAC refers to atmospheric carbonaceous particles that absorbs light within the visible wavelength spectrum and includes eBC, rBC, and EC.[23,26,27] While the latter two forms of BC employ direct methods of measuring BC, the former employs an indirect method for BC estimation by converting the light absorption of carbonaceous particles to mass by way of a mass absorption coefficient (MAC).

The MAC varies according to wavelength, particle size, source, space, and time, likely representing the influence that the mixing state of aerosols have on the MAC – where the MAC is approximately 1.5 times higher for aged aerosols due to coating than it is for freshly emitted aerosols.[27–30] The MAC can be empirically derived by taking the linear regression slope or average ratio of light attenuation (b_{att}) versus EC.[29] For ambient measures of BC-containing particles, the MAC is estimated to be 10 m^2/g at a wavelength of 550 nm and ranges from 14.9 – 19.9 at 880 nm.[27,31] For freshly emitted particles, however, the estimated MACs are lower. For example, at 550 nm, the MAC is commonly cited to be 7.5 m^2/g, according to Bond and Bergstrom (2006). Liu et al., 2020 reported a non-significantly higher MAC (8.0 m^2/g) at this same wavelength. Presler-Jur et al.[33] observed a range of MAC values (6.9 – 9.4 m^2/g) at 880 nm that varied primarily spatially by region. Garland et al.[34] estimated cookstove source-specific MAC values at 880 nm according to filter type and reported the MAC to be higher for Teflon compared to quartz filters (7.7 vs. 6.4 m^2/g).

There are a multitude of ways to measure particle absorption including photothermal (photo-acoustic and interferometric techniques) and filter-based techniques. Given the scope of my dissertation, I will focus on filter-based particle absorption measures for this review. The darkening of a filter as particles are deposited can be used to measure light absorption of particles. The intensities of light transmitted through unloaded (I_0) and post-sampled filters (I) are commonly used to produce an absorption coefficient according to the Beer-Lambert law,

$$I = I_0 * e^{(-b_{att} * L)}$$

where L is the optical path length incorporating the volume of air sampled (m^3) and the sampled filter area (m^2). According to Presler-Jur et al.[33], the b_{att} can be calculated, in inverse megameters (Mm^{-1}) units, at a specific wavelength after substituting the volume of air and filter area for L as follows:

$$b_{att} = \frac{Area(m^2)}{Volume(m^3)} * ln\frac{I_0}{I} * 10^6$$

The measured b_{att} can then be used to calculate the concentration of eBC by dividing the b_{att} by the corresponding MAC. A major source of uncertainty for filter-based measures is the filter substrate used for measuring particle absorption. For example, an evaluation of Emfab filters found a linear relation between I_0 values and filter weight, where I_0 values increased as the weight of the reference filter increased thereby potentially reducing the light attenuated when compared to a sampled filter.[35] The authors then recommended a correction factor that allows the use of any unexposed Emfab filter as a reference filter and reduces error in estimating attenuation by approximately 10%.[35] For Teflon filters however, findings from Presler-Jur et al.[33] suggest the I_0 for this substrate to be highly variable, but the variability is not systematic and therefore cannot be corrected using a conversion factor. Samples with low mass loadings may be disproportionately impacted due to the uncertainty of I_0 values. For example, Presler-Jur et al., 2017 observed negative eBC for filters with $PM_{2.5}$ mass concentrations up to 20 μg/m^3 using a MAC of 10 m^2/g at 880 nm. Therefore, the authors suggest conducting measurements of I_0 using the pre-sampled filters as opposed to a blank reference filter. To date, there are no studies, field based or otherwise, that evaluate the added benefit to eBC estimations from using filter specific I_0 values. This is a knowledge gap in the literature that my dissertation aims to address.

Another limitation of characterizing BC using filter-based techniques is the assumption that all particles that absorb light at the wavelength measured is associated with BC. Other constituents of carbonaceous atmospheric particles, namely brown carbon (BrC), can absorb light as well and are not fully accounted for when using this technique. BrC is defined as light-absorbing organic atmospheric aerosols typically originating from humic-like substances[26] (i.e., peat, soil, and coal) with a yellowish to light brown hue according to its non-uniform absorption over the visible wavelength range[23] For example, BrC lies in the middle of the absorptivity spectrum where the absorbance efficiency of these compounds is present to a lesser degree within the visible light wavelength range (~400 – 700 nm) but increases sharply within the UV wavelength range (100 – 400 nm).[26] In contrast, EC has high absorbance at longer visible light wavelengths, whereas colorless organic substances such as hydrocarbons are on the low end of the spectrum.[26] The issue of BrC being misrepresented as eBC may be of less importance when measures are conducted at longer wavelengths (i.e., 880 nm) where BrC is not expected to absorb light.[36] Further analysis should be done to elucidate the potential for misrepresentation of eBC as BrC at 880 nm vs. 550 nm. Potential findings may be of importance when trying to discern the source and differential toxicities of atmospheric carbonaceous particles.

KEY SOURCES AND SPATIAL VARIATION OF BC EMISSIONS

Air pollution source apportionment is a quantitative process that identifies emission sources and attributes air quality concentrations to these source contributions. Source apportionment studies provide important quantitative data on BC sources that is crucial for health policy implementation. These methods are typically characterized as either bottom-up or top-down. Bottom-up methods employ detailed emission estimates of anthropogenic sources and inputs this data into air dispersion models that can be used to evaluate control strategies.[37] Bottom-up methods, however, are limited with their reliance on emission inventories that are often uncertain and possibly inaccurate, as well as their limited ability to account for pollution associated with long-range transport.[16] Therefore, top-down methods are more commonly used as they identify and quantify relative source contributions using real-world air monitoring data as a key input in source receptor models.[16] There are generally five different top-down methods including the aethalometer method, macro-tracer method, radiocarbon method, chemical mass balance (CMB) and positive matrix factorization (PMF). The first three methods can only identify two source categories: (1) fossil fuels and (2) biomass burning. CMB and PMF, on the other hand, can identify a multitude of source categories, but require a much more robust dataset than the previously mentioned methods.[16]

The latest literature on BC inventories suggest that there are over 73 different BC emission sources in major sectors that can be generally broken down into natural and anthropogenic sources.[38] Natural sources consist of wildfires (i.e., deforestation fires, forest fires, and savanna fires) and are commonly referred to as open biomass burning sources. Anthropogenic sources include a mix of sectors including transportation (i.e., vehicles using diesel, gasoline, and liquid biofuels), industry (i.e., waste incineration, brick production, coke ovens, etc.), power generation using diesel, coal, waste and other forms of biomass, and residential sources including domestic burning of all kinds of biomass, coal, and waste for household energy needs.

Bond et al.[39] based its global inventory estimates of BC and OC according to fuel use data from 1996. The study found open biomass burning to be the largest contributor of both BC and OC with higher contributions from both savannah fires in Africa and the burning of forests in South America.[39] BC contributions from transport was most dominant in North America, Latin America, and Europe while residential contributions were dominant in Africa and Asia.[39] Beyond residential contributions in Asia, transport and industry provided significant contributions to BC emissions as well.[39] Bond et al.[39] estimated global contributions of BC to be highest for open biomass burning (35%), followed by residential (25%), transport (20%), and industrial (10%) sources.

Briggs & Long.[16] conducted a review of source apportionment methods in Europe and the United States. The authors found contributions from fossil fuels to be much higher than that from wildfires in both Europe (78%) and the U.S. (70%). Within the fossil fuels source category, transportation source contributions were dominant in both Europe and the U.S., followed by industrial and residential contributions. On the contrary, wildfires contribute a dominant proportion (83%) of total BC emissions in Russia with additional contributions from flaring gas (burning of unprocessed petroleum gas from oil production) and on-road transportation emissions.[40] Policies aiming to mitigate emissions from flaring and on-road vehicles (e.g., increased fines for flared methane emissions and increased market access to petroleum gas) have appeared to be successful.[40] In China, estimated contributions of BC was highest for wildfires (42%), followed by residential and transportation sources (24%), and industry (10%).[36] In India, however, BC emissions are dominated by residential sources (47%) and industrial sources (22%).[41]

The most up to date assessment of BC emission trends details emission factors from 1960 to 2017 with a 0.1° x 0.1° spatial resolution.[38] The study found substantial temporal variation in BC contributions among the most common sources. For example, in 1960, transportation contributed a small proportion (5%) of global BC but by 2017, this proportion increased to

26%.[38] During this time span, pollution from residential combustion of coal reduced due to energy switching primarily in Russia (coal to natural gas), China (coal to gas/electricity), and Poland (coal to firewood).[38] Concurrently, BC emissions from residential combustion of biomass, especially wood, has been steadily increasing, likely due to growing rural populations in LMICs.[38] To date, residential biomass burning remains the largest contributor of global BC emissions at approximately 35%.[38] The expected increase in BC emissions from residential biomass burning highlights the importance of characterizing the health risks associated with BC exposures in LMICs.

HEALTH EFFECTS FROM RESIDENTIAL BURNING

Differential Toxicities of BC and $PM_{2.5}$

To date, there are limited reviews and/or meta-analyses of the health effects associated with household air pollution (HAP) exposure from residential biomass burning. Earlier reviews on BC-associated health effects have largely been constrained to more developed nations in North America and Europe[13,18,42], where transportation sources are most dominant. These reviews aimed to tease out the differential toxicities of $PM_{2.5}$ and its constituents as combustion related particles have been posed as a better indicator for adverse health outcomes than $PM_{2.5}$ from non-combustion sources (i.e., crustal species like dust and soil). Overall, a unit increase in BC was found to be a more robust indicator for adverse health effects (reported as mortality, hospital admissions, and emergency department visits) than a corresponding unit increase in $PM_{2.5}$.[13,42] However, effect estimates for BC and $PM_{2.5}$ were comparable when reported as interquartile range (IQR) increases.[13,18,42] This distinction is important because there is less bias in comparing IQR increases for BC and $PM_{2.5}$ since the proportion of BC in $PM_{2.5}$ is highly variable according to source type.[38] Moreover, the effect of BC, after adjusting for $PM_{2.5}$ in two-pollutant models, remained robust while that for $PM_{2.5}$ was consistently non-significant after adjusting for BC.[18] These findings highlight the importance of including BC as an additional indicator of adverse health from exposure to air pollution.

There are two more recent review articles whose objectives were to assess whether BC has a stronger association with cardiovascular (CVD) responses than $PM_{2.5}$; however, these reviews mostly included studies conducted in developed nations.[43,44] Luben et al.[44] evaluated general CVD responses such as CVD emergency department visits, hospital admissions, and mortality; while Kirrane et al.[43] evaluated more targeted responses like autonomic nervous system tone, heart arrhythmia, blood pressure and vascular functions, and ischemia. Both studies observed no consistent pattern of results, suggesting that neither BC nor $PM_{2.5}$ could be referred to as the better indicator of adverse CVD responses.[43,44] These studies, along with those mentioned in the previous paragraph, typically used ambient air pollution measures as inputs which could have major implications for their suggested findings since BC is known to vary more spatially than $PM_{2.5}$.[45] The higher spatial variability of BC likely results in greater measurement error for BC, since ambient measures will likely not capture the full scope of the spatial variability of BC relative to $PM_{2.5}$. The increased measurement error for BC could subsequently lead to a disproportionate attenuation of its association with adverse health effects, relative to $PM_{2.5}$.[46] Therefore, more health risk studies including personal exposures to both BC and $PM_{2.5}$, in LMICs, are needed to fully tease out the differential toxicities of these pollutants from residential biomass burning.

Respiratory Health Outcomes

Gordon et al.[7] provided an extensive review of HAP-related adverse respiratory effects and found HAP to be associated with upper/lower respiratory infections, nasopharyngeal and lung cancer, and chronic lung diseases such as chronic obstructive pulmonary disorder (COPD) and bronchiectasis. Smith et al.[47] conducted a comparative risk assessment for HAP exposure and reported pooled odds ratios (OR) for child acute lower respiratory infection (ALRI) (OR: 1.7; 95% confidence interval (CI): 1.45, 2.18), COPD (OR: 1.94; CI: 1.62, 2.33), and lung cancer from coal (OR: 1.98; CI: 1.16, 1.36) and biomass (OR: 1.18; CI: 1.03, 1.35]) smoke exposure. Chen et al.[48] conducted a meta-analysis of 16 studies and found an increased risk of childhood pneumonia (e.g., ALRI) (OR: 1.66; CI: 1.36, 2.02) from solid fuel use in LMICs. Kinney et al.[49] conducted an exposure-response analysis for 1,141 infants enrolled in GRAPHS and found a positive association between personal exposure to CO and increased risk for pneumonia and severe pneumonia in the first year of life[50], thereby presenting a potential mode of action as impaired lung function may increase the risk of pneumonia in the first year of life.

As part of the Prospective Urban Rural Epidemiology Air Pollution (PURE) study, Wang et al.[51] analyzed 48-hr household and personal measures of BC and $PM_{2.5}$ from 870 individuals across 8 LMICs. Although all exposure response relationships for personal BC included the null, the study found an IQR increase in household BC to be positively associated with both wheeze (OR: 1.20; CI: 1.03, 1.39) and sputum (OR: 1.19; CI: 1.00, 1.41). These findings are in line with that from Coker et al[52], which found HAP indicators-including fuel type, number of hours burning solid fuels, cooking indoors, etc. to be positively associated with persistent cough.[52]

There is ample evidence pointing to an association between exposures to HAP and adverse respiratory outcomes. However, a causal link between the two has yet to be established and experimental analyses of these associations are scarce. The Cardiopulmonary outcomes and Household Air Pollution (CHAP) trial assessed the effectiveness of an LPG stove intervention on cardiopulmonary outcomes among 180 adult women in Puno, Peru. The study found no significant differences in for peak expiratory flow (PEF) and respiratory symptoms between treatment groups. Additionally, Jack et al.[53] evaluated the effectiveness of an ethanol stove intervention on various child health endpoints and found no significant improvements in severe childhood pneumonia among the intervention.

Cardiovascular related health outcomes

Grahame et al.[17] built upon the work in Janssen et al.[13] and WHO[42], by providing a review of health effects associated with HAP in LMICs in addition to that associated with diesel and traffic emissions in more developed nations. Overall, the authors' findings suggest that BC from various sources is causally associated with all-cause, lung cancer, and cardiovascular (CVD) mortality and morbidity.[17] Other adverse CVD endpoints found to be associated with BC include CVD hospital admissions; biomarkers of oxidative stress, inflammation, and vascular function; increased blood pressure, liperoxidation, HRV changes, greater ST-segment depression; and reduced production of t-PA.[17]

Norris et al.[54] enrolled 45 women (ages 25 – 66 years) from rural India and assessed the association between personal BC and blood pressure after adjusting for body mass index, socioeconomic status, and salt intake. The study observed IQR increases in personal BC to be associated with an increase in systolic and diastolic blood pressure from -0.4 mm Hg (95% CI: -2.3, 1.5) to 1.9 mm Hg (95% CI: -0.8, 4.7) and -0.9 mm Hg (95% CI: -1.7, -0.1) to -0.4 mm Hg (95% CI: -1.6, 0.8), respectively.

Fandiño-Del-Rio[55] conducted a cross-sectional analysis of the effect HAP has on biomarkers of inflammation among 180 adult women (ages 25 – 64 years) in Puno, Peru. The study observed positive and negative associations between kitchen BC and the pro-inflammatory markers TNF-α and IL-1β, respectively. The study also found a negative association between kitchen BC and the anti-inflammatory maker IL-10. Additionally, the biomarkers of inflammation were more strongly associated with kitchen area concentrations of BC than $PM_{2.5}$.

Ye et al.[56] conducted an intention-to-treat and exposure-response analysis of gestational blood pressure among the 3,195 pregnant women (ages 18 – 35 years) enrolled in the Household Air Pollution Intervention Network (HAPIN) trial, a randomized controlled trial of LPG gas and stoves in Guatemala, India, Peru, and Rwanda. Unexpectedly, the study found blood pressure to be higher in the intervention group compared to that in the control group while Checkley et al., 2020, found no significant differences in adult women (ages 25 – 64) blood pressure between treatment groups. Many exposure-response estimates for the association between exposures to HAP and blood pressure among adult women also included the null.[57–59] However, Ye, Thangavel, et al.[60] did observe a significant positive association between exposures to carbon monoxide and diastolic blood pressure (DBP) where a 1-log $\mu g/m^3$ increase in CO exposure was associated with 0.36 mmHg higher DBP (95% CI: 0.02 – 0.70).

Adverse Birth and Nervous System Outcomes

There remains limited information on the relationship between BC and adverse birth outcomes, cognitive development, and other central nervous system (CNS) effects.[17] The specific mechanism for deleterious effects to the CNS from air pollutants is the induction of radical oxidative species (ROS) that increases oxidative stress thereby enhancing neuroinflammatory responses that can lead to unusual behaviors such as weak social interaction and communication deficits.[61] Although not BC-specific, both clinical and experimental analysis support the hypothesis that a link exists between atmospheric particles and other forms of CNS impairment such as adult cognition, major depressive disorders, dementia, and Parkinson's disease.[62]

Bové et al.[63] presented a biological mode of action for adverse health from early life exposures to combustion-related particles as BC particles were found to accumulate on the fetal side of the human placenta. The specific mechanism for adverse birth outcomes from HAP exposure remains unclear; however, it has been suggested that oxidative stress, DNA methylation, mitochondrial DNA content alteration, endocrine disruption, and male reproductive toxicity play a major role in which atmospheric particles potentially impact the health of both pregnant women and their fetuses.[64] Generally, exposure to atmospheric particles is associated with low birthweight (LBW), preterm birth (PTB), and stillbirth.[64] A recent global assessment of the exposure-response relationship between HAP and birth outcomes (n = 3,060 live births in Guatemala, India, Peru, and Rwanda) observed a stronger association for an IQR increase in BC, relative to that for $PM_{2.5}$, for both birthweight and weight-for-gestational age z-scores.[5] Similar to findings from two-pollutant models stated above[13,18], the exposure-response relationship was more robust for BC than it was for $PM_{2.5}$ as the confidence interval for the effect estimate for $PM_{2.5}$ included the null.

Summary

In totality the previously mentioned articles all point to strong associations between exposure to HAP, and in some instances BC, from residential solid fuel burning and numerous adverse cardiovascular and respiratory effects with less information on the relationship between HAP exposure and adverse birth and CNS outcomes. Many studies included in this review point to BC as a better indicator for health risks than, yet there remains no clear consensus of the relative toxicities of BC and $PM_{2.5}$, likely due to the limited amount of two-pollutant health risk modelling studies that use personal measures of BC and $PM_{2.5}$ as inputs. Overall, the majority of studies mentioned in this section are observational and suggest an association between HAP and adverse health, yet evidence from experimental analyses is lacking. To fill this knowledge gap, intention-to-treat analyses, in the form of randomized control cookstove intervention programs, have been implemented in LMICs.

EFFECTIVENESS OF COOKSTOVE INTERVENTION TRIALS IN REDUCING HAP EXPOSURES

In 2019, HAP accounted for 2.3 million deaths and greater than 90 million disability-adjusted life years (DALYs) with disproportionate adverse health effects in low- and middle-income countries (LMICs) where people predominantly use traditional biomass cookstoves (TCS) for heating and cooking.[3] In 2021, the World Health Organization (WHO) proposed an updated air quality guideline (AQG) for $PM_{2.5}$ (5 $\mu g/m^3$).[65] No such guideline exists for BC exposures. The purpose of the AQG is to provide an evidence-based metric to inform legislation and policy where exceedance of the proposed threshold is associated with important risks to public health. Due to limited resources and infrastructure, cookstove interventions are not expected to reduce personal $PM_{2.5}$ levels below the AQG in LMICs, therefore the WHO provided interim targets (IT) to guide air pollution reduction efforts.[65] For example, IT-1, IT-2, IT-3, and IT-4, for annual $PM_{2.5}$, are associated with a 24%, 16%, 8% and 4% respective increases in mortality relative to the AQG. These interim targets are more attainable in LMICs and can be used as a metric to gauge the effectiveness of cookstove interventions in reducing exposures to HAP. The following provides a brief summary of the literature on the effectiveness of cookstove trials in reducing exposures and adverse health outcomes associated with pollutants from cookstove sources.

The most recent review and meta-analysis on the effectiveness of cookstove trials in reducing personal exposures to $PM_{2.5}$ consisted of 13 studies with 20 estimates in total.[11] Of these 13 studies, nine included improved biomass cookstoves (ICS) with vents (ICS-vent), two included advanced biomass cookstoves (ACS) with a gasifier, one included electric stoves, and eight included liquefied petroleum gasoline (LPG) stoves. The weighted post-intervention mean reduction was 75.5% for ICS with vents, 32.3% for ACS, 69.2% for electric stoves, and 74.3% for LPG stoves. None of the post-intervention exposures for both ICS and ACS stoves were close to the annual WHO IT-1 target (35 $\mu g/m^3$), while six of the eight studies implementing LPG stoves had exposures at or below WHO guidelines. These findings from Pope et al.[11] highlight the potential for clean burning alternatives to provide the exposure contrast needed for significant health benefits. Therefore, I focus on cookstove interventions of cleaner burning fuel alternatives for the remainder of this section.

Alexander et al.[66] assessed the impact of an ethanol cookstove intervention on birth outcomes among 324 pregnant women in Ibadan, Nigeria. The study found the intervention group to have significantly higher birthweight than the control group by 88g; however, the study was not powered to detect a difference in mean birthweight less than 250g. Moreover, there was no significant difference in $PM_{2.5}$ exposure levels between treatment groups. Therefore, the significant difference in mean birthweight between groups cannot be attributed to lower exposures among the intervention group. The authors cited significant seasonal effects (i.e., noticeably higher $PM_{2.5}$ exposures in the dry versus summer months, $p < 0.001$) and high ambient air pollution as potential reasons why the study did not detect significant differences in exposure levels between groups.

Chillrud et al.[67] evaluated the effect of an LPG stove intervention on exposures among 1414 pregnant women in rural Ghana. The intention to treat analysis for this intervention trial found no significant difference in risks for pneumonia or low birth between treatment groups.[68,69] This finding is in line with Chillrud et al.[67], which estimated only a 32% reduction in $PM_{2.5}$ exposures due to the intervention with approximately 66% of the post-intervention mean maternal 48-h $PM_{2.5}$ exposures being above the annual WHO IT-1 guideline (35 $\mu g/m^3$). Additionally, more than 85% of the mean maternal 24-h $PM_{2.5}$ exposures exceeded the daily WHO IT-4 guideline (25 $\mu g/m^3$) and virtually all mean exposures were above the annual 5 $\mu g/m^3$ guideline. The authors cited ambient air pollution, population density, and stove stacking (i.e., the use of clean fuels in the control group) as potential reasons why the intervention did not reduce maternal exposures to health relevant standards.

As a part of the Cardiopulmonary outcomes and Household Air Pollution (CHAP) trial, Checkley et al.[70] evaluated the effectiveness of an LPG stove and fuel distribution, education, and behavioral messaging intervention on cardiopulmonary outcomes among 180 adult women in Puno, Peru. To our knowledge, this is the first clean fuel intervention study to report BC measures in conjunction with $PM_{2.5}$. The study found no evidence of improved benefits to blood pressure, lung function, or respiratory symptoms from the intervention; yet post-randomization personal exposures to both $PM_{2.5}$ (30 vs. 98 $\mu g/m^3$; Kolmogorov-Smirnov $p < 0.001$) and BC (2 vs. 16 $\mu g/m^3$; $p < 0.001$) were lower among intervention participants when compared to control participants, respectively. Fandiño-Del-Rio[71] conducted a follow-up exposure assessment by conducting a one-year crossover trial of this study population. In the second year of the study, after the control group was provided the intervention, the exposure reductions in the control group were similar to those found in the intervention during the first year of the intervention.[71] Findings from both Checkley et al.[70] and Fandiño-Del Rio et al.[71], show the ability of a free LPG stove and fuel intervention, coupled with behavioral messaging, to reduce HAP exposures to levels below WHO guidelines. However, the reduction in HAP exposures did not coincide with significant reductions in primary health outcomes between treatment groups.[70] The authors cited stove stacking, low population density, high elevation, and low ambient air pollution as key limitations of the study that may impact the portability of study results to other settings as well as the effectiveness of the intervention in providing meaningful health benefits.

Clasen et al.[72] reported health effects from the Household Air Pollution Intervention Network (HAPIN) trial, a randomized, controlled trial of LPG stoves and fuels enrolling 3,200 pregnant women in Guatemala, India, Peru, and Rwanda. The study found no significant difference in birthweight between treatment groups. This finding is particularly interesting given the contrasts in median post randomization exposures between groups reported for $PM_{2.5}$ (-61%), BC (-62%), and carbon monoxide (CO; -81%).[73] In total, 69% of $PM_{2.5}$ exposures were below the WHO IT-1 target, highlighting the potential for LPG interventions to reduce exposures to levels at or below health relevant targets.

Given the results of aforementioned studies, there is little evidence of a causal relationship between HAP and various adverse health effects, even with the introduction of cleaner burning fuel alternatives. For the trials conducted in Ghana and Nigeria, the non-significant differences in exposures between groups most likely attributed to stove stacking, high ambient air pollution, and population density.[66,67] CHAP and HAPIN, on the other hand, significantly reduced exposures between treatment groups; however, no significant differences in health outcomes were observed. A subgroup analysis in HAPIN, suggests that the timing of the intervention can impact its effectiveness as the mean difference in birthweights were greater when the intervention was implemented before 18 weeks of gestation (+33.8 g; 95% CI: -2.6 – 70.2) compared to when the intervention was implemented after 18 weeks of gestation (+5.3 g; 95% CI: -31.0 – 41.7).[72] Although the CI for the sub-group with the earlier intervention timing included the null, future cookstove interventions may want to consider ways to implement interventions earlier during pregnancy. Furthermore, there still remains an uncertainty in the exposure contrast needed to exhibit clear benefits to health from cookstove intervention. To maximize the effectiveness of cookstove interventions we must fully understand the factors associated with HAP exposures. Therefore, I provide a synopsis of factors associated with personal exposures to $PM_{2.5}$, CO, and BC, -related BC in the following section.

FACTORS ASSOCIATED WITH BC EXPOSURE IN LMICs

This section includes seven observational studies that assessed the influence that certain individual- and group-level factors have on BC exposures in LMICs via regression analysis. The majority of these studies were conducted in Asia (3) followed by Latin America (2) and Sub-Saharan Africa (SSA) (1). There was only one global study that explored the relationships between BC and select community, household, and behavioral factors.

Asia

Lee et al.[74] analyzed 2,246 24-h BC measurements from 787 adults in China. Daily BC exposures ranged from 0.00 – 12.0 μg/m^3 with significantly lower averages (-78%, 95% confidence interval CI: -[84.2, -69.3]) during the non-heating (1.1 μg/m^3, CI: [1.0, 1.1]) versus heating (1.7 μg/m^3, CI: [1.6, 1.8]) season.[74] The study also found BC exposures to be lower when comparing non-smokers with no smokers in the home to smokers (-12.8%, CI: [-24.2, 0.3]), exclusive clean fuel users to solid fuel users (-14.8%, CI: [-23.1, -5.6]), use of an outdoor solid fuel heating stove to that of an indoor solid fuel heating stove (-19.9%, CI: [-35.0, -0.5]). BC exposures increased by 7.9% (CI: 6.6, 9.2) and 5.5% (CI: 4.1, 7.0) for every 10 μg/m^3 and 1% unit increase in outdoor $PM_{2.5}$ and relative humidity, respectively. The model explained 17% of the variation in personal BC exposures.

Sanchez et al.[75] collected 569 BC exposure samples from 349 adults (207 men and 142 women) in India . Geometric means of personal BC exposures were slightly higher among women (6.06 μg/m^3 [standard deviation (SD): 9.63]) than men (4.61 μg/m^3 [SD: 7.04]). Predictive models for women highlight the effect cooking and socioeconomic status have on exposure. For example, women with biomass primary stoves had 62% higher BC exposures than women with clean stoves, BC exposures increased by 20% for every hour spent cooking with biomass, and BC exposures were 26% lower for women in households that owned a motorcycle compared to those that did not. For men, BC exposures were positively associated with their occupation, stove type used in the home and ambient $PM_{2.5}$ at their residence. The models explained 10% and 57% of the variation between men and women, respectively and approximately 20% of the variation in personal BC exposures was attributable to the variation between all participants.

Downward et al.[76] collected 414 personal particulate matter absorbance (PM_{abs}) measurements from 163 homes in China as proxies for BC estimation. Wood was found to have the highest PM_{abs} compared to all other biomass fuels including different types of coal and plant products. Portable stove users had lower PM_{abs} exposures than those with unventilated stoves.

Personal measures were higher in winter months compared to summer months and homes without stairways had lower PM_{abs} than those with stairways in the main cooking room. The model explained 17% of the variation in personal PM_{abs}.

Sub-Saharan Africa

Curto et al.[77] analyzed personal BC measures from 187 adult women in semi-rural Mozambique. Mean (SD) 24-hr personal BC exposure was 15.3 μg/m^3 (19.4) and ranged from 0.4 – 108.8 μg/m^3. Cookstove type was not shown to be a predictor of personal BC due to the lack of variation in cookstove types in this study population. Instead, key predictors of personal exposure to BC were lighting source, kitchen type, ambient EC levels, and temperature. Women who used kerosene as their primary lighting source had 81% (CI: 34, 147) higher BC exposure than women that used electricity as a primary lighting source. Women with enclosed or partially enclosed kitchens had 61% (CI: 17, 122) higher personal BC than those with open air kitchens or those that did not have a kitchen at all. Personal BC exposure increased by 44% (CI: 11, 87) for every five degree Celsius and increase in temperature and decreased by 24% (CI: 0.7, 42) for every 1 μg/m^3 increase in ambient EC. Lastly, the authors observed a modest, yet significant, negative association between personal BC exposure and unit increase in the number of children in the household (-9%, CI: [-16, -2]). Overall, the multivariable model explained 21% of the variation in personal BC.

Latin America

Fandiño-Del-Rio et al.[78] collected personal BC samples from 180 adult women in Puno, Peru. Mean BC exposures were significantly lower for women with LPG (-28%, CI: [-45, -7])

and chimney (-40%, CI: [-60, -9]) stoves compared to those with biomass and unventilated stoves, respectively. Women with metal roofs had higher BC exposures (57%, CI: [23, 99]) than women with roofs made with straw materials. Women with kitchens with adjacent wall to the main residence had higher BC exposures (31%, CI: [3, 66]) compared to those with kitchens that don't have walls adjacent to the main residence. BC exposures increased by 19% (CI: [1, 40]) for each additional open window in the residence. The model explained 17% of the variability in personal BC exposures.

Witinok-Huber et al.[79] analyzed six repeated measures of personal BC from 228 adult women enrolled in a stepped-wedge randomized controlled cookstove intervention trial in rural Honduras. BC exposures ranged from 0.01 – 2364 $\mu g/m^3$ with higher geometric mean (GSD) exposures among traditional (11.6 $\mu g/m^3$ (4.5)) compared to both Justa (4.2 $\mu g/m^3$ (6.2)) and mixed (3.3 $\mu g/m^3$ (6.4)) stove users. On average, traditional stove users had 107% (CI: [79 – 139]) higher BC exposures than Justa stove users. Women with no electricity in the household had 26.8% (CI: [3, 44]) higher BC exposures than those with electricity in the home. The highest category of stove use time (>80%) was associated with elevated BC exposures compared to the lowest category of time (0 – 60%) and BC exposures were lower in the rainy season compared to the dry season.

Global

Wang et al.[80] analyzed 48-hr BC measures from 1,187 people across 8 LMICs in Bangladesh, Chile, China, Colombia, India, Pakistan, Tanzania, and Zimbabwe. The study observed an increasing gradient in household BC according to cooking fuels where, compared to gas, BC increased 53% (CI: 30, 79) for coal, 142% (CI: 117, 169) for wood, and 190% (CI: 149,

238) for other biomass. Personal BC was negatively associated with increasing socioeconomic status as individuals in the high wealth index category had 26% (CI: 9, 40) lower personal BC exposures than individuals in the low wealth index category. Personal BC increased by 4% (CI: 2, 7) for every hour spent in the cooking area. Personal BC was 40% (CI: 29, 49) lower in summer months compared to winter months and increased by 3% (CI: 2, 3) for every 10 $\mu g/m^3$ increase in ambient $PM_{2.5}$. Overall, the model explained 54% of the variation in personal BC.

Summary of Factors Associated with Personal BC

As expected, cookstove and fuel use are the most consistent predictors of personal BC exposure. Beyond contributions from these sources, certain house characteristics and behavioral patterns have been shown to influence BC exposures as well. Housing characteristics like the type of kitchen, location of the cookstove, roof type, and other type of dynamics that impacts the ventilation of the home like (e.g., number of windows) point to how the built environment plays a role in an individual's exposure pattern. Additional factors like access to electricity (or lack thereof), lighting source, time spent cooking or in the cooking area, temperature, and humidity may highlight how a person's behavior and time activity patterns influences their exposure. Overall, the predictive models including estimates of ambient and/or area measurements of exposures performed much better than those that solely used questionnaire data to predict exposures. Only three out of the seven studies included in this synopsis assessed repeated measures of BC exposure and each was relegated to one region of the world. The global study, arguably the study with the best portability of results, was cross-sectional. Therein lies a need for a longitudinal analysis of the many space- and time-dependent factors that impact exposures to BC in HAP settings on a global scale.

SUMMARY

In summary, there is no universally accepted definition for the term BC. Instead, BC is typically characterized by the techniques used to measure it. The light absorbing properties of BC provides a cheap, cost-effective, and more practical way to measure its exposure levels, as eBC, in LMICs where residential biomass burning is most prevalent. Measures of personal BC exposure will become more paramount in health risk studies since these measures have consistently been shown to be more robust as predictors of adverse health than $PM_{2.5}$ alone. To date, cookstove intervention trials have not provided the evidence needed to establish a causal link between residential biomass burning which begs the questions, “How low can exposures be reduced from cookstove interventions in LMICs?” and “Is this a reduction enough to provide significant benefits to health?”. Moreover, few cookstove interventions have assessed personal exposures to BC. As stakeholders continue to grapple with the aforementioned questions, two things are certainly clear: (1) more comprehensive exposure assessments are needed to provide a better understanding of the levels and drivers of BC and (2) health risk studies can benefit from more accurate BC exposure measures.

CHAPTER 3

COMPARISON OF BLACK CARBON MEASUREMENTS USING FILTER SPECIFIC REFERENCE TRANSMITTANCE TO THOSE USING LAB BLANKS OR AN AVERAGE OF UNLOADED FILTERS[1]

[1] Campbell D, Johnson M, Pillarisetti A, Piedrahita R, Balakrishnan K, Peel J, Underhill L, Steenland K, Rosa G, Kirby M, Diaz-Artiga A, McCracken JP, Clark M, Waller L, Thompson LM, Sarnat JA, Nicolaou L, Kearns K, Kremer J, Mollinedo E, Checkley W, Clasen TF, Naeher LP & the Household Intervention Network (HAPIN) Trial Investigators. To be submitted to Journal of Atmospheric Environment.

Abstract

Background

Equivalent black carbon (eBC) measures from filter-based optical techniques compare light transmission intensities between loaded (I) and unloaded (I_0) filters as a measure of the light-absorbing mass on the filter. Conventionally, I_0 values are estimated by taking the average transmission of many unloaded filters. However, the transmission between individual unloaded Teflon filter media is highly variable, implying a need to evaluate how filter specific I_0 values may impact eBC measurements. In this study, we assess the influence that different methods of I_0 estimations have on eBC.

Methods

We analyzed 5,379 15mm Teflon filters collected from Enhanced Children MicroPEMs (ECMs) used to measure 24-hour personal exposure in the Household Air Pollution Intervention Network (HAPIN) trial. We compared eBC measurements using filter specific I_0 values (Method 1) to those using three other methods of I_0 estimation: (1) an unloaded lab blank from a given scanning session (Method 2), (2) the average of all pre-sample filter scans (Method 3), and (3) the average of all scans of unloaded lab blanks (Method 4). We assessed the agreement between Method 1 and the alternative methods using Bland-Altman analysis. We also compared the relationship between Method 1 and the alternative methods across the complete measurement range and after stratifying exposure data into quartiles according to Method 1 eBC exposures. Paired observations for the alternative methods were placed into Method 1 exposure quartiles regardless of exposure level.

Results

Mean (SD) personal eBC exposure using Method 1 was 7.8 μg/m^3 (5.9). Overall, the average difference (limits of agreement) between the Method 1 and Methods 2, 3, and 4, was 0.7 μg/m^3 (-1.4 – 2.8), 0.1 μg/m^3 (-1.9 – 2.1), and 0.7 μg/m^3 (-1.4 – 2.7), respectively. The performances of linear regression models between Method 1 and all other methods were moderate to strong (R^2 range: 0.42 – 0.93) in the second, third, and fourth quartiles; however, the models in the first quartile (eBC range: 1.3 – 2.9 μg/m^3), performed poorly (R^2 = 0.25 – 0.26) with error approximately 25% of the mean.

Conclusion

Our findings suggest that I_0 values from lab blanks or an average of unloaded filters result in an over-reporting of eBC measures compared to those from filter specific I_0 values. Although eBC measurements from archived filters are sufficient in many settings, the additional effort of analyzing filter media before sampling adds to the accuracy of BC estimations, particularly in lower concentration settings.

INDEX WORDS: Household Air Pollution, Exposure Assessment, Comparison, Intervention, Black Carbon, Biomass, Liquefied Petroleum Gas

Introduction

Health-based air quality guidelines have long been established for aerosol particles primarily characterized via fine particulate matter ($PM_{2.5}$) measurements.[65] The most light-absorbing fraction of these particles, black carbon (BC), is a by-product of incomplete combustion and has been gaining regulatory attention due to its detrimental impact on both climate [22] and human health.[42] BC can be estimated directly through thermal desorption of elemental carbon (EC) or incandescence from refractory black carbon (rBC) and indirectly through the conversion of light absorption of particulate mass measurements to estimate equivalent black carbon (eBC).[81]

The residential combustion of polluting fuels (e.g., kerosene, coal, or biomass such as wood, dung, and charcoal) contributes a substantial proportion (~35%) of global BC emissions.[38] Current trends suggest that these emissions will likely increase as a result of growing populations rural in low- and middle-income countries (LMICs) where the daily use of polluting fuels for household needs is most prevalent. Field-based measures of BC concentrations and exposures in LMICs are becoming more important in fully estimating the burden of disease from residential biomass burning, health benefits associated with switching to cleaner-burning fuel alternatives, as well as the contribution of this pollutant source to global warming. Direct methods of BC estimation, like thermal desorption, are costly and resource-intensive, limiting the practical use of these methods in exposure studies in LMICs where residential combustion of biomass is most prevalent.

Particle light attenuation (b_{att}) measurements on polytetrafluoroethylene (PTFE, Teflon™) filter media and EC measured on quartz filters correlate well with each other ($\rho > \sim 0.80$), demonstrating the potential for b_{att} measures to serve as a quick, efficient, and cost-effective way to indirectly measure EC (as eBC) without destroying filter media.[29] Light attenuation can also be reported in attenuation of light (ATN) units at a specific wavelength where the intensity of light through an unloaded filter (I_0) is compared to that through a loaded filter (I) (equation 1). The b_{att} is then estimated with the ATN, the volume of air sampled (m^3), and the sample area on the filter[33] (equation 2). The mass concentration of eBC ($\mu g/m^3$) is then calculated by dividing b_{att}, in inverse megameter (Mm^{-1}) units, by the mass absorption coefficient $\boldsymbol{\sigma}_{att}$ ($m^2/\mu g$) which is derived by taking the slope of the linear regression of b_{att} on EC measurements [29] (equation 3).

