

Article

Air Quality Modeling of Cooking Stove Emissions and Exposure Assessment in Rural Areas

Yucheng He ¹, Sanika Ravindra Nishandar ¹, Rufus David Edwards ^{2,*} and Marko Princevac ¹

¹ Department of Mechanical Engineering, Marlan and Rosemary Bourns College of Engineering, University of California Riverside, Riverside, CA 92521, USA; yucheng.he@email.ucr.edu (Y.H.); sanika.nishandar@email.ucr.edu (S.R.N.)

² Department of Epidemiology and Biostatistics, Program in Public Health, University of California Irvine, Irvine, CA 92697, USA

* Correspondence: edwardsr@hs.uci.edu

Abstract: Cooking stoves produce significant emissions of PM_{2.5} in homes, causing major health impacts in rural communities. The installation of chimneys in cooking stoves has been documented to substantially reduce indoor emissions compared to those of traditional open fires. Majority of the emissions pass through chimneys to the outdoors, while some fraction of the emissions leak directly into the indoor air, which is defined as fugitive emission. Indoor PM_{2.5} concentrations are then the result of such fugitive emissions and the infiltration of outdoor neighborhood pollutants. This study uses a combination of the one-contaminant box model and dispersion models to estimate the indoor PM_{2.5} household concentration. The results show that the contributions of outdoor infiltration to indoor PM_{2.5} concentrations increase with higher packing densities and ventilation rates. For a case study, under WHO recommended ventilation conditions, the 24 h average mass concentration is ~21 µg/m³, with fugitive concentration accounting for ~90% of the total exposure for highly packed communities. These results help to identify the potential benefits of intervention strategies in regions that use chimney stoves.

Keywords: dispersion model; health risk assessment; particulate matter; indoor air quality; cook stove; biomass burning



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Citation: He, Y.; Nishandar, S.R.; Edwards, R.D.; Princevac, M. Air Quality Modeling of Cooking Stove Emissions and Exposure Assessment in Rural Areas. *Sustainability* **2023**, *15*, 5676. <https://doi.org/10.3390/su15075676>

Academic Editors: José Carlos Magalhães Pires, Álvaro Gómez-Losada and Vincenzo Torretta

Received: 13 January 2023

Revised: 11 March 2023

Accepted: 15 March 2023

Published: 24 March 2023



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1. Introduction

Globally, many rural communities rely on traditional biomass burning stoves to meet household energy demands, both indoors [1] and outdoors [2], which result in a significant burden of disease [2–4]. WHO announced that an estimated 3.2 million deaths per year were attributed to household air pollution in 2020 [5]. In addition, many studies have reported high health risk associated with traditional open-fire cooking [6,7]. A large portion of the rural population utilizes traditional stoves for household needs because of socioeconomic factors including availability of fuel, the cost of the stove, and a lack of alternative energy sources such as LPG [3]. The necessity of reducing pollutants associated with stove burning has led to the development of technologies to improve combustion efficiency. The change of behavior in the kitchen can also reduce PM_{2.5} exposure. Although there are many different stove types, the combination of a burning chamber and a flue/chimney are commonly used in many areas of the world, in which majority of the emissions are exhausted from the chimney to the outdoor environment. Indoor PM_{2.5} concentrations are the result of the fraction of the emissions that leak directly into the indoor air [8], combined with the outdoor infiltrated pollutants. The outdoor neighborhood pollutants quantified in this study are from chimney emissions of an individual home and upstream neighborhood homes [9]. Often, the stacks for household chimneys installed in these regions are short, resulting in neighborhood pollution, and substantial emissions accumulate in the vicinity of homes, which enables infiltration back indoors [10,11]. The current study focuses on the

contributions of neighborhood pollution to indoor air pollution associated with different housing packing densities, which have not previously been well characterized.

Epidemiological studies have indicated that $PM_{2.5}$ exposure can cause adverse health impacts through heart, respiratory, and other chronic diseases [12]. Biomass fuel burning is recognized as a major cause of chronic obstructive pulmonary disease, especially for individuals in developing countries [13]. Estimating the risk of exposure that can lead to health problems is vital to inform risk abatement strategies. The current analysis evaluates the outdoor neighborhood pollution distributions, outdoor to indoor infiltration, indoor stove fugitive contamination, and the associated $PM_{2.5}$ exposure risk [14]. Subsequently, EPA health risk assessment [15] was applied to quantify the inhalation risk of $PM_{2.5}$ to the female population of different age groups to contribute to the development of control strategies for air quality management in and around rural communities where solid fuels are used for cooking.

2. Materials and Methods

2.1. Background

The disability-adjusted life years (DALYs) represent the sum of years of populations living in a status of less than good health resulting from specific causes. Figure 1 presents the household attributed DALYs per 100,000 people according to WHO released data for the year 2019 [16]. The DALYs are substantially higher in developing countries, where biomass combustion supplies the majority of household primary energy [17].

