

Reviewer #2

This study revealed significant urban-rural exposure inequality in terms of airborne microplastics (MPs) and plasticizers in Northern China. Collected PM_{2.5} samples were analyzed across indoor and outdoor environments. It was found that rural residents face substantially higher contamination levels compared to urban residents. The 24-hour time-weighted exposure for rural residents was 3.6 times higher for MPs and 6.8 times higher for plasticizers compared to those living in urban areas. Specifically, the concentration of plasticizer DEHP reached roughly 600 ng/m³ in rural indoor environments, which is over ten times higher than those measured in urban areas. In addition, rural populations were found to endure consistently higher cancer/non-cancer risk and OPDTT. As revealed by source apportionment, plastic products using was found to contribute the high exposure risk of rural residents. A number of different assessments were performed here. However, current conclusions lack novelty, and the findings need to be more firmly supported.

Response:

We sincerely thank the reviewer for the careful evaluation of our manuscript and for identifying several important issues regarding the representativeness of the study, indoor environmental processes, uncertainty in source apportionment, interpretation of health risks, and the distinction between environmental inequality and equity. We have substantially revised the manuscript accordingly. In particular, we have narrowed the spatial and temporal scope of the conclusions, expanded the discussion of indoor accumulation processes, tempered the interpretation of the PMF results, added cellular ROS studies of representative MP polymers removed correlation-based attribution of DTT activity to MPs and plasticizers, revised the health-risk assessment and its associated limitations, and replaced broad policy claims with evidence-based but non-quantitative recommendations.

Beyond these revisions, two substantive additions have been made to strengthen the novelty of the present work. First, we have conducted cellular ROS assays using representative MP polymers (PET, PE, and PS) to more directly evaluate their intrinsic oxidative potential, thereby providing toxicological evidence beyond environmental concentration measurements. Second, we have added a dedicated section (Section 3.6) that systematically examines the indoor-dominated drivers underlying urban–rural exposure inequality, synthesizing source apportionment, questionnaire data, and indoor environmental factors to clarify why rural indoor environments exhibit persistently higher MP and plasticizer levels. Our point-by-point responses are provided below.

1. Current title of this manuscript was “exposure inequality for rural residents in Northern China”, but the sampling sites involved in this work were very limited to draw such conclusion. Many

important cities in the BTH region were not mentioned. Also, only samples from January were collected and analyzed. But there is a lack of discussion on the temporal variation. Thus, the representativeness of the conclusion should be reconsidered.

Response:

We thank the reviewer for this important comment. We agree that our sampling is geographically and temporally limited. We have therefore revised the title to: *Indoor-Driven Urban–Rural Inequalities in Time-Weighted Exposure to PM_{2.5}-Bound Microplastics and Plasticizers: A Case Study in the Guanzhong Plain, China* and narrowed the interpretation of the conclusions to a case study in the Guanzhong Plain. We have also explicitly acknowledged the constrained geographic scope and winter-only sampling as limitations as the following: *The relatively small sample size, the constrained geographic scope, and the winter-only sampling season may limit the generalizability of our findings to other regions and seasons, such as summer conditions with different ventilation and source patterns.*

2. The authors reported the comparison between the concentration of microplastics and plasticizers between indoor and outdoor environments. However, there is a lack of discussion on many characteristic impacts of indoor environments. For example, the indoor-outdoor air exchange rates and indoor resuspension effects. Differences in building air exchange rates (rural vs. urban) could substantially influence indoor MP/plasticizer levels in addition to the source influences.

Response:

We are very grateful for the reviewers' important comments. We have made the following major revisions to the main text, and the manuscript structure has changed. Sections 3.1-3.3 remain unchanged; section 3.4 is now "Influencing factors and sources of MPs and plasticizers"; and section 3.5 is "Health effects of MPs and plasticizers in PM_{2.5}"; section 3.6 (newly added) is "Dominated drivers underlying urban–rural exposure inequality". The specific content of newly added section 3.6 is shown below.