1) $\mathrm{ATN} = \ln\left(\frac{I_0}{I}\right) * 100$
2) $b_{att}(Mm^{-1}) = \frac{Area(m^2)}{Volume(m^3)} * ATN * 10^4$
3) $[eBC]\left(\frac{\mu g}{m^3}\right) = b_{att}(Mm^{-1})/\sigma_{att}\left(\frac{m^2}{\mu g}\right)$

To our knowledge, few exposure studies in LMICSs have measured I_0 using pre-sampled filters.[82] Instead, most exposure studies measure I_0 as the average intensity of light transmitted through many unloaded filters. The average light transmission value of unloaded filters; however, does not likely represent an individual Teflon filter's I_0 value because individual filters are heterogeneous, with light intensities (IR) that can vary by as much as 60,000 units.[33] Presler-Jur et al. recommended that, for BC estimation from newly collected filters, a filter-specific I_0 measurement should be made (i.e., using the pre-sampled filter to estimate I_0) to provide a more comparable measure of light attenuated when passing through the post-sampled filter. To date, there are no field-based studies that assess the potential for improved accuracy of eBC measures from filter-specific I_0 values.

To address this knowledge gap, we compared personal eBC measures from the Household Air Pollution Intervention Network (HAPIN) trial, which used filter specific I_0 values for eBC measures in Guatemala, Peru, and Rwanda, to corresponding eBC measures using other methods to estimate I_0. The findings from this study provide information on the added benefit of using filter specific I_0 values, as opposed to using I_0 values from blank lab filters or the average of unloaded filters to estimate eBC.

Methods

Study Setting and Participants

The study design, site descriptions, and related exposures are all described in detail elsewhere.[14,73,83–86] Briefly, we conducted a randomized controlled trial to evaluate the effectiveness of a liquefied petroleum gas (LPG) fuel and stove intervention versus the continued use of a traditional biomass cookstove (Clasen et al., 2020). The trial had study sites located in Guatemala, India, Peru, and Rwanda. In each site, ~800 households were enrolled (split equally between control and intervention arms) totaling 3,195 pregnant women**.** Study site selection criteria included but were not limited to population density, fuel use, socioeconomic status, and other sources of ambient and/or indoor air pollution.

Exposure Assessment

Aerosol particles were collected onto Teflon filters using the RTI Enhanced Children's MicroPEM^{TM} (ECM, RTI International, Research Triangle Park, USA) to measure time weighted gravimetric $PM_{2.5}$. The ECM is small and lightweight and uses a 2.5-micron size-cut impactor at a flow rate of 0.3 liters per minute for gravimetric measures. The ECM also measures real-time

$PM_{2.5}$ concentrations using a nephelometer and logs temperature, relative humidity, and triaxial accelerometry. Personal exposure measures among pregnant women were collected at baseline (gestational age between 9-20 weeks) and at two follow up periods after randomization at 24-28 (post-intervention visit 1, "P1") and 32-36 (post-intervention visit 2, "P2") weeks of gestation.[73] All participants were asked to wear aprons fitted with an ECM in their breathing zone. If the participants were to conduct any activities that may damage the equipment (e.g., sleeping, bathing, heavy washing), they were instructed to keep the apron with the ECM nearby but not on their person. Field blank filters were collected for approximately 3% of all samples to adjust for mass changes associated with filter handling and processing.

Light Absorption Measurements

We used the Magee Scientific Sootscan model OT21 transmissometer (Magee Scientific Corporation, Berkeley, CA) **(Figure 3.1)** to measure the absorbance of a sampled filter relative to a blank reference filter at 880 nm wavelength. The Sootscan OT21 has a filter tray with two slots. The top slot holds the adapter for the blank reference filter, and the bottom slot holds the adapter for the sample filter. Sandwiched between either adapter is the filter on top of a diffuser. Both the filter and diffuser were made of the same filter substrate material.

Filters were analyzed at the University of Georgia (UGA, Athens, GA, USA) before deployment to the study sites in Guatemala, Peru, and Rwanda. **Figure 3.2** shows the IRs for all reference filter scans. The intensity of light transmitted through the pre-sampled filter was used as the filter specific I_0 value (Method 1: gray dots). Method 1 represents our "gold standard" for I_0 estimation. After filters were returned to UGA for post-sampling, lab blank reference filter scans were used to estimate I_0 values (Method 2: black dots). We also used the average of all pre-sample (Method 3: red line) and lab blank (Method 4: blue dashed line) filter scans as estimates for I_0. We did not calculate site-specific average I_0 values for Methods 3 and 4 since all filters were scanned before deployment with the same instrument at UGA. For all study measurements, we used a derivation of equations 1 – 3 to estimate eBC exposures. First, we calculated the ATN using each method of I_0 estimation (Methods 1 – 4). ATN values below 1 or above 125 were removed from our analyses upon recommendation from Magee Scientific [87]. We then multiplied the ATN by both the effective filter area (0.000079 m^2) and the unit conversion constant (10^4) and divided by the $\boldsymbol{\sigma}_{att}$ reported in Garland et al., 2017 (13.7 $\mu g/cm^2$ or 7.6e-6 $m^2/\mu g$) for Teflon filters collected from similar sources to estimate the mass deposited on the filter (μg). Concentrations of eBC ($\mu g/m^3$) were then calculated as the mass deposited on the filter (μg) divided by the total sample volume (m^3) collected over the 24-hour sampling period. Data from India were not included in this manuscript due to a slightly modified Sootscan procedure conducted at Sri Ramachandra Institute for Higher Education and Research (SRIHER, Chennai, India) that impacts the direct comparison of measures between India and the other study sites.

Statistical Analysis

We applied the conventional approach for estimating site-specific limit of detections (LoDs), in μg units, as three times the standard deviation (SD) of field blank results.[88] Values below the LoD were replaced by the $LoD/\sqrt{2}$. We applied all further analyses to site-specific LoD-adjusted measures. We conducted a descriptive summary to characterize the between method variations in eBC exposures and Bland-Altman analyses[89,90] to estimate the average differences (mean bias) and limits of agreement (LoA) between method-specific measurements. We also used paired t-test to test the null hypothesis that the means of eBC for methods 2 – 4 were the same as that for

Method 1. Finally, we evaluated the relationship between measures using Method 1 and those using methods 2 – 4 via linear regression analysis of non-transformed eBC exposures across the entire measurement range as well as after stratifying the data into quartiles according to Method 1 eBC exposures. Paired observations for the alternative methods were placed into Method 1 quartiles regardless of exposure level. We performed all analyses in R (versions 3.6 and 4.0, R Foundation for Statistical Computing, Vienna, Austria).

Results

Summary of Filter I_0 Values

A summary of I_0 values from all filters from the current study is provided by study site and filter type in **Table 3.1**. There is much variation in I_0 values for unloaded filters as the SD for all pre-sample and field blank filters were above 9,000 units with the difference between minimum and maximum I_0 values being upwards of ~68,000 units. The SDs of lab blank filter scans were considerably smaller than that for the other filter types because only two lab blank filters were used throughout the duration of the study period.

Limit of Detection Analysis

Method- and site-specific LoDs (μg) are provided in **Table 3.2**. Across all study sites, we observed consistently higher LoDs for Methods 2 – 4, compared to those for Method 1.

Summary and Agreement of Method-Specific eBC Exposures

Method-specific personal eBC exposures and agreement between methods are provided in **Table 3.3** and **Figure 3.3**, respectively. The overall Method 1 mean (SD) eBC exposure was 7.8 μg/m^3

(5.9) with noticeably higher exposures among pregnant women in Guatemala and Rwanda compared to women in Peru. For all measures, the mean eBC exposures were higher for Methods 2 – 4 compared to Method 1 (all $p < 0.001$), with mean differences (limits of agreement: LoA) of 0.7 μg/m^3 (-1.4 – 2.8), 0.1 μg/m^3 (-1.9 – 2.1), and 0.7 μg/m^3 (-1.4 – 2.7), respectively **(Figure 3.3)**. This trend was consistent within each study site for both methods using lab blanks (Method 2 & Method 4). However, there was no significant differences in mean eBC exposures between Method 3 and Method 1 in Guatemala and Rwanda (both $p > 0.05$), while the mean exposure level for Method 3 was higher in Peru compared to that for Method 1 ($p < 0.001$). Visual inspection of the Bland-Altman plots shows that there is no evidence of proportional bias between methods: the average difference between methods is constant and does not change as exposures increase.

Method Comparisons via Linear Regression

Given the systematic bias (overreporting eBC measures) associated with using lab blanks or an average of unloaded filters for I_0 estimations, compared to using filter specific I_0 values, we assessed the linear relationship between eBC exposures from Method 1 and all other methods after stratifying the data into quartiles to gauge the relative contribution of the systematic bias to measurement error at different levels of exposure. **Table 3.4** provides results from the linear regression models showing the relationship between Method 1 and methods 2 – 4. Overall, the linear regression models between Method 1 and all other methods fit well, as adjusted R^2 values were high (all 0.97) with RMSEs ranging from 1.0 – 1.1 μg/m^3. RMSEs decreased as exposures got smaller in magnitude; however, it is clear that the relative fit of the models decreased substantially in turn, as R^2 values went from 0.91 in the fourth quartile to as low as 0.22 in the first quartile. Our results show that, in the lowest quartile of Method 1 eBC exposures (range: 1.3 – 2.9 μg/m^3), our models do not perform well and that all other methods have errors that are approximately 25% of the mean in the lowest quartile.

Discussion

The variation between individual Teflon filters in the current study was similar to that reported in Presler-Jur et al[33] as our I_0 values varied by as many as 68,000 units. The LoDs using Methods 2 – 4 were consistently higher than that using Method 1, reflecting the increased uncertainty of I_0 values estimated from session-specific lab blank or the average of unloaded filter scans[33] versus the true I_0 value estimated by scanning the filter media before sampling. Our LOD analysis results suggest that there is less variance in ATN units when the field blank was compared to itself (within-filter variance) than when it was compared to lab blanks or an average of unloaded filter scans (between-filter variance). Our current findings provide further empirical evidence that conducting pre-sample analysis of filter media allows us to account for the variability in transmittance between individual Teflon filters[33] resulting in more accurate representations of I_0 values as well as lower LoDs. We observed consistent overreporting of eBC exposures for all alternative methods of I_0 estimation, relative to Method 1. We also observed the relationships between Method 1 and all other methods to be moderate to strong across most exposures; however, these relationships were weak in the lowest eBC exposure quartile range.

To the best of our knowledge, the current study is the first field-based study to compare eBC estimations using pre-sampled (Method 1) versus session specific lab blanks, an average of unloaded filters, or an average of lab blanks (Methods 2 – 4) as a reference for the absorbance of post-sampled Teflon filters. Our findings have important implications for future exposure studies like those conducted in cookstove intervention trials in LMICs that aim to accurately characterize personal BC across a wide range of exposures. In instances where intervention programs are conducted by institutions with limited resources, Methods 2 – 4 provide a way to employ about half the laboratory labor hours as Method 1. However, the added effort of conducting pre-sample filter scans reduces the measurement error by accounting for the inherent variability between Teflon filters, thereby providing more accurate representations of I_0 values and subsequent eBC concentrations. The added accuracy of eBC measures using Method 1 may allow future studies to more accurately estimate coefficients of health risk associated with exposure to BC since exposure misclassification can bias exposure-response relationships towards the null.[46,91] Some exposure-response studies in developed nations comparing the magnitude of associations for BC and $PM_{2.5}$ with various cardiovascular health endpoints[43,44] found no difference between the two pollutants, while others in LMICs, have reported stronger associations for BC, relative to $PM_{2.5}$, with blood inflammatory markers[55], blood pressure[92], and adverse birth outcomes[5]. Moreover, constituents of $PM_{2.5}$, like BC, inherently have greater measurement error than $PM_{2.5}$ itself [45,93] which can further obscure the true relative strength of each pollutant as an indicator for health risks. Therefore, further improvements in the accuracy of eBC measurements, particularly at the low end of exposures, may allow future epidemiological studies to adequately elucidate the role that co-pollutant confounding plays when comparing health risk effect estimates for BC and $PM_{2.5}$ exposures.

Study Limitations

Our optical method of BC estimation assesses the existence of light-absorbing carbon particles in totality. Particle absorption can also be influenced by the presence of light-absorbing organic carbon (OC), also referred to as brown carbon or BrC. BrC can absorb light in the red and near-infrared spectral region[94], which can lead to an overestimation in BC, or EC, by approximately 2-3 fold using optical instruments.[95] Although this issue is beyond the scope of the current study, special considerations should be taken when characterizing BC with optical instruments in health and climate change studies. Future studies that assess the BrC fraction in biomass cookstove emissions are needed to fully understand the health risks from exposure to cookstove sources. Secondly, our eBC measures were not adjusted for the proposed reduction in measurement sensitivity with increasing filter loading.[96,97] In the current study and for all HAPIN studies detailing BC exposures, we applied the manufacturer's recommendation and restricted the dataset to ATN < 125. Therefore, we suspect that the filter loading effect did not impact our exposure measures significantly since this artifact has been found to minimally impact data restricted to ATN < 300.[98]

Conclusion

The study provides the first field-based comparison of eBC measures using filter-specific I_0 values versus those using lab blanks or the average unloaded filters. Through LOD analysis, we provide evidence that the variation in light transmission between unloaded filters is greater than that within the same filter. Our results suggest that eBC measurements from archived filters are sufficient in most settings; however, the additional effort of analyzing Teflon filter media before sampling adds substantially to the accuracy of BC estimation, particularly in low emission settings. Therefore, we too recommend using filter specific I_0 values for BC estimations, especially in studies that aim to accurately characterize the exposure contrast from cookstove interventions with the potential to reach levels expected to improve health.

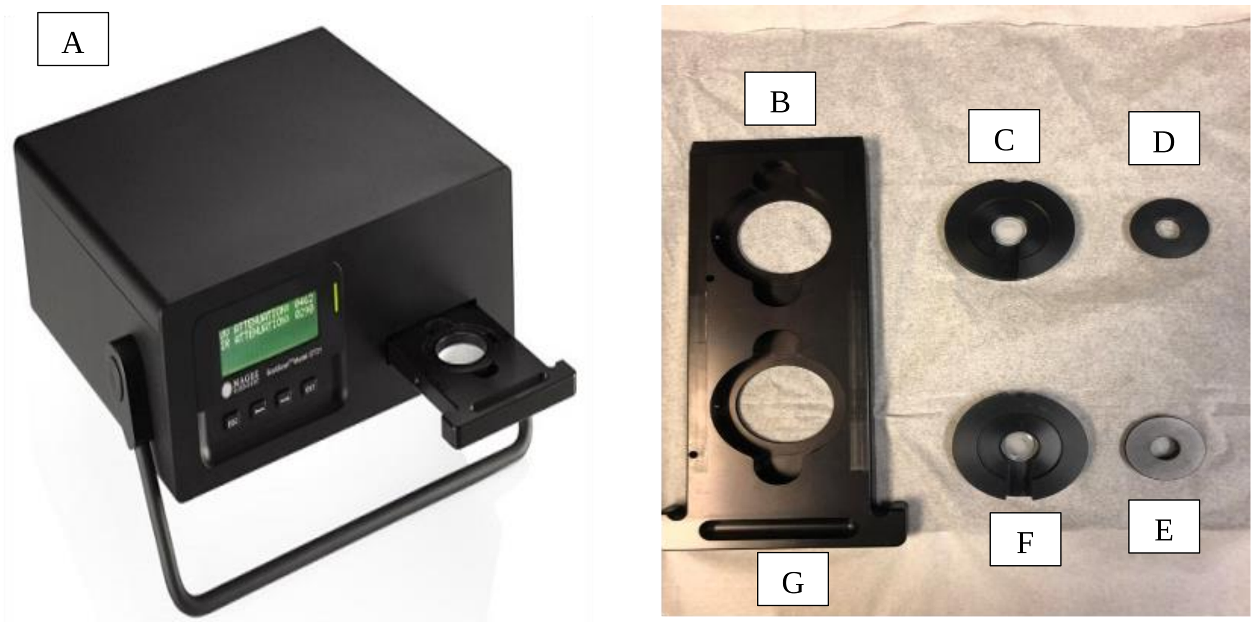


Figure 3.1. Magee Scientific Sootscan model OT21 transmissometer (A) (Magee Scientific Corporation, Berkeley, CA) and filter tray with two slots (B and G) that hold the adaptors for the lab blank or reference filter (C & D) and sample (F & E) filters. For Method 1, the reference filter scan is from the pre-analyzed sample filter as opposed to the lab blank filter.

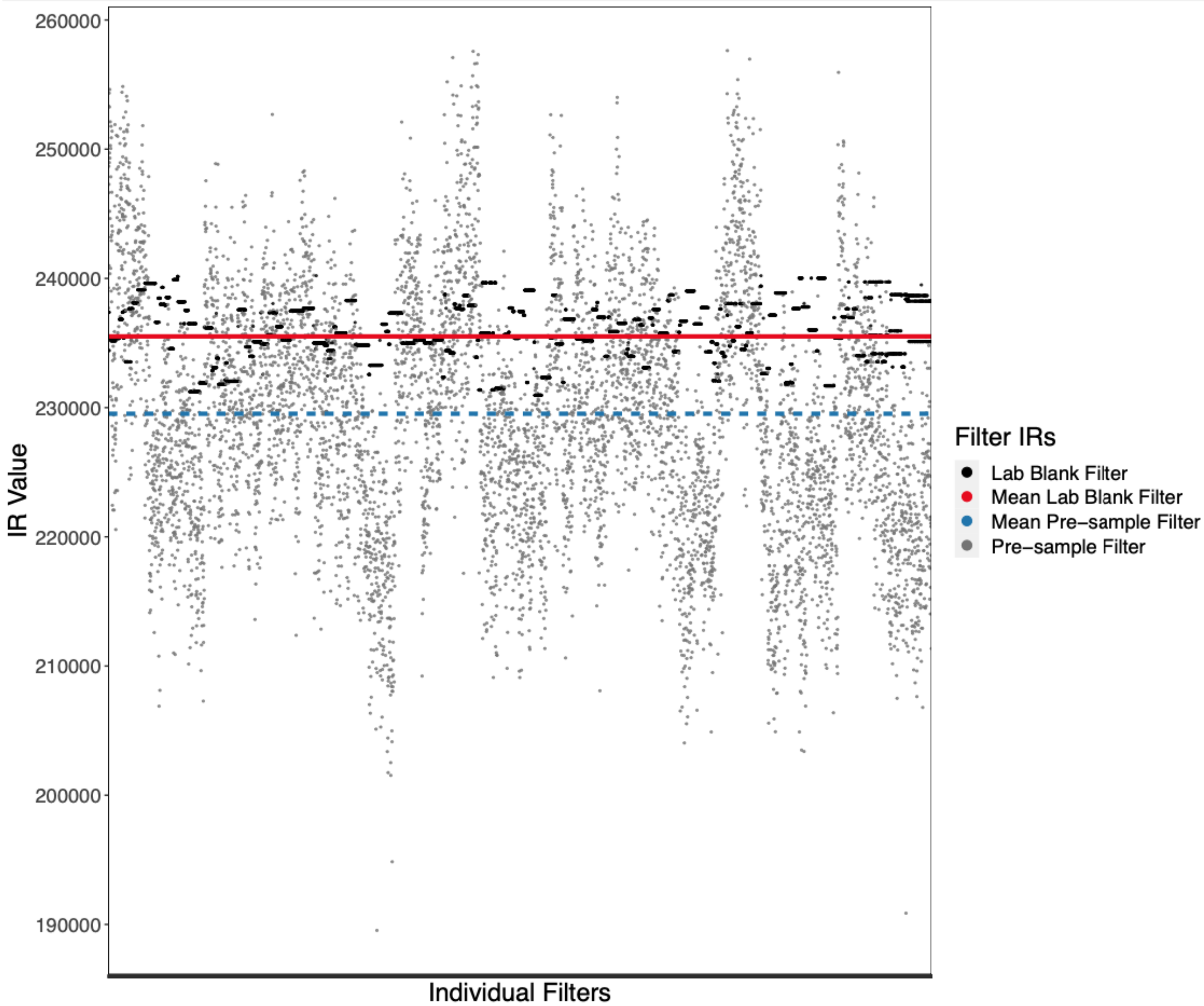


Figure 3.2. Light intensities (IR) for all lab blank and pre-sample filter scans. Pre-sample and blank filter scans represent I_0 estimations using Methods 1 and 2, respectively. Methods 3 and 4 estimate I_0 as the average IR value for all pre-sample ($I_0 = 229497.2$) and blank filter ($I_0 = 235510.6$) scans, respectively.

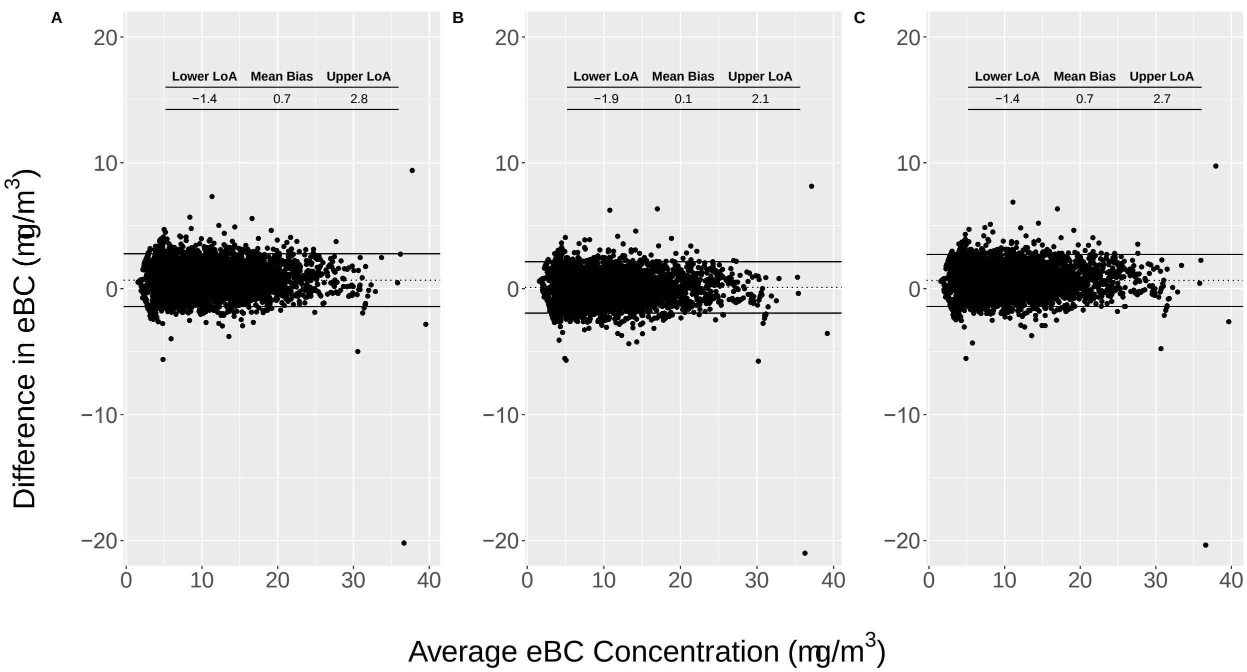


Figure 3.3. Bland-Altman plots comparing the agreement between Method 1 eBC measures and that from Method 2 (A: session specific lab blank scan), Method 3 (B: average of pre-sample filter scans), and Method 4 (C: average of lab blank scan). The dashed line reports the **average difference or mean bias (μg/m^3)** (Method (2 or 3 or 4) – Method 1) between measures. The bold lines show the limits of agreement (LoA) where 95% of the differences lie between the upper (ULoA) and lower (LLoA) limits of agreement. Bland-Altman statistics are provided in their corresponding panels.

Table 3.1. Summary of I_0 values by study site and filter type

I_0 Value Summary	Pre-sample Filter Scans	Lab Blank Filter Scans	Field Blank Filter Scans
Guatemala			
N	2345	72*	215
Average (SD)	229612 (9072.5)	235552.1 (2212.3)	228829 (9705.6)
Range	204045.5 - 257569.5	230956 - 239662.5	205561 - 255443.5
Median (IQR)	229233.5 (223249 - 235579)	235570.5 (234146.1 - 237586.2)	229130 (221308.2 - 234949.2)
Peru			
N	1719	50*	275
Average (SD)	226891.5 (10006.8)	236211 (2388.8)	227837.1 (9171.8)
Range	190870.5 - 257629	231695.5 - 240202	205959 - 251801
Median (IQR)	225903.5 (219688 - 233294.2)	236301.8 (234412.9 - 238195.21	226625.5 (220625.8 - 234191)
Rwanda			
N	2825	83*	86
Average (SD)	230180.6 (8846.2)	235758.5 (2127.4)	232780.6 (8016.5)
Range	189538 - 254853	231228 - 240208	217628.5 - 252389
Median (IQR)	230336 (224245 - 236261)	235378 (234553.2 - 237429.8)	232855 (227159.2 - 238540.6)
Overall			
N	6889	205*	576
Average (SD)	229166.3(9319.4)	235796.4 (2226.4)	228945.4 (9350.7)
Range	189538 - 257629	230956 - 240208	205561 - 255443.5
Median (IQR)	228986 (222531.5 - 235527)	235646 (234334.5 - 237628)	228720.8 (221616.8 - 235397)

*The N here represents the number of scans for the two lab blank filters used over the course of the study.

Table 3.2. Limit of Detections (LoDs) for each method and study site combination

Study site	$N_{field\ blanks}$	Method 1[a] (μg)	Method 2[b] (μg)	Method 3[c] (μg)	Method 4[d] (μg)
Guatemala	211	0.578	1.38	1.36	1.36
Peru	262	0.886	1.31	1.36	1.36
Rwanda	86	1.60	1.73	1.78	1.78

[a] Method 1: pre-sample scan for I_0 value
[b] Method 2: lab blank scan for I_0 value
[c] Method 3: average pre-sample I_0
[d] Method 4: average lab blank I_0

Table 3.3. Summary of mean (SD) eBC exposure measures (μg/m^3)

	N	Method 1	Method 2	Method 3	Method 4
Overall	6889	7.8 (5.9)	8.5 (5.9)	7.9 (5.8)	8.5 (5.9)
Guatemala	2345	9.2 (5.7)	9.8 (5.8)	9.2 (5.7)	9.8 (5.8)
Peru	1719	5.6 (6.4)	6.5 (6.5)	6.1 (6.2)	6.5 (6.4)
Rwanda	2825	8.0 (5.3)	8.6 (5.3)	8.0 (5.2)	8.6 (5.2)

Table 3.4. Model performance statistics for the linear regression between Method 1 and Method 2-4 measures

		Method 1 eBC (μg/m^3)			Method 2		Method 3		Method 4	
Quartile	N	Range	Mean	Median	R^2	RMSE	R^2	RMSE	R^2	RMSE
First Quartile	1799	1.3 - 2.9	2.0	1.9	0.26	0.49	0.25	0.49	0.26	0.49
Second Quartile	1629	2.9 - 6.3	4.4	4.6	0.45	0.80	0.42	0.82	0.45	0.80
Third Quartile	1746	6.3 - 11.4	8.7	8.7	0.66	0.85	0.67	0.85	0.67	0.84
Fourth Quartile	1715	11.4 - 46.8	16.1	14.8	0.92	1.21	0.93	1.20	0.93	1.20
Overall	6889	1.3 - 46.8	7.8	6.3	0.97	1.06	0.97	1.04	0.97	1.05

- All Method 1 eBC exposures were stratified into four quartiles; corresponding data for Methods 2 – 4 were placed in the matching quartile regardless of concentration level.
- RMSE represents the root mean square error between averaged Method 1 and Method 2, 3, and 4 measures.
- The Adjusted R^2 values represents the percentage of the variation in Method 1 measures that is explained by Method 2 – 4 measures.

CHAPTER 4

FACTORS ASSOCIATED WITH PERSONAL EXPOSURES TO BLACK CARBON AMONG RURAL PREGNANT WOMEN IN GUATEMALA, INDIA, PERU, AND RWANDA[2]

[2] Campbell D, Johnson M, Pillarisetti A, Piedrahita R, Waller L, Kearns K, Kremer J, Mollinedo E, Sarnat J, Clark M, Underhill L, McCracken JP, Diaz-Artiga A, Thompson L, Steenland K, Rosa G, Kirby M, Ndagijimana F, Dusabimana E, Balakrishnan K, Sambandam S, Mukhopadhyay K, Sendhil S, Natarajan A, Nicolaou L, Checkley W, Hartinger S, Peel J, Clasen T, & Naeher LP. To be submitted to Journal of Environmental Science & Technology.

Abstract

Background

Household air pollution (HAP) from residential biomass burning is an important source of black carbon (BC) exposure among rural communities in low- and middle-Income countries (LMICs). However, there is a limited understanding of the predictors of BC exposures among rural pregnant women.

Methods

We collected up to three repeated measures of 24-hour personal BC exposures via the OT-21 Magee Sootscan (total n = 7,165 observations) and individual/household level information from 3,103 pregnant women enrolled in the Household Air Pollution Intervention Network (HAPIN) trial conducted in 4 diverse LMICs (Guatemala, India, Peru, and Rwanda). Women assigned to the intervention arm received free liquefied petroleum gas (LPG) stoves throughout pregnancy, while women in the control arm continued to use their traditional biomass stoves. We developed mixed effects models to characterize major predictors of BC exposure and assess the potential role that each determinant plays in characterizing the BC exposure contrast between treatment arms post-randomization.

Results

The median (IQR) personal BC exposure level was 7.1 μg/m^3 (2.9 - 12.6) overall and 10.0 μg/m^3 (5.7 - 14.0) and 2.9 μg/m^3 (1.7 - 4.8) post-randomization in the control and intervention arm, respectively. We observed the most variation in BC exposures according to primary stove type: compared to traditional open fires, BC was lower for LPG (-70%, 95% CI: -71, -69), other improved biomass stoves (-29%, 95% CI: -35, -22), and chimney stoves (-22%, 95% CI: -28, -14). In addition to primary stove type, we identified secondary stove type, kerosene use, study site, whether the participant cooked or not, primary lighting source, other sources of smoke, kitchen location, roof material, participant occupation, hours of stove use, season, temperature, and relative humidity to be significant predictors of personal BC exposures across HAPIN. The HAPIN-wide personal BC exposure model performed moderately well (R^2 = 0.47). We also found evidence that the HAPIN-wide contrast in BC exposure between arms, post-randomization, differed according to study site, adherence to the assigned study stove, and the act of cooking.

Conclusion

Our study identified the potential impactful that various individual- and group-level factors have on personal BC exposure among pregnant women in rural communities in LMICs. We also provide information on the differential roles that selected factors play in reducing BC exposures via a randomized stove intervention trial. Our findings highlight the importance of both stove adherence as well as the mitigation of kerosene use in cookstove intervention trials to achieve more effective exposure contrasts between treatment groups.

INDEX WORDS: Household Air Pollution, Exposure Assessment, Exposure Models, Intervention, Black Carbon (BC), Biomass Fuel Stoves, Liquefied Petroleum Gas

Introduction

Approximately 2.4 billion people are exposed to elevated levels of household air pollution (HAP) due to the residential combustion of polluting fuels in the form of biomass (e.g., wood, dung, charcoal), coal and kerosene used for cooking and heating purposes.[2] HAP ranks among the top risk factors contributing to the global burden of disease and accounted for 2.3 million deaths and more than 90 million disability-adjusted life years (DALYs) in 2019 with disproportionate adverse health effects in low- and middle-income countries (LMICs).[3] Health outcomes associated with HAP include adverse respiratory (asthma, acute respiratory infection in adults and children, chronic obstructive pulmonary disease, lung cancer), cardiovascular (cerebrovascular disease, ischemic heart disease, and cardiovascular mortality), and birth outcomes.[4,5,8,9,47,99–102]

While HAP constitutes a mixture of many pollutants, it is primarily characterized via measurements of fine particulate matter with an aerodynamic diameter of ≤2.5 μm ($PM_{2.5}$) and carbon monoxide (CO).[103] $PM_{2.5}$ is a heterogeneous chemical mixture containing elemental carbon (EC), black carbon (BC), organic carbon (OC), polycyclic aromatic hydrocarbons (PAHs), and various metal species.[104] BC, the most light-absorbing portion of $PM_{2.5}$, is a by-product of incomplete combustion and is therefore a more specific marker of combustion-related sources than $PM_{2.5}$ alone. BC has been gaining more attention due to its global warming potential[22] and its associations with adverse human health.[17,51] Residential biomass burning is the top-ranked source of BC globally (~35%) with increasing emissions[38] due to growing rural populations in LMICs.[105] Therein lies is a need for more studies on BC in LMICs to better gauge its contribution to adverse climate and human health.

To date, BC remains largely understudied in settings where polluting fuels are used for daily cooking. The majority of epidemiology studies linking BC to health outcomes have been conducted in Europe and North America.[65,106] Many health and exposure studies in LMICs have reported the effectiveness of cookstove interventions in providing meaningful health benefits or reductions in HAP exposures.[10,11,66,67,107–109] Yet few cookstove intervention trials have assessed personal BC exposures.[71,108,110]

While several space- and time-dependent factors potentially impact HAP-related $PM_{2.5}$ exposures (e.g., stove type, kitchen location, time spent cooking), little is known about the how these factors are associated with personal BC exposures.[19] A recent global assessment of 870 individuals across 8 LMICs from the PURE-AIR cohort found stove and fuel type, hours spent cooking, and having a window in the kitchen as important factors for BC exposure.[80] The study did not collect longitudinal data, nor did it address potentially substantial contributions from kerosene used for lighting.[80] Another cross-sectional analysis of BC exposures among adult women in Peru identified stove stacking (i.e., the use of multiple stove type and/or fuel combinations within the same household), roof type, number of bedrooms, and ventilation to be significant predictors of personal BC.[78] There is a need for longitudinal analyses to fully elucidate how space- and time-dependent factors impact BC exposures in HAP settings.

The Household Air Pollution Intervention Network (HAPIN) trial is a randomized controlled trial of liquefied petroleum gas (LPG) stoves and continuous free fuel distribution in four LMICs.[14] Details on personal exposures to $PM_{2.5}$, BC, and CO among pregnant women in the HAPIN trial, both overall and by intervention arm and study site, have been published previously.[73]. Here, we explore the association between BC exposure and a variety of household characteristics, participant practices, and other factors assessed at baseline and follow-up visits. To complement this analysis, we also evaluate how select factors potentially modified the effectiveness of the LPG intervention in reducing personal BC exposures in the HAPIN trial.

Methods

Study Setting

The HAPIN study (registration NCT02944682) was a randomized controlled trial evaluating the health effects of an introduced LPG stove and free fuel intervention versus the continued use of traditional cookstoves in Guatemala, India, Peru, and Rwanda.[14] In each study site, ~800 households were enrolled totaling 3,195 pregnant women (aged 18 – 35 years) randomized to either control or intervention arm.[14] As part of HAPIN, we analyzed 24-hour personal BC exposures from the 3,103 (97%) pregnant women across the four study sites with at least one valid measurement (n = 7,165 BC measures).

HAPIN specifically chose rural communities without major ambient air pollution sources and a high proportion of homes that typically use traditional biomass cookstoves for cooking and heating purposes to reflect locations where a household air pollution intervention would be expected to provide maximum exposure reductions. Details regarding study site characteristics, inclusion and exclusion criteria, and overall study design are described elsewhere.[14,85] Pregnancy-related exposures as well as details on total versus valid samples collected are reported in our previous work.[73]. We provide a brief description of the HAPIN sampling plan during pregnancy below.

Exposure Measurement

Twenty-four-hour average exposure measurements were conducted at three times during pregnancy. Baseline ("BL") measures were collected at 9-20 weeks of gestation before the LPG stove was introduced. Two follow-up measurements were made after at 24-28 (post-intervention visit 1, "P1") and 32-36 (post-intervention visit 2, "P2") weeks of gestation. At each visit, participants were asked to wear customized vests or aprons fitted with air monitoring instrumentation near their breathing zone. If participants were to conduct activities that could damage the equipment, they were asked to remove the vest but keep it nearby (within 1-2m).

$PM_{2.5}$ aerosols were collected using the RTI Enhanced Children's MicroPEM™ (ECM, RTI International, Research Triangle Park, USA). The ECM uses a 2.5-micron size-cut impactor at a flow rate of 0.3 liters per minute for gravimetric samples on 15 mm polytetrafluoroethylene (Teflon) filters (Measurement Technology Laboratories, USA). The ECM also logs temperature, relative humidity, and triaxial accelerometry.

BC concentrations from $PM_{2.5}$ filter samples were estimated using the Sootscan Model OT21 Optical Transmissometers (Magee Scientific, USA) at either the University of Georgia (UGA, Athens, GA, USA), for samples collected in Guatemala, Peru, and Rwanda, or at Sri Ramachandra Institute for Higher Education and Research (SRIHER, Chennai, India), for samples collected in India. We measured the optical transmittance of filters and compared the intensity of light through a post-sampled and reference filter. At UGA, the reference filter was primarily the pre-sampled filter itself. At SRIHER, the reference filter was a blank laboratory filter.

To estimate BC concentrations, we used an equation derived from Presler-Jur et al., 2017 that measures particle light absorption (1):

$$[BC](\mu g/m^3) = \ ln(I_0/I) \ * \ A(m^2) \ / \ V(m^3) \ * \ 10^6 \ / \ \sigma(m^2/\mu g) \qquad (1)$$

Here, [BC] is the BC mass concentration, I_0 is the intensity of light through a reference filter, I is the intensity of light through a post-sampled filter, A is the area of the filter, V is the air sample volume, and σ is the attenuation cross-section. This equation relates absorbance and mass content. We used the attenuation cross-section value reported in Garland et al., 2017 ($\sigma = 13.7$ $\mu g/cm^2$) for Teflon filters collected from similar source types.

Data Collection on Individual and Household Characteristics

At the BL visit, trained local field technicians administered questionnaires to assess baseline participant characteristics (e.g., age, family size, and access to electricity) and sources of household exposures (e.g., primary stove and fuel types used for cooking, heating, and lighting). At all post-randomization exposure visits, technicians also collected information regarding behavioral practices that occurred during each 24-hour sampling period (e.g., whether the participant cooked or not, stove and fuels used, general kerosene use, etc.) as well as additional household characteristics such as food insecurity, kitchen location, roof type, and the presence of smoke from a source other than the participant's stove. Our questionnaires were not specific enough for us to discern whether kerosene fuel was used explicitly for lighting or cooking activities during the sampling period. Seasonality was dichotomized as summer (April to September) and winter (October to March) for study sites in the northern hemisphere and reversed for those in the southern hemisphere.