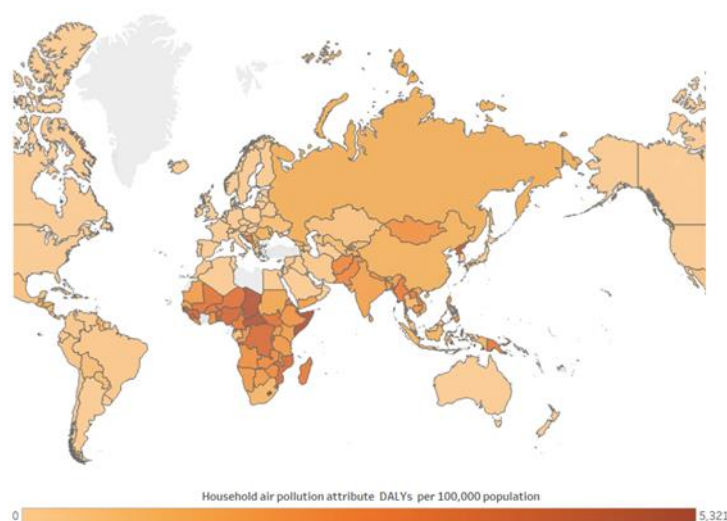


Figure 1. Household air pollution attributed DALYs per 100,000 people [16].

2.2. Evaluation Framework

The schematic of the study process is shown in Figure 2. The study is divided into the estimation of outdoor and indoor air quality. For the outdoor air quality study, meteorological parameters such as temperature, wind speed, direction, and cloud cover are required. These required micrometeorological parameters for the dispersion model inputs are derived from routine weather data [18,19]. Integrating the dispersion model with a meteorological preprocessing approximation is a good alternative when the field measurements are not available. The approximation details, together with field validation, are described in the Supplementary Material. Figure S1 shows a good agreement of the measured friction velocity with the approximation model output. A dispersion model is then deployed to quantify outdoor pollutant distribution. Such outdoor pollutants near the household are the infiltration source to the indoor environment. The indoor generated $PM_{2.5}$ is from indoor fugitive emissions. The infiltrated and the indoor-generated $PM_{2.5}$, combine to form

indoor pollution. Finally, a US EPA health risk assessment methodology [20] is utilized to quantify the potential dose and risk quotient for long-term exposure to such pollutants.

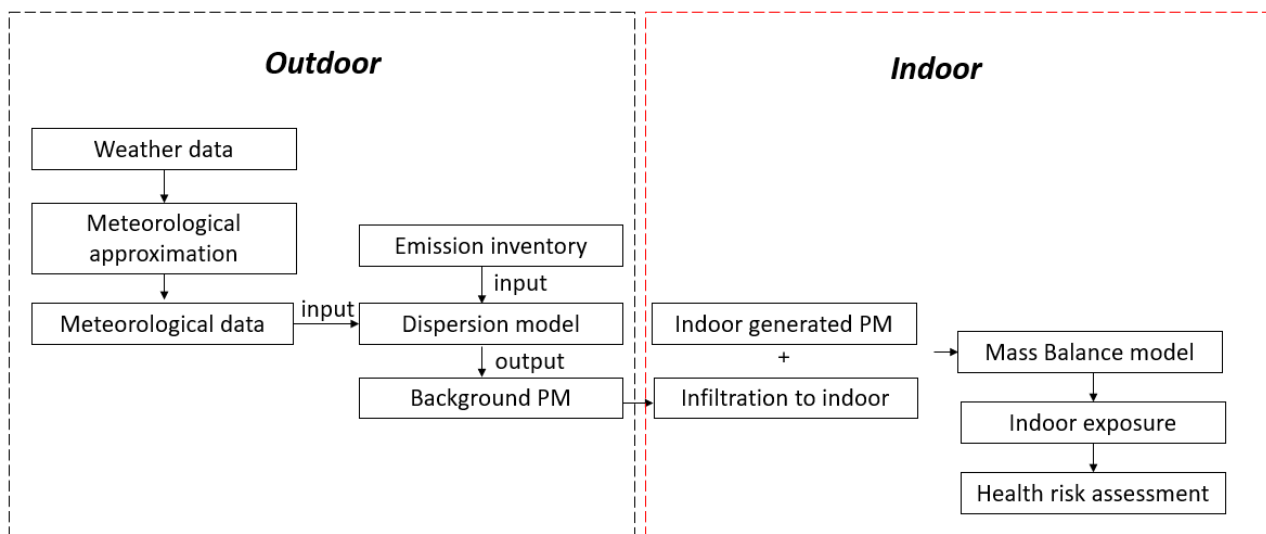


Figure 2. Analysis flow.