3.6 Dominated drivers underlying urban–rural exposure inequality

The indoor and outdoor concentration distributions (Figures 3 and 4) further indicated that pollutant concentrations were substantially elevated in the rural indoors, with a rural – urban concentration ratio of 3.9 for MPs and a nearly order-of-magnitude gap for DEHP and other typical PAEs; However, urban outdoor PAEs were slightly higher than rural levels, and urban MPs were slightly lower. This inversion demonstrates that differences in indoor source emission strength, combined with indoor accumulation and retention, constitute the core mechanism driving urban–rural exposure inequality. PMF source apportionment revealed that the core structural difference

in urban–rural exposure to MPs and plasticizers lies in the contribution gap from plastic-product-related sources: these sources accounted for 51.7% of the total in rural areas, roughly 1.9 times the urban value (27.6%).

The combined evidence from PMF source apportionment, indoor–outdoor concentration patterns, questionnaire responses, and field observations suggests that household activities and residential characteristics jointly drive the substantially higher indoor burden of MPs and plasticizers in rural households. The aging and resuspension of everyday plastic items contribute significantly: rural households tend to use plastic commodities (storage containers, tableware, table covers, etc.) for longer periods with higher reuse rates, leading to more severe material degradation and sustained release of MPs and plasticizers (Yu et al., 2023; Wang et al., 2024). Frequent turning of heavy winter clothing and synthetic bedding facilitates resuspension of fibrous MPs, while prolonged food storage in plastic bags and containers adds to plasticizer volatilization, together forming a baseline of indoor sources (Hernandez et al., 2017; Vassilenko et al., 2021; Wang et al., 2024; Bi et al., 2024). According to the questionnaire information, 60% of rural households in this survey burn plastic waste (such as apple bags, recycled mulch film, etc.) in indoor stoves (usually used as kindling for solid fuel) or nearby yards during the winter heating season. the incomplete combustion generates flue gases rich in MPs and plasticizers. Burning indoors directly retains the emissions in living spaces, while courtyard burning allows the fumes to infiltrate indoors through door/window gaps and ventilation openings (Velis & Cook, 2021; Wu et al., 2021; Růžicková et al., 2023).

Beyond source strength differences, the residential characteristics of rural areas in winter markedly reduce pollutant dispersion and thus amplify urban–rural exposure disparities. During the sampling period in January 2024, the average daily outdoor temperature for rural households was -5°C, and the average indoor temperature was 9°C (without central heating); while the average daily outdoor temperature for urban households was -1°C, and the average indoor temperature was 21°C. The cold winter climate prompts rural families to substantially reduce window opening for thermal insulation. Inadequate ventilation dilution is a primary factor for elevated indoor pollutant concentrations. Questionnaire showed that the composite score for “daily living habits” (including ventilation frequency, cleaning practices, and use of air purifiers) was lower in rural areas (1.50) than in urban areas (1.73), indicating generally lower ventilation and cleaning frequencies. Meanwhile, winter heat-retention practices in rural area in order to resist the cold, such as sealing doors and windows and installing door curtains, increase building airtightness and reduce natural ventilation and outdoor air infiltration, thereby lowering air exchange rates and weakening pollutant dilution and removal (Ghasemi et al., 2020; Jahanbin & Semprini, 2022; Mukai et al.,

2009; Wallace et al., 2002; Zhang et al., 2022). Moreover, there are the interconnected cooking, living, and sleeping areas in many rural houses (Feng et al., 2025). Cooking and heating activities in the kitchen can generate buoyancy-driven indoor airflow, altering the transport, dispersion, and decay of deposited particles; the stronger near-floor airflow and turbulence may increase particle resuspension.