Statistical Analysis

We conducted a descriptive analysis of the study population at BL. We report median (IQR) BC exposures according to select combustion-related factors expected to impact personal exposures to BC. For all further analyses via linear models, we natural log-transformed BC exposures to meet regression assumptions given the right-skewed distribution of the data. We conducted all analyses in R (versions 3.6 and 4.0, R Foundation for Statistical Computing, Vienna, Austria)

To assess the relationship between covariates and personal BC, we employed a series of mixed linear effects models with random intercepts for households to account for the correlation of repeated measurements made on the same participant. We ran mixed models for all HAPIN measures (adjusting for study site) and for each study site separately to look at univariate associations between covariates and exposure as a descriptive analysis (2). Here, y_{ij} is the j^{th} measurement of BC for participant i; β_0 is the overall intercept (mean); β_1 is coefficient for the study site adjustment (not included in the site-specific models); β_2 is the coefficient for the covariate (X_2) of interest in the model; α_i is the random intercept for participant i; and ε_{ij} is the random error. These models assume that α_i and ε_{ij} are independent and normally distributed with variances σ_b^2 (i.e., variance between individuals) and σ_w^2 (i.e., variance within individuals), respectively. We then ran multivariable mixed-effects regression analyses to assess this relationship after adjusting for other model covariates (3). Covariates were selected to be included in the model using a backward stepwise regression procedure which eliminated non-significant ($p > 0.05$) variables.

$$\ln(y_{ij}) = \beta_0 + \beta_1(IRC) + \beta_2(X_2) + a_{ij} + \varepsilon_{ij} \quad (2)$$

$$\ln(y_{ij}) = \beta_0 + \beta_1(IRC) + \cdots \beta_n(X_n) + a_{ij} + \varepsilon_{ij} \quad (3)$$

We estimated the root mean square error (RMSE), marginal R^2 value, and intraclass correlation (ICC) for the HAPIN-wide and site-specific models. We calculated the RMSE as a metric for the absolute fit of the model to the data by taking the square root of the average of the squared residuals. The marginal R^2 value represents the proportion of the variance in personal BC explained only by the fixed effects of the predictors included in the model. We determined the ICC by estimating the proportion of the total variance in personal BC attributable to the variance between individuals conditional on the predictors in the model (4).

$$\sigma_b^2/(\sigma_b^2 + \sigma_w^2) \quad (4)$$

As a sensitivity analysis, we imputed missing questionnaire data with the MICE package in R[111] and used a stepwise method similar to that posed in Brand[112] for imputed data to identify predictors of personal BC. Briefly, we imputed data for missing survey variables 10 times. Next, we performed stepwise elimination model selection for each imputed dataset separately, keeping all variables that were present in at least half of the 10 models. We then conducted a backward elimination procedure using the Wald statistic to test whether each variable should be in the final model. We removed each variable in turn and then compared models with and without the variable. If the Wald statistic had a p-value above 0.05, the variable was removed. This backward elimination procedure stops when all p-values are less than 0.05. Model results using the imputed dataset are provided in the supplemental material.

To test whether covariates included in the full models were confounders for the BC exposure contrast between arms post-randomization, we added an indicator variable for the control versus intervention arm to the full exposure models (5). Evidence of confounding was observed if the fixed effect of the treatment arm differed from the between group BC exposure contrast reported in Johnson et al.[73]. We then constructed linear mixed-effects models with an interaction term between arm and all covariates included in the full models to assess which predictors potentially modified the effectiveness of the intervention (6). Parameters with significant interaction coefficients were deemed to show potential effect modification. Next, we evaluated each predictor with significant interaction coefficients, separately, after controlling for all other covariates found to be associated with personal BC in previous models (7). Although not completely comparable to BC exposure contrasts between arms reported in Johnson et al.[73], which only controlled for study site in the HAPIN-wide model, deviations from Johnson et al.[73] results show how select predictors potentially modified the effect of the intervention in reducing exposures to BC. For this analysis, we categorized stove use hours by quantile (i.e., ≤ 25th percentile, middle 50 percent, ≥ 75th percentile) and replaced primary stove type with a dichotomous indicator for adherence to the stove assignment in each treatment arm. Participants in the control and intervention arm adhered to the stove assignment if they respectively used a biomass or LPG stove as their primary stove.

$$\ln(y_{ij}) = \beta_0 + \beta_1(IRC) + \beta_2(Study\ Arm) + \cdots \beta_n(X_n) + a_{ij} + \varepsilon_{ij} \quad (5)$$

$$\ln(y_{ij}) = \beta_0 + \beta_1(Study\ Arm) * (X_1) + \cdots \beta_n(Study\ Arm) * (X_n) + a_{ij} + \varepsilon_{ij} \quad (6)$$

$$\ln(y_{ij}) = \beta_0 + \beta_1(Study\ Arm) * (X_1) + \cdots \beta_n(X_n) + a_{ij} + \varepsilon_{ij} \quad (7)$$

Results

Summary of Study Population

Baseline participant and household characteristics are provided in **Table 4.1**. The majority (70%) of our participants ranged in age from 20 – 29 years. Virtually all participants used wood as a cooking fuel at baseline in Guatemala (99%) and India (100%). Cow dung was the most common cooking fuel used at baseline in Peru (87%), while in Rwanda, participants used both wood (73%) and charcoal (25%). Electricity was the predominant primary lighting source in all study sites with the exception of Rwanda where only 28% of the participants used electricity as their primary lighting source at BL.

Summary of Personal BC Exposures

Summary statistics for personal BC measurements overall and by primary stove type, act of cooking, use of kerosene, and reported smoke from other sources during the sampling period are provided in **Table 4.2**. Overall, our dataset consists of 7,165 personal BC measures with most of our measures coming from participants using either open fires (49%) or LPG stoves (34%) as their primary stove type. Daily (24-hour) personal exposures to BC ranged from 0.6 – 132.6 μg/m^3. Overall, the median (IQR) BC exposure level was 7.1 μg/m^3 (2.9 – 12.6). We observed substantial differences in personal BC exposures according to primary stove type, cooking activity, kerosene use, and other sources of reported smoke during the sampling period. Median (IQR) BC exposures were 10.0 μg/m^3 (5.7 - 14.0) and 2.9 μg/m^3 (1.7 - 4.8) in the control and intervention arm, respectively, after restricting to data after randomization **(Table 4.S1)**.

Major Predictors of Black Carbon Exposures

HAPIN-wide (adjusted for study site) and site-specific associations between personal BC and single household, individual, and environmental characteristics, unadjusted for other covariates, are provided in **Table 4.3**. Our complete dataset, comprising both questionnaire/field observation and exposure data, includes approximately 81% (n = 5838) of our total BC measures. We have missing data for kitchen volume (13%), roof type (12%), season (5%), temperature and relative humidity (5%), hours of stove use during the 24-hour sampling period (2%), food security (2%), kitchen location (2%), participant occupation (<1%), other sources of smoke (1%), lighting fuel (<1%), fuel type (<1%), and kerosene use (<1%) **(Figure 4.S1)**. Primary stove type and study site explained the most variation in personal BC among all other covariates in our HAPIN-wide analysis ($R^2 = 0.42$), and stove type remained important in each site-specific model (R^2 range 0.36 – 0.48). In HAPIN-wide models, fuel type at BL, cooking fuel, kitchen location, lighting source, other sources of smoke, participant occupation and education, access to electricity, food insecurity, age at BL, roof type, season, hours of stove use per day, relative humidity, and temperature all had significant associations with personal BC but had little to no additional explanatory power than the model including only study site with R^2 values ranging 0.06 – 0.1. Fuel type had considerable explanatory power at BL in Rwanda ($R^2 = 0.21$).

HAPIN-wide and site-specific multivariable mixed effects regression coefficients between personal BC and household, individual, and environmental characteristics are shown graphically in **Figures 4.1 and 4.S2**, respectively. The predictors selected in the full HAPIN-wide model were study site, primary stove type, secondary stove type, stove use hours, the act of cooking, other sources of smoke, lighting source, kerosene use, kitchen location, roof type, participant occupation, food insecurity, temperature, humidity, and season.

We observed significant variation between study sites as BC exposures were 17% lower (95% CI: -25, -9) and 53% lower (95% CI: -58, -46) in India and Peru, respectively, compared to Guatemala. However, the most variation in BC exposures was observed as a decreasing exposure gradient according to stove and fuel types: compared to traditional open fires, BC was lower for LPG (-70%, 95% CI: -71, -69), followed by other improved biomass stoves (-29%, 95% CI: -35, -22), and chimney stoves (-22%, 95% CI: -28 -14), respectively. General kerosene use was associated with a 47% (95% CI: 36, 58) increase in personal BC across HAPIN and was included in each site-specific model except Peru, which had minimal kerosene users. In these models, general kerosene use was associated with a 25% (95% CI: 3, 52), 52% (95% CI: 38, 68), and 75% (95% CI: 48, 108) increase in personal BC in Guatemala, India, and Rwanda, respectively.

Kerosene lamp users had 31% (95% CI: 13, 52) higher BC exposures than those that used electricity for lighting. Another combustion-related variable impacting BC exposures was smoke from a neighbor's kitchen, which was associated with elevated BC exposures (+14%, 95% CI: 5, 23) compared to when no other sources of outside smoke were reported.

We also observed significant associations between personal BC and other individual and household characteristics as well as environmental conditions. Participants that worked either in the household (-20%, 95% CI: -25, -15) or as vendors (-25%, 95% CI: -34, -15) had lower BC exposures compared to those working in agriculture. We observed a decreasing exposure gradient according to kitchen location: compared to participants with kitchens located inside the home, BC exposures were lower for those with outside enclosed kitchens (-8%, 95% CI: -12, -3), outside open-air kitchens (-15%, 95% CI: -25, -3), and kitchens that were not located at the residence (-65%, 95% CI: -84, -27) respectively. BC exposures were found to be lower for participants with primary (-7%, 95% CI: -12, -3) or secondary (-11%, 95% CI: -16, -5) school education, compared to those with no formal education. Participants with permeable roof types had lower BC exposures (-11%, 95% CI: -16, -5) than those with impermeable roof types; however, neither roof type nor participant education were included in any of the site-specific models. Every 5% increase in relative humidity was negatively associated with personal BC (-5%, 95% CI: -6, -4). We also observed every 5 degrees Celsius increase in temperature to be negatively associated with personal BC (-9%, 95% CI: -13, -5) and found BC exposures to be higher in winter months (+5%, 95% CI: 1, 9) compared to summer months.

Model Performance and Fit

Model performance and fit statistics are provided in **Table 4.4** and are comparable to those from models with imputed data for missing covariates **(Table 4.S2)**. Our HAPIN-wide model explained 47% of the variation in BC exposures among our study population with similar marginal R^2 values for each site-specific model. The within-individual variance in personal BC was much greater than the between-individual variance as our ICC was 0.20. The RMSE between the natural logged-transformed predicted and measured BC exposures was 7.8 $\mu g/m^3$.

Potential for Confounding and/or Effect Modification

As expected, due to randomization, we found no substantial change in the contrast between treatment arms upon the addition of potential confounders to the model (results not shown). The HAPIN-wide and site-specific BC exposure contrasts between arms post-randomization are shown in **Figure 4.2** by level of other covariates (assessing potential for effect modification). Sample sizes for the control and intervention measures for each comparison are provided in the supplemental material **(Table 4.S3)**. HAPIN-wide, we observed the exposure contrast between arms to differ according to study site, adherence to the assigned stove, the act of cooking, lighting source, and kitchen location.

As reported by Johnson et al.[73], BC exposures in the intervention group were 62% (95% CI: 60, 65) lower than those in the control group across all HAPIN sites. This contrast was altered when participants resided in Peru (-70%, 95% CI: -73, 66), India (-68%, 95% CI: -70, 65), and Rwanda (-53%, 95% CI: -58, -48), did not adhere to the assigned stove type (+54%, 95% CI: 19, 99), were not the primary cooks during sampling (-47%, 95% CI: -57, -36), or had kitchens located outside without an enclosure (-13%, 95% CI: -46, 39) as confidence intervals for these effect estimates did not overlap that from Johnson et al.[73]. The confidence interval for the HAPIN-wide BC exposure contrast among general kerosene users (-54%, 95% CI: -62, -45) overlapped with that from Johnson et al.[73]. This effect, however, was much more pronounced in India as the contrast between arms was lower among kerosene users (-50%, 95% CI: -62, -35) compared to the BC exposure contrast reported by Johnson et al.[73] in India (69%, 95% CI: -73, -66). In Peru, we estimated that the BC exposure contrast increased from -62% (95% CI: -66, -57) to -74% (95% CI: -77, -71) after restricting to those that adhered to the study stove assignment. We also observed the intervention in Peru to be more impactful as participants in both treatment arms spent more time using their cookstoves. In Rwanda, we observed the contrast between groups to be attenuated among participants that were in the other occupation category (-31%, 95% CI: -45, -12), reported smoke from a neighbor's kitchen (-38%, 95% CI: -49, -25), or were not food insecure (-44%, 95% CI: -50, -38).

Discussion

Study Overview

We conducted one of the largest and most comprehensive BC exposure assessment studies to date, comprising 7,165 24-hr measures from 3,103 pregnant women across four diverse countries. The HAPIN study site selection process created the opportunity to assess personal exposures from individuals using clean fuel alternatives in more or less ideal situations with few sources of air pollution other than the cookstove in the home. With questionnaires and technician observations, we were able to collect information on numerous factors that may potentially be associated exposure to household air pollutants.

We observed significant differences in personal exposures to BC according to primary stove type and kerosene use. In HAPIN-wide and site-specific analyses, we consistently observed a strong gradient where exposures were highest among participants using open fires, followed by those using improved biomass, and LPG stoves, respectively. Elevated exposures to BC were observed among kerosene users, compared to non-users, for participants in Guatemala, India, and Rwanda which highlights the potential importance of this BC emission source. The findings also support the notion that adherence to the HAPIN study stove assignment and general kerosene use potentially modified the BC exposure contrast between treatment arms post-randomization; these effects were most prominent for participants in Peru and India, respectively.

Our study-wide and country-specific models performed moderately well (R^2 range: 0.45 – 0.49) with R^2 values similar to that which was reported for predicting personal $PM_{2.5}$ exposures using only survey-based data in Kenya ($R^2 = 0.51$)[113] and higher than that reported for predicting global personal BC with survey data, ambient $PM_{2.5}$, and household concentration levels (R^2 range: 0.33 - 0.39)[80], which highlights the explanatory power of information from questionnaires.

We also observed larger within-participant variance compared to between-participant variance in BC exposures, even after adjusting for key predictors of exposure. The ICC in our HAPIN-wide model (0.20) was higher than those observed for personal $PM_{2.5}$ (0.16) and BC (0.11) exposures among adults in China[74], lower than that observed for carbon monoxide (CO) exposure among children in The Gambia (0.27)[114], and overlapped with ICCs observed for exposures among adult women in Guatemala[115] (CO: 0.17 – 0.33) and peri-urban India[75] ($PM_{2.5}$: 0.22; BC: 0.21). The direct comparison of ICCs across studies can be misleading given the differences in the predictors[116], target population, and pollutants assessed. Yet, our results, along with those from other longitudinal studies, suggest that a single 24-hr exposure measurement does not sufficiently characterize longer-term exposures to investigate chronic health effects.

Stove and Fuel as Predictors for Black Carbon Exposures

We observed substantial variation in personal BC according to stove and fuel types as shown in effect estimates from the overall **(Figure 4.1)** and site-specific **(Figure 4.S2)** plots of multivariable linear regression coefficients. A previous assessment of household BC in eight LMICs observed a concentration gradient for household BC comparing gas to coal (+53% increase), wood (+142% increase), and other biomass (+190% increase) fuels.[80] Although not completely comparable, we observed similar findings according to stove/fuel types where BC exposures increased going from LPG to other biomass stove users. The distinction between these stove types is driven primarily by the fuel type and combustion conditions associated with each stove. For example, in Rwanda, Imbabura stove users predominantly cooked with charcoal and had lower BC exposures than those burning wood with Rondereza or open fire stoves. Our finding of BC exposures from charcoal stoves being lower than that from wood stoves is in line with both field[34] and lab-based studies[117] that show charcoal to have lower BC emission factors (reported as quantities of emitted BC per unit of fuel used) than wood. We observed no significant differences in BC exposures among wood and cow dung users in Peru **(Table 4.3)**, possibly suggesting that these two fuel types have similar burning efficiencies.

Our assessment of the stove type used during sampling allowed us to characterize exposures more accurately beyond the assignment to arm. Thus, we were able to evaluate how the exposure contrast between arms changed after adjusting for adherence to the study stove assignment. HAPIN-wide, we observed adherence as a potential effect modifier for the BC exposure contrast between groups, but this effect was more pronounced in Peru. The percentage of Peruvian participants in control homes exclusively using an LPG stove in the past 24-hr at BL, P1, and P2 visits was 6%, 22%, and 23%, respectively.[118] Although these percentages are much smaller than those for corresponding participants in intervention homes that reported exclusive LPG use at follow-up (96-97%)[118], we estimated that the BC exposure contrast between arms would be greater by about 12% had participants adhered to stove assignment in Peru.

Kerosene Use as a Predictor for Personal Black Carbon

Kerosene was also a major predictor of BC exposures, likely due to the relatively high BC emission factor of this combustion source. Kerosene reportedly emits $PM_{2.5}$ comprised mostly of BC (88 – 100%)[119], which highlights the importance of studying the impact kerosene has on BC exposures in HAP settings. The percentage of women with access to electricity for our study population in Rwanda (~35%) was relatively higher than estimates in rural Sub-Saharan Africa (SSA) (12%)[120], yet much lower than that among women from the other study sites (>90%).[113] The lower access to electricity in Rwanda possibly suggests an increased reliance on fuel-based lighting sources like kerosene lamps and may explain why the magnitude of the association between kerosene use and BC exposure was greater than that seen in the other HAPIN IRCs. The elevated BC exposures seen among kerosene users in Rwanda were in line with those seen in Mozambique when comparing women using kerosene (+81%) to women using electricity as a lighting source.[77] In Uganda, interventions of solar lighting systems versus fuel-based lighting sources resulted in massive reductions (>90%) in BC exposures[121] which suggests that exposures from fuel-based lighting sources can reduce the impact of a clean burning cookstove intervention on BC exposures. Future intervention programs may benefit from aiming to mitigate both cookstove and lighting sources in tandem to comprehensively reduce exposures in HAP settings, particularly in SSA.

The role of kerosene use for pregnant women in India is less clear than it is for those in Rwanda. For example, virtually all of our participants in India either had access to electricity or used electricity as a primary lighting source[73]; however, ~20% of participants used kerosene during sampling at BL. While this percentage remained relatively constant in the control arm post-randomization, the percentage of those using kerosene nearly halved in the intervention arm post-randomization **(Table 4.S4)**. Additionally, unlike in Rwanda, both kerosene-related variables (e.g., general kerosene use during sampling and primary lighting source) were selected in the multivariable predictive model for India, which possibly alludes to multiple uses of kerosene in this study site. These assertions, along with technicians' observations of Indian mothers using kerosene as a fire starter in the control group for cooking, imply that the intervention had an unforeseen benefit in reducing the need to use kerosene in the intervention arm post-randomization. Both kerosene-related variables that we assessed have the potential to modify the BC exposure contrast between treatment arms; however, we observed the general kerosene use variable to impact the intervention substantially in India where the exposure contrast was sharply attenuated among kerosene users.

Study Findings in a Health-Risk Context

For our study population demographic, elevated blood pressure (BP) is a major concern since it is a leading risk factor in the global burden of disease, complicates an estimated 3-10% of pregnancies worldwide, and contributes to approximately 28,000 maternal deaths annually.[15,122,123] Although the positive association between HAP and BP is well established from studies on ambient pollution[124], information on the exposure-response relationship of personal BC and BP is currently lacking, particularly for pregnant women. In exposure-response analyses of pregnant women enrolled in HAPIN, we observed no significant association between personal BC and BP at baseline[58] or throughout pregnancy.[56] Our null findings may potentially be the result of a young and healthy study population as none of the participants smoked during the study period and only 6% of the participants were obese.[58] Exposures to air pollution during pregnancy can also detrimentally impact the health of the fetus since inhaled BC particles can cross the placenta and subsequently translocate into fetal organs during gestation.[63,125] In fact, we observed birthweight to decrease by 22g per interquartile increase in personal BC, highlighting the potential for adverse health of infants from prenatal exposures.[5] In the current study, we identified a plethora of factors that can impact exposures to BC and explored the potential of these factors in modifying the effect of an intervention of LPG stove and fuels. This information can be used in future exposure and health studies to estimate exposures among this demographic on a grander scale than personal monitoring would allow. Additionally, our findings may also provide the information needed to improve the effectiveness of intervention programs in reducing exposures to HAP and improving health.

Study Limitations

The HAPIN study design included many strengths, such as robust sample size, global representation amongst LMICs, clear distinctions of stove and fuel use, repeated measures of personal BC, and comprehensive assessment of individual and group-level factors that can potentially impact exposures; our study also has some weaknesses. We only reported BC exposures up to the second follow-up period, P2. Repeated postnatal measures may allow us to tease out any additional explanatory power of select questionnaire variables regarding the variation of exposures within and between individuals. The current study only has two repeated post-randomization measures for individuals in the intervention arm using the LPG stove, limiting our ability to assess the within-participant variance among this group. Second, we have missing survey data; however, after conducting the same analysis with imputed data for missing variables, both non-imputed and imputed model results were similar to one another. Third, our questionnaires were not specific enough to discern the different uses of kerosene among our study population. After consulting with our field exposure team, we suspect that participants in India additionally used kerosene to ignite biomass or as a backup cooking fuel as opposed to using kerosene solely as a lighting source like participants in the other study sites. We also provided evidence from our data that supports this. Future exposure assessments may benefit from administering more specific surveys to parse the potential impacts of exposure to kerosene in cooking and lighting practices. Fourth, we employed the same dichotomization scheme for seasons as in Shupler et al.[126], which may be too generic to fully capture the seasonal variations in each study site. For example, a potentially better way to capture seasonal variation may be according to rainy vs. dry seasons. Finally, as an efficacy trial, HAPIN purposefully selected study sites that reduced the contribution of air pollution from traffic sources. Additionally, ambient measures of exposures were not available during the time of our data analysis for this manuscript. Therefore, our results do not account for this source's contribution to exposures among our study population and may impact the portability of our findings to other settings with significant traffic- or industry-related source profiles.

Conclusion

Our study provides one of the most comprehensive assessments of personal BC exposures in LMICs. We identified the potential impact that cookstove and non-cookstove sources as well as individual and household characteristics have on personal BC exposure among rural pregnant women. Our predictive modeling may be beneficial to future studies aiming to quantify the health risks associated with exposure to BC in settings where household air pollution is dominated by cookstove sources. The study also provides information on the differential role that select factors play in reducing BC exposures in a liquefied petroleum gas stove intervention trial. Our findings highlight the importance of both stove adherence as well as the mitigation of kerosene use in cookstove intervention trials to achieve more effective exposure contrasts between treatment groups.

Table 4.1. Summary of baseline participant and household characteristics for the 3,103 women included in the study.

Characteristic	Category	HAPIN N (%)	Guatemala N (%)	India N (%)	Peru N (%)	Rwanda N (%)
Total		3103 (100)	791 (100)	787 (100)	753 (100)	772 (100)
Age	<20	388 (13)	119 (15)	126 (16)	94 (12)	49 (6)
	20-24	1161 (37)	320 (40)	380 (48)	264 (35)	197 (26)
	25-29	979 (32)	226 (29)	223 (28)	245 (33)	285 (37)
	30-35	568 (18)	122 (15)	58 (7)	149 (20)	239 (31)
Primary cooking fuel	Charcoal	190 (6)	---	---	---	190 (25)
	Cow dung	656 (21)	---	---	656 (87)	---
	Other	30 (1)	3 (0)	---	9 (1)	18 (2)
	Wood	2220 (72)	784 (99)	787 (100)	87 (12)	562 (73)
	Missing	7 (0)	4 (1)	---	1 (0)	2 (0)
Primary lighting source	Electricity	2362 (76)	697 (88)	757 (96)	688 (91)	220 (28)
	Kerosene lamp	81 (3)	6 (1)	16 (2)	1 (0)	58 (8)
	Other	179 (6)	72 (9)		30 (4)	77 (10)
	Solar light	275 (9)	2 (0)	3 (0)	22 (3)	248 (32)
	Torch (battery)	199 (6)	10 (1)	11 (1)	11 (1)	167 (22)
Participant's occupation	Agriculture	1479 (48)	6 (1)	333 (42)	568 (75)	572 (74)
	Commercial	126 (4)	17 (2)	4 (1)	23 (3)	82 (11)
	Household	1335 (43)	733 (93)	425 (54)	121 (16)	56 (7)
	Other	154 (5)	29 (4)	25 (3)	40 (5)	60 (8)
	Missing	9 (0)	6 (1)	---	---	
Family size	Small (<=4)	1994 (64)	386 (49)	572 (73)	386 (49)	617 (80)
	Medium (5-9)	1022 (33)	340 (43)	212 (27)	340 (43)	148 (19)
	Large (>10)	79 (3)	61 (8)	3 (0)	61 (8)	5 (1)
	Missing	8 (0)	---	---	4 (1)	---
Household food insecurity	None	1743 (56)	435 (55)	634 (81)	388 (52)	286 (37)
	Mild	835 (27)	249 (31)	113 (14)	259 (34)	214 (28)
	Moderate/Severe	474 (15)	93 (12)	36 (5)	95 (13)	250 (32)
	Missing	51 (2)	14 (2)	4 (1)	11 (1)	22 (3)
Participant's education	No complete formal education	804 (32)	311 (46)	235 (34)	24 (4)	234 (41)
	Primary school complete	861 (34)	268 (40)	195 (28)	168 (28)	230 (40)
	Secondary school or equivalent complete	873 (34)	95 (14)	269 (38)	404 (68)	105 (18)
	Missing	1 (0)	1 (0)	0 (0)	0 (0)	0 (0)

Table 4.2. Summary statistics for personal BC measurements (μg/m^3) overall and by primary stove used, act of cooking, use of kerosene, and reported smoke from other sources during the sampling period.

Variable	N (measures)	(%)	Median (IQR)	Mean (SD)	Range
Overall	7165	100	7.1 (2.9 - 12.6)	9.3 (9.5)	0.6 - 132.6
Primary stove					
Chimney	357	5	9.8 (5.6 - 13.6)	10.8 (8.3)	1.5 - 65.4
Imbabura	290	4	6.9 (4.8 - 9.4)	7.8 (5.1)	2.7 - 43.2
LPG	2443	34	2.7 (1.6 - 4.6)	4.0 (5.3)	0.6 - 131.5
Open fire	3515	49	10.8 (6.4 - 15.5)	12.6 (10.6)	0.6 - 132.6
Other	240	3	8.7 (4.5 - 13.8)	10.6 (9.7)	0.7 - 73.2
Rondereza	320	4	11.1 (7.8 - 14.5)	12.3 (8)	2.8 - 76.9
Participant cooked					
No	474	7	4.3 (2.3 - 8.4)	7 (8.6)	0.7 - 97.8
Yes	6672	93	7.3 (2.9 - 12.8)	9.5 (9.6)	0.6 - 132.6
Missing	19	0	4.2 (2.1 - 10.3)	6.1 (4.8)	1.1 - 14.6
Participant used kerosene fuel					
No	6704	94	6.9 (2.8 - 12.2)	8.8 (8.6)	0.6 - 132.6
Yes	432	6	11.3 (5.5 - 20.8)	16.5 (17.2)	0.7 - 122
Missing	29	0	3.7 (1.8 - 9)	5.6 (4.7)	1.1 - 14.6
Other sources of smoke reported by technician or participant					
None	6598	92	6.9 (2.8 - 12.5)	9.2 (9.6)	0.6 - 132.6
Neighbor kitchen	427	6	9.4 (5.2 - 13.7)	10.9 (9.4)	0.7 - 85.7
Other	35	0	3.8 (1.7 - 11.6)	6.3 (5.5)	1.1 - 23.3
Missing	105	1	8.7 (4.2 - 11.6)	10.3 (9.2)	0.8 - 66.1

Table 4.3. HAPIN-wide and IRC-specific association between personal BC and household, behavioral, and environmental characteristics. Effect estimates are transformed and represent the percent change in BC measurements.

	HAPIN[a]		Guatemala[b]		India[b]		Peru[b]		Rwanda[b]	
	Change (95%CI) in Personal BC	R^2 or Sample Size	% Change (95%CI) in Personal BC	R^2 or Sample Size	% Change (95%CI) in Personal BC	R^2 or Sample Size	% Change (95%CI) in Personal BC	R^2 or Sample Size	% Change (95%CI) in Personal BC	R^2 or Sample Size
IRC		**0.08**		---		---		---		---
Guatemala	Ref	2000	---	---	---	---	---	---	---	---
India	-26 (-31, -21)	1874	---	---	---	---	---	---	---	---
Peru	-48 (-52, -45)	1553	---	---	---	---	---	---	---	---
Rwanda	-3 (-9, 4)	1738	---	---	---	---	---	---	---	---
Cooking fuel[c]		**0.06**		---		---		**0.00**		**0.21**
Wood	Ref	1848	Ref	---	Ref	---	Ref	73	Ref	407
Charcoal	---	152	---	---	---	---	---	---	-47 (-52, -41)	152
Cow Dung	---	514	---	---	---	---	1 (-21, 30)	514	---	---
Other fuel[d]	-31 (-38, -23)	19	---	---	---	---	-2 (-53, 105)	8	-17 (-42, 20)	9
Primary stove used during sampling		**0.42**		**0.48**		**0.32**		**0.4**		**0.36**
Open fire	Ref	3515	Ref	1016	Ref	1229	Ref	808	Ref	462
Chimney	-19 (-26, -12)	357	-23 (-28, -16)	294	-37 (-75, 62)	3	2 (-17, 26)	60	---	0
Imbabura[f]	---	---	---	---	---	---	---	---	-41 (-46, -36)	290
Rondereza[f]	---	---	---	---	---	---	---	---	-10 (-17, -3)	320
Other stove[e]	-35 (-39, -30)	850	-36 (-46, -22)	29	-21 (-36, -3)	64	-51 (-63, -35	31	-19 (-27, -10)	116
LPG	-70 (-71, -69)	2443	-67 (-68, -65)	661	-73 (-75, -71)	578	-75 (-77, -73	654	-62 (-65, -59)	550
Participant cooked		**0.08**		**0**		**0**		**0**		**0.02**
No	Ref	474	Ref	39	Ref	90	Ref	202	Ref	143
Yes	20 (11, 30)	6672	-8 (-26, 14)	1954	22 (-3, 52)	1782	10 (-6, 30)	1346	41 (26, 58)	1590
Kitchen location		**0.1**		**0**		**0.01**		**0.02**		**0.23**
Inside	Ref	4192	Ref	1641	Ref	1477	Ref	473	Ref	601
Outside enclosed	29 (23, 36)	2209	9 (-1, 20)	317	26 (11, 43)	345	1 (-10, 14)	848	96 (84, 110)	699
Outside open-air	66 (52, 80)	638	43 (2, 99)	12	-1 (-36, 51)	22	57 (31, 86)	197	101 (86, 118)	402
Kitchen not at residence	-34 (-62, 15)	9	106 (-46, 685	1	---	---	-59 (-82, -6)	6	41 (-36, 213)	2
Primary lighting source		**0.08**		**0**		**0.01**		**0**		**0.09**
Electricity	Ref	5503	Ref	1770	Ref	1802	Ref	1424	Ref	507
Kerosene lamp	92 (65, 125)	183	15 (-24, 75)	16	107 (40, 203)	38	14 (-73, 1595	1	121 (92, 156)	128
Other	10 (-1, 22)	409	2 (-10, 16)	175	---	---	-6 (-30, 26)	59	38 (21, 57)	175
Solar light	15 (4, 27)	616	-9 (-59, 97)	4	-30 (-70, 65)	8	1 (-27, 40)	49	30 (19, 42)	555
Torch (battery)	6 (-5, 19)	436	-12 (-37, 23)	23	-11 (-45, 43)	26	-33 (-60, 11)	19	23 (12, 36)	368
Kerosene used during sampling		**0.1**		**0**		**0.05**		**0**		**0.05**
No	Ref	6704	Ref	1959	Ref	1546	Ref	1536	Ref	1663
Yes	82 (67, 99)	432	12 (-12, 43)	33	85 (63, 110)	322	83 (-16, 301)	7	110 (79, 146)	70
Other sources of smoke		**0.08**		**0**		**0**		**0**		**0.01**
None	Ref	6598	Ref	1720	Ref	1846	Ref	1498	Ref	1534
Neighbor kitchen	11 (2, 21)	427	4 (-5, 14)	252	10 (-36, 88)	14	34 (-32, 166)	9	22 (9, 35)	152
Other	-24 (-43, 1)	35	-31 (-55, 5)	10	-54 (-79, 4)	6	-11 (-49, 58)	13	2 (-39, 72)	6
Participant Occupation		**0.1**		**0.01**		**0.06**		**0**		**0.07**
Agriculture	Ref	3247	Ref	13	Ref	762	Ref	1187	Ref	1285
Commercial	-32 (-39, -23)	282	7 (-36, 77)	43	-25 (-63, 51)	11	8 (-21, 49)	51	-37 (-44, -29)	177
Household	-31(-36, -26)	3251	44 (-8, 125)	1859	-41 (-47, -35)	1042	-7 (-21, 8)	228	-31 (-40, -20)	122
Other	-20 (-28, -10)	363	63 (0, 164)	69	-20 (-41, 8)	59	2 (-20, 31)	86	-30 (-39, -20)	149
Family size		**0.08**		**0**		**0**		**0**		**0**
Small (<=4)	Ref	4622	Ref	977	Ref	1360	Ref	879	Ref	1406
Medium (5-9)	1 (-4, 6)	2333	-1 (-8, 7)	857	9 (-4, 23)	508	1 (-10, 13)	652	-6 (-15, 4)	316
Large (>10)	1 (-13, 19)	191	-9 (-21, 5)	154	6 (-59, 177)	6	71 (4, 183)	20	31 (-19, 112)	11
Access to electricity		**0.08**		**0**		**0**		**0**		**0.05**
No	Ref	1456	Ref	205	Ref	72	Ref	84	Ref	1095
Yes	-19 (-24, -12)	5632	-7 (-18, 4)	1783	-29 (-47, -5)	1802	11 (-13, 42)	1468	-27 (-32, -21)	579
Household food insecurity		**0.08**		**0**		**0**		**0**		**0.02**
None	Ref	4075	Ref	1115	Ref	1515	Ref	794	Ref	651
Mild	11 (5, 17)	1899	3 (-5, 13)	613	19 (1, 39)	266	7 (-6, 21)	540	23 (12, 35)	480
Moderate/Severe	12 (4, 20)	1073	12 (-1, 26)	234	14 (-13, 49)	84	5 (-12, 25)	200	17 (7, 28)	55
Age at baseline		**0.08**		**0**		**0**		**0.01**		**0**
<20	Ref	896	Ref	299	Ref	300	Ref	187	Ref	110
20-24	-9 (-16, -2)	2717	0 (-10, 12)	804	-8 (-22, 7)	902	-21 (-35, -6)	560	-15 (-28, 1)	451
25-29	-4 (-11, 4)	2243	8 (-4, 22)	580	-14 (-28, 2)	526	-4 (-21, 15)	504	-8 (-22, 8)	633
30-35	-6 (-14, 3)	1291	5 (-8, 21)	305	-21 (-37, 1)	146	-9 (-26, 12)	301	-8 (-22, 8)	539
Participant education		**0.09**		**0.01**		**0.02**		**0.01**		**0.06**
No complete formal education	Ref	2387	Ref	943	Ref	656	Ref	62	Ref	726
Primary school complete	-17 (-21, -11)	2490	-7 (-14, 1)	789	-26 (-35, -15)	540	-36 (-53, -15	463	-11 (-17, -3)	698
Secondary school or equivalent completed	-23 (-28, -18)	2285	-15 (-24, -5)	266	-28 (-37, -18)	678	-29 (-47, -6)	1027	-38 (-43, -31)	314
Roof material		**0.07**		**0**		**0.02**		**0**		**0.01**
Impermeable	Ref	5006	Ref	1894	Ref	981	Ref	888	Ref	1243
Permeable	-13 (-18, -7)	1332	-6 (-21, 13)	86	-26 (-34, -17)	877	8 (-6, 25)	348	49 (8, 105)	21
Season		**0.08**		**0**		**0**		**0**		**0**
Summer	Ref	2640	Ref	894	Ref	972	Ref	340	Ref	434
Winter	4 (0, 9)	4179	-8 (-14, -2)	959	11 (1, 22)	886	11 (-2, 26)	1077	6 (-1, 14)	1257
Hours of stove use per day	7 (5, 8)	**0.09** 7041	3 (2, 5)	**0.01** 1976	26 (21, 31)	**0.07** 1843	5 (1, 8)	**0** 1518	6 (4, 8)	**0.02** 1704
Relative humidity (per 5%)	-7 (-9, -7)	**0.1** 6829	-7 (-9, -5)	**0.02** 1858	-8 (-10, -6)	**0.03** 1859	-6 (-8, -4)	**0.02** 1418	-8 (-10, -7)	**0.04** 1694
Temperature (per 5 degrees Celsius)	7 (3, 12)	**0.08** 6829	24 (15, 34)	**0.02** 1858	-10 (-18, -2)	**0** 1859	7 (-2, 17)	**0** 1418	29 (19, 42)	**0.02** 1694
Kitchen volume (per 10m3)	0 (0, 0)	**0.07** 7048	-2 (-3, -1)	**0.01** 1976	-3 (-6, 0)	**0** 1844	0 (0, 0)	**0** 1524	-5 (-9, -1)	**0.01** 1704

[a] HAPIN-wide models are adjusted for IRC
[b] IRC-specific univariable analysis
[c] Cooking fuel analysis only includes baseline measures
[d] For HAPIN-wide analysis only, other fuel includes charcoal and cow dung
[e] For HAPIN-wide analysis only, other stove includes Imbabura and Rondereza stoves, as well as stoves reported as "other"
[f] Imbabura and Rondereza stoves included in Rwanda model analysis only
[g] Marginal R^2 (**bold**) represents the percentage of variation explained by fixed effects
[h] Sample size shows the number of observations per category with a valid personal BC measurement

Table 4.S1. Pooled post-randomization median (IQR) BC exposures

	Control			Intervention		
	N (measures)	%	BC (μg/m^3)	N (measures)	%	BC (μg/m^3)
Overall	2266	100	10.0 (5.7 – 14.0)	2360	100	2.9 (1.7 - 4.8)
IRC						
Guatemala	640	28	11.8 (8.9 - 14.6)	685	29	3.8 (2.6 - 5.4)
India	581	26	9.3 (4.9 - 14.3)	594	25	2.4 (1.5 - 4.2)
Peru	447	20	5 (1.6 – 11)	510	22	1.6 (1.6 - 1.6)
Rwanda	598	26	10.5 (7.1 - 13.6)	571	24	4.3 (2.9 - 5.9)

Table 4.S2. Sample size (N measures) of comparison groups (Control vs Intervention) after randomization

IRC	Model parameters	Sample Size	Median (µg/m^3)	Mean (µg/m^3)	SD (µg/m^3)	RMSE (µg/m^3)	ICC	Marginal R^2
HAPIN	Study site + primary stove type + secondary stove type + stove use hours + participant cooked + other sources of smoke + primary lighting source + general kerosene use + kitchen location + roof type + occupation + education + temperature + humidity + season	7165	7.4	7.8	4.5	6.7	0.20	0.47
Guatemala	Primary stove type + secondary stove type + other sources of smoke + kitchen location + kitchen volume + humidity	2000	9.7	8.9	4.1	6.5	0.24	0.49
India	Primary stove type + secondary stove type + stove use hours + cooked + primary lighting source + general kerosene use + occupation + temperature + humidity + season	1874	6.8	7.6	5.1	7.2	0.18	0.46
Peru	Primary stove type + secondary stove type + stove use hours + participant cooked + kitchen location + age at baseline + humidity	1553	5.3	5.6	3.9	6.0	0.20	0.49
Rwanda	Primary stove type + participant cooked + general kerosene use + occupation + education + humidity	1738	8.9	8.3	3.5	4.9	0.03	0.46