2.3. Dispersion Model

Dispersion models are effective, widely utilized tools to evaluate atmospheric pollution level when field measurements are not available. Such models have been utilized to quantify pollution emitted from different sources, such as cooking, traffic, and industry [17,21,22]. The ability to isolate emission sources, thus targeting the sole effect of one possible source, can inform source management and relevant policymaking. Most of the plume models do not consider complex effects of obstacles such as buildings on the dispersion of pollutants in urban or suburban areas [23]. The current study compares AERMOD with the Quick Urban and Industrial Complex (QUIC) results to examine the influence of building morphology. QUIC accommodates building influences, rapidly enables detailed modeling of the flow field around buildings, and applies this generated wind field in a particle dispersion model. The simulation results are utilized to quantify the neighborhood infiltration because it estimates pollution dispersion in the vicinity of the buildings. AERMOD is extensively used for regulatory purposes and plays a substantial role in decision making [24]. AERMOD does not explicitly solve the flow features in the vicinity of obstacles, but accounts for obstacles through a Plume Rise Model Enhancements (PRIME) model. The current analysis compares neighborhood pollution results using both approaches. The comparison of point source dispersion among Gaussian, QUIC, and water channel evaluation is presented in Figure S2. And the comparison of contours for the outdoor emission estimation of AERMOD output with QUIC output is given in Figure S3.

QUIC has broad applications, primarily in modeling wind flow and dispersion patterns in urban or suburban areas, providing building-scale results that can show pollutant concentrations and their interaction with eddies in the built environment [23,25]. QUIC is a fast-response dispersion model that is comprised of a wind field model QUIC-URB and a dispersion model QUIC-PLUME. The flow patterns modeled by QUIC-URB have been validated with USEPA wind tunnel measurements [26]. QUIC-PLUME has also been validated through many experiments and modeling comparisons. For example, Zajic et al. compared QUIC-PLUME output to a Gaussian plume model output at different atmospheric stability, with the results being in good agreement for unstable and neutral atmospheric stabilities [27].

AERMOD was developed by the U.S. Environmental Protection Agency (EPA) in conjunction with the American Meteorological Society (AMS) to incorporate scientific advances into a dispersion model for regulatory applications [28]. AERMOD is a regulatory

model with superior performance to other models in a 17 field study databases [29]. Many studies have integrated meteorological preprocessing to estimate AERMOD required inputs. Kumar et al. integrated a weather forecast model, using data-driven predicted meteorological data as inputs [30]. The current meteorological approximation has been validated for different built environments including relatively uniform terrain, as well as urban canopies [18,31].

2.4. Indoor $PM_{2.5}$

The indoor fugitive emission and infiltration of outdoor pollution are the two major sources of household pollutants (please see the schematic in Figure 3). Such external infiltration from the upstream community is determined by ventilation type, room volume, flow direction, and nature and size of openings in walls, windows, and doors. The ventilation in rural households is usually natural, wherein the leakage of airflow is through the openings in the building walls, windows, and doors. The dimensionless infiltration factor of outdoor pollution to the indoors is described by [14,32]:

$$F_{inf} = \frac{P * a}{a + k} \quad (1)$$

where a is the air changes per hour (ACH), P is the penetration coefficient that indicates the fraction of outdoor pollutant passing indoors [33], and k is the pollutant deposition rate per hour. ACH is the rate of indoor air replacement by outdoor air. It is an important parameter that determines air ventilation in microenvironments, thus affecting indoor air exposure [34]. When particles penetrate through the building envelope, gravity, diffusion, and inertial interaction are the major determinants of P . P shows a hill-shape distribution with respect to particle size, and it is assumed to be 0.8 in the study case [32]. K describes the flux of particulate matter deposited on the frames of windows, doors, walls, and other surfaces when traveling indoors and it is adapted to be 0.53 h^{-1} for particle size at $2.5 \mu\text{m}$, based on experiments in six naturally ventilated houses [32].

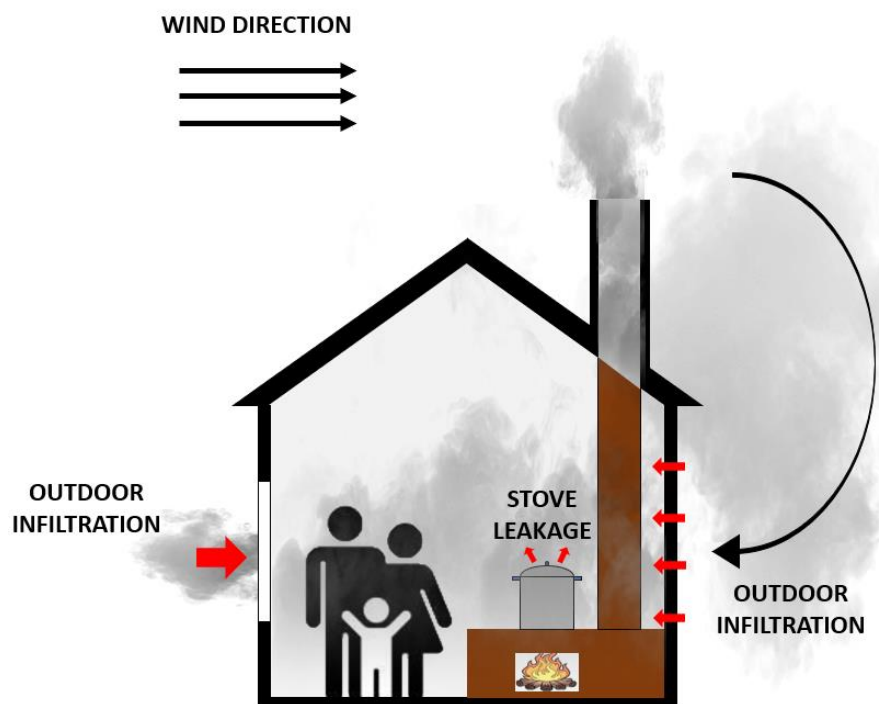


Figure 3. Household pollution sources from indoor fugitive emission and outdoor infiltration.

