
Reviewer 1:

This study presents a systematic investigation into the urban-rural disparities in exposure to microplastics (MPs) and plasticizers in PM_{2.5} in the Guanzhong Plain, northern China. The manuscript is generally well-written and provides valuable data on concentration levels, health risk assessments, and source apportionment. The topic is timely and relevant, particularly in the context of environmental inequality and emerging contaminants. However, several key scientific and methodological issues need to be addressed before the manuscript can be considered for publication.

Response:

Thanks very much for taking your time to review this manuscript. We have carefully considered each of your suggestions and have made the necessary revisions to improve the quality of our manuscript. Below are our detailed responses to your comments.

(1) The authors used the dithiothreitol (DTT) assay to assess the oxidative potential (OP) of PM_{2.5} and attributed a significant portion of OP to MPs and plasticizers. However, MPs are insoluble particles and do not directly contribute to DTT consumption in aqueous-phase assays. Similarly, most phthalates (PAEs) are hydrophobic and poorly soluble in water, making them unlikely to directly participate in DTT reactions. The observed correlations between OP and these analytes may be confounded by co-emitted species (e.g., metals, quinones, PAHs). The authors should discuss the mechanistic plausibility of MPs and PAEs contributing to DTT-based OP and acknowledge the limitations of the DTT assay in capturing the oxidative potential of insoluble or hydrophobic compounds.

Response:

We sincerely thank the reviewer for raising this important mechanistic concern. We agree with your opinion. We have removed all content related to DTT from the main text. And, 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 µg 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	BEQ _{chem}		Contribution to PM _{2.5}	
		Effect	(nmol mg ⁻¹ PM _{2.5})		ROS (%)	
		Potency	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 ($EC_{1.5} = 0.14 \text{ mg L}^{-1}$, $REP = 1.0$; 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.
- Hahladakis, J. N., Velis, C. A., Weber, R., Iacovidou, E., & Purnell, P. (2018). An overview of chemical additives present in plastics: Migration, release, fate and environmental impact during their use, disposal and recycling. *Journal of Hazardous Materials*, 344, 179–199.
- 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.
- Zhang, M., & Xu, L. (2022). Transport of micro- and nanoplastics in the environment: Trojan-horse effect for organic contaminants. *Critical Reviews in Environmental Science and Technology*, 52(5), 810–846.

(2) The bridge plot in Figure 2c appears to be incorrectly oriented. The current representation suggests that urban concentrations are higher than rural ones for certain MPs, which contradicts the data presented elsewhere. The authors should either reverse the direction of the difference or reorder the groups (e.g., urban on the left, rural on the right) to clearly illustrate the rural excess.

Response:

Thank you for your suggestion. We have modified the bridge diagram in Figure 2 in the main

text. In Figure 2c and figure 2d of the bridge, we follow the arrangement of rural areas on the left and cities on the right. Although it is true that the addition concentration of some micro plastics and plasticizers is higher in urban areas than in rural areas (such as PVC and BPA), positive values are used in the bridge diagram to express this situation, and the text also describes (e.g., in line 257 “Rural areas showed higher time-weighted exposure than urban areas for seven microplastic types, excluding PVC.”). The title of Figure 2 is modified as “Fig. 2 Urban-rural inequality in time-weighted personal exposure to (a) total MPs, (b) total plasticizers, (c) eight types of MPs, and (d) three types of plasticizers (In Fig. 2c and 2d, rural values are used as the reference; negative values indicate that weighted exposure concentration in rural areas is higher than in urban areas).”.