Table 4.S3. Sample size (N measures) of comparison groups (Control vs Intervention) after randomization

		HAPIN		India		Peru		Rwanda	
Predictors		N (Control measures)	N (Intervention measures)	N (Control measures)	N (Intervention measures)	N (Control measures)	N (Intervention measures)	N (Control measures)	N (Intervention measures)
	Johnson et al. 2022	2266	2360	581	594	447	510	598	571
Study site									
	Guatemala	640	685	---	---	---	---	---	---
	India	581	594	---	---	---	---	---	---
	Peru	447	510	---	---	---	---	---	---
	Rwanda	598	571	---	---	---	---	---	---
Adherence									
	No	129	90	9	26	119	18	---	---
	Yes	2137	2270	572	568	328	498	---	---
Participant cooked									
	No	142	153	---	---	---	---	---	---
	Yes	2118	2197	---	---	---	---	---	---
Hours of stove use during sampling									
	First Quartile	823	1023	---	---	305	330	---	---
	Middle 50%	611	524	---	---	8	8	---	---
	Third Quartile	832	813	---	---	134	172	---	---
Roof type									
	Impermeable	1523	1795	---	---	---	---	---	---
	Permeable	429	378	---	---	---	---	---	---
General kerosene use									
	No	2104	2261	459	539	---	---	---	---
	Yes	152	86	118	53	---	---	---	---
Kitchen location									
	Inside	1117	1801	---	---	---	---	---	---
	Outside enclosed	796	496	---	---	---	---	---	---
	Outside open-air	314	9	---	---	---	---	---	---
	Kitchen not at residence	3	5	---	---	---	---	---	---
Participant Occupation									
	Agriculture	---	---	233	253	---	---	478	394
	Commercial	---	---	5	2	---	---	45	68
	Household	---	---	324	321	---	---	28	54
	Other	---	---	19	18	---	---	45	53
Other sources of smoke reported by technician or participant									
	None	---	---	---	---	---	---	534	520
	Neighbor kitchen	---	---	---	---	---	---	44	35
	Other	---	---	---	---	---	---	2	1
Food insecurity									
	None	---	---	---	---	---	---	188	242
	Mild	---	---	---	---	---	---	177	155
	Moderate/Severe	---	---	---	---	---	---	216	157
Season									
	Summer	792	852	---	---	---	---	---	---
	Winter	1346	1388	---	---	---	---	---	---

Table 4.S4. Number (%) of baseline and post-randomization measures by treatment arm and kerosene use in India

Kerosene use	Control		Intervention	
	Baseline	Post-randomization	Baseline	Post-randomization
No	275 (78)	459 (80)	273 (78)	539 (91)
Yes	76 (22)	118 (20)	75 (22)	53 (9)
Total	351 (100)	577 (100)	348 (100)	592 (100)

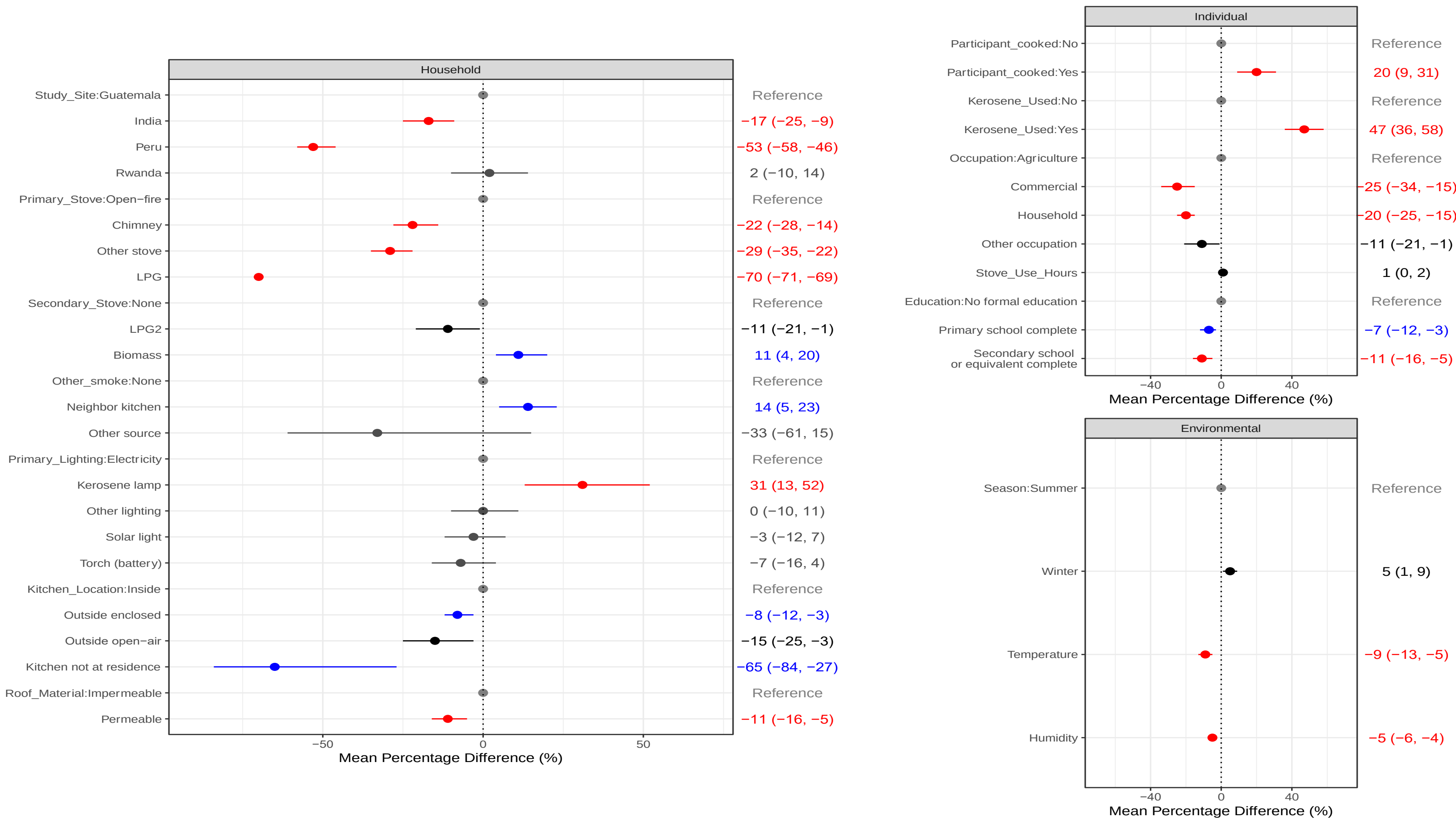


Figure 4.1. HAPIN-wide multivariable mixed-effects regression coefficients (with 95% confidence intervals) for the personal BC exposure model. Numeric coefficients represent the mean percentage change of the geometric mean in BC exposure compared to the reference category (light gray) based on the final multivariable linear regression models. Coefficients for relative humidity, temperature, and kitchen volume represent a 5 percentage point increase, a 5 degrees Celsius increase, and a 10 m^3 increase, respectively. Statistical significance is designated with asterisks (color coded) where n.s. is not significant ($p > 0.05$) (dark gray), * is $p < 0.05$ (black), ** is $p < 0.01$ (blue), and *** is $p < 0.001$ (red).

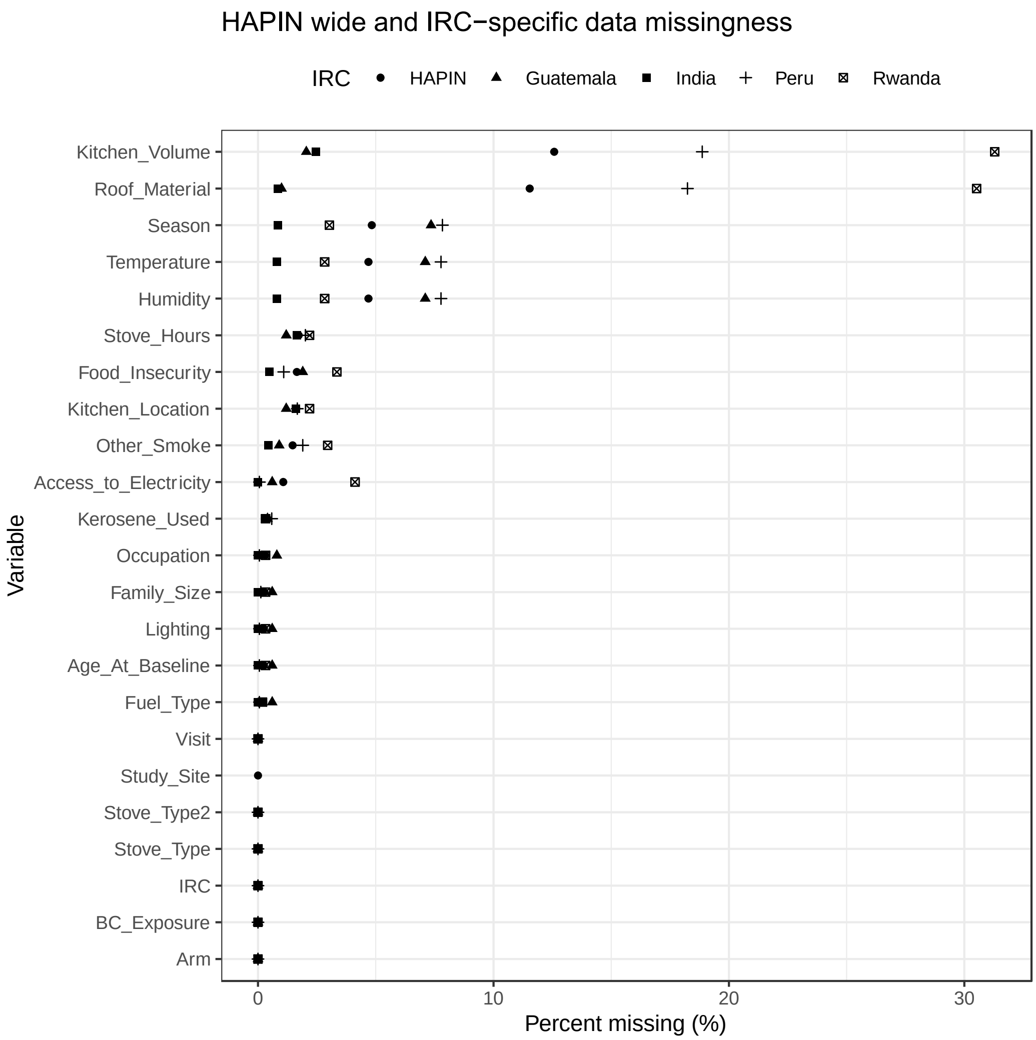


Figure 4.S1. Percent of missing data for each covariate in HAPIN (circle), Guatemala (square), Peru (plus), and Rwanda (box with a check).

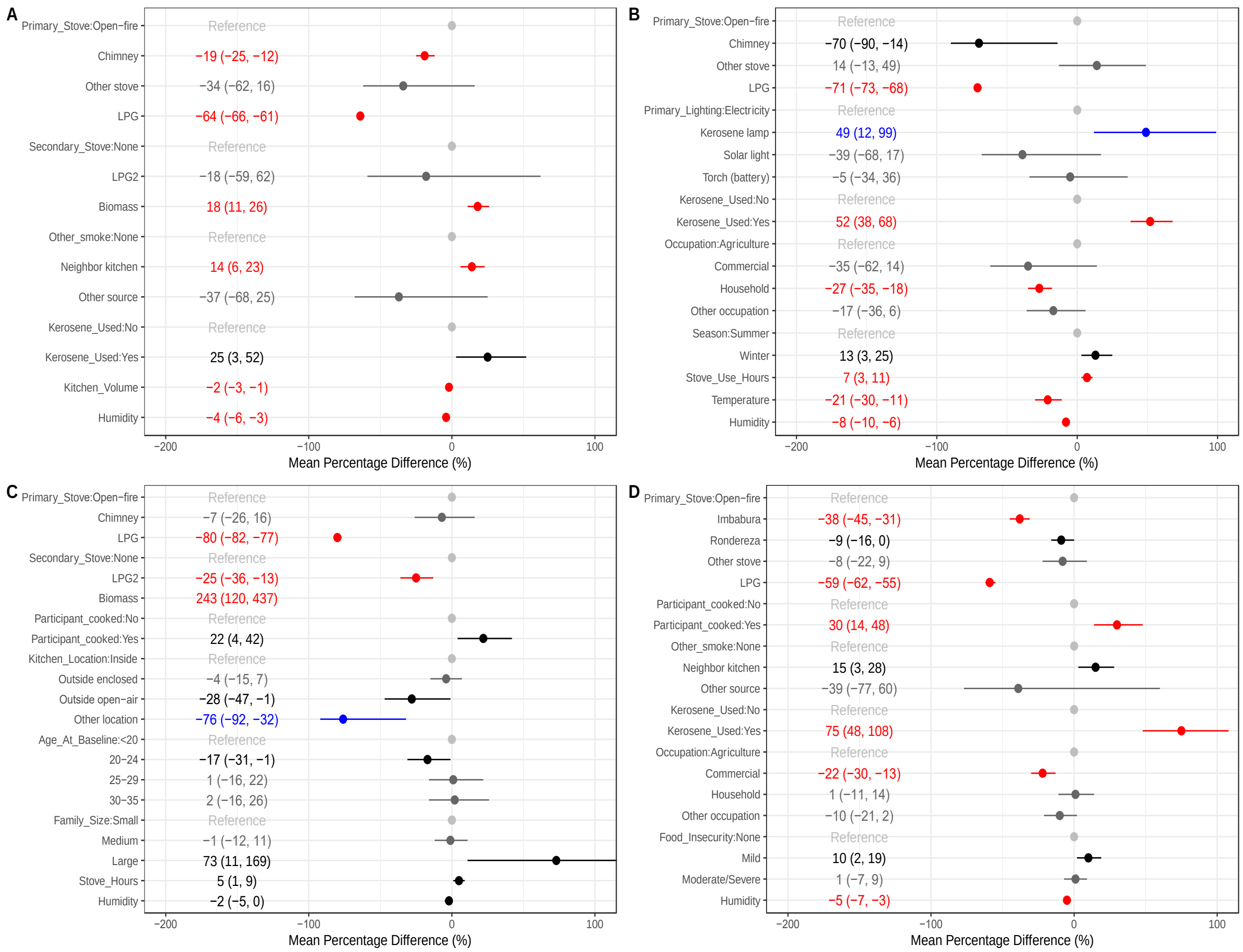


Figure 4.S2. IRC-specific multivariable linear regression coefficients (with 95% confidence intervals) for the personal BC models in Guatemala (A), India (B), Peru (C), and Rwanda (D). Numeric coefficients represent the mean percentage change of the geometric mean on respective BC exposures compared to the reference category based on the final multivariable linear regression models. Coefficients for relative humidity, temperature, and kitchen volume represent a 5 unit increase in percentage, a 5 unit increase in degrees Celsius, and a 10 unit increase in volume, respectively. Statistical significance is designated with asterisks (color coded) where n.s. is not significant ($p > 0.05$) (gray), * is $p < 0.05$ (black), ** is $p < 0.01$ (blue), and *** is $p < 0.001$ (red).

Chapter 5

Exposure Contrasts Of Adult Women During The Household Air Pollution Intervention Network Randomized Controlled Trial[3]

[3] Campbell D, Johnson M, Pillarisetti A, Piedrahita R, Balakrishnan K, Peel J, Underhill L, Steenland K, Rosa G, Kirby M, Diaz-Artiga A, McCracken JP, Clark M, Waller L, Chang HH, Wang J, Dusabimana E, Ndagijimana F, Sambandam S, Mukhopadhyay K, Kearns K, Kremer J, Mollinedo E, Sendhil S, Natarajan A, Nicolaou L, Checkley W, Hartinger S, Clasen T, Naeher LP & the Household Intervention Network (HAPIN) Trial Investigators. To be submitted to Science of the Total Environment.

Abstract

Background: The emission of household air pollution (HAP) from solid fuel use reportedly results in approximately 2.3 million deaths annually, with more than a third of these deaths occurring in women over the age of 50. Household energy interventions have had limited success in reducing HAP to levels below WHO interim guidelines. We report exposure reductions from a liquefied petroleum gas (LPG) stove and fuel intervention for non-pregnant women enrolled in the Household Air Pollution Intervention Network (HAPIN) trial.

Methods: We enrolled 418 non-pregnant adult women aged 40 – 75 years old in Guatemala, India, Peru, and Rwanda. Up to six repeated measures of daily exposures to particular matter ($PM_{2.5}$), black carbon (BC), and carbon monoxide (CO) were measured once before randomization into control and intervention (both n = 209) and up to five additional times totaling an 18-month period. We used linear mixed-effects models to estimate the impact of the intervention on exposure levels.

Results: Overall, median post randomization exposures to $PM_{2.5}$ in the intervention arm reduced by 72% from baseline (82.7 vs. 23.0 μg/m^3), resulting in 70% and 53% of the post-intervention $PM_{2.5}$ exposures falling below the annual WHO-IT1 (35 μg/m^3) and WHO-IT2 (25 μg/m^3) guidelines, respectively. From baseline, we report 58% (95% Confidence Interval (CI): 50%,65%), 59% (95% CI: 52%, 65%), and 72% (95% CI: 55%, 82%) reductions in $PM_{2.5}$, BC, and CO, respectively in the intervention arm after adjusting for the same reductions observed in the control arm.

Discussion: Our findings suggest that an LPG intervention can significantly and consistently reduce exposures below health relevant guidelines.

INDEX WORDS: Liquefied Petroleum Gas, Clean Cooking, Intervention, Fine Particulate Matter ($PM_{2.5}$), Black Carbon (BC), Carbon Monoxide (CO)

Introduction

Approximately 3-4 billion people, living primarily in low- and middle-income countries (LMICs), rely on polluting fuels like wood, dung, kerosene, and crop residues for daily household energy needs.[127] The incomplete combustion of these fuels results in exposures to household air pollution (HAP), including particulate matter with an aerodynamic diameter of ≤ 2.5 µm ($PM_{2.5}$), carbon monoxide (CO), black carbon (BC), and other hazardous emissions.[99] In 2019, 2.3 million premature deaths globally were attributed to HAP exposure[3], with more than a third of these deaths (~830,000) occurring in women over the age of 50.[128]

Health outcomes associated with exposures to HAP, among middle- and older-aged women, include adverse cardiovascular (cerebrovascular disease, ischemic heart disease, and cardiovascular mortality), respiratory (asthma, acute respiratory infection, chronic obstructive pulmonary disease, lung cancer), and cognitive (memory, orientation, and object naming) impacts effects.[4,7,8,47,102,129] Findings from a recent review suggest HAP may also be an important risk factor for hypertension[130], a precursor to cardiovascular disease and leading risk factor for adverse health for those aged >50 years.[15]

Randomized controlled trials (RCT) of improved or advanced biomass cookstoves and clean fuel cookstoves, meant to establish a causal link between various adverse health outcomes and HAP, have largely failed in providing meaningful health benefits or reductions in HAP exposures.[11,66,67,70,101,107,131] Clean fuel alternatives, like liquefied petroleum gas (LPG) and electricity, have been more impactful than improved biomass stoves in reducing HAP exposures;

however, contributions from mixed fuel use and ambient background air pollution have likely prevented these interventions from reaching health- relevant exposure targets.[11]

Personal measures of exposure are increasingly recognized as important to accurately quantify exposure-response relationships in HAP-related health studies.[132,133] To date, the majority of epidemiological studies linking HAP and hypertension have used either microenvironmental measures or stove and fuel use categories as proxies of HAP exposure, potentially providing effect estimates with less precision.[124,130] Within the context of cookstove interventions, personal HAP exposures are typically measured for pregnant women and children because these populations are particularly at risk given the significant amount of time they spend indoors. Less is known about the exposure distributions and associated health risks among middle- and older-aged women, despite the substantial estimated health impact experienced within this age group. More exposure information is needed to better characterize exposure-response relationships for this demographic, since they too bare a disproportionate percentage of the health burden from cookstove emissions.

As part of the multi-country Household Air Pollution Intervention Network (HAPIN) RCT of an LPG cookstove and fuel intervention, we performed an extensive personal exposure assessment at baseline (prior to intervention) and at an additional five timepoints throughout an 18 month period.[14] Here, we report the impact of HAPIN's LPG stove and fuel intervention on personal $PM_{2.5}$, BC, and CO exposures among non-pregnant adult women.

Methods

HAPIN Trial and Study Overview. The HAPIN RCT evaluated the health effects of an LPG stove and free fuel intervention versus continued use of traditional cookstoves in four intervention research centers (IRCs) located in Guatemala, India, Peru, and Rwanda. In Guatemala, households commonly engaged in indoor cooking practices utilizing wood fuel in chimney stoves and open fires. In India, traditional mud and clay stoves fueled with wood were predominantly utilized for indoor cooking. In Peru, households employed built-in open or chimney stoves, utilizing wood and cow dung as fuel sources. In Rwanda, indoor cooking was primarily carried out using three-stone fires or simple open stoves (ronderezas) that utilized wood, or portable charcoal-burning stoves (imbabura).

The HAPIN study design and site descriptions have been described in detail.[14,83,85,86] Briefly, we specifically selected rural sites in each country with low ambient air pollution, few additional air pollution sources, and a high proportion of homes that typically use traditional biomass stoves for cooking and heating purposes to reflect locations where an intervention would be expected to provide maximum exposure reductions from cookstove sources.[70,134,135] In each IRC, we enrolled ~800 households totaling 3,195 pregnant women, their resulting newborn children, and 418 non-pregnant other adult women, all randomized to either the control or intervention arm.

This paper presents exposure results for non-pregnant other adult women, which are aligned with one of the HAPIN trial's primary health outcomes – blood pressure among non-pregnant adult women. Sample sizes, specifically for the non-pregnant other adult women, were informed by power calculations for minimal detectable differences in mean blood pressure.[14]

Recruitment. Across all IRCs, non-pregnant other adult women aged 40 to <80 years residing in the same household as an enrolled pregnant woman were recruited (one per household) provided they did not fall within the exclusion criteria: currently using tobacco products or planning to move out of the current household in the next 12 months. Pregnant women were identified and enrolled through partnerships with local clinics and community health workers. Eligible women were aged 18-35 years old at 9 to <20 weeks gestation, with a viable singleton pregnancy (confirmed by ultrasound); primarily used biomass fuel for cooking; and agreed to participate via informed consent.

Intervention Design. Following a baseline assessment, participant households were randomly assigned (one-to-one) to receive an LPG stove, continuous fuel delivery, and regular behavioral messaging or to continue use of a biomass-burning stove. In India and Peru, additional stratified randomization was used to ensure balance between distinct geographical regions within each IRC (2 in India, 6 in Peru). No additional stratification was used in Rwanda and Guatemala, where the study areas were deemed homogenous. The intervention package was informed by formative research and described in detail previously.[86,134] Briefly, all LPG stoves had at least two burners with additional components to meet needs in each IRC (e.g., a flat griddle for cooking tortillas in Guatemala; a roasting device in Rwanda). LPG stoves and continuous fuel supply were distributed to intervention households at no cost throughout follow-up. Initial stove installation and LPG cylinder delivery were completed by field staff in Guatemala, Peru, and Rwanda, whereas a contracted local LPG distribution company conducted these services in India, per local regulations. Behavioral messaging included safety training, encouragement of exclusive use of LPG, discouraged use of traditional biomass stoves, and behavioral reinforcements on detection of traditional stove use via stove use monitors. Participants in intervention homes were asked to pledge to use the LPG stove for all cooking throughout the trial.

Air Pollutant Sampling Instrumentation. We measured exposures to $PM_{2.5}$ using the RTI Enhanced Children's MicroPEM (ECM, RTI International, Research Triangle Park, USA).[136] The ECM uses a 2.5-micron size-cut impactor at a flow rate of 0.3 liters per minute to collect gravimetric samples on 15 mm polytetrafluoroethylene (Teflon) filters (Measurement Technology Laboratories, USA). The ECM also measures real-time $PM_{2.5}$ concentrations via nephelometry and logs temperature, relative humidity, and triaxial accelerometry. The ECM is light (approximately 150 g), small (2:5×6:5×12:5 cm), and nearly silent during use. BC concentrations from $PM_{2.5}$ filter samples were estimated via transmissometry.

We measured 1-minute CO concentrations using the Lascar EL-USB-300 (Lascar Electronics), which is the size of a large pen (125×26:4×26:4 mm, 42 g), runs on one-half AA batteries, and has a sensing range between 0 and 300 ppm. The Lascar CO device has been used extensively in HAP assessment.[67,73,137]

Sampling Strategy. Personal exposures of the non-pregnant adult women to $PM_{2.5}$, BC, and CO reported in this manuscript were based on 24-h measurements at six visits at each HAPIN IRC. Baseline measurements were made at >9 and <20 weeks of gestation, prior to randomization. Follow-up, post randomization measurements were made during the mother's pregnancy (at 24–28 wk of gestation and 32–36 wk of gestation) and after the birth of the child (at <3 months, ~6 months, and ~12 months). At each visit, participants were asked to wear customized garments[85] fitted with air monitoring instrumentation near their breathing zone.[138,139] If participants were to conduct activities that could damage the equipment (e.g., sleeping, bathing, heavy washing, or work that soaks the participant), they were asked to remove the vest but keep it nearby (within 1-2m).

Determining $PM_{2.5}$ Mass Concentrations. We used 1 μg resolution microbalances (Sartorius Cubis, MSA6.6s-000-DF) at the University of Georgia (filters for Guatemala, Rwanda, and Peru) and at the Sri Ramachandra Institute for Higher Education and Research (for India) to assess mass changes pre- and post-sampling for 24-h gravimetric filter samples collected at each visit. Gravimetric data was validated with a three-staged approach: 1) field technicians evaluated pre- and postsample flow rates with a primary flowmeter at the field office to ensure flagging and removal of samples outside of expected ranges; 2) laboratory technicians invalidated damage filter samples; and 3) data analysts then removed data that did not meet criteria regarding sampling duration (24 h +/- 4 h), flow rate (300 +/- 100 mL/min, measured by the internal flow sensor), and inlet pressure (95th percentile <5 inches H2O). In instances where the gravimetric sample was invalidated (e.g., due to a missing or damaged filter or flow faults), instrument-specific nephelometric data were used to estimate personal PM2.5 exposure which required additional validity checks. For samples with invalid gravimetric but valid nephelometric measurements, we applied modeled correction factors obtained from regressions of all valid gravimetric and nephelometric pairs based on study arm and site to the adjusted 24-h average nephelometer values, resulting in instrument-specific nephelometric $PM_{2.5}$ concentrations normalized to field-based filter samples.

Quality Control and Assurance. Field blanks were collected at a rate of 4 blanks per 100 filter samples for IRC-specific median blank correction. The limit of detection (LOD) was conventionally calculated for each IRC separately as three times the standard deviation of the blank mass depositions[88], after removing blanks deemed to be invalid due to mislabeling or storage errors. We then replaced sample depositions below the LOD with LOD/($2^{0.5}$).[140]

BC. BC concentrations from $PM_{2.5}$ filter samples were estimated using the SootScan Model OT-21 Optical Transmissometer (Magee Scientific, USA) at either the University of Georgia (UGA, Athens, GA, USA), for samples collected in Guatemala, Peru, and Rwanda, or at Sri Ramachandra Institute for Higher Education and Research (SRIHER, Chennai, India), for samples collected in India. We converted filter absorbance to mass deposition per Garland et al.[34] using the BC attenuation cross-section value for similar Teflon filters ($\sigma = 13.7$ μg/m^2) collected from similar source types. All filters collected for the Guatemala, Peru, and Rwanda samples used both a pre- and postscan. For India, the average of blank filter postscan values was substituted for prescan values. LOD was calculated as it was for gravimetric mass (three times the blank standard deviation of field blank results). Values below the LOD were replaced with LOD/($2^{0.5}$).

CO. CO concentrations were calibrated using zero air and CO span gas (ranging between 40 and 80 ppm by IRC) and checked automatically and at regular intervals via a server-based quality assurance procedure as well as visually with a rating system similar to that applied in the Ghana Randomized Air Pollution and Health Study[67] to ensure quality of the data. CO loggers were calibrated every 1-3 months per Johnson et al.[85] using the temporally closest calibration coefficient. Data outside the range for sampling duration (24 h +/- 4 h) or otherwise flagged due to displayed response artifacts were removed. Duplicate monitors were deployed for a subset of households to evaluate LASCAR performance.

Statistical Analyses. All analyses were performed in R (versions 3.6 and 4.0; R Foundation for Statistical Computing). We provide summaries of baseline household and participant characteristics by treatment arm and IRC. Characteristic summaries for participants with and without missing exposure data are provided in the supplemental. We then calculated pollutant-specific descriptive statistics for valid measurements in control and intervention groups by study visit (baseline versus post-intervention rounds) and IRC. We evaluated Spearman correlations between measurements for the same pollutants collected at baseline and all post-intervention rounds as well as correlations between pollutants at each measurement round. These were evaluated overall and stratified by assigned stove type. Differences in pollutant levels between control and intervention groups by round were evaluated using nonparametric tests (Wilcoxon Rank Sum, Kruskal-Wallis, and Dunn's tests). All aforementioned exposure analyses, stratified by assigned stove type and study period (baseline versus pregnancy and post-birth periods), were subsequently applied to a subset of households with valid measurements for both non-pregnant and enrolled pregnant participants living in the same household for comparison.

We also evaluated the proportion of non-pregnant adult women samples that were less than or equal to WHO guidelines and targets. For $PM_{2.5}$, we compared our measurements to the Annual Interim Target 1 (WHO-IT1) value of 35 $\mu g/m^3$ because it represents an attainable target for

LMICs on the pathway toward achieving the final guideline value of 5 μg/m^3.[65] For CO, we compare our measurements with the WHO 24-h guideline value of 4 mg/m^3 (~ 3.5 ppm) since no annual guideline or target is provided.

Following approaches described in McCracken et al.[115], Chillrud et al.[67], and our previous work[73], we used statistical methods that leverage our study design and repeat measurements to assess the impact of an LPG intervention on exposure to $PM_{2.5}$, CO, and BC throughout an 18 month period. For these regression analyses, we natural log-transformed pollutant concentrations given the right-skewed distribution of measured data. We used linear mixed-effects models to assess the impact of the intervention on log-transformed personal exposures and included a random intercept to account for correlation among repeated measurements made on the same participants (i.e., at baseline and post-intervention visits 1 - 5). We also evaluated non-transformed models to estimate the absolute change in exposures. Finally, we used mixed-effect models with no covariates and a random effect for participant ID to partition variance into its within and between variance components, enabling subsequent intraclass correlation coefficient (ICC) estimations.

Using the same approach from our previous work[73], we fit four models to assess four distinct comparisons (e.g. before and after, between group, and comparison of changes by study phase and visit) for the association between the LPG intervention and exposures among non-pregnant other adult women. Model 1 estimates the difference between baseline and post-intervention exposures in each arm separately. Model 2 estimates the difference in exposures between arms post-intervention. Model 3 estimates the change in exposure in the intervention arm, between study phases (pre- vs. post-intervention), relative to that in the control arm over the same period. Model 4 estimates a similar comparison of changes by study visit. The parameters of interest are the fixed effect for treatment arm (Model 1), the respective fixed effect for study phase in each arm (Model 2), the "treatment arm x study phase" interaction term (Model 3), and the "treatment arm x study visit" interaction term (Model 4). We estimated the percent reduction in personal exposure due to the intervention by exponentiating the parameters of interest, subtracting them from 1, and multiplying by 100. Models were run for the entire data set and separately for each IRC.

The study protocol has been reviewed and approved by institutional review boards (IRBs) and Ethics Committees at Emory University (00089799), Johns Hopkins University (00007403), Sri Ramachandra Institute of Higher Education and Research (IEC-N1/16/JUL/54/49), the Indian Council of Medical Research – Health Ministry Screening Committee (5/8/4-30/(Env)/Indo-US/2016-NCD-I), Universidad del Valle de Guatemala (146-08-2016), Guatemalan Ministry of Health National Ethics Committee (11-2016), Asociación Benefica PRISMA (CE2981.17), the London School of Hygiene and Tropical Medicine (11664-5), the Rwandan National Ethics Committee (No.357/RNEC/2018), and Washington University in St. Louis (201611159). The study has been registered with ClinicalTrials.gov (Identifier NCT02944682).

Results

Household & OAW Characteristics

A total of 418 non-pregnant adult women aged 40 to <70 across the four IRCs were enrolled and completed randomization (209 in the control arm and 209 in the intervention arm). **Table 5.1** summarizes trial-wide and IRC-specific household and participant characteristics by study arms. The baseline characteristics of the intervention and control groups were similar. The mean (SD) age of participants at baseline was 51.8 (7.5) in the control group and 52.3 (8.2) in the intervention group. Most participants had no formal education or did not complete primary school in both controls (167, 79.9%) and intervention (168, 80.4%) groups. Households typically cook indoors. Wood and Charcoal are the primary fuels for households in Guatemala, India, and Rwanda, while cow dung is dominant in Peru.

Exposure Measurements, Data Completeness, Compliance, and QAQC

We visited each woman up to six times (once at baseline, BL, and five times post-intervention, twice before the pregnant woman in the home gave birth, P1 and P2, and three times post-delivery, B1, B2, B4). 100% of women had at least one valid $PM_{2.5}$ measurement. Among those, 80% had three or more valid $PM_{2.5}$ exposure during the 18-month study period. We observed a relatively low ICC of 0.19 (0.23 without baseline) for $PM_{2.5}$ measurements, indicating relatively high variability between measurements taken from the same individual. The ICCs are slightly higher for BC (0.22; 0.40 [without baseline]) and CO (0.31; 0.32 [without baseline] measurements. Approximately 3% of the samples with invalid gravimetric samples were replaced with device-specific adjusted nephelometer values using modeled correction factors.

The average percentage of daytime hours when motion was detected by the instrument, ranged from 19% in India to 82% in Rwanda **(Table 5.S1, Figure 5.S1)**. We did not exclude samples due to low percentages of time that motion was detected because participants were asked to keep the instrumentation nearby when they were conducting activities for which the equipment could not be safely worn.

For BC, 88% of the participants had a valid baseline measurement and at least two valid post-intervention measurements. For CO, 82% of the participants had a valid baseline sample and at least two valid post-intervention samples. All participants with valid baseline measurements had at least one valid post-randomization measurement for both BC and CO. The numbers and percentages of exposure samples successfully collected by visit and IRC are presented in **Table 5.S2**.

Exposure Summary

24-hour personal exposure to $PM_{2.5}$, BC, and CO for non-pregnant adult women participants by study arm, visit, and IRC are summarized in **Tables 5.2-4** and displayed graphically in **Figure 5.1** (IRC-specific plots are in **Figure 5.S2-S4** for $PM_{2.5}$, BC, and CO, respectively). Trial-wide at baseline, there was no statistically significant difference for $PM_{2.5}$ exposure (Wilcoxon rank sum, p = 0.90) between the control group (median: 89.6 μg/m^3; IQR: 44.3 – 130.6) and intervention group (median: 82.7 μg/m^3; IQR: 43.8 – 149.4). Baseline BC (Wilcoxon rank sum, p = 0.84) and CO (Wilcoxon rank sum, p = 0.61) exposures were also similar between the control and intervention arms. Median (IQR) exposures to BC and CO were 10.8 μg/m^3 (6.2 – 16.1) and 1.2 ppm (0.5 – 2.8) in the control group and 10.9 μg/m3 (6.4 – 16.0) and 1.3 ppm (0.3 – 2.6) in the intervention group. However, $PM_{2.5}$ and BC exposures were significantly higher among the controls in Rwanda at baseline. Their baseline median (IQR) for $PM_{2.5}$ and BC were 106 (67.1-

160.3) μg/m^3 and 12.1 (10.1-15.9) μg/m^3 compared to the interventions (PM2.5: 40.7 (29.3-61.3) μg/m^3; BC: 5.8 (4.1-8.5) μg/m^3) (Wilcoxon rank sum, $p < 0.01$).

Given that important life events such as pregnancy and the arrival of a child may have an impact on the exposure of the other adult woman in the same household through a shift of household and caregiving responsibilities, we also summarized exposure by study periods: pregnancy (P1 and P2) and post-birth (B1, B2, and B4) in addition to each study visits. Median post-intervention exposure to $PM_{2.5}$ in the intervention arm was 72% lower, compared to that in the control arm, during pregnancy (23.4 vs. 82.7 μg/m^3) and post-birth (22.8 vs. 82.7 μg/m^3). BC exposures in the intervention group were 78% lower during both pregnancy and post-birth periods (2.4 vs. 10.9 μg/m^3) in comparison with controls. CO exposures were lower in the intervention group by 87% (0.17 vs. 1.30 ppm) and 85% (0.19 vs. 1.30 ppm) during the pregnancy and post-birth period, respectively.

At baseline, 18.4% and 19.8% of $PM_{2.5}$ measurements were less than or equal to the annual WHO-IT1 for PM2.5 in control and intervention arms, respectively. During the post-intervention period, 26.6% of control exposures were at or below the target, while 70% of the intervention exposures fell below the annual WHO-IT1 of 35 μg/m3, and 53% were at or below WHO-IT2 of 25 μg/m3. For CO, 87% and 81% of the 24-hour exposures in the control and intervention arms, respectively, were below the WHO annual guideline value (3.5 ppm) at baseline. Post-intervention, 88% of control CO exposures were below the guideline value, whereas 92% of the intervention exposure were less than the guideline. The percentage below the guideline was slightly higher during pregnancy than the post-birth period for both control and intervention arms.

Exposure Over Time

We plotted personal exposures to $PM_{2.5}$ over time after randomization/intervention trial-wide (**Figure 5.2**) and in each IRC (**Figure 5.S5**). The plot highlights a similar distribution of $PM_{2.5}$ exposures at baseline ($p = 0.9$) but a distinct separation of exposures between the control and intervention groups post-intervention and over the entire ~18-month intervention period. IRC-specific personal $PM_{2.5}$ exposure trends follow a similar pattern, as shown in **Figure 5.2**, although the magnitude of exposures and exposure contrasts vary between sites.

Among the control households, we again observed a small but statistically significant reduction in $PM_{2.5}$ between baseline and the two post-intervention study periods; but the difference between post-intervention periods was not statistically significant. The median $PM_{2.5}$ exposures in the control group were 89.6 $\mu g/m^3$, 68.1 $\mu g/m^3$, and 64.6 $\mu g/m^3$ at baseline, pregnancy, and post-birth, respectively. We note similar trends for BC: 10.8 $\mu g/m^3$, 8.8 $\mu g/m^{3,}$ and 8.6 $\mu g/m^3$ at baseline, pregnancy, and post-birth study periods, respectively. Post-randomization median CO also decreased: 1.25 ppm, 0.83 ppm, and 0.95 ppm for baseline, pregnancy, and post-birth periods, respectively.

We observed a significant and substantial reduction in exposure to all three pollutants in the intervention group between baseline and post-intervention visits. For $PM_{2.5}$, the median exposures decreased from 82.7 $\mu g/m^3$ at baseline to 22.8 $\mu g/m^3$ and 23.4 $\mu g/m^3$ during pregnancy and post-birth periods. For BC, the median baseline exposure was 10.9 μg/m^{3,} and pregnancy and post-birth exposures were 2.43 μg/m^3 and 2.37 μg/m^3, respectively. For CO, the baseline median was 1.30 ppm, and the pregnancy and post-birth median CO were 0.19 ppm and 0.17 ppm, respectively.