The ACH in a region can vary substantially based on the local wind speed, location, and the closing/opening of the door and windows. Figure 4 shows different ACH measured

in rural areas in India [7], Bangladesh [34], China [35], and Nepal [36]. When the windows are closed, the ventilation is solely dependent on the leakage through gaps. The Literature states that the ACH in such conditions is around 1–2 for houses with solid walls. WHO suggested a default ACH of 21, for kitchens using biomass for cooking, to achieve the indoor air quality standard [37]. For current analysis, indoor pollution concentration and residents' potential dose were estimated for ACH range of 1–25.

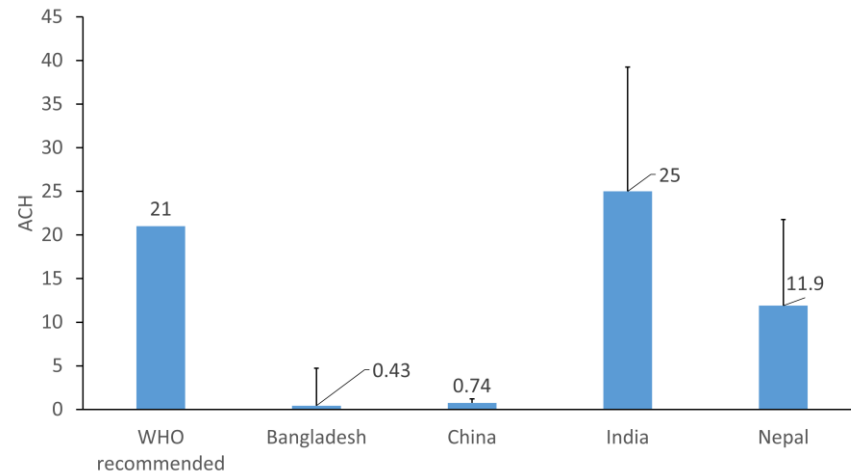


Figure 4. Reference rural air changes per hour in different regions.

Using the calculated dimensionless infiltration factor, the outdoor infiltration is then described by:

$$C_b = F_{inf} \times C_{out} \quad (2)$$

where C_{out} is the outdoor $PM_{2.5}$ concentration in $\mu g/m^3$, which is obtained from the dispersion model output.

Computational Fluid Dynamic (CFD) models are at times utilized for indoor air quality modeling [38,39]. In contrast, the current study deployed a computationally inexpensive single zone mass balance model [40] to estimate the indoor generated $PM_{2.5}$ concentration, which is described by:

$$C(t) = C_b + \frac{S}{V(a+k)} + (C_{ini} + C_b + \frac{S}{V(a+k)})e^{-(a+k)t} \quad (3)$$

where V is the kitchen volume, which is $\sim 40 m^3$ for multiple regions [41]. S is the indoor source emission rate in $\mu g/h$. In reality, the emission rate measurements vary widely, based on factors such as experimental methodology, combustion facilities, and fuel properties [8]. The indoor fugitive emission is around 2–5% of the total emission, based on the laboratory experiments, as well as field measurements [8,10,41]. Household energy needs are met mainly by biomass fuels, including crop residues, wood, coal, etc. Usage of different types of fuels impact the overall emission levels. Figure 5 presents the $PM_{2.5}$ emission factor (EF) of different fuel types, in the laboratory [42] and in field, in different regions including China [8], Nepal [42], Mexico [43], and India [42]. For the same type of fuel, the EF differences may be due to factors such as fuel shape, moisture content, and the stove differences. With the different emission factors, one can estimate the emission inventory based on the fuel consumption rate from the various cooking events [44].

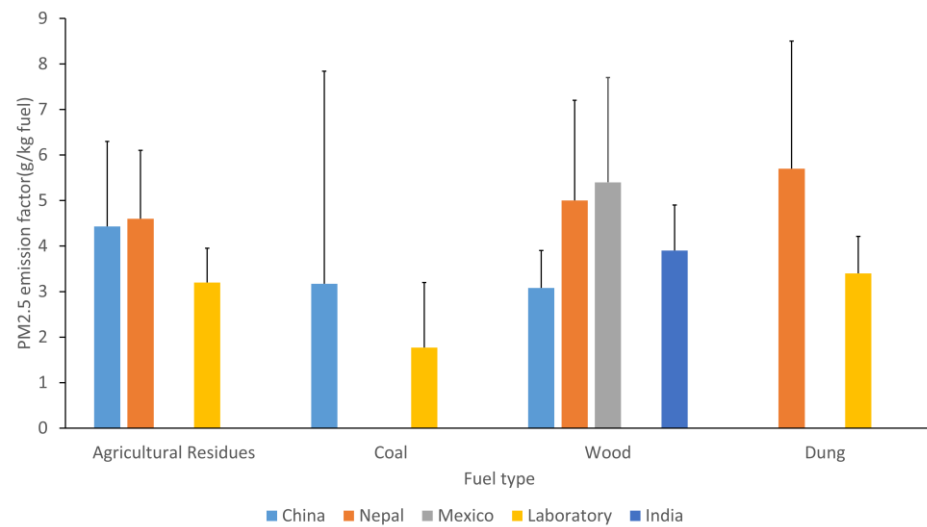


Figure 5. Reference emission factor for different fuel types.