In summary, the superposition of source strength and retention effects collectively elevates the indoor inhalation exposure burden of rural residents and provides a mechanistic explanation for the marked urban–rural differences in health risks. The stronger indoor sources and reduced pollutant removal in rural households resulted in substantially higher inhalation exposure, which was reflected in the greater ROS-inducing potential of rural PM_{2.5} samples and the elevated non-carcinogenic and carcinogenic risks associated with MPs and plasticizers compared with urban areas. Based on these contrasting source and exposure patterns, source-informed exposure-reduction priorities may differ between the investigated rural and urban settings. For the rural households, potential measures include reducing the repeated handling and indoor storage of disposable or aged plastic materials, improving plastic-waste collection and separation, avoiding uncontrolled waste burning, increasing ventilation when outdoor conditions permit, and adopting lower-emission alternatives for food storage and agricultural materials. For the investigated urban environment, priorities may include reducing traffic-related non-exhaust emissions, particularly tire-wear particles, and strengthening the control of emissions associated with consumer products and building materials. These recommendations represent source-informed priorities rather than quantitatively validated mitigation efficiencies.

Reference:

- Bi, C., Eichler, C. M. A., Wang, C., & Little, J. C. (2024). Association between emission parameters and material-phase concentrations of phthalate plasticizers and their alternatives. *Aerosol and Air Quality Research*, 24(1), 230163.
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- Wu, D., Li, Q., Shang, X., Liang, Y., Ding, X., Sun, H., Li, S., Wang, S., Chen, Y., & Chen, J. (2021). Commodity plastic burning as a source of inhaled toxic aerosols. *Journal of Hazardous Materials*, 416, 125820.
- Yu, Y., Craig, N., & Su, L. (2023). A hidden pathway for human exposure to micro- and nanoplastics—The mechanical fragmentation of plastic products during daily use. *Toxics*, 11(9), 774.
- Zhang, B., Cai, X., & Liu, M. (2022). Study on a new type of ventilation system for rural houses in winter in the severe cold regions of China. *Buildings*, 12(7), 1010.

3. In terms of the source apportionment analysis, the limited number of samples and the chemical species (no typical tracing components) measured in this work are likely to bring significant uncertainties in source identification in indoor and outdoor environments. Running Positive Matrix

Factorization (PMF) on only 22 urban and 19 rural indoor samples with 13 tracer species risks model overfitting and unstable source solutions.

Response:

We thank the reviewer for raising this important concern. We agree that the relatively limited sample size and tracer coverage may increase the uncertainty of PMF source interpretation. We would like to clarify, however, that the four-factor solution was not selected solely on the basis of factor interpretability. We evaluated three- to five-factor solutions using multiple diagnostic criteria, including Q-values, scaled residuals, signal-to-noise ratios, agreement between observed and predicted concentrations, factor interpretability, and Bootstrap reproducibility, following the EPA PMF 5.0 framework.

The 4-factor solution showed the best overall diagnostic performance. In 100 Bootstrap runs, the factor mapping rates were 87%, 86%, 76%, and 82% for the rural model and 84%, 95%, 92%, and 92% for the urban model. Thus, seven of the eight factors had mapping rates above 80%, indicating generally satisfactory reproducibility. We acknowledge that rural factor 3, with a mapping rate of 76%, was comparatively less stable and should be interpreted with greater caution. In comparison, the 3-factor model exhibited poor fitting performance ($Q(\text{Robust})/Q(\text{True})$ deviated far from 1), while the 5-factor solution produced highly scattered Bootstrap mapping results (most factors < 70%).