Comparison of Non-pregnant and Pregnant Adult Women Exposures

24-hour personal exposure to $PM_{2.5}$, BC, and CO for non-pregnant adult women and pregnant women participants living in the same households by study arm and, period, are summarized in **Table 5.5 (IRC-specific summaries are presented in Table 5.S3-5)** and displayed graphically in **Figure 5.S6-8** . In control households trial wide, median $PM_{2.5}$ exposures were statistically higher for non-pregnant women compared to pregnant women at both baseline (89.4 vs. 71.2 μg/m^3; Dunn's $p = 0.02$) and post-birth (75.6 vs. 52.6 μg/m^3; Dunn's $p = 0.04$) periods. We also observed higher median $PM_{2.5}$ exposures for non-pregnant compared to pregnant women in intervention households during the pregnancy (26.3 vs. 20.1 μg/m^3; Dunn's $p < 0.001$) and post-birth (25.9 vs. 19.1 μg/m^3; Dunn's $p <0.01$) periods. Other pollutant exposures were generally not statistically different between participant types in either treatment arm, except in the intervention arm post-birth, where median exposures for non-pregnant women were statistically higher than those for pregnant women for both BC (2.2 vs. 1.7 μg/m^3; Dunn's $p = 0.04$) and CO (0.3 vs. 0.2 μg/m^3; Dunn's $p < 0.01$). Statistically significant pairwise testing results between non-pregnant and pregnant adult women by IRC, study period, arm, and pollutant are shown in **Table 5.S6**.

Trial-wide, we observed moderate to strong correlations (Spearman's ρ range: 0.36 – 0.78) for all three pollutants between non-pregnant and pregnant participants living in the same household (**Table 5.S7**). At baseline, the correlations between participant types were 0.67, 0.66, and 0.55 for $PM_{2.5}$, BC, and CO exposures, respectively. During the pregnancy period, we observed slightly lower correlations for participants in LPG using households compared to those using biomass for $PM_{2.5}$ (0.52 vs 0.69), BC (0.66 vs 0.78), and CO (0.44 vs 0.50) alike. During the post-birth period, correlations between participant types remained lower among LPG users compared to those using biomass for $PM_{2.5}$ (0.41 vs 0.66) and CO (0.36 vs 0.45); however, the contrary was true for BC (0.71 vs 0.62).

Correlations Between Measurement Rounds and Between Pollutants

We observed moderate to low correlations (Spearman's ρ) of all three pollutants between measurement rounds (**Table 5.S8**). For $PM_{2.5}$ in the control group, the correlations ranged from 0.17 to 0.47. Correlations for BC (range: 0.11 – 0.58) in the control group were similar to $PM_{2.5}$ but were weaker for CO (range: 0.04 – 0.42). Generally, the correlations between measurement rounds in the intervention group are weaker compared to that in the control group, even between the post-intervention measurement rounds. Among intervention households, correlations between all measurement rounds ranged from 0.06 – 0.34, 0.07 – 0.52, and 0.00 – 0.31 for $PM_{2.5}$, BC, and CO exposures, respectively. Among the consecutive visits, the two follow-up visits during pregnancy (P1 and P2) tend to have stronger correlations for all three pollutants in the intervention group. In the control group, stronger correlations are observed between the last follow-up visit during pregnancy (P2) and the first follow-up visit post-birth (B1), indicating the non-pregnant adult women, usually the mother or mother-in-law of the pregnant women living in the same household, might have more consistent cooking-related behavior during these periods of time.

We observed a moderate correlation (trial-wide Spearman's $\rho = 0.53$) between $PM_{2.5}$ and CO among the traditional stove households (intervention group at baseline and all control group measurements), and the correlations are consistent across IRCs. This PM-CO exposure correlation is much stronger than in LPG stove households (intervention group post-intervention measurements) (overall Spearman's $\rho = 0.11$), and the relationship varies by IRC (**Figure 5.S10**). The correlation between BC and CO among traditional stove households was also moderate (trial-wide Spearman's $\rho = 0.49$). Again, the relationships were much weaker among the post-intervention measurements of LPG stove households (trial-wide Spearman's $\rho = 0.05$), as expected due to the lack of biomass combustion, the major source of HAP exposure in the homes. We found a stronger correlation between $PM_{2.5}$ and BC with a trial-wide Spearman's ρ of 0.77 in the traditional stove households and 0.60 in the LPG stove households, although some heterogeneity between countries was presented (**Table 5.S9**). Detailed summary and comparison of correlations between pollutants by stove type and IRC are present in **Table 5.S9 and Figure 5.S9-11**.

Modeling Results

We assessed the effect of the HAPIN LPG cookstove and fuel intervention on personal exposure using three different modeling strategies: "between groups," "before and after," and "comparison of changes." All models showed significant reductions in all three pollutants (**Table 5.6**). Visualization of the results across models for $PM_{2.5}$ is shown in **Figure 5.3** (results for BC and

CO are shown in **Figure 5.S12-13**). The three modeling approaches yield similar estimated percent reduction in $PM_{2.5}$ exposure due to the intervention: 58% (95% CI: 53%, 63%) for the "between groups" approach; 67% (95% CI: 63%, 71%) for the "before and after" approach; and 58% (95% CI: 50%, 65%) for the "comparison of changes" approach (**Table 5.6**). The reductions were similar for BC but more pronounced for CO (**Table 5.6**).

We also modeled untransformed exposures to show the absolute mean reductions. For $PM_{2.5}$, the absolute reductions were 66 (95% CI: 53, 78) $\mu g/m^3$, 83 (95% CI: 70, 97) $\mu g/m^3$, and 77 (95% CI: 55, 100) $\mu g/m^3$ for "between groups," "before and after" and "comparison of changes" strategies, respectively. The "before and after" approach also indicated a 21% (95% CI: 10%, 31%) reduction in exposure between baseline and post-intervention periods for the control group (6 [95% CI: -12, 25] $\mu g/m^3$). The visit-specific "comparison of changes" models (labeled Visit P1 through Visit B4) presented consistent percent reductions in personal $PM_{2.5}$/BC/CO exposures across visits, indicating the effectiveness of the LPG stove and fuel intervention in reducing exposures over time. IRC-specific reductions generally reflect the trial-wide pattern, although the magnitude varied (**Table 5.S10-13**).

Discussion

We contribute to the literature on exposures to other adult women between the ages of 40 and <75 in four diverse LMIC settings. The main results from the present study show that the 18-month HAPIN intervention of an LPG cookstove and continued fuel supply led to substantial and significant reduction of personal exposures to $PM_{2.5}$, BC, and CO in other adult women participants receiving the intervention compared to the controls. The overall median post-intervention $PM_{2.5}$ exposure was 23.0 μg/m^3, representing a 72% reduction from baseline (82.7 μg/m^3). 70% of the post-intervention $PM_{2.5}$ exposures fell below the annual WHO-IT1 of 35 μg/m^3, and 53% were at or below WHO-IT2 of 25 μg/m^3. The overall median BC and CO exposures in the intervention group were 78% and 86% lower, respectively, in comparison with baselines. We conducted six 24-hour personal exposure measurements over the 18-month intervention period, approximately three months apart and varied by 7 μmg/m^3 or less (**Table 5.2; Figure 5.2**) indicating a stable effect in exposure reduction throughout the study period.

Our findings demonstrated the large reductions in personal exposures to three major household air pollutants, $PM_{2.5}$, BC, and CO. The three modeling approaches yield similar estimated percent reduction in $PM_{2.5}$ exposure due to the intervention: 58% (95% CI: 53%, 63%) for the "between groups" approach; 67% (95% CI: 63%, 71%) for the "before and after" approach; and 59% (95% CI: 51%, 66%) for the "comparison of changes" approach (**Table 5.6**). The reductions were similar for BC but more pronounced for CO (**Table 5.6**).

Nonpregnant and Pregnant Women Exposure Relationships

We compared the exposures of younger pregnant and older nonpregnant adult women living in the same household. Differences in exposure distributions are possibly due to various behavioral changes associated with pregnancy, such as dietary requirements, physical activity, time spent at home, cooking activity, occupation, and child rearing.[19,141] Although we observed moderate to strong correlations between pollutant exposures of pregnant and nonpregnant women in our study, these correlations varied considerably across IRCs. Specifically, weak, and consistently weaker correlations were found in Peru compared to Guatemala and India. This variation in correlations suggests that differences in time-activity patterns between participant groups may influence the levels of exposure observed in each demographic. In general, we found that nonpregnant women had higher exposures compared to pregnant women. These differences were statistically significant for all pollutants among intervention households during pregnancy. Our findings align with previous studies on cookstove interventions, which demonstrated statistically significant reductions in CO emissions for pregnant women in Guatemala[142] and India[143] but not for their nonpregnant counterparts. The results of our study, particularly in intervention households, were mainly driven by participants from Peru. This is the only IRC where significant differences in exposure were observed for all pollutants during each post-intervention period. The low correlations between pregnant and nonpregnant women in Peru suggest the presence of distinct time-activity patterns that contribute to exposure differences. However, further comparative analyses are necessary to better understand and characterize these differential behavioral patterns.

Exposure Comparisons with Previous Studies

Several recent HAP studies provide notable yet imperfect comparisons for pollutant exposures seen in the current study considering the near-exclusive use of LPG in HAPIN.[118] Our previous work characterized exposure reductions associated with LPG use among pregnant women enrolled in HAPIN.[19] The study reported statistically significant exposure reductions, after adjusting for those seen in the control group ("comparison of changes"), of 62%, 62%, and 82% for PM, BC, and CO, respectively, which are similar to exposure reductions (58% for $PM_{2.5}$, 59% for BC, 72% for CO) observed in the current study for nonpregnant women in HAPIN. Median postintervention exposures for pregnant women in HAPIN intervention households (15 –

34 μg/m^3 for $PM_{2.5}$; 2.7 – 2.8 μg/m^3 for BC; and 0.2 ppm for CO) were well within the ranges we report for corresponding nonpregnant women (19 – 26 μg/m^3; 1.7 – 3.0 μg/m^3; and 0.1 – 0.3 ppm). These findings together suggest that the HAPIN intervention package additionally provided improved air quality for individuals who typically were not the primary cooks in the household.

Comparisons with exposure estimates from studies in similar regions also highlight the effectiveness of the HAPIN intervention in reducing exposures to household air pollutants. In Peru, the Cardiopulmonary outcomes and Health and Air Pollution (CHAP) trial conducted an LPG intervention for 180 adult women and reported statistically significant reductions in pollutant exposures for participants in the intervention group (range for $PM_{2.5}$: 66 – 78%; range for BC: 65 – 89%; range for CO: 48 – 71%) as well as those in the control group (range for $PM_{2.5}$: 66 – 78%; range for BC: 65 – 89%; range for CO: 48 – 71%) after implementation of the intervention.[71] The study also reported median postintervention PM2.5 exposures of 15 – 25 μg/m^3 for those in the intervention arm (in comparison with medians of 12 - 22 μg/m^3 in the intervention arm for the HAPIN Peru site). Similar distributions of the other pollutant exposures were observed in both studies as well. In India, the Tamil Nadu Air Pollution and Health Effects (TAPHE) cohort study of pregnant women estimated median $PM_{2.5}$ exposures of 75 μg/m^3 for biomass stove users and 46 μg/m^3 for those using primarily LPG (in comparison with 46 – 79 and 18 – 39 μg/m^3 in the control and intervention arms, respectively, for the HAPIN site in Tamil Nadu, India).[138] A panel study of 45 women in rural India observed geometric mean exposures to BC during cooking sessions ranging from 40 – 56 μg/m^3 (in comparison with 6.9 – 11.7 μg/m^3 in the control arm for the HAPIN site in Tamil Nadu, India).[54] In Guatemala, an exposure assessment of 218 pregnant women reported median exposures to $PM_{2.5}$ at 148 and 55 $\mu g/m^3$ for participants owning biomass and LPG stoves, respectively (in comparison with 55 – 106 and 23 – 29 $\mu g/m^3$ in the control and intervention arms, respectively, for the HAPIN site in Guatemala), resulting in an estimated 38% reduction in $PM_{2.5}$ for LPG owners.[144] In Rwanda, a trial of rocket-style cookstoves and water filters reported median exposures of 146 and 158 $\mu g/m^3$ in the control and intervention arms, respectively, for the primary cook (in comparison with 69 – 106 $\mu g/m^3$ in the control arm and 20 – 57 $\mu g/m^3$ postintervention for the HAPIN site in Rwanda).[145]

Special considerations are necessary when comparing exposure estimates from HAPIN to those from other relevant HAP studies. As an efficacy trial, HAPIN's study design aimed to eliminate contributions from pollution sources by implementing strategies to support exclusive LPG use, such as free provision and delivery of stoves and fuel supply, delivering behavior modification messaging, and ensuring stove maintenance.[86] Furthermore, study sites with low background air pollution and relatively low-density housing were carefully selected[14,85,134,135] to accurately assess the reduction in air pollutant exposures under "ideal" conditions. The high adherence to the HAPIN intervention may explain the lower exposures among LPG users in HAPIN relative to those reported in the PURE study as LPG users in PURE likely utilized biomass stoves as well. Additionally, previous studies, such as Pope et al.[11] and Alexander et al.[66], have cited mixed fuel use and ambient air pollution as potential reasons for consistently elevated among LPG users, exceeding health-relevant targets. Although GRAPHS did not measure ambient pollution, they found a positive association between air pollution exposure and population density[67], highlighting a "neighboring effect" that could attenuate the exposure contrast between treatment groups. This effect, coupled with contributions from other sources increasing background pollution levels, may also explain the higher exposures observed among biomass users in other studies relative to those in HAPIN, given the contextual differences of our study sites.

Study Strengths

The current study demonstrates several notable strengths. Firstly, it stands as one of the largest endeavors examining the impact of a household intervention on personal exposures specifically focusing on a previously underrepresented demographic of women. This research addresses an important gap in the literature. Secondly, extensive pretrial testing allowed us to develop targeted strategies aimed at promoting exclusive LPG use. This, in turn, resulted in high adherence (>96%) to the cookstove intervention implemented throughout HAPIN.[118] The careful planning and implementation of these interventions contribute to the robustness of our findings. Additionally, we established best practices for data collection, cleaning, and analysis, ensuring the internal and external credibility of our exposure estimates.[70,134,135] This meticulous approach enhances the reliability and validity of our study results. Thirdly, we conducted a comprehensive exposure assessment by collecting up to six repeated measurements of multiple pollutant exposures per participant with relatively high data completeness. This longitudinal design allowed us to capture exposure dynamics over time. The availability of high-quality longitudinal exposure data is crucial for subsequent epidemiological analyses and accurate quantification of exposure-response relationships. Overall, these strengths contribute to the scientific rigor and value of our research, providing a solid foundation for further investigation and informing future interventions.

Study Limitations

Our study also has some limitations. First, as an efficacy trial, HAPIN provided free LPG cookstoves and continued fuel supply over the entire study period. Combined with behavioral reinforcement activities as needed, the trial achieved high fidelity and exclusive use of the intervention. A similar exposure contrast between the LPG and biomass cookstove might be hard to observe in contexts without such support. Moreover, to assess the largest achievable exposure reduction from LPG cookstoves in the field, we deliberately selected study sites without major air pollution point sources (low contribution of background air pollution levels to personal exposures).[14,73,135] This could limit the implication of exposure reduction from clean household energy intervention to areas with exposure sources other than burning solid fuels, such as garbage burning, road traffic, and industry pollution.

Second, although the HAPIN trial collected a large number of repeated personal exposure measurements in HAP research, up to six 24 h measurements over 18-month study period (roughly three months apart), it may still prove inadequate to fully capture the impacts of behavioral and environmental factors on personal exposures over time, resulting in some risk of exposure misclassification. A recently published intensive field sampling indicated that >48 h sampling duration substantially reduces measurement variation, and repeated sampling per week or month led to a higher probability of being closer to the "true" long-term average concentrations.[146] Still, our findings showed that the high adherence to intervention resulted in stable exposure reduction (**Figures 5.2 and 5.3**), suggesting that our measurements provided a reasonable estimate of long-term average exposures over the study period. Another source of exposure measurement error may come from the wearing compliance of exposure instruments.

We could not rule out the possibility of participants' behavior changing when wearing the exposure instrument during the sampling period, leading to a departure from their "true" exposure.

Third, although the exposure levels among the controls remained high, we observed ~20% exposure reduction simultaneously in the control group post-intervention. This reduction might be due to the nature of the intervention and study design. Participants and field workers were not blinded to the study arm, and the frequent interactions between participants and the field team for exposure monitoring and health checkups may have improved the awareness of harmful HAP exposures and led to behaviors changes associated with solid fuel uses. If this was the case, the exposure contrast between LPG and biomass cookstoves could have been more prominent, and the observed percentage reduction in the current analysis is underestimated.

Additionally, with a large number of participants being followed over a long study period and during the COVID-19 pandemic, some sample loss was inevitable. The trial suspended data collection due to the pandemic in March 2020 and resumed household visits during the fifth year of the trial.[147] The lockdown impacted some post-birth (i.e., B1, B2, and B4) exposure assessments. Among 418 enrolled, on average, 88% had successful exposure visits pre-pandemic compared to 66% post-birth and during the pandemic **(Table 5.S2)**.

Conclusions

This analysis suggests that an 18-month LPG cookstove/fuel intervention can substantially and consistently reduce personal HAP exposure among non-pregnant women (primary cook or non-cook) living in households that rely on solid fuels. The trial yielded up to six personal $PM_{2.5}$, BC, and CO exposure measurements per participant and is one of the largest and more comprehensive personal air pollution exposure monitoring efforts in the context of clean cooking and HAP to date. The presented exposure contrast between women using biomass and LPG cookstoves/fuel is among the largest of all other household energy intervention studies. As an efficacy trial with high fidelity and adherence to the intervention, HAPIN showed the highest observable exposure reductions from using LPG for cooking in the field of four LMICs characterized by diverse socioeconomic, cultural, behavioral, and environmental factors. Our findings provide evidence that implementing a clean household energy intervention can effectively reduce personal air pollution exposure and achieve levels below the annual WHO IT-1 target of 35 $\mu g/m^3$.

Table 5.1. Household and other adult women participants characteristics at baseline, by IRC and study arm.

Variable	Guatemala		India		Peru		Rwanda	
	Control	Intervention	Control	Intervention	Control	Intervention	Control	Intervention
	(n = 67)	(n = 71)	(n = 51)	(n = 53)	(n = 67)	(n = 66)	(n = 24)	(n = 19)
Household and kitchen characteristics								
Household Size								
Mean (SD)	7.8 (3.0)	7.7 (2.8)	4.3 (1.4)	4.4 (1.3)	5.8 (1.9)	5.4 (1.7)	6.1 (2.2)	5.7 (2.2)
Range	3 - 18	3 - 17	2 - 8	2 - 9	3 - 12	2 - 10	3 - 10	2 - 10
Missing	0	0	0	0	0	0	0	0
Access to electricity								
No	2 (3%)	5 (7%)	2 (3.9%)	1 (1.9%)	6 (9%)	2 (3%)	16 (66.7%)	12 (63.2%)
Yes	65 (97%)	66 (93%)	49 (96.1%)	52 (98.1%)	61 (91%)	64 (97%)	7 (29.2%)	5 (26.3%)
Missing	0	0	0	0	0	0	1 (4.2%)	2 (10.5%)
Kitchen volume (m3)								
Mean (SD)	45.1 (27.6)	41.8 (23.1)	18.3 (13)	18.7 (13.7)	24.3 (21.3)	54.2 (166)	12.4 (5.7)	12.4 (4.5)
Range	12.2-174	10.3-174	4.2-64.6	1.8-61.2	4.4-150	4-1148	4.8-23.5	5.5-23.5
n	66	69	51	53	51	56	18	16
Missing (n)	1	2	0	0	16	10	6	3
Roof in the kitchen								
No	0	1 (1.4%)	0	0	15 (22.4%)	10 (15.2%)	6 (25%)	3 (15.8%)
Yes	66 (98.5%)	70 (98.6%)	51 (100%)	53 (100%)	52 (77.6%)	56 (84.8%)	18 (75%)	16 (84.2%)
Missing	1 (1.5%)	0	0	0	0	0	0	0
Number of stoves								
One	12 (17.9%)	14 (19.7%)	29 (56.9%)	32 (60.4%)	19 (28.4%)	18 (27.3%)	14 (58.3%)	10 (52.6%)
Two	43 (64.2%)	40 (56.3%)	21 (41.2%)	20 (37.7%)	41 (61.2%)	44 (66.7%)	8 (33.3%)	8 (42.1%)
Three or more	12 (17.9%)	17 (23.9%)	1 (2%)	1 (1.9%)	7 (10.4%)	4 (6.1%)	1 (4.2%)	1 (5.3%)
Missing	0	0	0	0	0	0	1 (4.2%)	0
Primary stove has a chimney								
No	51 (76.1%)	55 (77.5%)	51 (100%)	52 (98.1%)	38 (56.7%)	41 (62.1%)	22 (91.7%)	19 (100%)
Yes	16 (23.9%)	16 (22.5%)	0	1 (1.9%)	29 (43.3%)	25 (37.9%)	1 (4.2%)	0
Missing	0	0	0	0	0	0	1 (4.2%)	0
Primary fuel type								
Cow dung	0	0	0	0	62 (93.9%)	54 (81.8%)	0	0
Wood	67 (100%)	70 (100%)	51 (100%)	53 (100%)	4 (6.1%)	8 (12.1%)	23 (95.8%)	16 (84.2%)
Charcoal	0	0	0	0	0	0	0	3 (15.8%)
Other	0	0	0	0	0	4 (6.1%)	0	0
Missing	0	0	0	0	0	0	1 (4.2%)	0
Primary stove location								
In participant's	1 (1.5%)	1 (1.4%)	7 (10%)	11 (20.8%)	0	0	0	1 (5.3%)
Room immediately adjacent to the participant's bedroom	21 (31.3%)	18 (25.4%)	22 (43.1%)	13 (24.5%)	1 (1.5%)	4 (6.1%)	0	0
Separated from the participant's bed-room but inside the house	27 (40.3%)	32 (45.1%)	9 (17.6%)	11 (20.8%)	9 (13.4%)	13 (19.7%)	0	2 (10.5%)
Outside the house (outdoors)	1 (1.5%)	1 (1.4%)	0	0	18 (26.9%)	10 (15.2%)	6 (25%)	3 (15.8%)
In a separate building detached from the bedroom-main home	17 (25.4%)	19 (26.8%)	13 (25.5%)	18 (34%)	39 (58.2%)	39 (59.1%)	17 (70.8%)	13 (68.4%)
Missing	0	0	0	0	0	0	1 (4.2%)	0
Primary light source								
Torch (battery)	0	0	0	1 (1.9%)	0	0	4 (16.7%)	5 (26.3%)
Kerosene lamp	0	0	3 (5.9%)	3 (5.7%)	0	0	1 (4.2%)	1 (5.3%)
Solar light	0	0	0	0	3 (4.5.%)	3 (4.5%)	10 (41.7%)	7 (36.8%)
Electricity	65 (97%)	65 (91.5%)	48 (94.1%)	49 (92.5%)	60 (89.6%)	60 (90.9%)	6 (25%)	4 (21.1%)
Other	2 (3%)	6 (8.5%)	0	0	4 (6.0%)	3 (4.5%)	2 (8.3%)	2 (10.5%)
Missing	0	0	0	0	0	0	1 (4.2%)	0
Presence of a smoker in home								
No	58 (86.6%)	64 (90.1%)	32 (62.7%)	35 (66%)	67 (100%)	66 (100%)	22 (91.7%)	17 (89.5%)
Yes	9 (13.4%)	7 (9.9%)	19 (37.3%)	18 (34%)	0	0	1 (4.2%)	2 (10.5%)
Missing	0	0	0	0	0	0	1 (4.2%)	0
Participant characteristics								
Age (year)								
Mean (SD)	53.3 (6.7)	53.8 (9.1)	49.5 (5.7)	48.6 (7.2)	52.3 (8.7)	52.8 (6.9)	51 (8.3)	55.2 (8.9)
Range	41.5-73.8	40.4-74.2	40.6-68	40.2-71.6	40.1-73.5	40.9-73.6	40.5-66.1	42.6-74.3
Missing	0	0	0	0	0	0	0	0
Occupation								
Agriculture	0	1 (1.4%)	41 (80.4%)	38 (71.7%)	9 (13.4%)	14 (21.2%)	21 (87.5%)	12 (63.2%)
Commercial	1 (1.5%)	3 (4.2%)	1 (2%)	2 (3.8%)	2 (3.0%)	2 (3.0%)	1 (4.2%)	3 (15.8%)
Household	65 (97%)	66 (93%)	4 (7.8%)	7 (13.2)	53 (79.1%)	46 (69.7%)	0	1 (5.3%)
Other	1 (1.5%)	1 (1.4%)	2 (3.9%)	2 (3.8%)	3 (4.5.%)	3 (4.5%)	2 (8.3%)	1 (5.3%)
Unemployed	0	0	3 (5.9%)	4 (7.5%)	0	0	0	2 (10.5%)
Missing	0	0	0	0	0	1 (1.5%)	0	0

Table 5.2. Summary of personal exposure to $PM_{2.5}$ of other adult women participants by IRC and study group.

$PM_{2.5}$ Exposure	Guatemala		India		Peru		Rwanda		Overall	
	Control	Intervention	Control	Intervention	Control	Intervention	Control	Intervention	Control	Intervention
Baseline										
N	63	67	46	43	51	50	19	12	179	172
Average (SD)	125.1 (100.6)	146.3 (117.8)	102.7 (93.3)	118.8 (152.4)	90.7 (90.7)	107.6 (113.2)	117.9 (66.8)	48.3 (26.2)	108.8 (93.2)	121.3 (124.3)
Range	13.7 - 661.2	17.3 - 622.4	9.7 - 531.3	9.9 - 759.4	12.5 - 427.4	12.5 - 477.7	14.7 - 297.1	19.8 - 95.4	9.7 - 661.2	9.9 - 759.4
Median (IQR)	105.8 (69.3-135.7)	115.8 (75.9-176.1)	78.7 (46.3-125.4)	71.1 (40.4-122.9)	60.3 (26.4-121.5)	67.1 (31.5-123)	106 (67.1-160.3)	40.7 (29.3-61.3)	89.6 (44.3-130.6)	82.7 (43.8-149.4)
Post-intervention Visit 1										
N	54	58	39	46	40	47	17	12	150	163
Average (SD)	131.6 (120.8)	30.3 (29.7)	89.2 (96)	35.8 (66.1)	65.4 (70.7)	44.6 (56.3)	101 (84.4)	33.4 (21.9)	99.5 (101.6)	36.2 (49.9)
Range	11.8 - 602.3	9.7 - 176.1	10.2 - 407.2	10.1 - 464.1	13.1 - 300.8	12.4 - 289.8	23 - 318.1	14.6 - 92.2	10.2 - 602.3	9.7 - 464.1
Median (IQR)	96.4 (54.1-171.1)	24.6 (11.8-30)	46 (31.9-113.7)	19.5 (17.7-34)	40.7 (14.8-87.3)	15.7 (14.8-53.1)	81.1 (56.6-110.4)	20.9 (20.2-41.6)	64.6 (33.8-130.8)	21.2 (15-35)
Post-intervention Visit 2										
N	47	59	35	32	33	41	17	12	132	144
Average (SD)	118 (112.5)	37.7 (36)	99 (89.5)	25.8 (12.2)	66.1 (110.9)	49.7 (111)	78.9 (68.6)	31.9 (11.3)	94.9 (102.6)	38 (64)
Range	11.8 - 502.1	9.7 - 222.7	9.5 - 392.9	9.7 - 59.7	12.3 - 541.2	12.5 - 724.8	30.5 - 295.5	20.1 - 49.1	9.5 - 541.2	9.7 - 724.8
Median (IQR)	77.6 (47.2-147.1)	24.7 (11.9-47.3)	78.4 (41.3-111.4)	18 (17.8-35.6)	29.9 (15-57)	22.1 (14.8-45.9)	61.1 (43.7-69.7)	29.3 (21.5-42)	63.4 (32.5-108.7)	23.9 (15.2-41.9)
Post-intervention Visit 3										
N	44	46	29	34	18	31	13	7	104	118
Average (SD)	133.4 (113.7)	35.8 (31.3)	92 (98.7)	47.7 (119.8)	40.6 (33)	28.8 (40.5)	88.6 (71.2)	35.2 (11.3)	100.2 (99.7)	37.4 (70)
Range	21.5 - 520.3	11.2 - 171.4	9.8 - 412.2	13.1 - 718.2	12.7 - 100.3	12.5 - 233.6	18.8 - 275.2	20.8 - 55.5	9.8 - 520.3	11.2 - 718.2
Median (IQR)	104 (56.3-170.6)	29.4 (12.6-40.5)	71.9 (26.1-101.1)	18.2 (17.8-33)	24.5 (15-59.1)	14.9 (14.2-27.8)	77.9 (38.2-92.2)	34.3 (29.5-38.4)	72.2 (36.6-118.8)	24.4 (14.3-35.2)
Post-intervention Visit 4										
N	42	32	32	38	23	30	16	5	113	105
Average (SD)	115 (101.9)	41.1 (38.1)	119.6 (249)	27.3 (27)	116.6 (219.6)	71.8 (181.1)	100.7 (81)	45.1 (19.8)	114.6 (177.1)	45 (100.9)
Range	11.8 - 457.9	11.1 - 162.9	11.8 - 1447.3	13.4 - 138.3	10.6 - 1002.2	11 - 968.9	23.5 - 355.6	19.8 - 72.8	10.6 - 1447.3	11 - 968.9
Median (IQR)	80.3 (53-150.6)	27.8 (15.2-43.1)	71.2 (36.8-104.7)	18 (14.9-27.6)	29.2 (15-96)	15.1 (12.7-43.3)	83.3 (50.9-113.2)	48.2 (34.1-50.6)	69.4 (32.2-122.3)	18.9 (14.3-38.4)
Post-intervention Visit 5										
N	40	45	40	40	17	29	9	5	106	119
Average (SD)	95.2 (86.1)	34.9 (32.9)	106.3 (101.3)	50.6 (43.7)	163.3 (297.7)	24.9 (29.1)	85.5 (40.2)	43.6 (22.2)	109.5 (144.2)	38.1 (36.8)
Range	12 - 314.4	11 - 137.4	13.2 - 434.5	4.2 - 227	11 - 1062	10.5 - 136	47.5 - 158.2	14 - 64.3	11 - 1062	4.2 - 227
Median (IQR)	55.2 (35.8-129.8)	22.6 (14.4-34.9)	63.8 (36.7-137.3)	39.3 (17.9-56.1)	19.8 (12.9-144.3)	12.4 (10.9-26.3)	69.3 (59.2-89.6)	56.6 (25.9-57.3)	58 (31.5-134.5)	25.9 (13.1-43.9)

Table 5.3. Summary of personal exposure to BC of other adult women participants by IRC and study group.

BC Exposure	Guatemala		India		Peru		Rwanda		Overall	
	Control	Intervention	Control	Intervention	Control	Intervention	Control	Intervention	Control	Intervention
Baseline										
N	54	63	45	43	44	45	16	10	159	161
Average (SD)	12.3 (5.8)	13.4 (6.7)	14.4 (11.7)	12.4 (10.4)	11.5 (14.3)	14.5 (16.8)	12.9 (4.1)	6.4 (3.1)	12.8 (10.4)	13 (11.3)
Range	1.1 - 29	4.3 - 46.8	1.6 - 69.2	1.5 - 47.8	1.4 - 72.3	1.3 - 93.3	6.2 - 20.1	2.8 - 12.3	1.1 - 72.3	1.3 - 93.3
Median (IQR)	11.5 (8.8-15.7)	12.4 (9.8-14.8)	11.7 (7.2-18.3)	8 (3.8-20.6)	6 (2.1-15.4)	10.1 (3.4-16.5)	12.1 (10.1-15.9)	5.8 (4.1-8.5)	10.8 (6.2-16.1)	10.9 (6.4-16)
Post-intervention Visit 1										
N	53	56	38	46	34	44	17	11	142	157
Average (SD)	11.8 (6.3)	4 (3.2)	9.8 (8.1)	2.8 (2)	7.1 (6.7)	1.9 (1.5)	11.1 (7.9)	4.1 (1.5)	10 (7.3)	3.1 (2.5)
Range	1 - 33.1	0.9 - 16.9	1.7 - 36.9	1.4 - 9.5	1.3 - 24.8	1.3 - 10.6	2.9 - 37.7	2.8 - 6.6	1 - 37.7	0.9 - 16.9
Median (IQR)	11 (8-13.5)	2.9 (1.7-5.5)	6.9 (3.9-16.6)	1.7 (1.7-3.2)	4.1 (1.5-11.3)	1.5 (1.5-1.5)	8.8 (7.6-13.8)	3 (2.9-5.3)	8.7 (4.5-13.8)	1.7 (1.5-3.5)
Post-intervention Visit 2										
N	46	58	34	32	28	40	17	12	125	142
Average (SD)	11.5 (5.4)	5.4 (4.9)	12.3 (11.7)	3.2 (2.9)	8.4 (17)	2.3 (2)	6.9 (2.8)	4 (1.4)	10.4 (10.7)	3.9 (3.8)
Range	1 - 26.5	0.9 - 28.5	1.7 - 59.9	1.6 - 16.1	1.4 - 83.9	1.4 - 11.9	3.4 - 15.9	2.8 - 6.6	1 - 83.9	0.9 - 28.5
Median (IQR)	11.1 (8.5-13.5)	3.6 (2.4-7.3)	9 (4.4-14.3)	1.7 (1.7-3.8)	1.5 (1.4-6.5)	1.5 (1.5-2.3)	6.2 (5.3-8)	3 (2.9-5.1)	8.2 (4.1-13.2)	2.7 (1.6-4.7)
Post-intervention Visit 3										
N	37	42	29	34	18	31	13	7	97	114
Average (SD)	14.2 (8.1)	5.7 (4.6)	13.3 (15.3)	4.4 (6.6)	6.1 (6.6)	4.5 (12.4)	10 (8.1)	3.6 (1.2)	11.9 (10.9)	4.8 (7.9)
Range	5.5 - 52.3	0.9 - 19.6	1.6 - 75.9	1.6 - 36.5	1.4 - 22.8	1.4 - 70.6	2.8 - 33.3	2.9 - 5.9	1.4 - 75.9	0.9 - 70.6
Median (IQR)	12.5 (9.8-16.8)	3.6 (2.5-9.3)	8 (4.4-15.4)	1.7 (1.7-3.6)	2.6 (1.5-8.9)	1.5 (1.5-2.2)	8.8 (4.6-11.1)	3 (3-3.7)	9.6 (4.7-15)	2.5 (1.6-4.5)
Post-intervention Visit 4										
N	36	27	32	38	23	30	16	5	107	100
Average (SD)	11.4 (6)	6.5 (6.4)	10.2 (8)	2.2 (1.3)	7.8 (8.7)	2.6 (2.9)	9.8 (4.3)	4.8 (1.6)	10 (7.1)	3.6 (4.1)
Range	1 - 27.7	0.9 - 24.4	1.5 - 31.7	1.5 - 8.9	1.4 - 31.9	1.4 - 13.2	3.7 - 16.9	2.8 - 7.2	1 - 31.9	0.9 - 24.4
Median (IQR)	11.3 (7.8-13.8)	3.5 (2.2-7.6)	7.7 (5-12.4)	1.7 (1.7-1.9)	3.7 (1.5-10.2)	1.5 (1.5-2.2)	10 (5.6-12.7)	4.6 (4.5-4.7)	8.5 (4.7-13.6)	1.7 (1.6-3.2)
Post-intervention Visit 5										
N	29	37	38	40	16	28	9	5	92	110
Average (SD)	10.1 (5.7)	5.6 (4.8)	11.1 (10.4)	5.8 (9.7)	4.6 (5)	3.2 (3.7)	9.5 (4.5)	5.2 (1.8)	9.5 (8.1)	5 (6.8)
Range	1 - 18.9	0.9 - 17.4	1.6 - 56.4	1.5 - 53.2	1.4 - 15.2	1.4 - 20.4	4.1 - 17.6	2.9 - 7.8	1 - 56.4	0.9 - 53.2
Median (IQR)	9.9 (5.5-15.7)	3.4 (2.1-8.3)	8.4 (4-14.5)	2.4 (1.7-5.2)	1.6 (1.5-5.8)	2.3 (1.5-3)	8.9 (5.9-12.4)	5.1 (4.3-6.1)	7.9 (3.7-13.3)	3 (1.7-5.2)

Table 5.4. Summary of personal exposure to CO of other adult women participants by IRC and study group.

CO Exposure	Guatemala		India		Peru		Rwanda		Overall	
	Control	Intervention	Control	Intervention	Control	Intervention	Control	Intervention	Control	Intervention
Baseline										
N	54	63	40	37	37	40	17	11	148	151
Average (SD)	1.7 (1.7)	1.6 (1.3)	1.2 (1.4)	1.6 (2.5)	4 (3.9)	3.2 (3.8)	1.2 (1)	0.8 (1.1)	2.1 (2.6)	2 (2.6)
Range	0.1 - 7.3	0 - 5	0 - 6.3	0 - 11	0 - 18.1	0 - 16.5	0.1 - 3.8	0 - 3.8	0 - 18.1	0 - 16.5
Median (IQR)	1.2 (0.6-2.2)	1.4 (0.5-2.5)	0.6 (0.2-2.1)	0.4 (0.1-2.1)	3.1 (1.3-5.8)	1.8 (0.9-3.3)	0.9 (0.5-1.7)	0.3 (0.1-0.9)	1.2 (0.5-2.8)	1.3 (0.3-2.6)
Post-intervention Visit 1										
N	52	52	35	44	27	35	16	10	130	141
Average (SD)	1.7 (1.4)	0.6 (0.9)	1.6 (2)	0.3 (1.4)	2 (2.5)	1.7 (3)	1.2 (1)	0.4 (0.5)	1.7 (1.8)	0.8 (1.8)
Range	0 - 6.6	0 - 4.6	0 - 6.9	0 - 9.5	0 - 10.3	0 - 14.5	0 - 2.9	0 - 1.4	0 - 10.3	0 - 14.5
Median (IQR)	1.3 (0.6-2.7)	0.3 (0-0.6)	0.7 (0.1-2.1)	0 (0-0.1)	1.3 (0.2-2.5)	0.5 (0.2-1.9)	0.8 (0.3-1.8)	0.3 (0.1-0.4)	1 (0.4-2.5)	0.1 (0-0.6)
Post-intervention Visit 2										
N	41	51	29	29	26	35	15	9	111	124
Average (SD)	1.4 (1.4)	0.7 (0.9)	1.5 (2)	0.4 (1.3)	1.7 (2.9)	1.8 (2.8)	0.6 (0.9)	1.7 (4)	1.4 (1.9)	1 (2.1)
Range	0 - 6.6	0 - 3.6	0 - 8.9	0 - 6.7	0 - 13.9	0 - 13.7	0 - 3.3	0 - 12.3	0 - 13.9	0 - 13.7
Median (IQR)	1 (0.6-1.8)	0.2 (0-0.9)	0.9 (0.2-1.8)	0 (0-0.1)	0.9 (0.1-1.4)	0.7 (0.2-2)	0.2 (0.1-0.5)	0 (0-1)	0.9 (0.2-1.7)	0.2 (0-1)
Post-intervention Visit 3										
N	37	36	25	29	10	26	11	4	83	95
Average (SD)	1.8 (2)	0.6 (1)	1.5 (1.8)	0.4 (1)	3 (4.8)	3.4 (5.2)	1 (1.1)	0.9 (0.7)	1.7 (2.4)	1.3 (3.1)
Range	0.1 - 8.6	0 - 3.9	0 - 5.6	0 - 4.1	0 - 15.5	0.1 - 18.9	0.1 - 2.9	0 - 1.6	0 - 15.5	0 - 18.9
Median (IQR)	1.1 (0.3-2.5)	0.1 (0-0.5)	0.6 (0.2-2.5)	0.1 (0-0.1)	1.5 (0.1-2.7)	0.8 (0.5-3.7)	0.3 (0.2-1.9)	1 (0.4-1.5)	0.7 (0.2-2.5)	0.3 (0.1-1.1)
Post-intervention Visit 4										
N	39	27	32	36	18	23	14	5	103	91
Average (SD)	1.6 (1.7)	0.7 (2)	1.4 (1.8)	0.5 (1.1)	4 (4.9)	4.1 (4.9)	1.8 (3.7)	1.2 (1.1)	2 (3)	1.5 (3.1)
Range	0 - 6.5	0 - 10.1	0 - 6.4	0 - 4.7	0 - 14.8	0.1 - 17.8	0 - 13.5	0.2 - 3	0 - 14.8	0 - 17.8
Median (IQR)	0.9 (0.4-2.3)	0.2 (0-0.5)	0.7 (0.2-1.5)	0.1 (0-0.4)	1.7 (0.3-5.9)	2.8 (0.6-5.1)	0.4 (0.2-0.8)	0.9 (0.6-1.3)	0.8 (0.2-2.3)	0.3 (0-1.1)
Post-intervention Visit 5										
N	35	42	38	37	11	17	8	4	92	100
Average (SD)	1.5 (2.3)	0.4 (0.7)	1.5 (2)	0.7 (1.9)	2.7 (2.9)	3.1 (4.8)	0.9 (0.5)	0.2 (0.3)	1.6 (2.2)	1 (2.5)
Range	0 - 12.2	0 - 3	0 - 8.6	0 - 10.7	0 - 8.7	0 - 15.2	0.1 - 1.6	0 - 0.6	0 - 12.2	0 - 15.2
Median (IQR)	0.5 (0.2-2)	0.2 (0-0.4)	0.9 (0.3-2)	0 (0-0.3)	1.3 (0.7-4.5)	0.6 (0.3-3.8)	1 (0.5-1.2)	0.1 (0.1-0.3)	0.9 (0.3-1.9)	0.1 (0-0.6)

Table 5.5. Comparison of personal exposures to $PM_{2.5}$, BC, and CO between pregnant and non-pregnant adult women from the same households by study period.