2.5. Health Risk Assessment

The potential dose of stove-induced PM_{2.5} for individuals under long-term exposure to cooking emissions is assessed using US EPA risk assessment [20]. The potential dose (I) in $\mu\text{g}/\text{kg}\cdot\text{day}$ of PM_{2.5} can be quantified as [15,45,46]:

$$I = C_A \frac{IR \times ET \times EF \times ED}{BW} \times \frac{1}{AT} \quad (4)$$

where C_A is the concentration of PM_{2.5} in $\mu\text{g}/\text{m}^3$. ET is the exposure time in h/day, which in this study, is assumed to be 3 h exposure per day [47]. EF is the exposure frequency (days/year). ED is the exposure duration in the study period. AT is the average time of exposure in a day, which is $ED \times 365$. BW is the body weight (kg). IR is the inhalation rate (m^3/h), which represents the volume of air inhaled over a specified timeframe. The inhalation rates are typically indexed to activity levels. The inhalation rate for different age groups, segregated by gender and the average body weight, is referred from the EPA Exposure Factors Handbook [48]. Note that for the male gender, the potential dose results are very similar, within 2–7%. For brevity, here we present only results calculated using available female parameters, as in many rural locations, the primary coking activities are carried out by women. Under moderate activity levels, for age groups from 0.5–3, 3–10, and 10–18 years old, the average body weight is 11 kg, 23 kg, and 50 kg, and the average inhalation rate is $0.6 \text{ m}^3/\text{h}$, $0.9 \text{ m}^3/\text{h}$, and $1.26 \text{ m}^3/\text{h}$, respectively, while for the adult age groups from 18–30, 30–60, and above 60 years old, the average body weight is 62 kg, 68 kg, and 67 kg, and the average inhalation rate is $1.32 \text{ m}^3/\text{h}$, $1.32 \text{ m}^3/\text{h}$, and $1.2 \text{ m}^3/\text{h}$, respectively.

The risk quotient is often used to inform the health implications due to pollutant exposure. The risk quotient is described by:

$$RQ = \frac{I}{RfD} \quad (5)$$

where RfD is the reference dose of PM_{2.5} ($\mu\text{g}/\text{kg}\cdot\text{day}$) and represents the safe average daily dose. RfD is calculated from Equation (4), with a reference concentration of $5 \mu\text{g}/\text{m}^3$. If $RQ < 1$, the exposure is not considered adverse to public health; if $RQ > 1$, the exposure is considered detrimental to public health.

3. Results

3.1. Outdoor Pollution Level

A case study of neighborhood pollution attributed to chimney vented PM_{2.5} emissions is conducted using QUIC model, with a wind speed of 2 m/s (at 10 m height) and a south-southwest wind direction. The QUIC modeling results of ambient PM_{2.5} pollution during the steady cooking state is shown in Figure 6. The channeling effect caused by wind flow encountering the building obstacles will lead to accumulation of PM_{2.5} in the building wakes [23]. The QUIC model enables the detection of severely impacted regions, considering the building morphology. A typical rural village consists of regions with different building densities. In each of these building densities, the neighborhood pollution level varies because the effects of trapping PM_{2.5} near buildings differ. As indicated in Figure 6, the maximum ground level concentration occurred in the high building density region. Although the prevailing wind is from the south-southwest, the buildings in the high-density region substantially disturb the flow and consequently, distribute the emissions in different directions, while in the low-density region, the rarely disturbed flow quickly dilutes PM_{2.5}.