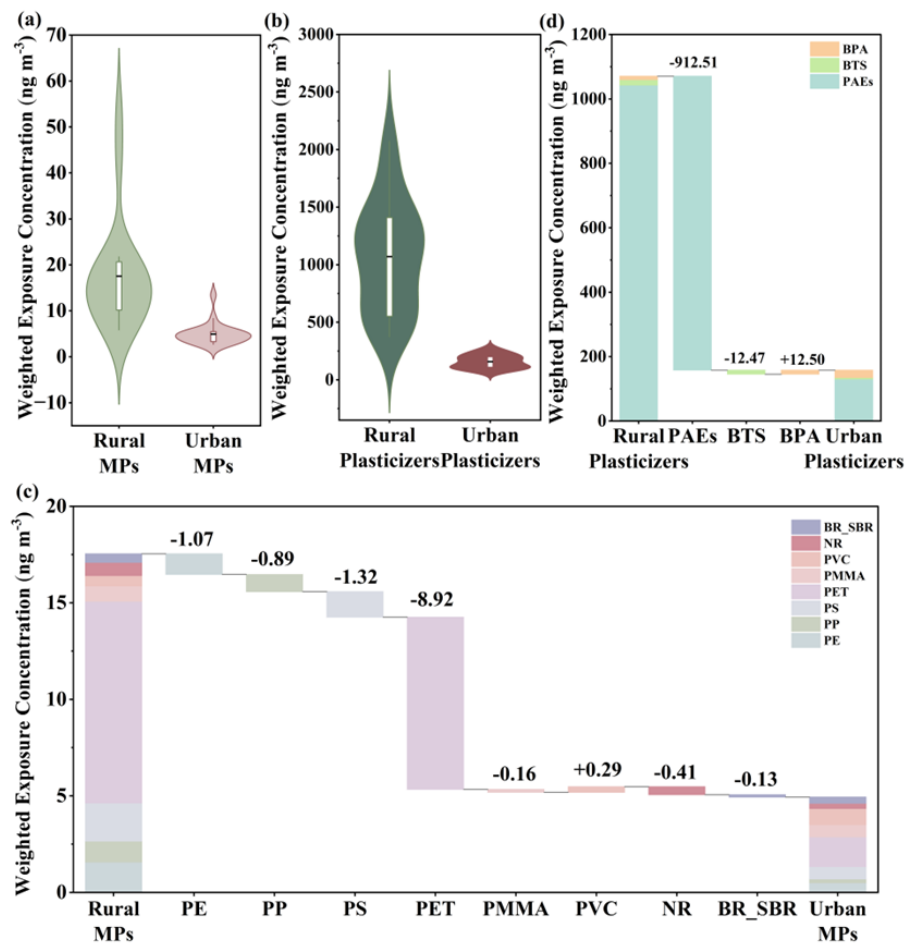


Fig. 2 Urban-rural inequality in time-weighted personal exposure to (a) total MPs, (b) total plasticizers, (c) eight types of MPs, and (d) three types of plasticizers (In Fig 2c and 2d, rural values are used as the reference; negative values indicate that weighted exposure concentration in rural areas is higher than in urban areas).

(3) Figure 4 reveals an important but underexplored pattern: outdoor concentrations of MPs

and PAEs are slightly higher in urban areas, while indoor concentrations are substantially higher in rural areas—by nearly an order of magnitude. This suggests that indoor sources dominate rural exposure. The authors should provide a more detailed discussion of potential indoor emission sources in rural homes (e.g., plastic sheeting, food storage, waste burning, consumer products), such as considering the role of winter ventilation practices in trapping indoor pollutants. If available, include data on housing characteristics (e.g., building materials, heating methods) to support the interpretation.

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.
- Feng, R., Wang, K., Xu, H., Gu, Y., Yang, L., Sun, J., & Shen, Z. (2025). Pulmonary function and influencing factor investigation for rural homemakers in the Fenwei Plain, China. *Toxics*, 13(12), 1061.
- Ghasemi, M., Toghraie, D., & Abdollahi, A. (2020). An experimental study on airborne particles dispersion in a residential room heated by radiator and floor heating systems. *Journal of Building Engineering*, 32, 101677.
- Hernandez, E., Nowack, B., & Mitrano, D. M. (2017). Polyester textiles as a source of microplastics from households: A mechanistic study to understand microfiber release during washing. *Environmental Science & Technology*, 51(12), 7036–7046.
- Jahanbin, A., & Semprini, G. (2022). Integrated effects of the heat recovery ventilation and heat source on decay rate of indoor airborne particles: A comparative study. *Journal of Building Engineering*, 50, 104156.
- Mukai, C., Siegel, J. A., & Novoselac, A. (2009). Impact of airflow characteristics on particle resuspension from indoor surfaces. *Aerosol Science and Technology*, 43(10), 1022–1032.
- Růžicková, J., Raclavská, H., Kucbel, M., Pfeifer, C., Juchelková, D., Hrbek, J., Šafář, M., Slamová, K., Švédová, B., & Kantor, P. (2023). Incidence and spread of additives from co-combustion of plastic waste in domestic boilers in indoor and outdoor environments around the family