	PM2.5				BC				CO			
	Control		Intervention		Control		Intervention		Control		Intervention	
	OAW	PW	OAW	PW	OAW	PW	OAW	PW	OAW	PW	OAW	PW
Baseline												
N	169	169	163	163	145	145	142	142	150	150	155	155
Average (SD)	109.8 (103.9)	92.1 (84.3)	117.5 (120.9)	129.1 (159.7)	12.7 (10.4)	11.8 (9.7)	12 (8.7)	13 (10.5)	2.2 (2.7)	1.8 (2.5)	2.3 (4)	2.6 (4.5)
Range	9.7 - 885.3	11 - 448	9.9 - 759.4	9.4 - 1381.9	1.1 - 72.3	1.6 - 66.1	1.4 - 49.3	0.6 - 77.2	0 - 18.1	0 - 16.8	0 - 38.7	0 - 32.4
Median (IQR)	89.4 (43.9-129)	71.2 (35.2-114.1)	80.1 (43.2-145.2)	82 (42.4-153.1)	11 (6.3-16.1)	10.5 (6.4-14.4)	10.7 (6.5-15.3)	11.3 (6.2-16.2)	1.3 (0.5-2.8)	1 (0.4-2)	1.3 (0.3-2.7)	1.3 (0.4-2.8)
Pregnancy												
N	164	164	172	172	154	154	168	168	162	162	173	173
Average (SD)	98.9 (101.5)	111.2 (137.9)	38.2 (49.6)	25.3 (15.3)	10.2 (8.8)	10.5 (8.2)	3.5 (2.6)	3.2 (2.4)	2 (2.9)	1.9 (2.9)	0.9 (1.9)	0.5 (1)
Range	11.8 - 602.3	10.7 - 1116.8	9.7 - 464.1	9.6 - 102.2	1 - 83.9	1.4 - 65.3	0.9 - 16.6	0.7 - 19.1	0 - 20.7	0 - 24.7	0 - 14.1	0 - 6.7
Median (IQR)	67.8 (37.3-120.4)	66.8 (37.6-130.2)	26.3 (17.4-39.9)	20.1 (14.6-32.3)	8.9 (4.7-13.4)	9.5 (4.9-13.8)	2.7 (1.6-4.6)	2.6 (1.6-4.1)	1.2 (0.4-2.6)	1.1 (0.4-2.2)	0.2 (0-1)	0.2 (0-0.6)
Post Birth												
N	153	153	153	153	108	108	120	120	149	149	145	145
Average (SD)	95.2 (86.5)	83.9 (98.2)	38.7 (42.5)	32.5 (46.5)	9.8 (7.8)	8.6 (10.1)	3.9 (5.5)	2.8 (2.8)	2 (2.6)	1.7 (2.4)	1.3 (2.7)	0.6 (1.3)
Range	11 - 520.3	10.8 - 788.5	10.7 - 368.1	10.4 - 392.6	1.4 - 49.9	1.4 - 91.2	0.9 - 44.9	0.9 - 25.4	0 - 15.8	0 - 16.7	0 - 17.8	0 - 12.4
Median (IQR)	75.6 (35.4-110.1)	52.6 (28.2-105.7)	25.9 (15-43)	19.1 (14.8-29.4)	7.9 (4.3-13.7)	6.5 (2.8-11.7)	2.2 (1.6-4)	1.7 (1.5-3)	1.2 (0.4-2.7)	0.8 (0.3-2.2)	0.3 (0.1-1.2)	0.2 (0-0.7)

Note: OAW – non-pregnant other adult women, PW – pregnant women.

Table 5.6. Percent decreases in $PM_{2.5}$, BC, and CO exposure associated with LPG intervention.

Model Type	Details	% Decrease in $PM_{2.5}$ exposure	% Decrease in BC exposure	% Decrease in CO exposure
		Estimate (CI)	Estimate (CI)	Estimate (CI)
Between Groups	—	58 (53, 63)	60 (55, 64)	73 (64, 79)
Before and After	Control	21 (10, 31)	27 (17, 35)	25 (-2, 44)
	Intervention	67 (63, 71)	70 (66, 73)	78 (68, 84)
Comparison of Changes	Overall	58 (50, 65)	59 (52, 65)	72 (55, 82)
	Visit P1	60 (50, 69)	65 (57, 72)	77 (58, 87)
	Visit P2	56 (44, 65)	58 (47, 66)	74 (51, 86)
	Visit B1	60 (48, 69)	62 (52, 70)	73 (45, 86)
	Visit B2	59 (47, 69)	62 (52, 70)	66 (34, 83)
	Visit B4	53 (39, 63)	45 (30, 57)	67 (36, 83)

Note: —, no data; BC, black carbon; CI, confidence interval; CO, carbon monoxide; LPG, liquefied petroleum gas.

Table 5.S1. Wearing compliance, defined as the fraction of time motion, was detected during daytime hours.

IRC	N	Mean	Median	SD	Min	Max
Guatemala	535	0.7	0.77	0.22	0.01	0.98
India	387	0.19	0.12	0.19	0	0.84
Peru	472	0.62	0.69	0.26	0	0.99
Rwanda	155	0.82	0.87	0.16	0.16	0.98

Table 5.S2. Overall exposure data completeness of other adult women participants in HAPIN Trial.

Visit	IRC	Enrolled	Exposure Visit Made	Exposure Data Available	% Total Enrolled with Exposure	% Total Visit Made with Exposure
BL	Guatemala	138	138	130	94.2	94.2
	India	104	104	89	85.6	85.6
	Peru	133	132	101	75.9	76.5
	Rwanda	43	43	31	72.1	72.1
P1	Guatemala	138	125	112	81.2	89.6
	India	104	96	85	81.7	88.5
	Peru	133	106	87	65.4	82.1
	Rwanda	43	33	29	67.4	87.9
P2	Guatemala	138	120	106	76.8	88.3
	India	104	81	67	64.4	82.7
	Peru	133	89	74	55.6	83.1
	Rwanda	43	37	29	67.4	78.4
B1	Guatemala	138	100	90	65.2	90.0
	India	104	84	63	60.6	75.0
	Peru	133	63	49	36.8	77.8
	Rwanda	43	26	20	46.5	76.9
B2	Guatemala	138	87	74	53.6	85.1
	India	104	84	70	67.3	83.3
	Peru	133	70	53	39.8	75.7
	Rwanda	43	31	21	48.8	67.7
B4	Guatemala	138	100	85	61.6	85.0
	India	104	91	80	76.9	87.9
	Peru	133	70	46	34.6	65.7
	Rwanda	43	22	14	32.6	63.6

Note: BL - baseline, P1 – prenatal visit 1, P2 – prenatal visit 2, B1 – post-birth visit 1, B2 – post-birth visit 2, and B4 – post-birth visit 3.

Table 5.S3. Comparison of personal exposures to $PM_{2.5}$ between pregnant and non-pregnant adult women in the same households by arm and study period.

PM2.5 Exposure	Guatemala				India				Peru				Rwanda				Overall			
	Control		Intervention		Control		Intervention		Control		Intervention		Control		Intervention		Control		Intervention	
	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM
Baseline																				
N	62	62	64	64	39	39	41	41	49	49	46	46	19	19	12	12	169	169	163	163
Average (SD)	130.4 (121.9)	110.5 (89.9)	147.9 (119.8)	135.4 (91.1)	99.4 (98.2)	99.5 (91)	112.7 (141.6)	143.9 (244.1)	90.1 (92)	55.6 (57.7)	87.7 (98.5)	92 (106.6)	114.9 (68.7)	111.1 (85.3)	86.4 (102.2)	186.8 (233.3)	109.8 (103.9)	92.1 (84.3)	117.5 (120.9)	129.1 (159.7)
Range	13.7 - 885.3	16.2 - 374.7	17.3 - 622.4	9.9 - 381.1	9.7 - 531.3	11.5 - 448	9.9 - 759.4	9.4 - 1381.9	12.5 - 427.4	11 - 310.7	12.5 - 477.7	11 - 512.6	14.7 - 297.1	33.5 - 408.8	19.8 - 388.4	23.7 - 849.9	9.7 - 885.3	11 - 448	9.9 - 759.4	9.4 - 1381.9
Median (IQR)	107.2 (71.5-135.8)	82.3 (46.6-154)	118 (76.8-173.8)	129.3 (66.8-179.9)	69.6 (42.5-119.3)	73.6 (49.9-112.4)	71.1 (40.7-117.5)	78.9 (37.6-119.9)	59.2 (28-123)	34.9 (14.9-81.8)	58.6 (23.8-93.5)	51.9 (19.4-115.5)	101.8 (56.6-160.3)	85.3 (52.4-147.3)	51 (36.1-83.4)	91.8 (50.8-239)	89.4 (43.9-129)	71.2 (35.2-114.1)	80.1 (43.2-145.2)	82 (42.4-153.1)
Pregnancy																				
N	57	57	60	60	41	41	48	48	46	46	49	49	20	20	15	15	164	164	172	172
Average (SD)	138.1 (126.5)	130.4 (120.6)	34.3 (25.9)	26 (14.9)	98.1 (79.4)	121 (99.7)	35.3 (64.2)	29.6 (15.9)	62.5 (86)	84.5 (196.7)	46.6 (60.9)	18.2 (13.7)	72.9 (44.5)	97.8 (68.3)	35.5 (18.8)	32.5 (11)	98.9 (101.5)	111.2 (137.9)	38.2 (49.6)	25.3 (15.3)
Range	11.8 - 602.3	10.7 - 670.8	9.7 - 120.2	9.6 - 62.6	13.6 - 375.7	25.5 - 506	10.1 - 464.1	10.6 - 87	12.3 - 541.2	11 - 1116.8	11.5 - 370	10.7 - 102.2	23 - 203	37.7 - 266.7	19.2 - 92.2	17.4 - 55.7	11.8 - 602.3	10.7 - 1116.8	9.7 - 464.1	9.6 - 102.2
Median (IQR)	94.7 (58.3-177.1)	90.9 (45.8-161.1)	27.1 (17.1-39.2)	21.3 (13.4-32.6)	77.2 (37.4-121.8)	86.9 (51.6-154.1)	26 (17.8-33.5)	27.1 (16.9-38.5)	39.9 (16-68.1)	26.4 (14.7-48.3)	24.6 (14.8-53.8)	14.6 (13.1-17.8)	65.4 (41.8-84.5)	60.2 (45.7-136.5)	31.6 (21.1-41.8)	31.7 (23.6-37.3)	67.8 (37.3-120.4)	66.8 (37.6-130.2)	26.3 (17.4-39.9)	20.1 (14.6-32.3)
Post Birth																				
N	59	59	49	49	47	47	48	48	32	32	49	49	15	15	7	7	153	153	153	153
Average (SD)	125.1 (102.6)	102.9 (115.5)	39.4 (33.8)	23.9 (20.7)	90.2 (70.9)	109.4 (102.2)	47 (59.3)	48.8 (55.3)	55.1 (72.5)	19.8 (10.1)	29.7 (30.7)	24 (54.4)	78.5 (40.3)	65.7 (36.8)	39.7 (17)	40.8 (15.4)	95.2 (86.5)	83.9 (98.2)	38.7 (42.5)	32.5 (46.5)
Range	12 - 520.3	20.2 - 788.5	11.1 - 171.4	11.6 - 158.1	13.7 - 292.4	13.1 - 459.1	13.4 - 368.1	13.5 - 260.8	11 - 381.4	10.8 - 52.6	10.7 - 174.4	10.4 - 392.6	21.2 - 150.9	21.2 - 152.9	19.9 - 64.7	15 - 54.8	11 - 520.3	10.8 - 788.5	10.7 - 368.1	10.4 - 392.6
Median (IQR)	91.4 (64.2-161.7)	69.7 (41.3-115.7)	27.9 (15.5-44.7)	19.8 (15.9-26.1)	67.9 (35.9-115.2)	86.1 (41-131)	27.4 (18.5-51)	28.7 (21.1-46.9)	28.5 (15.1-65.1)	14.9 (13-25.6)	15.2 (13.2-33.5)	14.1 (12.1-15.7)	72.5 (48.6-94.7)	58.7 (39-80.1)	37.7 (26.1-51.7)	46.6 (32.8-52)	75.6 (35.4-110.1)	52.6 (28.2-105.7)	25.9 (15-43)	19.1 (14.8-29.4)

Note: OAW – non-pregnant other adult women, MOM – pregnant women.

Table 5.S4. Comparison of personal exposures to BC between pregnant and non-pregnant adult women in the same households by arm and study period.

BC Exposure	Guatemala				India				Peru				Rwanda				Overall			
	Control		Intervention		Control		Intervention		Control		Intervention		Control		Intervention		Control		Intervention	
	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM
Baseline																				
N	52	52	60	60	39	39	38	38	39	39	38	38	15	15	6	6	145	145	142	142
Average (SD)	12.7 (5.6)	11.9 (7.8)	13.2 (6.8)	13.1 (9.8)	13.8 (11.6)	13.6 (11)	11.3 (8.5)	12.7 (11.3)	11.7 (15)	10 (11.7)	11.9 (11.6)	12.8 (10.9)	12.9 (4.2)	11.8 (4.5)	6 (2.5)	14.8 (11.5)	12.7 (10.4)	11.8 (9.7)	12 (8.7)	13 (10.5)
Range	1.1 - 29	2.6 - 55	4.3 - 46.8	2.6 - 77.2	1.6 - 69.2	1.6 - 54	1.6 - 33	0.6 - 48.8	1.4 - 72.3	1.6 - 66.1	1.4 - 49.3	1.5 - 38.8	6.2 - 20.1	4.8 - 20.3	2.9 - 8.9	2.9 - 31.9	1.1 - 72.3	1.6 - 66.1	1.4 - 49.3	0.6 - 77.2
Median (IQR)	11.6 (9.2-15.9)	10.9 (7.8-13.3)	12.4 (9.4-14.1)	11.5 (8.9-14.4)	11.7 (6.5-17.7)	10.5 (8.9-15.6)	7.9 (4.1-17.3)	7.5 (4.8-19.2)	5.8 (1.6-15.3)	6 (1.9-12.9)	9.2 (2.9-15.8)	11.8 (2.1-19.5)	11.1 (10.1-15.9)	11.8 (9.4-14.8)	5.8 (4.3-8.1)	12.9 (5.8-21.9)	11 (6.3-16.1)	10.5 (6.4-14.4)	10.7 (6.5-15.3)	11.3 (6.2-16.2)
Pregnancy																				
N	57	57	60	60	39	39	47	47	40	40	47	47	18	18	14	14	154	154	168	168
Average (SD)	11.5 (4.8)	11 (4.6)	4.6 (3.3)	4.3 (2.8)	11.3 (8.1)	12.2 (7.6)	3.1 (2)	3 (2.5)	8.2 (13.7)	8 (12.3)	2.2 (1.6)	1.8 (0.9)	8 (4.5)	11 (6.6)	4.2 (1.5)	4 (1.3)	10.2 (8.8)	10.5 (8.2)	3.5 (2.6)	3.2 (2.4)
Range	1 - 23.7	2.6 - 20.7	0.9 - 16.6	2.6 - 19.1	1.7 - 40.9	2.7 - 37.2	1.6 - 9.1	0.7 - 11.9	1.4 - 83.9	1.4 - 65.3	1.4 - 10.6	1.5 - 7.7	3.4 - 21.5	4.3 - 28	2.8 - 6.6	2.7 - 6.2	1 - 83.9	1.4 - 65.3	0.9 - 16.6	0.7 - 19.1
Median (IQR)	11.3 (8.6-14.1)	11.2 (7.4-14.1)	3.5 (2.2-6.1)	3.2 (2.6-4.8)	9.4 (5.2-15.5)	11.8 (6.1-14.5)	2.2 (1.7-3.9)	2.2 (1.4-3.7)	4.6 (1.5-9.7)	3.3 (1.6-8.3)	1.5 (1.5-2.1)	1.6 (1.6-1.6)	7.2 (5.1-9.6)	8.5 (6.7-14)	3.8 (2.9-5.1)	3.8 (2.9-5)	8.9 (4.7-13.4)	9.5 (4.9-13.8)	2.7 (1.6-4.6)	2.6 (1.6-4.1)
Post Birth																				
N	12	12	16	16	47	47	48	48	32	32	49	49	17	17	7	7	108	108	120	120
Average (SD)	14.5 (4.6)	11.2 (4.8)	5 (4.6)	3.4 (3)	10.9 (9.4)	12.4 (13.6)	4.2 (6.5)	3.6 (3.7)	6.6 (6.2)	2.7 (1.9)	3.2 (5.2)	1.7 (0.6)	9.2 (4.3)	7.6 (3.4)	4.2 (1.3)	4.2 (1.1)	9.8 (7.8)	8.6 (10.1)	3.9 (5.5)	2.8 (2.8)
Range	7.7 - 23	5 - 20.1	0.9 - 17.4	0.9 - 12.1	1.7 - 49.9	1.4 - 91.2	1.6 - 44.9	1.6 - 25.4	1.4 - 25.1	1.4 - 7.5	1.4 - 36.1	1.4 - 4	2.8 - 17.9	2.8 - 14.4	2.9 - 6.6	2.9 - 5.7	1.4 - 49.9	1.4 - 91.2	0.9 - 44.9	0.9 - 25.4
Median (IQR)	13.1 (12-16)	10.8 (7.4-13.2)	3.3 (1.7-6.5)	2.2 (1.5-3.9)	8.1 (4.5-14.8)	8.8 (4.8-15.4)	2.3 (1.7-3.9)	2.4 (1.7-4.4)	4.4 (1.5-8.2)	1.5 (1.5-3.5)	1.5 (1.5-2.6)	1.5 (1.5-1.5)	8.9 (5.1-12.4)	7.2 (5.6-9.4)	4 (3.2-4.8)	4.4 (3.4-4.9)	7.9 (4.3-13.7)	6.5 (2.8-11.7)	2.2 (1.6-4)	1.7 (1.5-3)

Note: OAW – non-pregnant other adult women, MOM – pregnant women.

Table 5.S5. Comparison of personal exposures to CO between pregnant and non-pregnant adult women in the same households by arm and study period.

CO Exposure	Guatemala				India				Peru				Rwanda				Overall			
	Control		Intervention		Control		Intervention		Control		Intervention		Control		Intervention		Control		Intervention	
	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM	OAW	MOM
Baseline																				
N	57	57	61	61	41	41	38	38	38	38	40	40	14	14	16	16	150	150	155	155
Average (SD)	1.9 (2)	1.6 (2.1)	1.7 (1.5)	1.6 (1.5)	1.3 (1.4)	1.5 (1.8)	2 (3.8)	2.6 (4)	4.1 (4)	2.7 (3.6)	3.8 (6.4)	4 (6.7)	1.1 (1)	0.9 (1.1)	1.4 (2.5)	2.5 (5.5)	2.2 (2.7)	1.8 (2.5)	2.3 (4)	2.6 (4.5)
Range	0.1 - 9.5	0 - 12.6	0 - 7.5	0 - 8.3	0 - 6.3	0 - 10	0 - 21.4	0 - 19.9	0 - 18.1	0 - 16.8	0 - 38.7	0 - 32.4	0.1 - 3.8	0 - 3.5	0 - 8.9	0 - 22.5	0 - 18.1	0 - 16.8	0 - 38.7	0 - 32.4
Median (IQR)	1.3 (0.6-2.4)	0.7 (0.4-1.9)	1.4 (0.5-2.6)	1.2 (0.5-2.2)	0.7 (0.3-2.1)	1.2 (0.3-2)	0.8 (0.2-2.2)	1.4 (0.2-3.3)	2.7 (1.3-6.6)	1.4 (0.6-2.5)	1.9 (1-3.3)	1.4 (0.8-3.8)	0.8 (0.4-1.6)	0.5 (0.2-1.1)	0.3 (0.1-1.3)	0.3 (0.2-2.4)	1.3 (0.5-2.8)	1 (0.4-2)	1.3 (0.3-2.7)	1.3 (0.4-2.8)
Pregnancy																				
N	59	59	64	64	45	45	49	49	35	35	47	47	23	23	13	13	162	162	173	173
Average (SD)	2.2 (2.9)	1.7 (1.6)	0.6 (0.8)	0.5 (0.8)	2.4 (3.6)	2.6 (4.2)	0.4 (1.4)	0.2 (0.3)	2.1 (2.4)	1.7 (3.1)	1.8 (2.5)	0.9 (1.1)	0.9 (1)	1.4 (1.6)	1.3 (3.3)	0.8 (1.8)	2 (2.9)	1.9 (2.9)	0.9 (1.9)	0.5 (1)
Range	0 - 20.7	0 - 8	0 - 3.6	0 - 4.5	0 - 17	0 - 24.7	0 - 9.5	0 - 1.4	0 - 10.6	0 - 17.9	0 - 14.1	0 - 5	0 - 3.3	0 - 5.9	0 - 12.3	0 - 6.7	0 - 20.7	0 - 24.7	0 - 14.1	0 - 6.7
Median (IQR)	1.5 (0.7-2.6)	1.3 (0.6-2.2)	0.3 (0.1-0.8)	0.2 (0.1-0.6)	1.1 (0.3-2.8)	1.2 (0.4-2.8)	0 (0-0.1)	0 (0-0.3)	1.3 (0.4-3.4)	1 (0.3-1.7)	0.8 (0.3-2.5)	0.4 (0.1-1.2)	0.5 (0.1-1.3)	0.8 (0.3-1.9)	0.3 (0.1-0.7)	0.1 (0.1-0.3)	1.2 (0.4-2.6)	1.1 (0.4-2.2)	0.2 (0-1)	0.2 (0-0.6)
Post Birth																				
N	56	56	47	47	47	47	50	50	29	29	36	36	17	17	12	12	149	149	145	145
Average (SD)	1.5 (1.7)	1.2 (1.2)	0.5 (0.6)	0.3 (0.6)	1.6 (1.5)	2.2 (3.3)	0.5 (0.9)	0.3 (0.4)	3.8 (4.4)	2.3 (2.6)	3.4 (4.5)	1.4 (2.2)	1.7 (1.8)	0.8 (0.8)	1.6 (1.8)	1 (1.6)	2 (2.6)	1.7 (2.4)	1.3 (2.7)	0.6 (1.3)
Range	0 - 8.6	0 - 5	0 - 2.1	0 - 3.2	0 - 5.1	0 - 16.7	0 - 4.1	0 - 2	0 - 15.8	0 - 8.2	0 - 17.8	0 - 12.4	0 - 6.8	0 - 2.5	0 - 5.4	0 - 5.9	0 - 15.8	0 - 16.7	0 - 17.8	0 - 12.4
Median (IQR)	0.9 (0.4-2)	0.8 (0.3-1.6)	0.3 (0.1-0.6)	0.1 (0-0.5)	1.2 (0.3-2.3)	1.1 (0.5-2.7)	0.2 (0-0.6)	0.1 (0-0.3)	2.7 (0.5-5.2)	0.8 (0.3-3.3)	1.4 (0.5-5)	0.7 (0.2-1.9)	1.2 (0.4-2.4)	0.6 (0.1-1.5)	0.9 (0.5-2.1)	0.4 (0.1-1)	1.2 (0.4-2.7)	0.8 (0.3-2.2)	0.3 (0.1-1.2)	0.2 (0-0.7)

Note: OAW – non-pregnant other adult women, MOM – pregnant women.

Table 5.S6. Detailed pairwise testing results between non-pregnant and pregnant adult women by IRC, study period, arm, and pollutant.

IRC	Study Period	Group1	Group2	Pollutant	Arm	dunn's p value	wilcox p value
Guatemala	Post_birth	Mother	OAW	PM2.5	Intervention	0.0129	0.0065
Peru	Baseline	Mother	OAW	PM2.5	Control	0.0165	0.0081
Peru	Post_birth	Mother	OAW	PM2.5	Control	0.0045	0.0023
Peru	Pregnancy	Mother	OAW	PM2.5	Intervention	0.0000	0.0000
Peru	Post_birth	Mother	OAW	PM2.5	Intervention	0.0305	0.0152
Peru	Baseline	Mother	OAW	CO	Control	0.0407	0.0204
Peru	Pregnancy	Mother	OAW	CO	Intervention	0.0189	0.0095
Peru	Post_birth	Mother	OAW	CO	Intervention	0.0236	0.0117
Peru	Post_birth	Mother	OAW	BC	Control	0.0036	0.0019
Peru	Pregnancy	Mother	OAW	BC	Intervention	0.0205	NA
Peru	Post_birth	Mother	OAW	BC	Intervention	0.0305	0.0152
Study-wide	Baseline	Mother	OAW	PM2.5	Control	0.0230	NA
Study-wide	Post_birth	Mother	OAW	PM2.5	Control	0.0400	NA
Study-wide	Pregnancy	Mother	OAW	PM2.5	Intervention	0.0009	NA
Study-wide	Post_birth	Mother	OAW	PM2.5	Intervention	0.0079	NA
Study-wide	Post_birth	Mother	OAW	CO	Intervention	0.0018	NA
Study-wide	Post_birth	Mother	OAW	BC	Intervention	0.0351	NA

Note:

Table 5.S7. Correlation (Spearman's ρ) of non-pregnant and pregnant adult women by pollutant, stove-type, IRC, and study period

	PM2.5		BC		CO	
Study site	**LPG**	**Traditional**	**LPG**	**Traditional**	**LPG**	**Traditional**
Baseline						
All	---	0.67	---	0.66	---	0.55
Guatemala	---	0.73	---	0.54	---	0.48
India	---	0.79	---	0.71	---	0.54
Peru	---	0.54	---	0.70	---	0.57
Rwanda	---	0.20	---	0.12	---	0.56
Pregnancy						
All	0.52	0.69	0.66	0.78	0.44	0.50
Guatemala	0.75	0.76	0.50	0.70	0.29	0.70
India	0.57	0.85	0.72	0.88	0.15	0.72
Peru	0.30	0.35	0.34	0.64	0.47	0.09
Rwanda	0.36	0.50	0.66	0.43	0.62	0.28
Post-birth						
All	0.41	0.66	0.71	0.62	0.36	0.45
Guatemala	0.36	0.68	0.84	0.22	0.21	0.47
India	0.51	0.83	0.87	0.80	0.31	0.49
Peru	0.17	-0.13	0.41	0.01	0.13	0.32
Rwanda	0.75	0.55	0.79	0.59	0.21	0.61

Table 5.S8. Correlations (Spearman's) of personal exposure to $PM_{2.5}$, BC, and CO between measurement rounds.

Group 1	Group 2	Correlation Coefficient	p-value	Correlation Coefficient	p-value
$PM_{2.5}$		*Control*		*Intervention*	
P1	BL	0.37	0	0.24	0.004
P2	BL	0.47	0	0.02	0.808
B1	BL	0.32	0.002	0.06	0.551
B2	BL	0.39	0	0.1	0.354
B4	BL	0.17	0.094	0.18	0.063
P2	P1	0.39	0	0.34	0
B1	P1	0.47	0	0.15	0.146
B2	P1	0.39	0	0.12	0.252
B4	P1	0.27	0.015	0.09	0.361
B1	P2	0.42	0	0.29	0.008
B2	P2	0.37	0.001	0.23	0.043
B4	P2	0.31	0.007	0.22	0.039
B2	B1	0.43	0	0.14	0.231
B4	B1	0.33	0.007	0.28	0.016
B4	B2	0.22	0.085	0.26	0.022
BC		*Control*		*Intervention*	
P1	BL	0.22	0.019	0.15	0.094
P2	BL	0.4	0	0.13	0.201
B1	BL	0.34	0.002	0.07	0.487
B2	BL	0.11	0.333	0.07	0.557
B4	BL	0.43	0	0.36	0
P2	P1	0.34	0.001	0.52	0
B1	P1	0.58	0	0.43	0
B2	P1	0.32	0.005	0.36	0.001
B4	P1	0.39	0.001	0.22	0.041
B1	P2	0.57	0	0.47	0
B2	P2	0.36	0.002	0.23	0.055
B4	P2	0.49	0	0.42	0
B2	B1	0.35	0.005	0.29	0.014
B4	B1	0.39	0.004	0.34	0.006
B4	B2	0.29	0.038	0.47	0
CO		*Control*		*Intervention*	
P1	BL	0.11	0.259	0.22	0.025
P2	BL	0.23	0.036	0.27	0.008
B1	BL	0.29	0.022	0.25	0.03
B2	BL	0.11	0.321	-0.05	0.661
B4	BL	0.27	0.022	-0.06	0.578
P2	P1	0.39	0	0.31	0.003
B1	P1	0.33	0.007	0.39	0.002
B2	P1	0.13	0.259	0.31	0.013
B4	P1	-0.09	0.51	0.18	0.13
B1	P2	0.42	0.001	0.23	0.085
B2	P2	0.26	0.043	0.36	0.004
B4	P2	0.39	0.002	0.35	0.004
B2	B1	0.04	0.786	0	0.997
B4	B1	0.22	0.131	0.21	0.128
B4	B2	0.38	0.005	0.09	0.523

Note: BL - baseline, P1 – prenatal visit 1, P2 – prenatal visit 2, B1 – post-birth visit 1, B2 – post-birth visit 2, and B4 – post-birth visit 3.

Table 5.S9. Correlations (Spearman's) between pollutants by stove type and IRC

	PM-BC		PM-CO		BC-CO	
IRC	**LPG**	**Traditional**	**LPG**	**Traditional**	**LPG**	**Traditional**
All	0.60	0.77	0.11	0.53	0.05	0.49
Guatemala	0.86	0.67	0.40	0.65	0.34	0.50
India	0.40	0.72	-0.03	0.61	0.16	0.54
Peru	0.36	0.80	0.06	0.56	0.13	0.62
Rwanda	0.79	0.85	-0.07	0.54	-0.23	0.54

Note: Traditional stove includes measurements from the intervention group at baseline and all measurements from the control group. LPG stove includes all post-intervention measurements from the intervention group.

Table 5.S10. Percent decreases in $PM_{2.5}$, BC, and CO exposure associated with LPG intervention in Guatemala.

Model Type	Details	% Decrease in PM2.5 exposure	% Decrease in BC exposure	% Decrease in CO exposure
		Estimate (CI)	Estimate (CI)	Estimate (CI)
Between Groups		69 (61, 75)	62 (53, 69)	80 (69, 87)
Before and After	Control	16 (0, 30)	11 (-3, 24)	21 (-17, 47)
	Intervention	76 (71, 80)	69 (62, 74)	85 (75, 91)
Comparison of Changes	Overall	72 (63, 78)	65 (55, 72)	81 (63, 90)
	Visit P1	77 (68, 83)	72 (63, 79)	83 (61, 93)
	Visit P2	70 (59, 79)	64 (51, 73)	73 (36, 89)
	Visit B1	74 (63, 82)	67 (55, 76)	86 (64, 94)
	Visit B2	68 (53, 78)	58 (40, 71)	86 (64, 95)
	Visit B4	63 (48, 74)	51 (31, 66)	75 (37, 90)

Table 5.S11. Percent decreases in $PM_{2.5}$, BC, and CO exposure associated with LPG intervention in India.

Model Type	Details	% Decrease in PM2.5 exposure	% Decrease in BC exposure	% Decrease in CO exposure
		Estimate (CI)	Estimate (CI)	Estimate (CI)
Between Groups		58 (50, 65)	66 (59, 73)	86 (76, 92)
Before and After	Control	16 (-10, 35)	31 (13, 46)	-7 (-112, 46)
	Intervention	63 (53, 71)	70 (62, 76)	89 (75, 95)
Comparison of Changes	Overall	57 (39, 70)	58 (43, 70)	90 (71, 97)
	Visit P1	56 (30, 72)	57 (36, 72)	92 (69, 98)
	Visit P2	65 (42, 78)	61 (40, 75,)	96 (83, 99)
	Visit B1	52 (20, 71)	58 (34, 73)	87 (42, 97)
	Visit B2	65 (43, 78)	67 (49, 78)	87 (44, 97)
	Visit B4	45 (13, 66)	46 (18, 64)	82 (26, 95)

Table 5.S12. Percent decreases in $PM_{2.5}$, BC, and CO exposure associated with LPG intervention in Peru.

Model Type	**Details**	**% Decrease in PM2.5 exposure**	**% Decrease in BC exposure**	**% Decrease in CO exposure**
		Estimate (CI)	**Estimate (CI)**	**Estimate (CI)**
Between Groups		37 (21, 50)	49 (37, 59)	15 (-42, 49)
Before and After	Control	33 (8, 52)	38 (11, 57)	50 (0, 75)
	Intervention	64 (53, 72)	76 (69, 81)	44 (1, 68)
Comparison of Changes	Overall	46 (17, 64)	63 (43, 75)	0 (-142, 58)
	Visit P1	40 (-2, 65)	72 (52, 83)	21 (-152, 75)
	Visit P2	31 (-23, 61)	58 (27, 76)	5 (-208, 71,)
	Visit B1	52 (6, 76)	68 (39, 83)	0 (-380, 79)
	Visit B2	51 (5, 74)	71 (45, 84)	-32 (-429, 67)
	Visit B4	62 (25, 81)	45 (-6, 71)	36 (-229, -87)

Table 5.S13. Percent decreases in $PM_{2.5}$, BC, and CO exposure associated with LPG intervention in Rwanda.

Model Type	Details	% Decrease in PM2.5 exposure	% Decrease in BC exposure	% Decrease in CO exposure
		Estimate (CI)	Estimate (CI)	Estimate (CI)
Between Groups		54 (40, 66)	51 (39, 60)	52 (3, 77)
Before and After	Control	24 (-4, 44)	33 (14, 48)	36 (-27, 68)
	Intervention	20 (-7, 41)	30 (9, 47)	35 (-124, 81)
Comparison of Changes	Overall	-6 (-67, 33)	-8 (-59, 27)	-3 (-282, 72)
	Visit P1	12 (-58, 52)	7 (-50, 42)	39 (-205, 88)
	Visit P2	-12 (-104, 38)	-30 (-109, 19)	-1 (-447, 81)
	Visit B1	-21 (-136, 38)	5 (-62, 44)	-144 (-1770, 68)
	Visit B2	-19 (-143, 42)	-11 (-95, 37)	-311 (-2610, 38)
	Visit B4	-16 (-146, 46)	-30 (-136, 28)	66 (-178, 96)

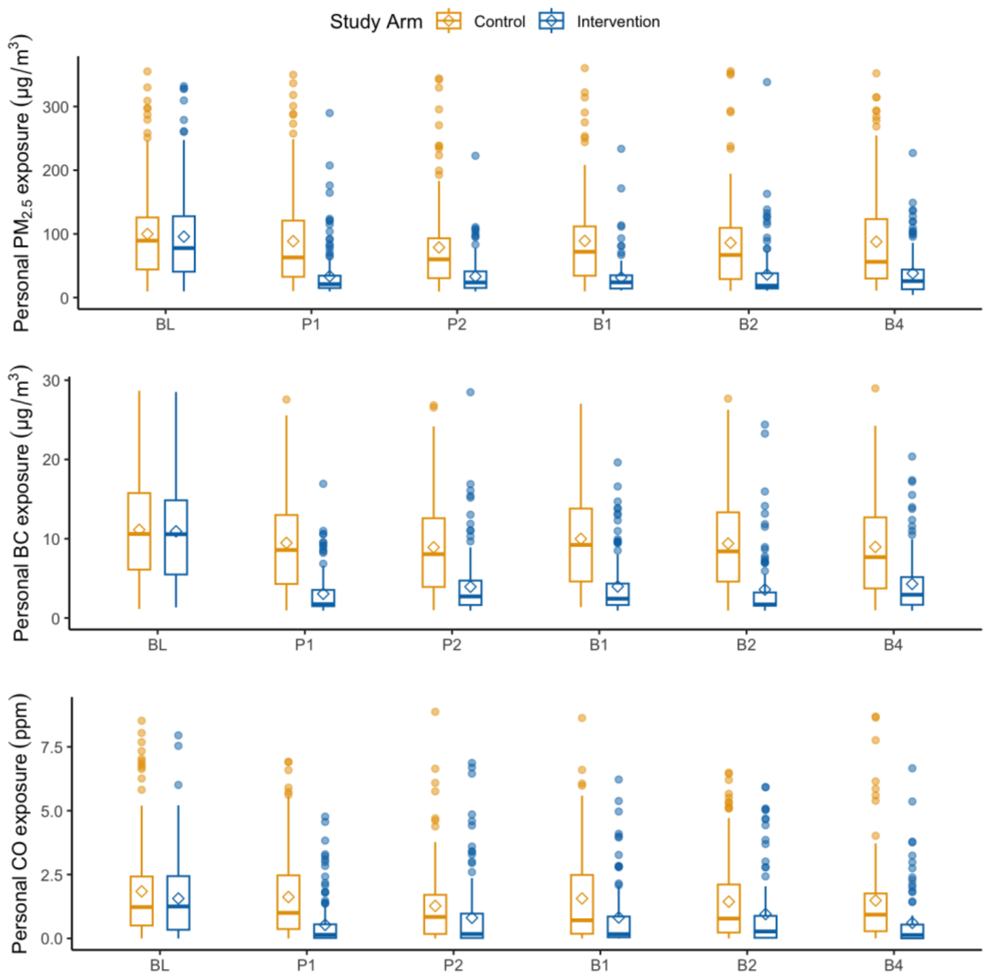


Figure 5.1. HAPIN other adult women participants personal exposure to $PM_{2.5}$, BC, and CO by study group.