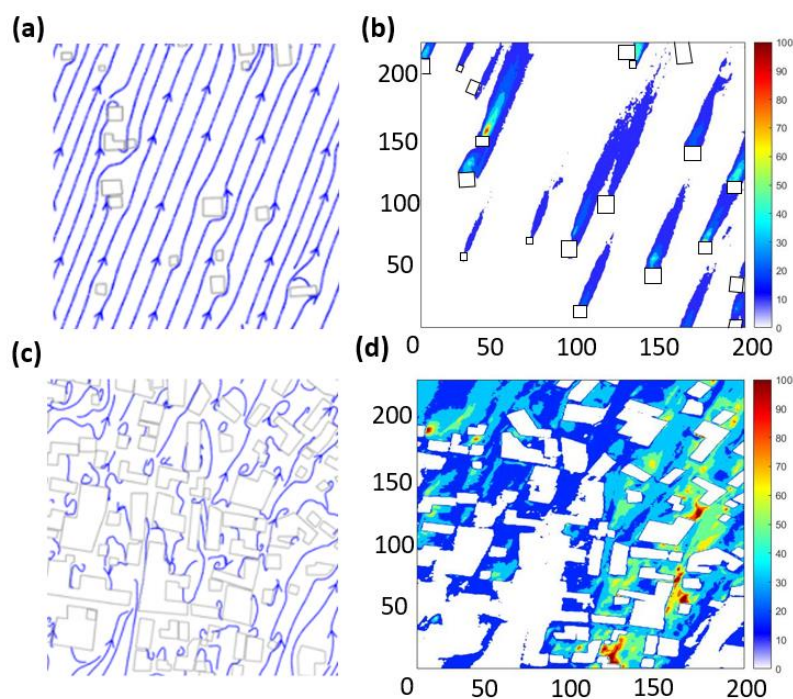


Figure 6. Outdoor dispersion modeling results: (a,c) flow trajectory for a low- and high-density region, respectively; (b,d). PM_{2.5} mass concentration distribution in a low- and high-density region, respectively.

The current analysis used an emission rate of 52 mg/min for the outdoor pollution modeling based on field measurements, with 96% of total emissions from chimneys [41]. The mean PM_{2.5} in the high building density region is $21.2 \pm 4.26 \mu\text{g}/\text{m}^3$. The mean PM_{2.5} in the low building density region is $4.57 \pm 2.8 \mu\text{g}/\text{m}^3$. The high-density area is more impacted, regardless of the wind direction, due to the building density. Hence, this region has the highest level of PM_{2.5} in the communities.

Other factors, such as seasonal relative humidity, temperature, and precipitation, also play a significant role in outdoor pollution dispersion across seasons [49]. In the Brazilian rainforest, during the dry season, exposures to PM_{2.5} can be 6 times higher than during the rainy season [15]. This lower exposure can be attributed to the leaching of air pollution to the ground as a result of higher precipitation in the rainy season [50].

3.2. Indoor Pollution Level

Instead of completely switching to clean fuel, which may be impractical, many studies have also recommended alternative actions to reduce household air pollution and exposure. Other studies have also recommended alternative actions to reduce household air pollution and exposure, such as increasing the natural ventilation rates [51]. Combining chimneys with improved combustion chambers in stoves can also result in substantially reduced overall emissions, although chimney maintenance is necessary to maintain these reductions [41]. Leakage emissions from well-maintained stoves were shown to be substantially lower than from those in need of repair, such as the filling of cracks and the cleaning of chimneys [52].

Despite the up to 90% reductions in indoor air concentrations of $PM_{2.5}$ associated with the installation of chimneys [53], fugitive concentration, combined with outdoor infiltrations, still contributes to poor indoor air quality. The indoor pollutant concentration is determined by the fugitive emission rates, room volume, particulate matter decay rate, and the infiltration of pollution from outdoors due to ventilation. Figure 7 shows the modeled indoor generated $PM_{2.5}$ mass concentrations during 1 h of cooking under different air exchange rates, incorporating contributions of both fugitive emissions and neighborhood infiltration for high-packing density. The fugitive emission rate ranges from 0.26–2.6 mg/min, based on direct field measurements in different regions [8,10,41,54]. The ACH ranges from 1–25 h^{-1} , from poor ventilated cases, to WHO default ventilation rates for ISO tiers.

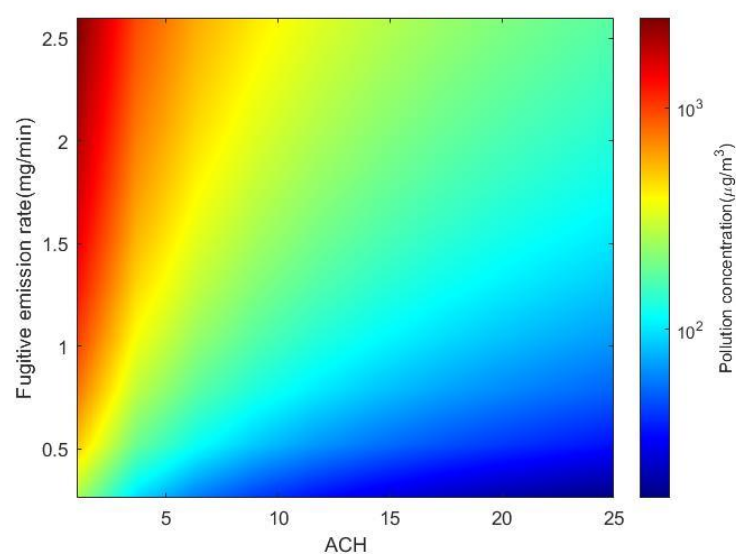


Figure 7. Indoor $PM_{2.5}$ concentrations under different ACH incorporating fugitive emission and neighborhood infiltration for a high-packing density neighborhood.

Infiltration of pollution from outdoors consists of both neighborhood pollution and regional background $PM_{2.5}$. The neighborhood pollution contribution to indoor concentrations depends on the packing density of upstream homes. Figure 8 shows the percentage of fugitive contribution to the total indoor air $PM_{2.5}$ concentrations. At 25 ACH, for homes in the high packing density area, the indoor generated $PM_{2.5}$ accounts for 90% of the total concentration, while for a low packing density region, the number is 98%.