Second, the 13 input species were not selected arbitrarily. They were consistently detected across the indoor samples and were selected to represent several relevant polymer and additive groups. The input matrix included eight polymers—PET, PP, PE, PS, PMMA, PVC, NR, and SBR/BR—and five plastic-associated chemicals—HOBt, 24MoBT, DEHP, DnOP, and BPA. NR and SBR/BR are widely used indicators of tire- and rubber-related emissions based on their characteristic pyrolysis products, while 24MoBT has been applied as a molecular tracer of tire-wear-related traffic emissions (Panko et al., 2019; Pan et al., 2012; Miller et al., 2022). HOBt is also abundant in tire-wear particles and other rubber-derived materials, although it is less source-specific than 24MoBT (Reddy and Quinn, 1997; Zhang et al., 2018). DEHP and DnOP provide source information related to plasticizers, building materials, and household furnishings, whereas BPA is indicative of emissions associated with polycarbonate plastics, epoxy resins, and other consumer products (Loganathan and Kannan, 2011; Li et al., 2021). Variations in the relative profiles of PET, PE, PP, PS, PVC, and PMMA may further reflect differences in the use, abrasion, weathering, and fragmentation of plastic-containing materials (Liao et al., 2021). We acknowledge that several of these species are source-informative rather than uniquely source-specific; therefore, factor

identification was based on the combined chemical profile, questionnaire data, field observations, and known material applications rather than on any individual marker alone.

We have revised Section 3.4 to explicitly discuss these limitations and the validation results. The added text clarifies that (Line 343): *the source results should be interpreted at the level of broad source patterns rather than as uniquely resolved emission sources, and acknowledges the uncertainty associated with factor 3 in rural areas. The newly added content is as follows: Although the four-factor PMF solution showed generally satisfactory diagnostic performance and Bootstrap reproducibility, the source results should be interpreted at the level of broad source patterns rather than as uniquely resolved emission sources. Seven of the eight factors had Bootstrap mapping rates above 80%, while rural factor 3 (Home decoration materials) had a mapping rate of 76%, indicating greater uncertainty in this factor. In addition, several polymers and additives may be shared among different consumer and household sources. The PMF results were therefore interpreted together with questionnaire and field-observation data.*

In addition, we have added model selection and diagnostic methods in Section 2.4. The details are as follows: *Three- to five-factor solutions were evaluated for both the rural and urban datasets. The number of factors was selected by jointly considering $Q(\text{Robust})/Q(\text{True})$, scaled residual distributions, signal-to-noise ratios, agreement between observed and predicted concentrations, factor interpretability, and Bootstrap reproducibility. The four-factor solution provided the best overall balance between goodness of fit and factor stability. Bootstrap analysis was conducted for 100 runs, and the detailed diagnostic results are provided in Text S2 and Table S4 of the Supporting Information.*

In summary, the PMF results in this study have good reliability and can, to some extent, support the source contribution of microplastic-related pollution in rural and urban areas during winter.

Reference:

- Li, X., Zhang, W., Lv, J., Liu, W., Sun, S., Guo, C., & Xu, J. (2021). Distribution, source apportionment, and health risk assessment of phthalate esters in indoor dust samples across China. *Environmental Sciences Europe*, 33, 19.
- Liao, Z., Ji, X., Ma, Y., et al. (2021). Airborne microplastics in indoor and outdoor environments of a coastal city in Eastern China. *Journal of Hazardous Materials*, 417, 126007.
- Loganathan, S. N., & Kannan, K. (2011). Occurrence of bisphenol A in indoor dust from two locations in the Eastern United States and implications for human exposures. *Archives of Environmental Contamination and Toxicology*, 61, 68–73.
- Miller, J. V., Chan, K., & Unice, K. M. (2022). Evaluation of three pyrolyzer technologies for

quantitative pyrolysis–gas chromatography–mass spectrometry of tire tread polymer in an artificial sediment matrix. *Environmental Advances*, 8, 100213.

Pan, S., Sun, Y., Zhang, G., Li, J., Xie, Q., & Chakraborty, P. (2012). Assessment of 2-(4-morpholinyl) benzothiazole (24MoBT) and N-cyclohexyl-2-benzothiazolamine (NCBA) as traffic tracers in metropolitan cities of China and India. *Atmospheric Environment*, 56, 246–249.