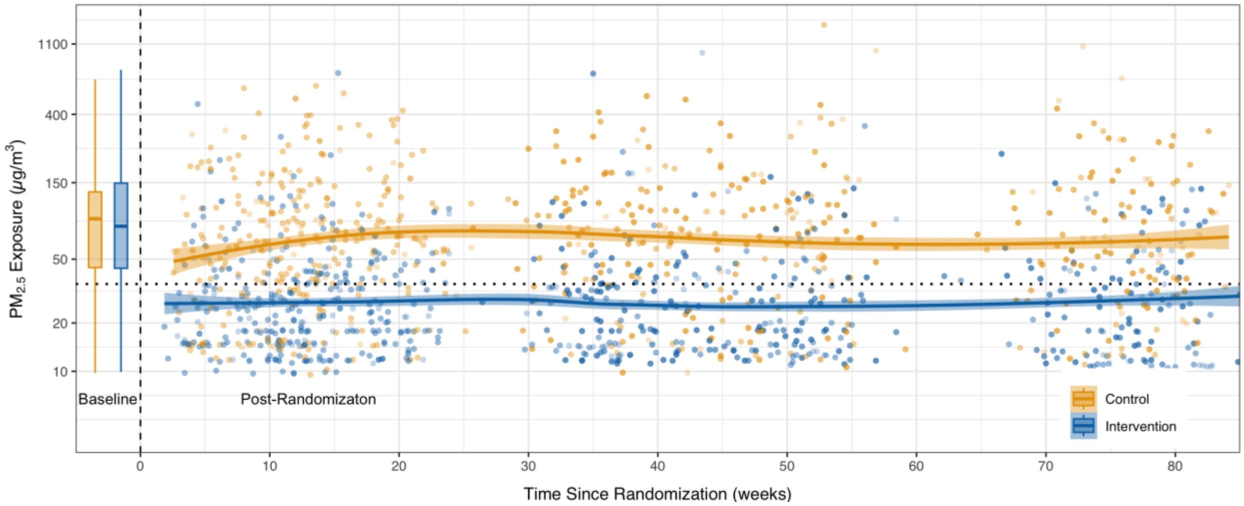


Figure 5.2. Trends in personal $PM_{2.5}$ exposure. The x-axis is the time since randomization (in weeks); time before 0 indicates the baseline period. Distribution of baseline exposures are presented as box plots. The lower and upper hinges correspond to the first and third quartiles (the 25th and 75th percentiles). The upper and lower whiskers extend 1:5 × IQR above and below the upper and lower hinges. Data beyond the whiskers are outliers. Solid lines are a locally weighted smoothing (LOWESS) function. Shaded areas are standard errors. Orange (lighter) points are individual data points from control households; Blue (darker) points are from intervention households. Note: IQR, interquartile range.

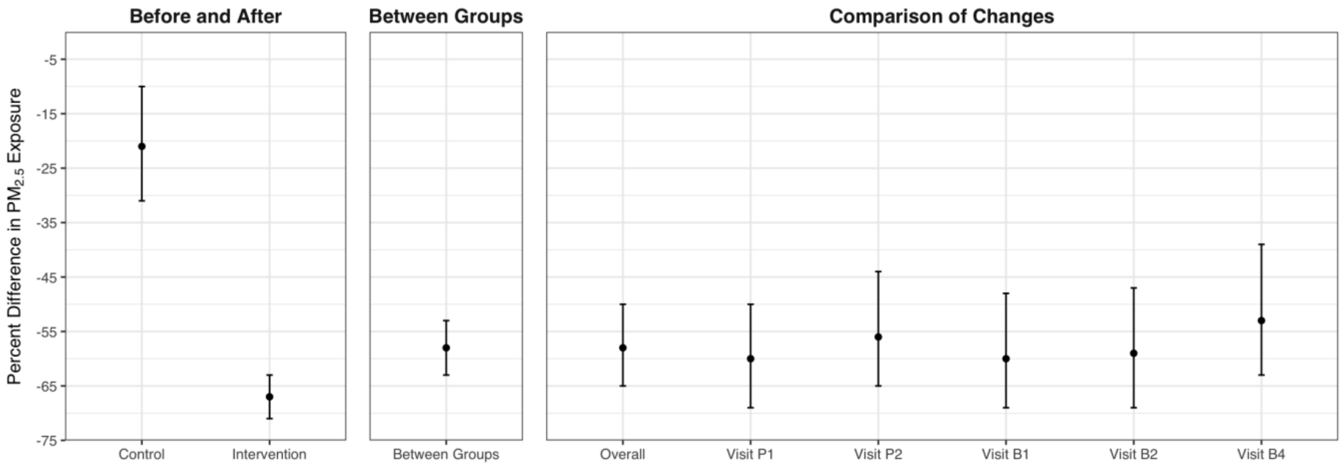


Figure 5.3. Estimated effects of the HAPIN LPG stove and fuel intervention on $PM_{2.5}$ exposure. All linear mixed -effects models used log-transformed $PM_{2.5}$ as the dependent variable. Whiskers indicate the 95% confidence intervals.

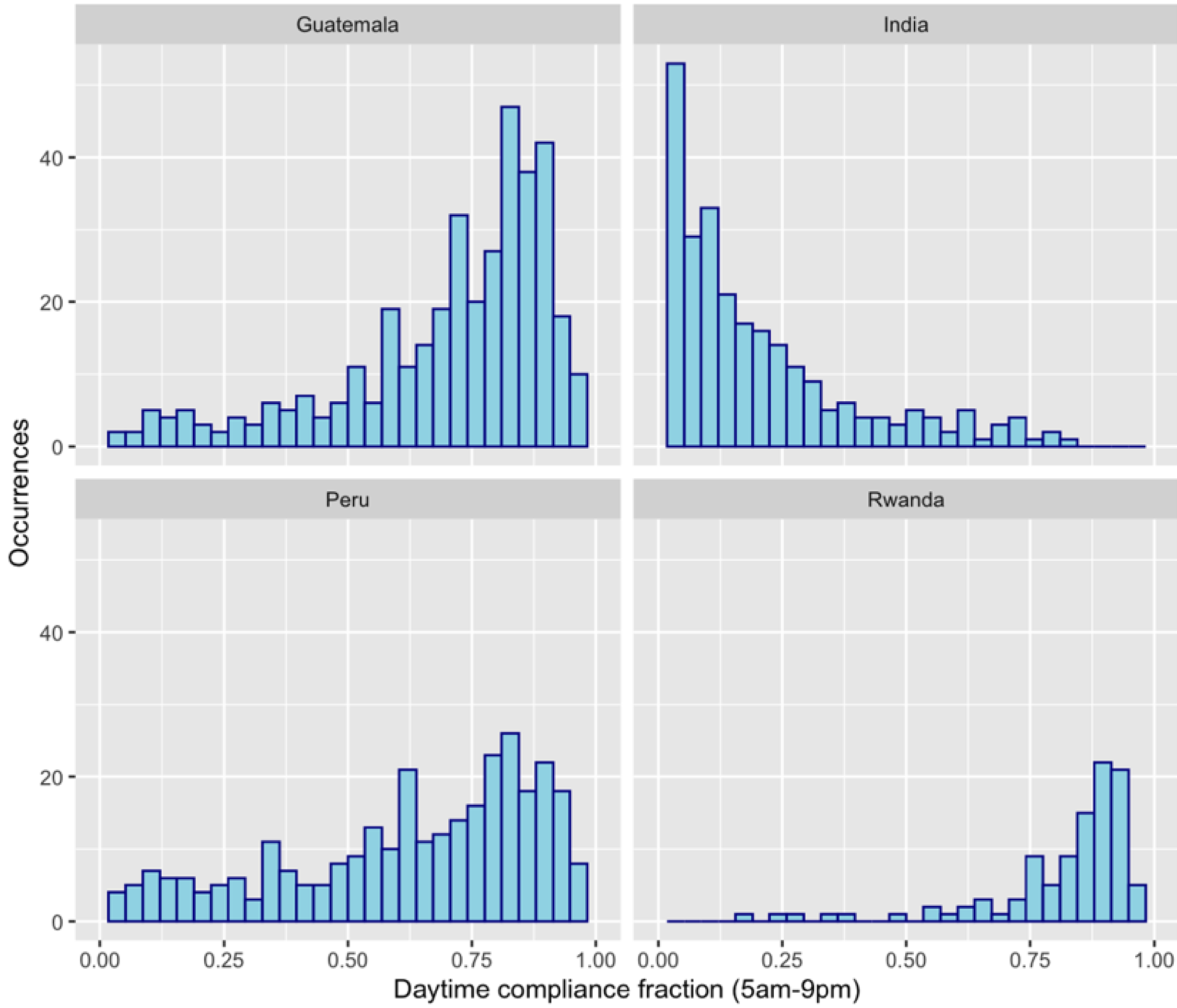


Figure 5.S1. Monitoring wearing compliance. Panels are individual IRCs. Bars are the number of measurements shown as wearing compliant for a given fraction of the day. Compliance is defined as the fraction of time motion detected during daytime hours.

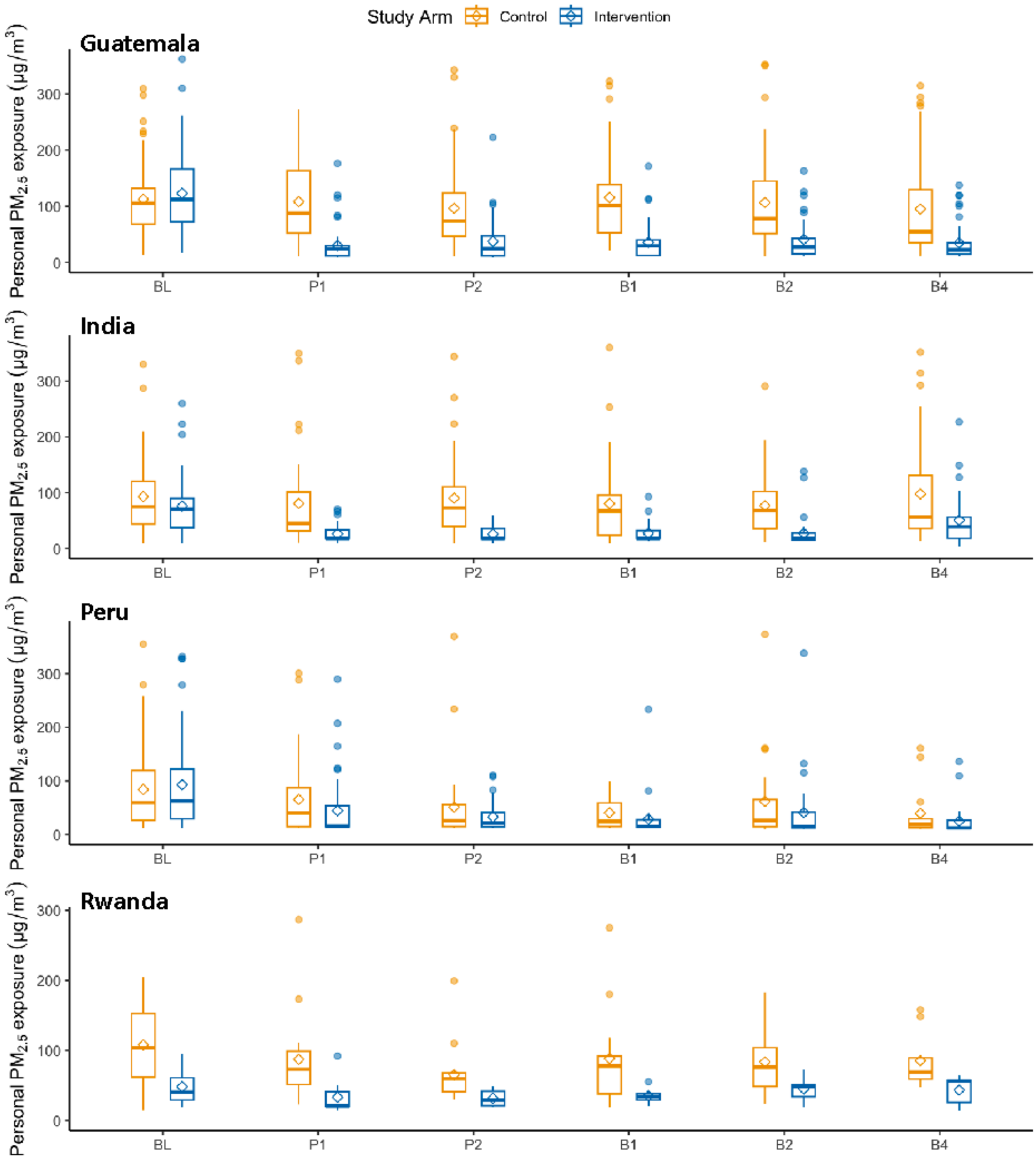


Figure 5.S2. Boxplot of personal exposure to $PM_{2.5}$ among non-pregnant adult women participants by IRC, study group and visit. The square in each box indicates the mean value. The highest 2.5% of the datapoints are not shown in the figure.

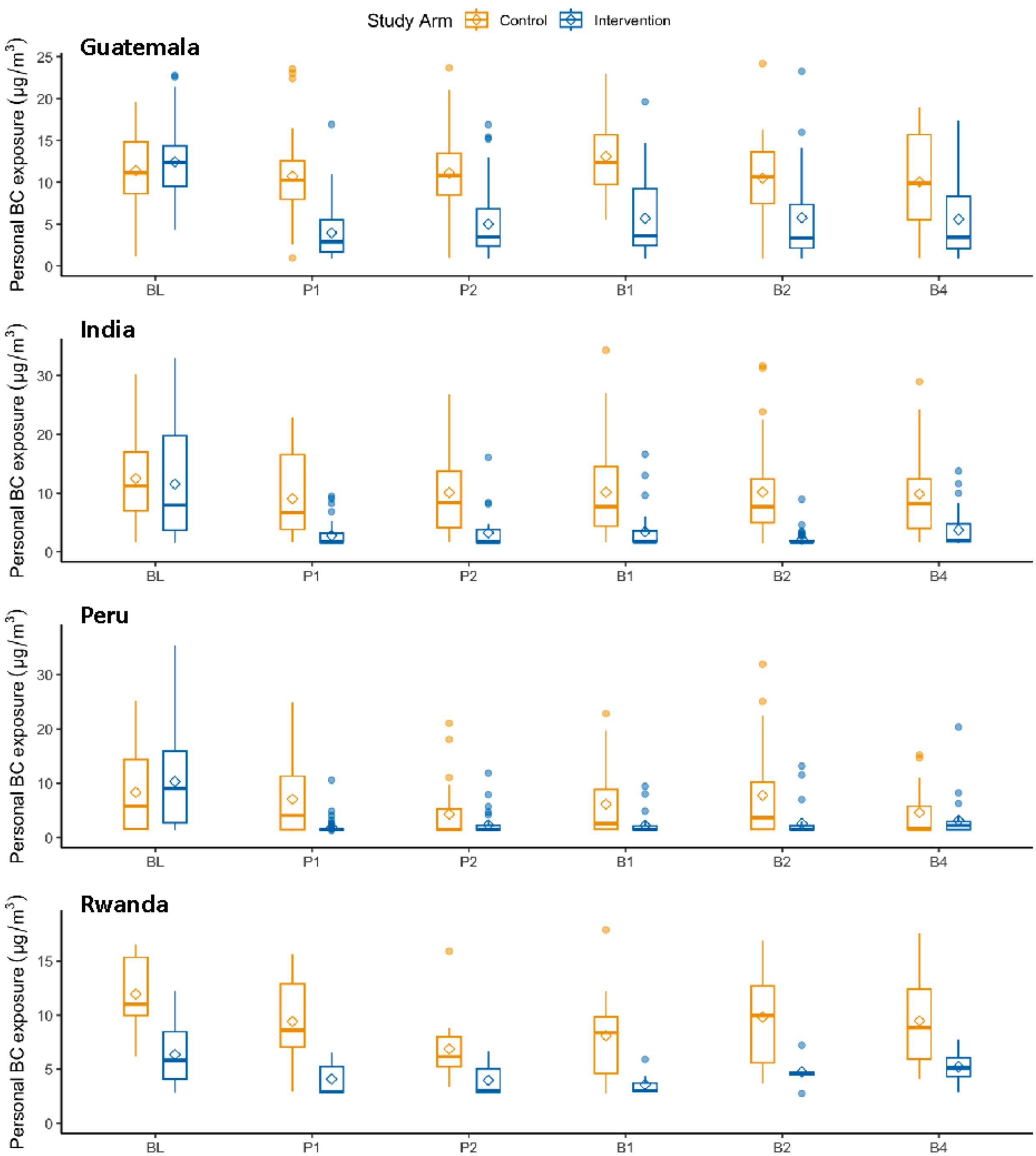


Figure 5.S3. Boxplot of personal exposure to BC among non-pregnant adult women participants by IRC, study group and visit. The square in each box indicates the mean value. The highest 2.5% of the datapoints are not shown in the figure.

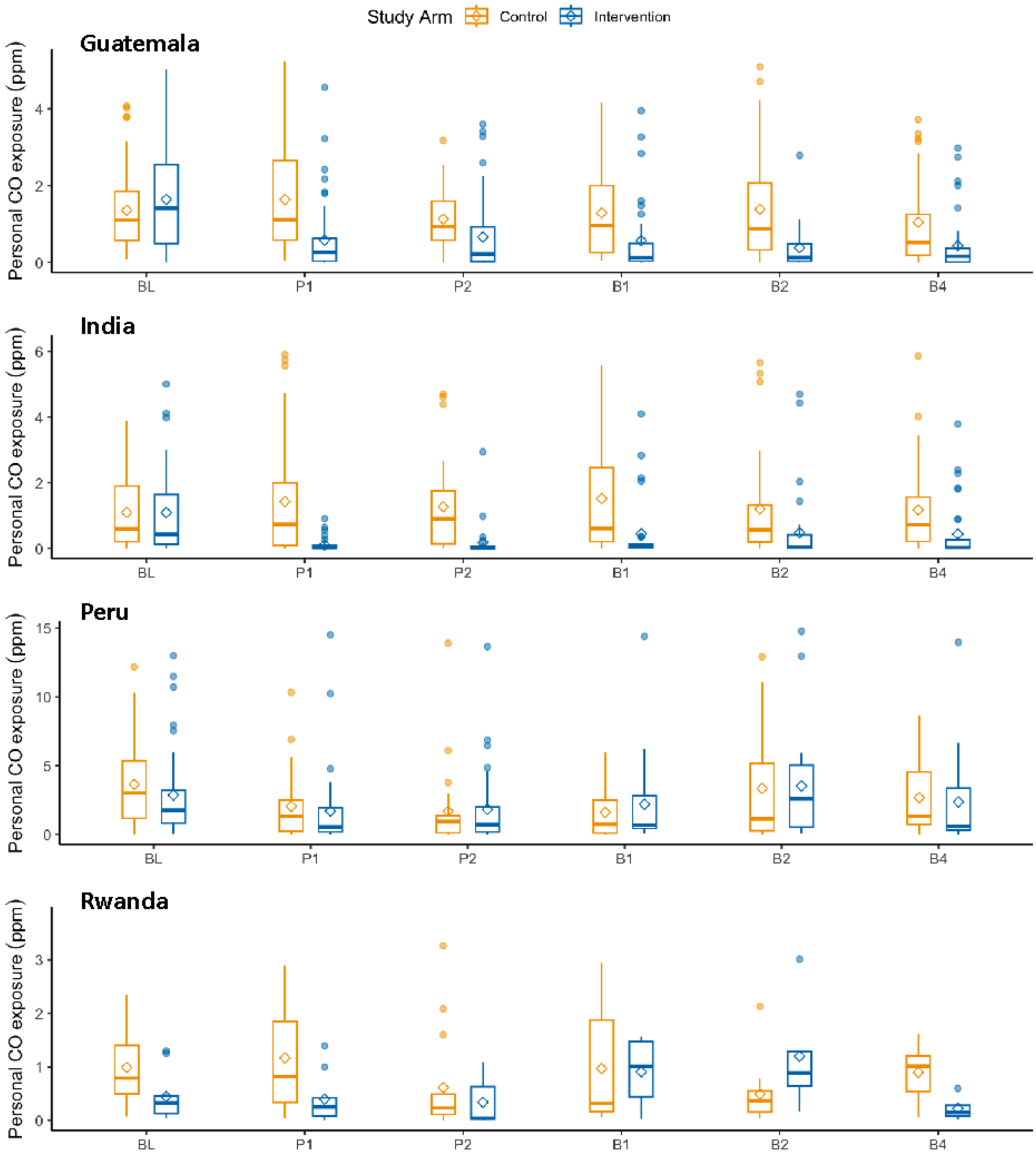
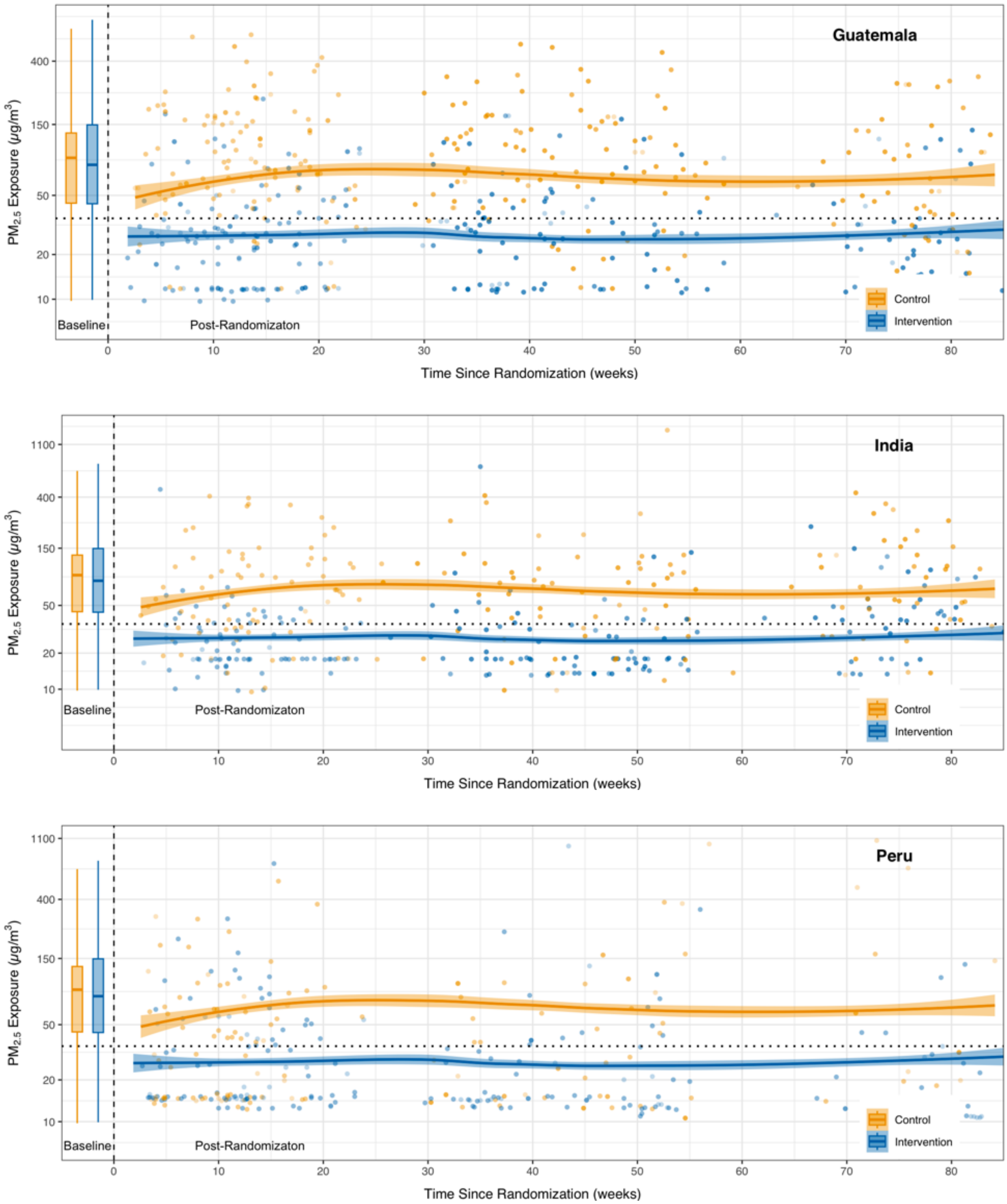


Figure 5.S4. Boxplot of personal exposure to CO among non-pregnant adult women participants by IRC, study group and visit. The square in each box indicates the mean value. The highest 2.5% of the datapoints are not shown in the figure.



Guatemala
India
Peru
PM2.5 Exposure (μg/m3)
Time Since Randomization (weeks)
Baseline
Post-Randomizaton
Control
Intervention

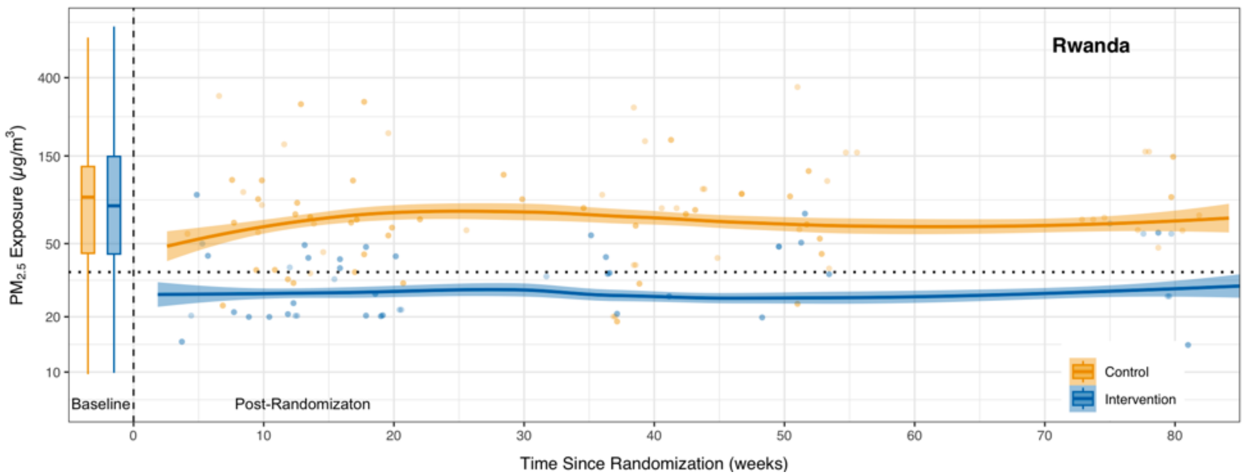


Figure 5.S5. Trends in personal $PM_{2.5}$ exposure among the HAPIN non-pregnant adult women participants in each IRC. The x-axis is the time since randomization (in weeks); time before 0 indicates the baseline period. Distribution of baseline exposures are presented as box plots. The lower and upper hinges correspond to the first and third quartiles (the 25th and 75th percentiles). The upper and lower whiskers extend 1:5 × IQR above and below the upper and lower hinges. Data beyond the whiskers are outliers. Solid lines are a locally weighted smoothing (LOWESS) function. Shaded areas are standard errors. Orange (lighter) points are individual data points from control households; Blue (darker) points are from intervention households. Note: IQR, interquartile range.

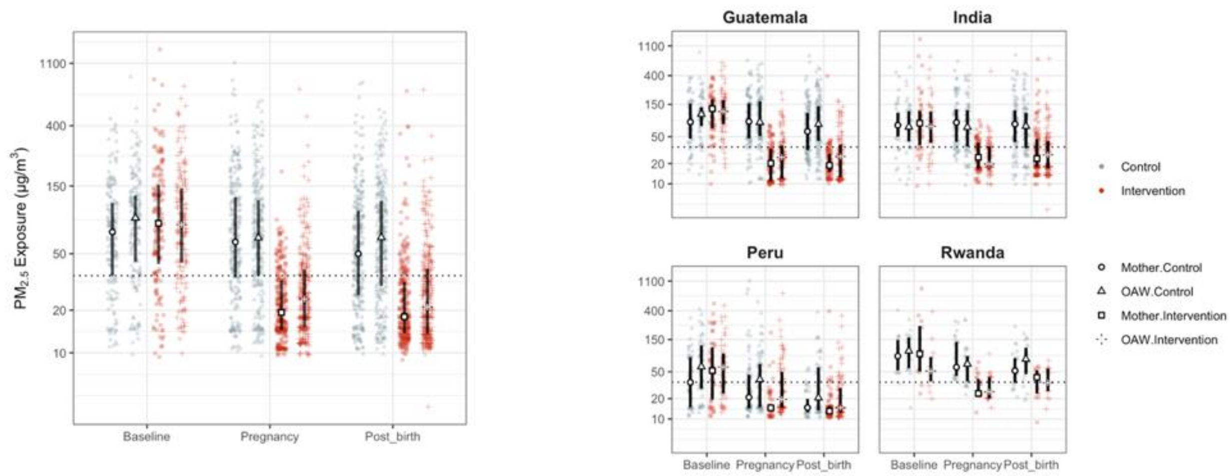


Figure 5.S6. Trial-wide and IRC-specific comparison of personal exposures to $PM_{2.5}$ between pregnant and non-pregnant adult women in the same households by arm and study period.

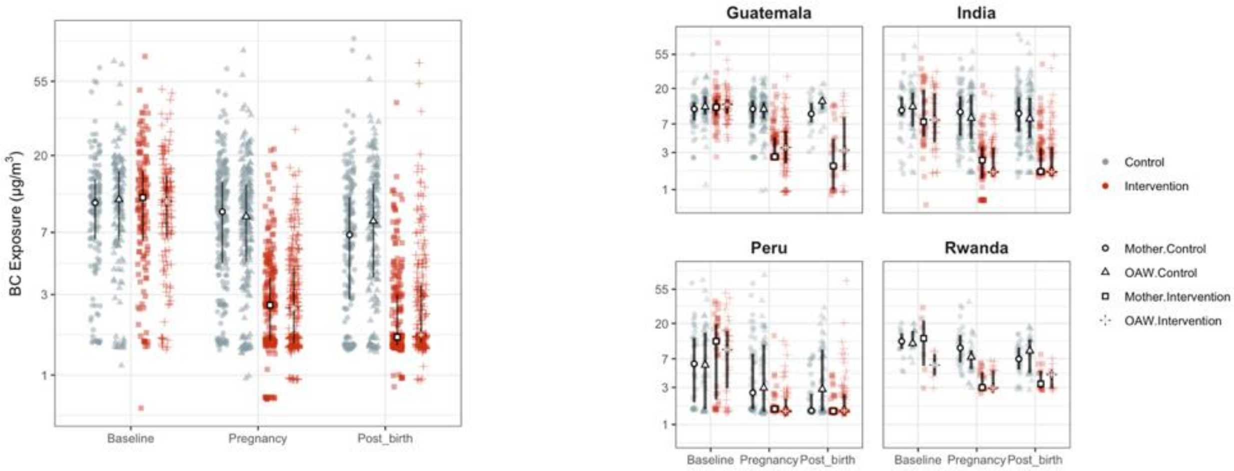


Figure 5.S7. Trial-wide and IRC-specific comparison of personal exposures to BC between pregnant and non-pregnant adult women in the same households by arm and study period.

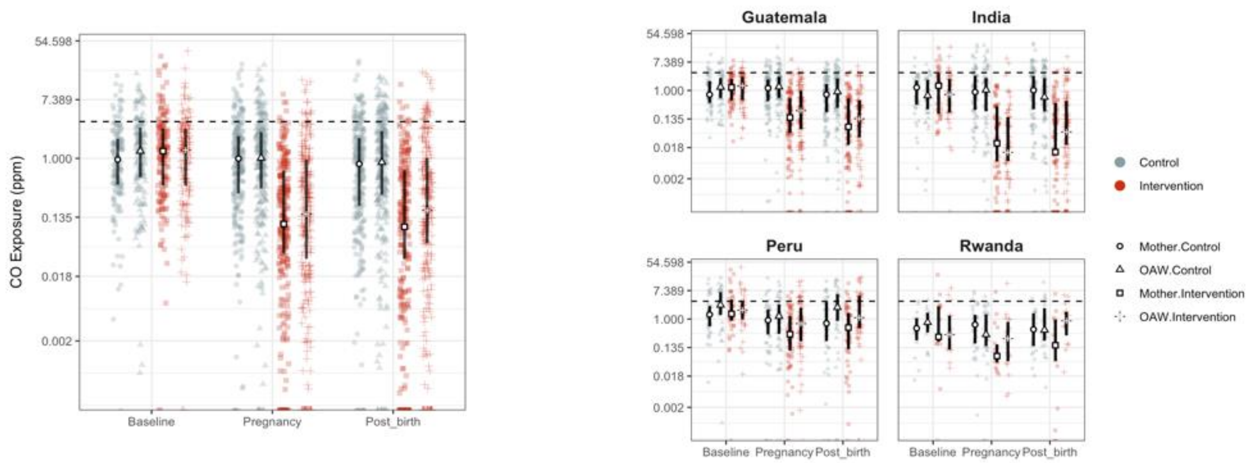


Figure 5.S8. Trial-wide and IRC-specific comparison of personal exposures to CO between pregnant and non-pregnant adult women in the same households by arm and study period.

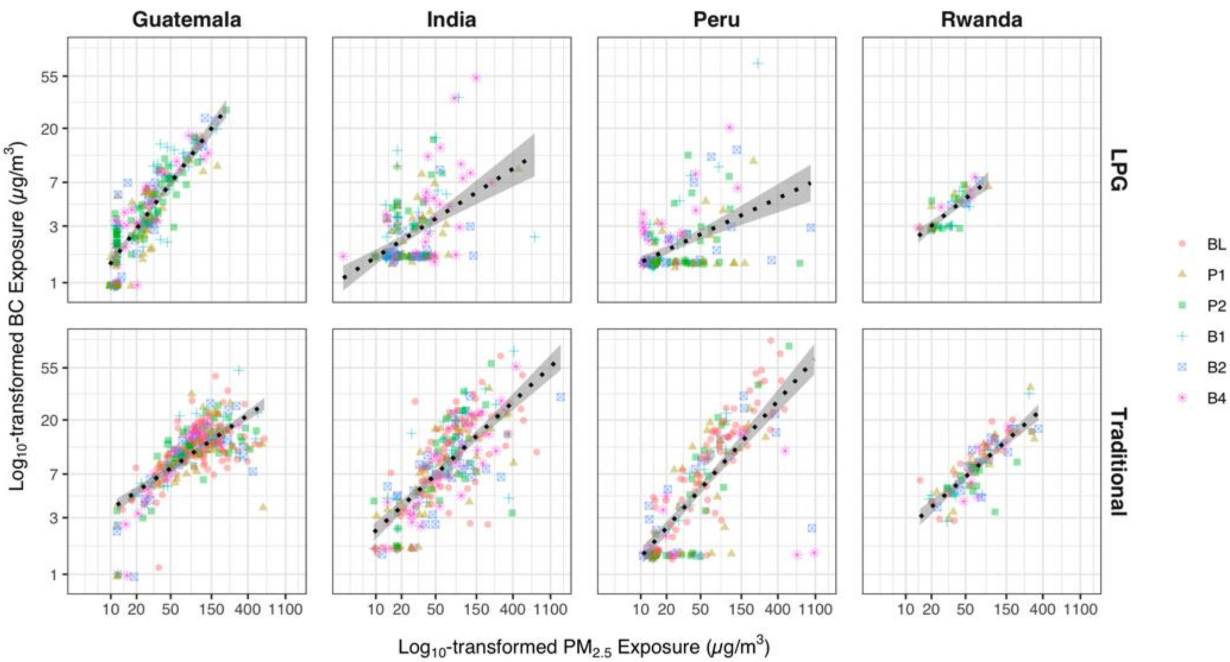


Figure 5.S9. Correlation between $\log_{10}$-transformed $PM_{2.5}$ and BC exposure by stove type and IRC

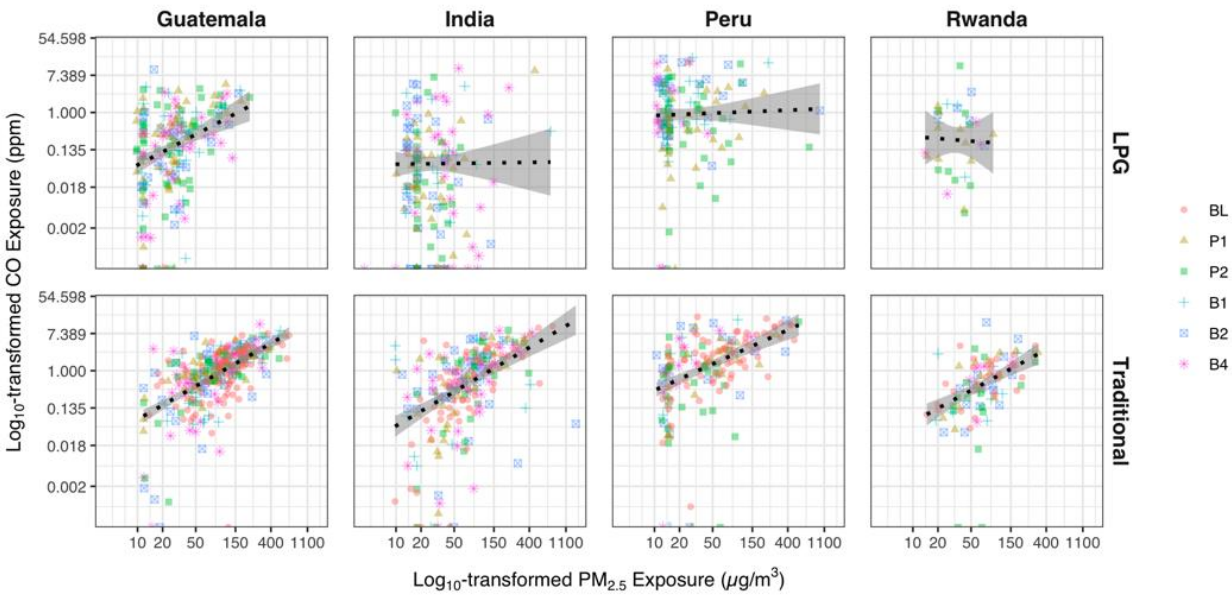


Figure 5.S10. Correlation between log_{10}-transformed $PM_{2.5}$ and CO exposure by stove type and IRC

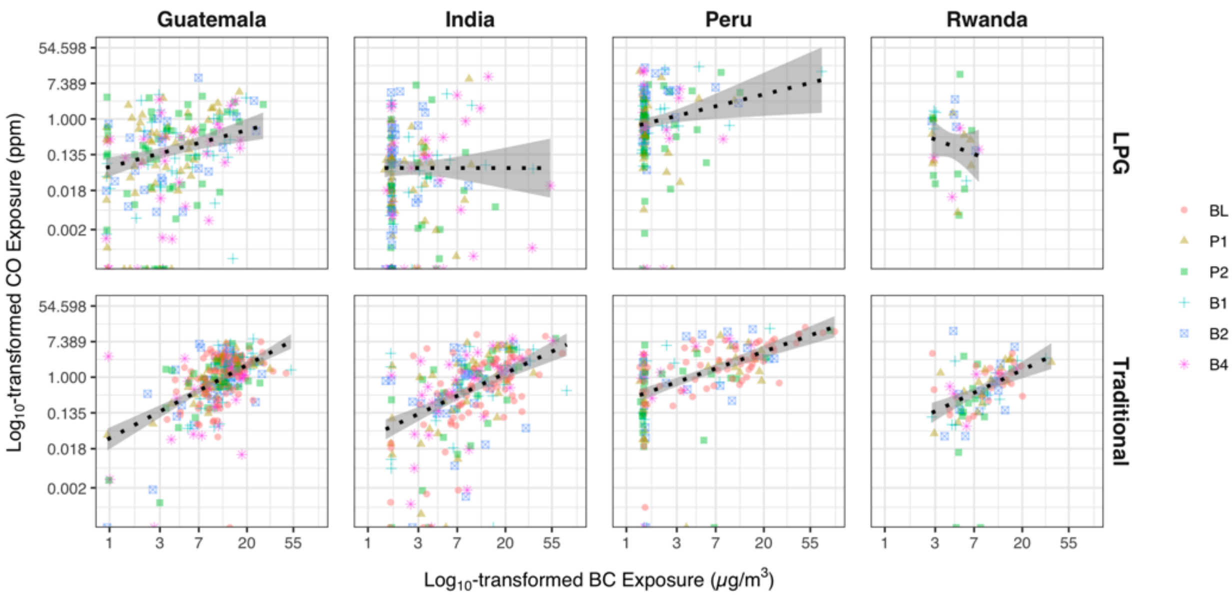


Figure 5.S11. Correlation between $\log_{10}$-transformed BC and CO exposure by stove type and IRC

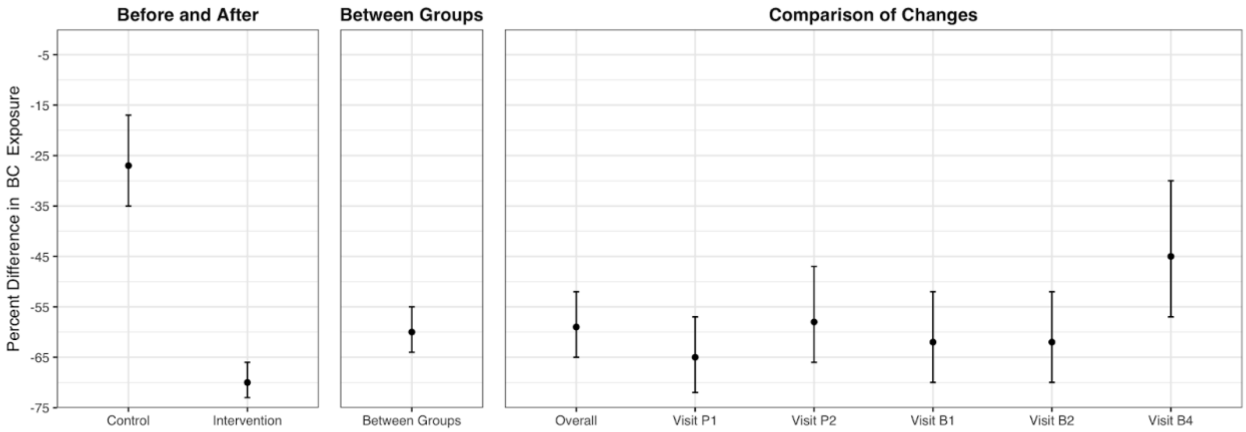


Figure 5.S12. Estimated effects of the HAPIN LPG stove and fuel intervention on BC exposure. All linear mixed -effects models used log-transformed BC as the dependent variable. Whiskers indicate the 95% confidence intervals.