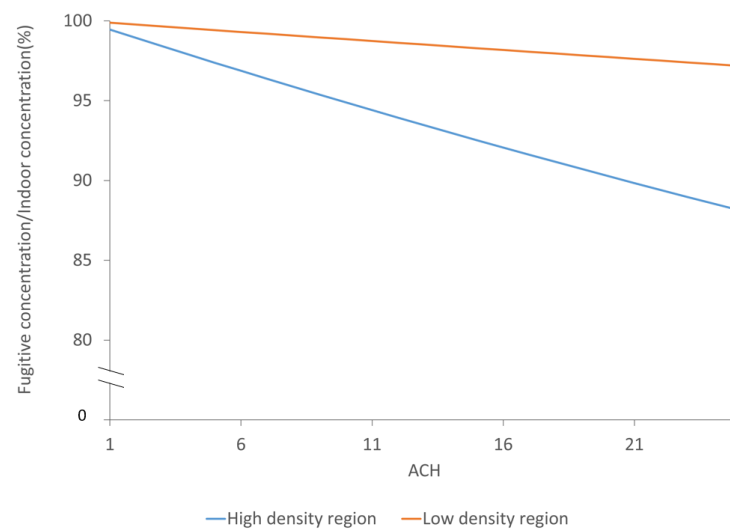


Figure 8. Contribution of fugitive emission to indoor air PM_{2.5} concentrations.

After the cooking concludes, the decay trend of the relative mass concentration ($C(t)/C_{max}$) under ACH = 1, 5, 10, 20, and 25 are shown in Figure 9. Studies indicated that the building characteristics, including ventilation, orientation, the morphology of the streets, wall construction, eave spaces, open-closed windows, etc., dominate the air exchange rate [50,55].

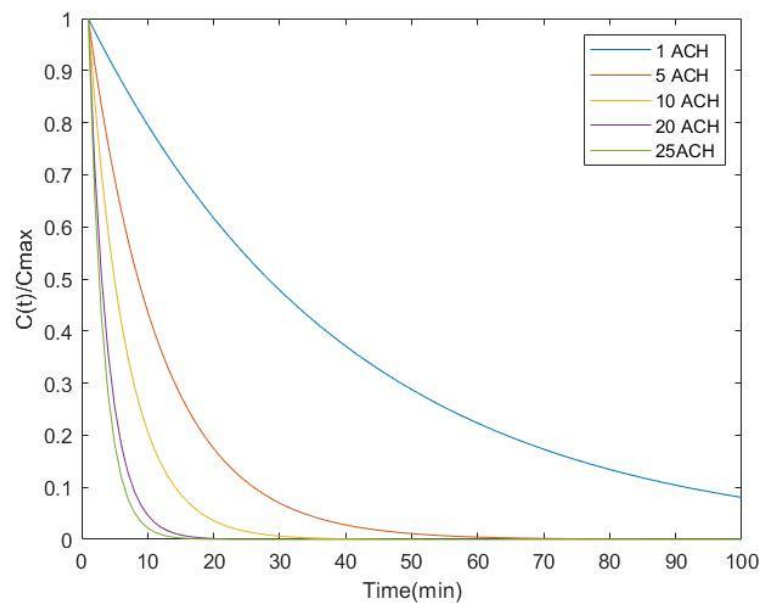


Figure 9. Influence of different ACHs on relative indoor concentration decay trend after steady cooking events.

This case study uses a fugitive emission rate of 2.1 mg/min, which is directly measured in rural Mexico using nested hoods to capture all emissions [41]. Even at the recommended 21 ACH, the indoor pollution concentration during the 1 h steady cooking event is 174 $\mu\text{g}/\text{m}^3$, with fugitive emissions contributing 90% to indoor concentrations. The corresponding 24 h average PM_{2.5} level is $\sim 21 \mu\text{g}/\text{m}^3$ under the assumption that each household conducts 3 h cooking each day [47]. This exceeds the 2021 WHO air quality guideline of 15 $\mu\text{g}/\text{m}^3$ for 24 h.

PM_{2.5} concentration levels in different rooms can be significantly lower than in other rooms [56] and can vary by season [49]. Zuk et al. found that the kitchen concentrations

were two times that of other rooms [56]. In addition, in many homes cooking-generated $PM_{2.5}$ may readily spread to the adjacent rooms in the house [49,57]. Since people spend the most time in bedrooms and living rooms, having a separate kitchen can help reduce exposures, although, room ventilation and location relative to the kitchen have been reported to impact the $PM_{2.5}$ level in the room [58]. Behavior changes, such as opening doors and windows [59] and the use of extraction fans, may also reduce indoor concentrations.