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- Vassilenko, E., Watkins, M., Chastain, S., Mertens, J., Posacka, A. M., Patankar, S., & Ross, P. S. (2021). Domestic laundry and microfiber pollution: Exploring fiber shedding from consumer apparel textiles. *PLOS ONE*, 16(7), e0250346.
- Wallace, L. A., Emmerich, S. J., & Howard-Reed, C. (2002). Continuous measurements of air change rates in an occupied house for 1 year: The effect of temperature, wind, fans, and windows. *Journal of Exposure Analysis and Environmental Epidemiology*, 12(4), 296–306.
- Wang, L., Gao, J., Wu, W.-M., Luo, J., Bank, M. S., Koelmans, A. A., Boland, J. J., & Hou, D. (2024). Rapid generation of microplastics and plastic-derived dissolved organic matter from food packaging films under simulated aging conditions. *Environmental Science & Technology*, 58(45), 20147–20159.
- 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.

(4) The current title implies a general conclusion about urban-rural disparities. However, the main findings are driven by indoor exposure differences, which are not reflected in the title. To avoid misleading readers, the authors should either modify the title to include the indoor context (e.g., “Indoor Exposure Disparities...”), or clearly state in the abstract and introduction that the study focuses on personal exposure integrating indoor and outdoor time-activity patterns.

Response:

We thank the reviewer for this important suggestion. To more accurately reflect the study design and avoid overgeneralization, we have 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*. We have also revised Lines 18–22 of the Abstract to “*This study examined MPs and plasticizers in PM_{2.5} across urban-rural environments in Northern China's Guanzhong Plain by integrating indoor-outdoor time-activity weighted exposure. The 24 h time-weighted exposure concentrations for MPs and plasticizers were 3.6 and 6.8 times higher,*

respectively, in rural areas than in urban areas.”.

(5) While the authors report strong correlations between OP and MPs/BTs, they do not sufficiently control for other known redox-active species in PM_{2.5} (e.g., transition metals, quinones, EC/OC). This weakens the claim that MPs and plasticizers are key drivers of OP. The authors should include a discussion of potential confounders.

Response:

We thank the reviewers for pointing out this issue. We take it very seriously. In the revised manuscript, we have completely removed all content related to the correlation analysis between the DTT method and MPs/plasticizers (including the corresponding results, discussion, and figures). Therefore, the potentially confusing statement in the original text regarding "MPs being a key driver of OP" is no longer present. Currently, we have only determined the efficacy of specific MPs (PET, PE, PS) in inducing ROS individually through cell experiments and estimated the contribution of these three polymers to the overall ROS response of PM_{2.5} using a CA approach. The results show that their contribution rates in rural and urban samples are only 0.52% and 0.15%, respectively. We have clearly stated in the discussion that this contribution is a screening-level estimate, and compared to known oxidatively active species such as transition metals and PAHs, the contribution of MPs is very limited. Please refer to the first comment of this reply for detailed information. Thank you again for the reviewers' rigorous comments.

(6) The study is conducted in January 2024 in the Guanzhong Plain, a region with distinct winter climate and heating practices. The authors should explicitly acknowledge that the findings may not be generalizable to other seasons (e.g., summer with different ventilation and source patterns).

Response:

We thank the reviewer for this important comment. We agree that the sampling campaign, conducted exclusively in January 2024, represents winter conditions in the Guanzhong Plain and should not be generalized to other seasons or interpreted as an annual exposure profile.

We really should add this part to the limitation description in the conclusion section. We Change the content in lines 580-576 to “*Although there are some limitations in this study, such as the relatively small sample size, the constrained geographic scope, the lack of physical methods to determine the morphology of MPs, and the winter-only sampling season which may limit the generalizability of our findings to other seasons (e.g., summer with different ventilation and source*

patterns), this is the first study of atmospheric MPs and plasticizers conducted in both indoor and outdoor environments across urban and rural areas in northern China.”.