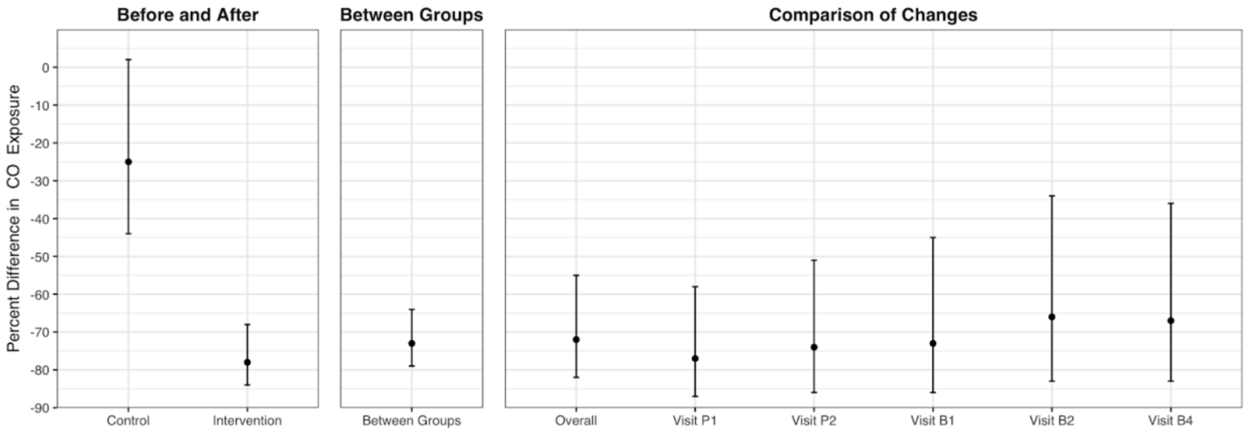


Figure 5.S13. Estimated effects of the HAPIN LPG stove and fuel intervention on CO exposure. All linear mixed -effects models used log-transformed CO as the dependent variable. Whiskers indicate the 95% confidence intervals.

CHAPTER 6

SUMMARY AND CONCLUSION

SUMMARY

Fuel switching campaigns in LMICs have resulted in reduced reliance of solid fuels for household energy needs; however, the percentage of solid fuel users in these countries remain remarkably higher than that in developed countries.[2] The resulting combustion of these polluting fuels results in HAP which is a leading risk factor in the global burden disease with disproportionate impacts among children and the elderly in rural parts of LMICs.[15] Major adverse health impacts include asthma, acute respiratory infection in adults and children, chronic obstructive pulmonary disease, lung cancer, cerebrovascular disease, ischemic heart disease, and cardiovascular mortality, low birth weight and stunting.[4–9] Cookstove intervention trials, to date, have not yet provided sufficient information for the exposure contrast needed to provide significant health benefits. To do so, we would need a much better understanding of the factors that can affect not only the magnitude of personal exposure levels, but the accurate measurement of exposure as well. Therefore, this dissertation sought to examine personal HAP exposures among pregnant and non-pregnant adult women enrolled in HAPIN, who bare a large health burden from HAP.

Summary findings from the first manuscript included in this dissertation showed that optical measures of BC, reported as eBC, correlate well overall regardless of the initial filter used as a reference for the attenuation of light through the filter after sampling. This trend, however, is not consistent across all exposure ranges reported in the study. In low exposure settings, the relationship between methods were poor, suggesting that pre-analyzing that initial measure of light intensity of the sample filter before deployment provides a more accurate estimate of eBC. This finding may have important implications for future studies that want to accurately measure the contrast in BC exposures between treatment groups, when the intervention is expected to drastically reduce exposures.

Summary findings from the second manuscript presented in this dissertation showed that there were significant variations in personal BC exposures according to stove type and use of kerosene among other important behavioral and household factors. These findings highlight the importance of participants adhering to the study stove assignment as well the mitigation of kerosene use for future cookstove interventions. These factors have the capacity to modify the contrast in HAP exposures, namely BC, between treatment arms.

Summary findings from the third manuscript suggests that implementing a clean cookstove intervention can effectively reduce personal air pollution exposure and achieve levels below the annual guidelines. We present significant and sustained reductions in multiple pollutant exposures from an 18-month LPG intervention, resulting in approximately 70% and 53% of post-intervention $PM_{2.5}$ levels in the intervention arm falling below WHO-IT1 and WHO-IT2 thresholds, respectively. Our reported reductions are among the largest of all other cookstove interventions to date.

CONCLUSION

In totality, results from this dissertation provides information that can be used for future health and exposure studies to implement more impactful and effective cookstove interventions. From the first manuscript, our data describes the enhanced measurement accuracy from conducting prescans of sample filters before deployment. In the second manuscript, we describe various factors that can influence the magnitude of an individual's exposure across four diverse rural LMIC settings. From these two manuscripts, the literature gains knowledge and understanding from multiple aspects of exposure assessments that get at the heart of estimating the true "typical" level of BC exposure. This knowledge is integral in providing the maximal exposure reductions from cookstove interventions as well as accurately characterizing the exposure contrasts associated with clean fuel alternatives. In the third manuscript, we report significant reductions in multiple pollutant exposures among non-pregnant women, but the information gleaned from the second manuscript suggests that there may be more parameters that future cookstove interventions can set to potentially reduce exposures even more.

FUTURE DIRECTIONS

As an efficacy trial, the selection of the HAPIN study sites was primarily contingent upon minimal contributions from sources other than the cookstove in the home, like vehicle emissions, that potentially contribute to ambient air pollution. As information on ambient air pollution from HAPIN sites becomes available, we will be able to compare personal exposures to ambient levels in hopes of gaining an understanding of the contributions to personal exposures from sources of air pollution outside of the participants' homes in these rural LMIC settings.

REFERENCES

1. Oluwatosin, D., Ola, K. & Olayinka, A. *The Impacts of the Use of Biomass Solid Fuels for Household Cooking in Sub-Saharan Africa - A Review*. https://www.researchsquare.com/article/rs-1840692/v1 (2022) doi:10.21203/rs.3.rs-1840692/v1.
2. Global Air. Household Air Pollution Exposure | State of Global Air. https://www.stateofglobalair.org/air/hap#billions-exposed (2020).
3. Bennitt, F. B., Wozniak, S. S., Causey, K., Burkart, K. & Brauer, M. Estimating disease burden attributable to household air pollution: new methods within the Global Burden of Disease Study. *Lancet Glob. Health* **9**, S18 (2021).
4. Lee, K. K. *et al.* Adverse health effects associated with household air pollution: a systematic review, meta-analysis, and burden estimation study. *Lancet Glob. Health* **8**, e1427–e1434 (2020).
5. Balakrishnan, K. *et al.* Exposure–response relationships for personal exposure to fine particulate matter (PM2·5), carbon monoxide, and black carbon and birthweight: Results from the multi-country Household Air Pollution Intervention Network (HAPIN) trial. 2022.08.06.22278373 Preprint at https://doi.org/10.1101/2022.08.06.22278373 (2022).
6. Bruce, N. *et al.* Does household use of biomass fuel cause lung cancer? A systematic review and evaluation of the evidence for the GBD 2010 study. *Thorax* **70**, 433–41 (2015).
7. Gordon, S. B. *et al.* Respiratory risks from household air pollution in low and middle income countries. *Lancet Respir. Med.* **2**, 823–860 (2014).

8. Simkovich, S. M. *et al.* The health and social implications of household air pollution and respiratory diseases. *Npj Prim. Care Respir. Med.* **29**, 1–17 (2019).

9. Younger, A., Alkon, A., Harknett, K., Jean Louis, R. & Thompson, L. M. Adverse birth outcomes associated with household air pollution from unclean cooking fuels in low- and middle-income countries: A systematic review. *Environ. Res.* **204**, 112274 (2022).

10. Pope, D., Bruce, N., Dherani, M., Jagoe, K. & Rehfuess, E. Real-life effectiveness of 'improved' stoves and clean fuels in reducing PM 2.5 and CO: Systematic review and meta-analysis. *Environ. Int.* **101**, 7–18 (2017).

11. Pope, D. *et al.* Are cleaner cooking solutions clean enough? A systematic review and meta-analysis of particulate and carbon monoxide concentrations and exposures. *Environ. Res. Lett.* **16**, 083002 (2021).

12. Naeher, L. P. *et al.* Woodsmoke Health Effects: A Review. *Inhal. Toxicol.* **19**, 67–106 (2007).

13. Janssen, N. A. H. *et al.* Black Carbon as an Additional Indicator of the Adverse Health Effects of Airborne Particles Compared with PM_{10} and $PM_{2.5}$. *Environ. Health Perspect.* **119**, 1691–1699 (2011).

14. Clasen, T. *et al.* Design and rationale of the HAPIN study: a multicountry randomized controlled trial to assess the effect of liquefied petroleum gas stove and continuous fuel distribution. *Environ. Health Perspect.* **128**, 047008 (2020).

15. Murray, C. J. L. *et al.* Global burden of 87 risk factors in 204 countries and territories, 1990–2019: a systematic analysis for the Global Burden of Disease Study 2019. *The Lancet* **396**, 1223–1249 (2020).

16. Briggs, N. L. & Long, C. M. Critical review of black carbon and elemental carbon source apportionment in Europe and the United States. *Atmos. Environ.* **144**, 409–427 (2016).

17. Grahame, T. J., Klemm, R. & Schlesinger, R. B. Public health and components of particulate matter: The changing assessment of black carbon. *J. Air Waste Manag. Assoc.* **64**, 620–660 (2014).

18. Janssen, N. A. H. *et al.* Health Effects of Black Carbon. *World Health Organ. Reg. Off. Eur*. (2012).

19. Clark, M. L. *et al.* Health and Household Air Pollution from Solid Fuel Use: The Need for Improved Exposure Assessment. *Environ. Health Perspect.* **121**, 1120–1128 (2013).

20. Schwartz, S. E. & Lewis, E. R. Atmospheric Chemistry and Physics Discussions. (2012).

21. Petzold, A. *et al.* Recommendations for reporting 'black carbon' measurements. *Atmospheric Chem. Phys.* **13**, 8365–8379 (2013).

22. Bond, T. C. *et al.* Bounding the role of black carbon in the climate system: A scientific assessment. *J. Geophys. Res. Atmospheres* **118**, 5380–5552 (2013).

23. Lack, D. A., Moosmüller, H., McMeeking, G. R., Chakrabarty, R. K. & Baumgardner, D. Characterizing elemental, equivalent black, and refractory black carbon aerosol particles: a review of techniques, their limitations and uncertainties. *Anal. Bioanal. Chem.* **406**, 99–122 (2014).

24. Moteki, N. & Kondo, Y. Dependence of Laser-Induced Incandescence on Physical Properties of Black Carbon Aerosols: Measurements and Theoretical Interpretation. *Aerosol Sci. Technol.* **44**, 663–675 (2010).

25. Watson, J. G., Chow, J. C. & Chen, L.-W. A. Summary of Organic and Elemental Carbon/Black Carbon Analysis Methods and Intercomparisons. *Aerosol Air Qual. Res.* **5**, 65–102 (2005).

26. Andreae, M. O. & Gelencser, A. Black carbon or brown carbon? The nature of light-absorbing carbonaceous aerosols. *Atmos Chem Phys* 18 (2006).

27. Bond, T. C. & Bergstrom, R. W. Light Absorption by Carbonaceous Particles: An Investigative Review. *Aerosol Sci. Technol.* **40**, 27–67 (2006).

28. Bond, T. C., Habib, G. & Bergstrom, R. W. Limitations in the enhancement of visible light absorption due to mixing state. *J. Geophys. Res. Atmospheres* **111**, (2006).

29. Chow, J. C., Watson, J. G., Green, M. C. & Frank, N. H. Filter Light Attenuation as a Surrogate for Elemental Carbon. *J. Air Waste Manag. Assoc.* **60**, 1365–1375 (2010).

30. Wang, Y. *et al.* Temporal Variation of Mass Absorption Efficiency of Black Carbon at Urban and Suburban Locations. *Aerosol Air Qual. Res.* **13**, 275–286 (2013).

31. Knox, A. *et al.* Mass Absorption Cross-Section of Ambient Black Carbon Aerosol in Relation to Chemical Age. *Aerosol Sci. Technol.* **43**, 522–532 (2009).

32. Liu, F. *et al.* Review of recent literature on the light absorption properties of black carbon: Refractive index, mass absorption cross section, and absorption function. *Aerosol Sci. Technol.* **54**, 33–51 (2020).

33. Presler-Jur, P., Doraiswamy, P., Hammond, O. & Rice, J. An evaluation of mass absorption cross-section for optical carbon analysis on Teflon filter media. *J. Air Waste Manag. Assoc.* **67**, 1213–1228 (2017).

34. Garland, C. *et al.* Black carbon cookstove emissions: A field assessment of 19 stove/fuel combinations. *Atmos. Environ.* **169**, 140–149 (2017).

35. Sreekanth, V., Tonne, C., Salmon, M., Arulselvan, S. & Marshall, J. D. The role of blank filter mass in attenuation measurements using an off-line transmissometer. *J. Aerosol Sci.* **131**, 41–47 (2019).

36. Ni, M. *et al.* A review on black carbon emissions, worldwide and in China. *Chemosphere* **107**, 83–93 (2014).

37. Abulude, F., Moez, B., Olubayode, S. & Adeoya, E. A review on top-down and bottom-up approach for air pollution studies. *World J. Environ. Res.* **7**, 52–60 (2018).

38. Xu, H. *et al.* Updated Global Black Carbon Emissions from 1960 to 2017: Improvements, Trends, and Drivers. *Environ. Sci. Technol.* **55**, 7869–7879 (2021).

39. Bond, T. C. *et al.* A technology-based global inventory of black and organic carbon emissions from combustion. *J. Geophys. Res. Atmospheres* **109**, (2004).

40. Evans, M. *et al.* Black carbon emissions in Russia: A critical review. *Atmos. Environ.* **163**, 9–21 (2017).

41. Paliwal, U., Sharma, M. & Burkhart, J. F. Monthly and spatially resolved black carbon emission inventory of India: uncertainty analysis. *Atmospheric Chem. Phys.* **16**, 12457–12476 (2016).

42. WHO. Health Effects of Black Carbon. (2012).

43. Kirrane, E. F. *et al.* A systematic review of cardiovascular responses associated with ambient black carbon and fine particulate matter. *Environ. Int.* **127**, 305–316 (2019).

44. Luben, T. J. *et al.* A systematic review of cardiovascular emergency department visits, hospital admissions and mortality associated with ambient black carbon. *Environ. Int.* **107**, 154–162 (2017).

45. Clougherty, J. E., Wright, R. J., Baxter, L. K. & Levy, J. I. Land use regression modeling of intra-urban residential variability in multiple traffic-related air pollutants. *Environ. Health* **7**, 17 (2008).

46. Kim, H. G.-M., Richardson, D., Loomis, D., Van Tongeren, M. & Burstyn, I. Bias in the estimation of exposure effects with individual- or group-based exposure assessment. *J. Expo. Sci. Environ. Epidemiol.* **21**, 212–221 (2011).

47. Smith, K. R. *et al.* Millions Dead: How Do We Know and What Does It Mean? Methods Used in the Comparative Risk Assessment of Household Air Pollution. *Annu. Rev. Public Health* **35**, 185–206 (2014).

48. Chen, T. *et al.* Indoor air pollution from solid fuel on children pneumonia in low- and middle-income countries: a systematic review and meta-analysis. *Environ. Sci. Pollut. Res.* (2022) doi:10.1007/s11356-021-18293-6.

49. Kinney, P. L. *et al.* Prenatal and Postnatal Household Air Pollution Exposures and Pneumonia Risk. *Chest* **160**, 1634–1644 (2021).

50. Lee, A. G. *et al.* Prenatal Household Air Pollution is Associated with Impaired Infant Lung Function with Sex-Specific Effects: Evidence from GRAPHS, a Cluster Randomized Cookstove Intervention Trial. *Am. J. Respir. Crit. Care Med.* **199**, 738–746 (2018).

51. Wang, Y. *et al.* Personal and household PM2.5 and black carbon exposure measures and respiratory symptoms in 8 low- and middle-income countries. *Environ. Res.* **212**, 113430 (2022).

52. Coker, E., Katamba, A., Kizito, S., Eskenazi, B. & Davis, J. L. Household air pollution profiles associated with persistent childhood cough in urban Uganda. *Environ. Int.* **136**, 105471 (2020).

53. Jack, D. W. *et al.* A cluster randomised trial of cookstove interventions to improve infant health in Ghana. *BMJ Glob. Health* **6**, e005599 (2021).

54. Norris, C. *et al.* A panel study of the acute effects of personal exposure to household air pollution on ambulatory blood pressure in rural Indian women. *Environ. Res.* **147**, 331–342 (2016).

55. Fandiño-Del-Rio, M. *et al.* Household air pollution and blood markers of inflammation: A cross-sectional analysis. *Indoor Air* **31**, 1509–1521 (2021).

56. Ye, W. *et al.* Effects of a Liquefied Petroleum Gas Stove Intervention on Gestational Blood Pressure: Intention-to-Treat and Exposure-Response Findings From the HAPIN Trial. *Hypertension* **79**, 1887–1898 (2022).

57. Tawiah, T. *et al.* The Association between Household Air Pollution and Blood Pressure in Obuasi Municipality, Ghana. *Atmosphere* **13**, 2033 (2022).

58. Ye, W. *et al.* Baseline associations between household air pollution exposure and blood pressure among pregnant women in the Household Air Pollution Intervention Network (HAPIN) multi-country randomized controlled trial. 2023.01.23.23284847 Preprint at https://doi.org/10.1101/2023.01.23.23284847 (2023).

59. Ye, W. *et al.* Association between personal exposure to household air pollution and gestational blood pressure among women using solid cooking fuels in rural Tamil Nadu, India. *Environ. Res.* **208**, 112756 (2022).

60. Ye, W. *et al.* Association between personal exposure to household air pollution and gestational blood pressure among women using solid cooking fuels in rural Tamil Nadu, India. *Environ. Res.* **208**, 112756 (2022).

61. Shabani, S. A mechanistic view on the neurotoxic effects of air pollution on central nervous system: risk for autism and neurodegenerative diseases. *Environ. Sci. Pollut. Res.* **28**, 6349–6373 (2021).

62. Sram, R. J., Veleminsky, M., Veleminsky, M. & Stejskalová, J. The impact of air pollution to central nervous system in children and adults. 8 (2017).

63. Bové, H. *et al.* Ambient black carbon particles reach the fetal side of human placenta. *Nat. Commun.* **10**, 3866 (2019).

64. Li, Z. *et al.* Impact of ambient PM2.5 on adverse birth outcome and potential molecular mechanism. *Ecotoxicol. Environ. Saf.* **169**, 248–254 (2019).

65. WHO. *WHO global air quality guidelines: particulate matter (PM2.5 and PM10), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide*. (World Health Organization, 2021).

66. Alexander, D. A. *et al.* Pregnancy outcomes and ethanol cook stove intervention: A randomized-controlled trial in Ibadan, Nigeria. *Environ. Int.* **111**, 152–163 (2018).

67. Chillrud, S. N. *et al.* The effect of clean cooking interventions on mother and child personal exposure to air pollution: results from the Ghana Randomized Air Pollution and Health Study (GRAPHS). *J. Expo. Sci. Environ. Epidemiol.* (2021) doi:10.1038/s41370-021-00309-5.

68. Asante, K. *et al.* The Ghana Randomized Air Pollution and Health Study (GRAPHS): a cluster randomized trial of Liquified petroleum gas (LPG) and efficient biomass cookstoves delivered during pregnancy. *Environ. Epidemiol.* **3**, 296 (2019).

69. Wylie, B. J. 31: The ghana randomized air pollution and health study (GRAPHS): A cluster-randomized trial of clean cookstoves to improve obstetric outcomes. *Am. J. Obstet. Gynecol.* **216**, S23 (2017).

70. Checkley, W. *et al.* Effects of a Household Air Pollution Intervention with Liquefied Petroleum Gas on Cardiopulmonary Outcomes in Peru. A Randomized Controlled Trial. *Am. J. Respir. Crit. Care Med.* **203**, 1386–1397 (2021).

71. Fandiño-Del-Rio, M. *et al.* Household Air Pollution Concentrations after Liquefied Petroleum Gas Interventions in Rural Peru: Findings from a One-Year Randomized Controlled Trial Followed by a One-Year Pragmatic Crossover Trial. *Environ. Health Perspect.* **130**, 057007 (2022).

72. Clasen, T. F. *et al.* Liquefied Petroleum Gas or Biomass for Cooking and Effects on Birth Weight. *N. Engl. J. Med.* **387**, 1735–1746 (2022).

73. Johnson, M. *et al.* Exposure Contrasts of Pregnant Women during the Household Air Pollution Intervention Network Randomized Controlled Trial. *Environ. Health Perspect.* **130**, 097005 (2022).

74. Lee, M. *et al.* Determinants of personal exposure to PM2.5 and black carbon in Chinese adults: A repeated-measures study in villages using solid fuel energy. *Environ. Int.* **146**, 106297 (2021).

75. Sanchez, M. *et al.* Personal exposure to particulate matter in peri-urban India: predictors and association with ambient concentration at residence. *J. Expo. Sci. Environ. Epidemiol.* **30**, 596–605 (2020).

76. Downward, G. S. *et al.* Outdoor, indoor, and personal black carbon exposure from cookstoves burning solid fuels. *Indoor Air* **26**, 784–795 (2016).

77. Curto, A. *et al.* Predictors of personal exposure to black carbon among women in southern semi-rural Mozambique. *Environ. Int.* **131**, (2019).

78. Fandiño-Del-Rio, M. *et al.* Household air pollution exposure and associations with household characteristics among biomass cookstove users in Puno, Peru. *Environ. Res.* **191**, 110028 (2020).

79. Witinok-Huber, R. *et al.* Effects of household and participant characteristics on personal exposure and kitchen concentration of fine particulate matter and black carbon in rural Honduras. *Environ. Res.* **214**, 113869 (2022).

80. Wang, Y. *et al.* Measuring and predicting personal and household Black Carbon levels from 88 communities in eight countries. *Sci. Total Environ.* 151849 (2021) doi:10.1016/j.scitotenv.2021.151849.

81. Sharma, S. *et al.* An evaluation of three methods for measuring black carbon in Alert, Canada. *Atmospheric Chem. Phys.* **17**, 15225–15243 (2017).

82. Young, B. N. *et al.* Reduced Black Carbon Concentrations following a Three-Year Stepped-Wedge Randomized Trial of the Wood-Burning Justa Cookstove in Rural Honduras. *Environ. Sci. Technol. Lett.* **9**, 538–542 (2022).

83. Barr, D. B. *et al.* Design and Rationale of the Biomarker Center of the Household Air Pollution Intervention Network (HAPIN) Trial. *Environ. Health Perspect.* **128**, 047010 (2020).

84. Clasen, T. F. *et al.* Liquefied Petroleum Gas or Biomass for Cooking and Effects on Birth Weight. *N. Engl. J. Med.* (2022) doi:10.1056/NEJMoa2206734.

85. Johnson, M. A. *et al.* Air Pollutant Exposure and Stove Use Assessment Methods for the Household Air Pollution Intervention Network (HAPIN) Trial. *Environ. Health Perspect.* **128**, 047009 (2020).

86. Williams, K. N. *et al.* Designing a comprehensive behaviour change intervention to promote and monitor exclusive use of liquefied petroleum gas stoves for the Household Air Pollution Intervention Network (HAPIN) trial. *BMJ Open* **10**, e037761 (2020).

87. Hansen, A. D. A. The Aethalometer. https://reader.elsevier.com/reader/sd/pii/S0973082608604299?token=3AF02A89A3E59663FE3303701A0C7717025FDCAEE656DBC27A30D8D0B11F6AC7CFEE0BFF4B63E03A519CE73C1765CBCE&originRegion=us-east-1&originCreation=20220628164909 doi:10.1016/S0973-0826(08)60429-9.

88. EPA. Detection Limit/Quantitation Limit Summary Table.

89. Bland, J. M. & Altman, D. G. Measuring agreement in method comparison studies. *Stat. Methods Med. Res.* **8**, 135–160 (1999).

90. Giavarina, D. Understanding Bland Altman analysis. *Biochem. Medica* **25**, 141–151 (2015).

91. Armstrong, B. G. Effect of measurement error on epidemiological studies of environmental and occupational exposures. *Occup. Environ. Med.* **55**, 651–656 (1998).

92. Baumgartner, J. *et al.* Highway proximity and black carbon from cookstoves as a risk factor for higher blood pressure in rural China. *Proc. Natl. Acad. Sci.* **111**, 13229–13234 (2014).

93. Mostofsky, E. *et al.* Modeling the Association Between Particle Constituents of Air Pollution and Health Outcomes. *Am. J. Epidemiol.* **176**, 317–326 (2012).

94. Hoffer, A., Tóth, Á., Pósfai, M., Chung, C. E. & Gelencsér, A. Brown carbon absorption in the red and near-infrared spectral region. *Atmospheric Meas. Tech.* **10**, 2353–2359 (2017).

95. Tasoglou, A., Subramanian, R. & Pandis, S. N. An inter-comparison of black-carbon-related instruments in a laboratory study of biomass burning aerosol. *Aerosol Sci. Technol.* **52**, 1320–1331 (2018).

96. Drinovec, L. *et al.* The filter-loading effect by ambient aerosols in filter absorption photometers depends on the coating of the sampled particles. *Atmospheric Meas. Tech.* **10**, 1043–1059 (2017).

97. Good, N., Anna, M., Peel, J. L. & Volckens, J. An accurate filter loading correction is essential for assessing personal exposure to black carbon using an Aethalometer. *J. Expo. Sci. Environ. Epidemiol.* **27**, 409–416 (2017).

98. Greilinger, M., Drinovec, L., Močnik, G. & Kasper-Giebl, A. Evaluation of measurements of light transmission for the determination of black carbon on filters from different station types. *Atmos. Environ.* **198**, 1–11 (2019).

99. Bruce, N. *et al.* WHO indoor air quality guidelines on household fuel combustion: Strategy implications of new evidence on interventions and exposure–risk functions. *Atmos. Environ.* **106**, 451–457 (2015).

100. Gordon, S. B. *et al.* Respiratory risks from household air pollution in low and middle income countries. *Lancet Respir Med* **2**, 823–60 (2014).

101. Quansah, R. *et al.* Effectiveness of interventions to reduce household air pollution and/or improve health in homes using solid fuel in low-and-middle income countries: A systematic review and meta-analysis. *Environ. Int.* **103**, 73–90 (2017).

102. Thakur, M. *et al.* Impact of improved cookstoves on women’s and child health in low and middle income countries: a systematic review and meta-analysis. *Thorax* **73**, 1026–1040 (2018).

103. Naeher, L. P. *et al.* Woodsmoke Health Effects: A Review. *Inhal. Toxicol.* **19**, 67–106 (2007).

104. Secrest, M. H., Schauer, J. J., Carter, E. M. & Baumgartner, J. Particulate matter chemical component concentrations and sources in settings of household solid fuel use. *Indoor Air* **27**, 1052–1066 (2017).

105. Ritchie, H. & Roser, M. Urbanization. *Our World Data* (2018).

106. US EPA, O. Integrated Science Assessment (ISA) for Particulate Matter. https://www.epa.gov/isa/integrated-science-assessment-isa-particulate-matter (2019).

107. Aung, T. W. *et al.* Effect on blood pressure and eye health symptoms in a climate-financed randomized cookstove intervention study in rural India. *Environ. Res.* **166**, 658–667 (2018).

108. Checkley, W. *et al.* Effects of a household air pollution intervention with liquefied petroleum gas on cardiopulmonary outcomes in Peru. A randomized controlled trial. *Am. J. Respir. Crit. Care Med.* **203**, 1386–1397 (2021).

109. Katz, J. *et al.* Impact of Improved Biomass and Liquid Petroleum Gas Stoves on Birth Outcomes in Rural Nepal: Results of 2 Randomized Trials. *Glob. Health Sci. Pract.* **8**, 372–382 (2020).

110. Young, B. N. *et al.* Reduced Black Carbon Concentrations following a Three-Year Stepped-Wedge Randomized Trial of the Wood-Burning *Justa* Cookstove in Rural Honduras. *Environ. Sci. Technol. Lett.* acs.estlett.2c00098 (2022) doi:10.1021/acs.estlett.2c00098.

111. Buuren, S. van & Groothuis-Oudshoorn, K. mice: Multivariate Imputation by Chained Equations in R. *J. Stat. Softw.* **45**, 1–67 (2011).

112. Brand, J. *Development, implementation and evaluation of multiple imputation strategies for the statistical analysis of incomplete data sets*. (1999).

113. Johnson, M. *et al.* Modeling approaches and performance for estimating personal exposure to household air pollution: A case study in Kenya. *Indoor Air* (2021).

114. Dionisio, K. L. *et al.* The exposure of infants and children to carbon monoxide from biomass fuels in The Gambia: a measurement and modeling study. *J. Expo. Sci. Environ. Epidemiol.* **22**, 173–181 (2012).

115. McCracken, J. P. *et al.* Combining individual-and group-level exposure information: child carbon monoxide in the Guatemala woodstove randomized control trial. *Epidemiology* 127–136 (2009).

116. Keller, J. P. & Clark, M. L. Estimating long-term average household air pollution concentrations from repeated short-term measurements in the presence of seasonal trends and crossover. *Environ. Epidemiol.* **6**, e188 (2022).

117. MacCarty, N. *et al.* Stratification of particulate matter in a kitchen: A comparison of empirical to predicted concentrations and implications for cookstove emissions targets. *Energy Sustain. Dev.* **54**, 14–24 (2020).

118. Quinn, A. K. *et al.* Fidelity and Adherence to a Liquefied Petroleum Gas Stove and Fuel Intervention during Gestation: The Multi-Country Household Air Pollution Intervention Network (HAPIN) Randomized Controlled Trial. *Int. J. Environ. Res. Public. Health* **18**, 12592 (2021).

119. Lam, N. L. *et al.* Household Light Makes Global Heat: High Black Carbon Emissions From Kerosene Wick Lamps. *Environ. Sci. Technol.* **46**, 13531–13538 (2012).

120. Crousillat, E., Hamilton, R. & Antmann, P. Addressing the electricity access gap. (2010).

121. Wallach, E. S. *et al.* Effect of a solar lighting intervention on fuel-based lighting use and exposure to household air pollution in rural Uganda: A randomized controlled trial. *Indoor Air* **32**, e12986 (2022).

122. von Dadelszen, P. & Magee, L. A. Preventing deaths due to the hypertensive disorders of pregnancy. *Best Pract. Res. Clin. Obstet. Gynaecol.* **36**, 83–102 (2016).

123. Wang, W. *et al.* Epidemiological trends of maternal hypertensive disorders of pregnancy at the global, regional, and national levels: a population-based study. *BMC Pregnancy Childbirth* **21**, 364 (2021).

124. Giorgini, P. *et al.* Air Pollution Exposure and Blood Pressure: An Updated Review of the Literature. *Curr. Pharm. Des.* **22**, 28–51 (2016).

125. Bongaerts, E. *et al.* Maternal exposure to ambient black carbon particles and their presence in maternal and fetal circulation and organs: an analysis of two independent population-based observational studies. *Lancet Planet. Health* **6**, e804–e811 (2022).

126. Shupler, M. *et al.* Household and personal air pollution exposure measurements from 120 communities in eight countries: results from the PURE-AIR study. *Lancet Planet. Health* **4**, e451–e462 (2020).

127. Stoner, O. *et al.* Household cooking fuel estimates at global and country level for 1990 to 2030. *Nat. Commun.* **12**, 5793 (2021).

128. GBD Compare. *Institute for Health Metrics and Evaluation* http://vizhub.healthdata.org/gbd-compare.

129. Dakua, M., Karmakar, R. & Barman, P. Exposure to indoor air pollution and the cognitive functioning of elderly rural women: a cross-sectional study using LASI data, India. *BMC Public Health* **22**, 2272 (2022).

130. Li, L. *et al.* Indoor air pollution from solid fuels and hypertension: A systematic review and meta-analysis. *Environ. Pollut.* **259**, 113914 (2020).

131. Katz, J. *et al.* Impact of Improved Biomass and Liquid Petroleum Gas Stoves on Birth Outcomes in Rural Nepal: Results of 2 Randomized Trials. *Glob. Health* **8**, 11 (2020).

132. Northcross, A. L., Hwang, N., Balakrishnan, K. & Mehta, S. Assessing Exposures to Household Air Pollution in Public Health Research and Program Evaluation. *EcoHealth* **12**, 57–67 (2015).

133. Smith, K. R. Fuel Combustion, Air Pollution Exposure, and Health: The Situation in Developing Countries. *Annu. Rev. Energy Environ.* **18**, 529–566 (1993).

134. Liao, J. *et al.* LPG stove and fuel intervention among pregnant women reduce fine particle air pollution exposures in three countries: Pilot results from the HAPIN trial. *Environ. Pollut.* **291**, 118198 (2021).

135. Sambandam, S. *et al.* Exposure contrasts associated with a liquefied petroleum gas (LPG) intervention at potential field sites for the multi-country household air pollution intervention network (HAPIN) trial in India: results from pilot phase activities in rural Tamil Nadu. *BMC Public Health* **20**, 1799 (2020).

136. Chartier*, R., Newsome, R. & Rodes, C. Development and Evaluation of an Enhanced Children's Micropem (ECM) to Support Indoor Air Pollution Studies. *ISEE Conf. Abstr.* (2014) doi:10.1289/isee.2014.P3-802.

137. Fandino-Del-Rio, M. *et al.* Effects of a liquefied petroleum gas stove intervention on pollutant exposure and adult cardiopulmonary outcomes (CHAP): study protocol for a randomized controlled trial. *Trials* **18**, 518 (2017).

138. Balakrishnan, K. *et al.* Exposures to fine particulate matter (PM2.5) and birthweight in a rural-urban, mother-child cohort in Tamil Nadu, India. *Environ. Res.* **161**, 524–531 (2018).

139. Delapena, S. *et al.* Using personal exposure measurements of particulate matter to estimate health impacts associated with cooking in peri-urban Accra, Ghana. *Energy Sustain. Dev.* **45**, 190–197 (2018).

140. Hornung, R. W. & Reed, L. D. Estimation of Average Concentration in the Presence of Nondetectable Values. *Appl. Occup. Environ. Hyg.* **5**, 46–51 (1990).

141. Nethery, E., Brauer, M. & Janssen, P. Time–activity patterns of pregnant women and changes during the course of pregnancy. *J. Expo. Sci. Environ. Epidemiol.* **19**, 317–324 (2009).

142. WHO. Indoor air pollution from solid fuels and risk of low birth weight and stillbirth. (2005).

143. Balakrishnan, K. *et al.* Household Air Pollution Exposures of Pregnant Women Receiving Advanced Combustion Cookstoves in India: Implications for Intervention. *Ann. Glob. Health* **81**, 375–385 (2015).

144. Grajeda, L. M. *et al.* Effectiveness of Gas and Chimney Biomass Stoves for Reducing Household Air Pollution Pregnancy Exposure in Guatemala: Sociodemographic Effect Modifiers. *Int. J. Environ. Res. Public. Health* **17**, (2020).

145. Kirby MA, N. C. Effects of a large-scale distribution of water filters and natural draft rocket-style cookstoves on diarrhea and acute respiratory infection: A cluster-randomized controlled trial in Western Province, Rwanda. *PLoS Med* **16**, (2019).

146. Pillarisetti, A. *et al.* Repeated assessment of PM2.5 in Guatemalan kitchens cooking with wood: Implications for measurement strategies. *Atmos. Environ.* **295**, 119533 (2023).

147. Simkovich, S. M. *et al.* Resources and Geographical Access to Care for Severe Pediatric Pneumonia in Four Resource-Limited Settings. *Am. J. Respir. Crit. Care Med.* rccm.202104-1013OC (2021) doi:10.1164/rccm.202104-1013OC.