Panko, J. M., Hitchcock, K. M., Fuller, G. W., & Green, D. (2019). Evaluation of tire wear contribution to PM_{2.5} in urban environments. *Atmosphere*, 10, 99.

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Zhang, J., Zhang, X., Wu, L., Wang, T., Zhao, J., Zhang, Y., Men, Z., & Mao, H. (2018). Occurrence of benzothiazole and its derivatives in tire wear, road dust, and roadside soil. *Chemosphere*, 201, 310–317.

4. The discussion of health risk is not comprehensive enough. Both OPDTT and cancer/noncancer risks were evaluated. But the relationships among MP/plasticizers, OP, and cancer risks were not discussed clearly. Correlations between DTTm (mass-normalized oxidative potential) and MPs/plasticizers were very weak ($r = 0.2\text{--}0.4$, some near 0). Also, the US EPA has no established RfD (reference dose) or SF (slope factor) values for inhaled microplastics as particulate polymers. The extrapolation of parameters should be clearly justified in terms of their toxicity parameter or routes of exposure.

Response:

We sincerely thank the reviewer for this critical comment. We agree that the previous manuscript did not clearly distinguish among three conceptually different endpoints: DTT-based acellular oxidative potential of bulk PM_{2.5}; cellular oxidative-stress responses induced by selected MPs; and chronic non-carcinogenic and carcinogenic risk estimates derived from chemical-specific toxicity parameters. We have placed great emphasis on distinguishing this confusion in the revised manuscript. In the revised manuscript, all DTT-related content has been removed, including the experimental method, DTTv/DTTm results, correlation analyses, corresponding figures, and statements linking MPs or plasticizers to DTT-based oxidative potential. Consequently, no relationship between DTT activity and cancer or noncancer risk is inferred.

We also agree that established inhalation RfD or slope-factor values are unavailable for particulate microplastic polymers, PS and PET. Therefore, the HQ calculations, the corresponding

rows in Table 1, and the related discussion of PS and PET have been removed. The retained HQ and ILCR assessments are limited to chemicals for which toxicity parameters are available.

we conducted intracellular ROS experiments using A549 human lung epithelial cells exposed to MP standards to more directly assess whether MPs can induce a biological oxidative-stress response independently. The selection of polyethylene terephthalate (PET), polyethylene (PE), and polystyrene (PS) was primarily based on their environmental representativeness and toxicological relevance. All three were the main polymers detected in the PM_{2.5} samples in this study, with PET dominating the indoor samples and exhibiting the most significant urban-rural exposure differences; PS and PE also showed high detection levels. Furthermore, these three MPs represent different chain structures: PET with an ester-based backbone, PE with a fully aliphatic chain, and PS with aromatic side chains. These structures can preliminarily reflect the differences in ROS induction among different polymer structures in a limited experimental system. The results showed that all three polymers induced concentration-dependent intracellular ROS formation. Their EC_{1.5} (inducing reactive oxygen species to reach 1.5 times the effective concentration of the control group) values were 3.03, 2.89, and 2.20 $\mu\text{g mL}^{-1}$ for PET, PE, and PS, respectively. When expressed as repeat-unit-equivalent molar concentrations, the corresponding EC_{1.5} values were 1.58×10^{-5} , 1.03×10^{-4} , and 2.11×10^{-5} mol L⁻¹ (Table 3 below).

These EC_{1.5} values were compared with selected PM_{2.5}-associated metals and PAHs reported in a standardized lung-cell toxicity dataset (Chen et al., 2025). The ROS-inducing potencies of the different substances were highly heterogeneous, with EC_{1.5} values spanning more than six orders of magnitude. The carcinogenic PAHs, such as dibenzo[a,l]pyrene (DBaP) and benzo[a]pyrene (BaP) and the redox-active transition metals, like Mn(II) and Cu(II) showed the high-potency of ROS, whereas the relatively redox-inactive crustal elements Rb(I), Mg(II), Al(III), and Ti(IV) exhibited substantially lower potencies. The three MPs molar effect potencies were approximately 1.3-2.2 orders of magnitude lower than those of Cu(II) and Mn(II). PET and PS were generally approximately one to two orders of magnitude more potent than Rb(I), Mg(II), Al(III), and Ti(IV), whereas PE was only approximately 2- to 15-fold more potent than these four elements.