3.3. Potential Dose Assessment

The inhaled dose is determined by individual behaviors and the distance from the emission source. A number of studies have shown that personal exposures are ~50% of indoor kitchen concentrations [52,60], as personal exposures include time spent away from the kitchen in other environments. Figure 10 shows the potential dose estimated under personal exposure. The risk quotient (RQ) for exposed residents is 26, 7.93, 3.96, and 2.19 under 1, 5, 10, and 20 ACH, respectively. This shows that even under high ventilation rates, household emissions moderately contribute to total chronic exposure and may induce respiratory health problems. Besides, the potential intake of pollutants from cooking activity is high, and thus, the long-term exposure can significantly impact individuals who perform the cooking.

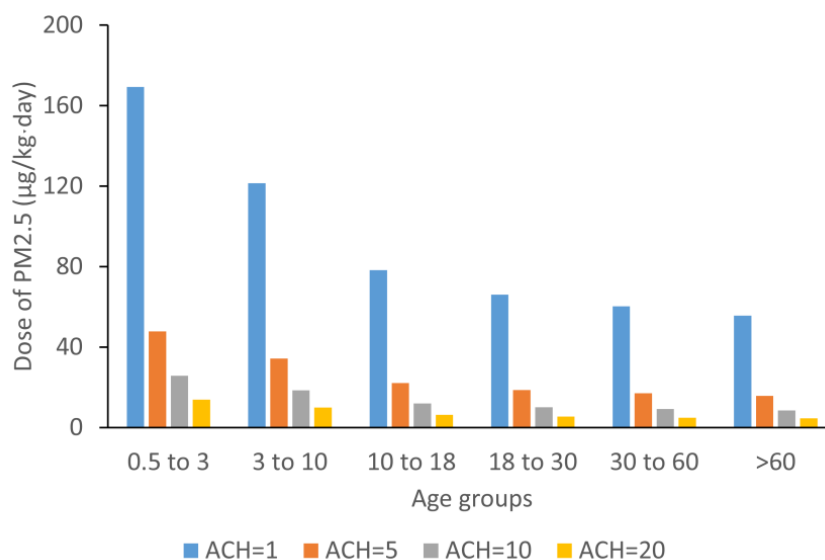


Figure 10. Case study of potential dose of $PM_{2.5}$ for different age groups under different ventilation parameters.

In general, potential $PM_{2.5}$ doses decreased with age groups, and children under 3 years had the highest potential dose, in agreement with other studies [15,50], because younger children are more active and breathe more per unit of body weight.

A study of chimney stove impact conducted over a period of 12 months by Chakraborty et al. found that the median value of $PM_{2.5}$ RQ was 1.63 [61]. Although the utilization of a chimney stove has adverse health effects, the results show a lower $PM_{2.5}$ potential dose compared to that from a traditional open fire, for which the observed RQ can be as high as 5.57 [61].

Single-zone models tend to overestimate concentrations, as the model assumes a well-mixed environment, which may attribute to the discrepancies in the calculated and measured concentration [62]. The overestimation of potential dose can thus be a limitation of this approach.

4. Conclusions

The aim of this study is to provide a full-cycle analysis integrating air quality models, infiltration models, and risk assessment models to better understand the impacts of

neighborhood pollution and stove fugitive emissions on the potential dose. The main conclusions of this study are:

1. PM_{2.5} pollutants tend to accumulate in the wake of buildings, and the pollutant infiltration can contribute significantly to poor indoor air quality. The major contributor to indoor pollution, however, is fugitive emissions from cooking stoves.
2. The contribution of chimney stoves to infiltration increases with higher packing densities and may contribute to indoor pollutant concentrations.
3. Indoor concentrations from chimney stoves exceed WHO air quality guideline values for PM_{2.5}. The associated health risk assessment shows that the risk quotient (RQ) is 2.19, despite good ventilation conditions.

Supplementary Materials: The following supporting information can be downloaded at: <https://www.mdpi.com/article/10.3390/su15075676/s1>, Figure S1: Comparison of meteorological approximation model output and field measurement; Figure S2: Comparison of point source dispersion among Gaussian, QUIC, and water channel evaluation; Figure S3: Comparison of (left) AERMOD output with (right) QUIC output for one-hour outdoor emission estimation. References [18,19,63–65] are cited in the supplementary materials.

Author Contributions: Conceptualization, Y.H., S.R.N., R.D.E. and M.P.; data curation, Y.H. and S.R.N.; formal analysis, Y.H. and S.R.N.; investigation, Y.H. and S.R.N.; software, Y.H. and S.R.N.; validation, Y.H. and S.R.N.; visualization, Y.H. and S.R.N.; methodology, Y.H. and S.R.N.; funding acquisition, R.D.E.; project administration, R.D.E. and M.P.; supervision, M.P. and R.D.E.; writing—original draft, Y.H. and S.R.N.; writing—review and editing, Y.H., S.R.N., R.D.E. and M.P. All authors have read and agreed to the published version of the manuscript.

Funding: This work was funded by Clean Stacking Options and Regional IAP Scenarios for Rural Mexico, NIH-5585744.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: Not applicable.

Acknowledgments: We are grateful to the Los Alamos National Laboratory for providing the QUIC model under license number: LIC-20-04147. Special thanks to Hannah Lee, Rebecca Albano, and Arthor Bernal for running the QUIC simulation.

Conflicts of Interest: The authors declare no conflict of interest.

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