Then, we applied a concentration-addition-based (CA) equivalent approach to estimate the contributions of PET, PE, and PS to the intracellular ROS response induced by rural and urban PM_{2.5} (Jin et al., 2019). The three MP polymers together accounted for approximately 0.52% of the PM_{2.5}-induced ROS response in rural samples and 0.15% in urban samples (Table S5). PET made the largest contribution, mainly because of its substantially higher environmental abundance. These results clarify the distinction between intrinsic hazard and mixture-level contribution. PET, PE, and PS possessed measurable cellular ROS-inducing activity at experimental concentrations, but their

direct contributions to the overall PM_{2.5}-induced ROS response were small at the measured environmental abundances. Nevertheless, their environmental persistence, broad diversity in polymer and additive composition, potential for deposition and prolonged retention in the distal respiratory tract, and capacity to interact with co-occurring toxicants warrant continued investigation of the health effects associated with chronic, low-dose inhalation exposure (Hahladakis et al., 2018; Prata, 2018; Amato-Lourenço et al., 2021; Zhang & Xu, 2022). We have also added an explicit limitation stating that the cellular experiments used pristine polymer standards and that the CA analysis represents a screening-level estimate under an additivity assumption. Environmentally aged MPs with altered surface chemistry or adsorbed redox-active contaminants may exhibit different biological effects.

We did not conduct corresponding cellular ROS experiments for PAEs in the revised version. Because the main purpose of the newly added experiments was to address the difficulty of characterizing the bio-oxidative effects of insoluble microplastic particles using the DTT method. Therefore, the focus was on verifying whether representative MPs could directly induce ROS in A549 cells. PAEs contain a variety of compounds with significantly different physicochemical properties and toxicological effects. Cellular experiments with these compounds require independent experimental designs tailored to monomer composition, environmental concentration range, solvent carrier, and mixed exposure methods. The cellular oxidative stress effects of PAEs and their mixed toxicity will be investigated in our future studies.

Table 3 Comparison of EC_{1.5} values for ROS induction between MPs and typical PM_{2.5}-bound metals and PAHs.

Category	Substance	EC _{1.5} (mol L ⁻¹)	Source
Metal	Rb(I)	1.51×10 ⁻³	Chen et al., 2025
Metal	Mg(II)	3.98×10 ⁻⁴	Chen et al., 2025
Metal	Al(III)	3.70×10 ⁻⁴	Chen et al., 2025
Metal	Ti(IV)	2.29×10 ⁻⁴	Chen et al., 2025
MPs	PE	1.03×10⁻⁴	This study
MPs	PS	2.11×10⁻⁵	This study
MPs	PET	1.58×10⁻⁵	This study
Metal	Cu(II)	8.28×10 ⁻⁷	Chen et al., 2025
Metal	Mn(II)	6.03×10 ⁻⁷	Chen et al., 2025
PAHs	BaP	1.60×10 ⁻⁸	Chen et al., 2025
PAHs	DBaP	1.20×10 ⁻⁹	Chen et al., 2025

Note: EC_{1.5} represents the concentration required to induce a 1.5-fold increase in intracellular ROS

relative to the control; lower EC_{1.5} values indicate greater oxidative stress potency. Molar concentrations of PET, PE, and PS are expressed as repeat-unit-equivalent concentrations, with corresponding repeat-unit molar masses of 192.2, 28.05, and 104.2 g mol⁻¹, respectively.

Table S5 Concentration-addition (CA) model estimates of ROS contribution from representative MPs in indoor PM_{2.5}.

Substance	EC _{1.5} (mg L ⁻¹)	Relative Effect Potency	BEQ _{chem} (nmol mg ⁻¹ PM _{2.5})		Contribution to PM _{2.5} ROS (%)	
			Rural	Urban	Rural	Urban
PET	3.03	0.0526	0.0374	0.00695	0.38	0.090
PE	2.89	0.0081	0.00411	0.00092	0.042	0.012
PS	2.20	0.0393	0.00964	0.00399	0.098	0.052
Three MPs	/	/	0.0511	0.0119	0.52	0.15

Notes: t-BHQ was used as the reference (EC_{1.5} = 0.14 mg L⁻¹, REP = 1.0; data from Jin et al., 2019 and Chen et al., 2025). The EC_{1.5} values for rural and urban PM_{2.5} samples were 84.1 and 107 mg L⁻¹, respectively, as determined by the A549 cell ROS assay in this study.

Reference:

- Amato-Lourenço, L. F., Carvalho-Oliveira, R., Júnior, G. R., dos Santos Galvão, L., Ando, R. A., & Mauad, T. (2021). Presence of airborne microplastics in human lung tissue. *Journal of Hazardous Materials*, 416, 126124.
- Chen, X., Wu, D., & Li, Q. (2025). A unified cellular toxic potency dataset of PM_{2.5} across chemicals, emission sources, and regions in China. *Scientific Data*, 12(1), 1901.
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- Jin, L., Xie, J., Wong, C. K., Chan, S. K., Abbaszade, G. I., Schnelle-Kreis, J. r., . . . Fu, P. (2019). Contributions of city-specific fine particulate matter (PM_{2.5}) to differential in vitro oxidative stress and toxicity implications between Beijing and Guangzhou of China. *Environmental Science & Technology*, 53(5), 2881-2891.
- Prata, J. C. (2018). Airborne microplastics: Consequences to human health? *Environmental Pollution*, 234, 115–126.
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5. In the conclusion part, the authors stated that “equity-aware frameworks and interventions” but provides no quantitative assessment of potential mitigation measures (e.g., improved rural waste management, ventilation changes, plastic alternatives). The recommendation needs to be clear with specific analysis. Also, inequality and equity are different terms.

Response:

We thank the reviewer for this important conceptual and policy-related comment. We agree that inequality and equity are distinct concepts. The present study quantified differences in pollutant exposure and estimated health risks between the investigated urban and rural communities; it did not evaluate whether resource allocation, procedural fairness, or intervention access was equitable. Therefore, “inequality” is the appropriate term for the empirical differences examined in this study, whereas “equity” would require additional social, institutional, and policy analyses. We have consequently removed or revised the terms “equity-aware framework,” “environmental equity,” and “equitable intervention” throughout the manuscript unless they are used explicitly as broader future policy considerations. The title, Abstract, Results, and Conclusion now use “urban–rural inequality,” “disparity,” or “disproportionate exposure burden” to describe the measured findings.

We also agree that the study did not conduct intervention experiments or quantitative mitigation scenarios. Therefore, we have removed language implying that the effectiveness of specific interventions was quantitatively demonstrated. And the following suggestion was added at the end of newly section 3.6: *Based on these contrasting source and exposure patterns, source-informed exposure-reduction priorities may differ between the investigated rural and urban settings. For the rural households, potential measures include reducing the repeated handling and indoor storage of disposable or aged plastic materials, improving plastic-waste collection and separation, avoiding uncontrolled waste burning, increasing ventilation when outdoor conditions permit, and adopting lower-emission alternatives for food storage and agricultural materials. For the investigated urban environment, priorities may include reducing traffic-related non-exhaust emissions, particularly tire-wear particles, and strengthening the control of emissions associated with consumer products and building materials. These recommendations represent source-informed priorities rather than quantitatively validated mitigation efficiencies.*