Environmental Systems Research

RESEARCH

Open Access

Occurrence and inhalation health risk of neonicotinoid pesticides in outdoor air particulate matters from 2019 to 2021 in China

Hairong Du[1*], Xiaoling Zhu[2], Yunying Yao[1] and Wei Yao[1]

Abstract

Neonicotinoid insecticides (NNIs), as a new type of insecticide, are widely used in agriculture and daily life. Because of the low volatility of NNIs, few studies have evaluated them in atmospheric particulate matter. In this study, 101 outdoor $PM_{2.5}$ samples were collected from the Wuhan urban area from 2019 to 2021, leading to the detection of seven NNIs and three of their metabolites. The detection frequencies of all 10 substances were more than 60%, with DIN (dinotefuran) and IMI (imidacloprid) reaching 100%. DIN (52.4 pg/m^3) and IMI (43.0 pg/m^3) had higher median concentrations than other substances. Concentration distributions of the four substances ACE (acetamiprid), DIN, IMI, and 5-OH-IMI (5-Hydro-Imidacloprid) exhibited statistically significant differences ($P<0.05$) across the four seasons. Concentration levels of ACE, CLO (clothianidin), and IMI were statistically different between the three years ($P<0.05$). The median concentration of imidacloprid-equivalent total neonicotinoids (IMIeq: generated by the relative potency factor method) was 256.1 pg/m^3. Finally, the estimated daily intake (EDI) of NNIs via respiration was greater in infants and young children than in the rest of the population, suggesting that infants and young children were more likely to be exposed to the health effects of airborne residual $PM_{2.5}$.

Keywords Neonicotinoid pesticides, $PM_{2.5}$, Health risk estimation

Introduction

Neonicotinoids (NNIs) are novel pesticides that function primarily through the selective control of insect nervous system nicotinic acetylcholine esterase receptors, thereby disrupting the normal conduction of the insect central nervous system and resulting in paralysis and mortality of the pest (Matsuda et al. 2001). NNIs include acetamiprid (ACE), clothianidin (CLO), dinotefuran (DIN), imidacloprid (IMI), thiacloprid (THCP), thiamethoxam (THM) and nitenpyram (NIT) (Chi-Hsuan Chang et al. 2018). Compared with conventional insecticides, NNIs possess notable advantages, such as safety, efficacy, and high selectivity, allowing them to be extensively utilized in agriculture and even in our daily lives (Chen et al. 2020). There were 12 kinds of NNIs registered in China as of September 2021; more than 120 countries were reported to have NNIs registered, and NNIs accounted for 25% of the global insecticide market in 2014 (Bass et al. 2015).

*Correspondence:
Hairong Du
dhrjack@163.com
[1]Guangming District Center for Disease Control and Prevention, Shenzhen, Guangdong 518016, PR China
[2]Shenzhen Hospital for the Prevention and Treatment of Occupational Diseases, Shenzhen 518000, PR China

However, excessive use of NNIs has been reported to have adverse effects on ecosystems and human health (Christen et al. 2016). IMI can cause colony collapse syndrome, which reduces the ability of bees to forage (Cox-Foster et al. 2007). In addition, NNIs are lethal to aquatic and terrestrial invertebrates and potentially dangerous to mammals (Li et al. 2020), including humans (Vijver and van den Brink 2014). Environmental media such as food, water, and air have been extensively contaminated by the overuse of NNIs, suggesting that humans could be exposed to NNIs through a variety of routes, such as ingestion, drinking, and breathing (Han et al. 2018).

China is the largest producer and pesticide consumer in the world (Xiao 2003). As the largest sale and application amount among NNIs in China, IMI's annual production reached 13,500 tons in 2010 (Shao et al. 2013). However, the mass production and use of NNIs have also brought certain disadvantages. In China, pesticides are usually sprayed, which inevitably lead to missed targets and released material into the air. It is reported that the amount of pesticides lost into the air during spraying can reach 30–50% (Coscollà et al. 2008). Ambient air particulate matter, especially small particles, are likely to adsorb toxic and hazardous substances (Valavanidis et al. 2008). However, it has been shown that NNI residues in atmospheric dust account for 0.01–0.40% of the actual amount applied (Montiel-León et al. 2019). While NNIs are present in the atmosphere at low concentrations, the environmental health risks posed should not be underestimated. Atmospheric NNIs not only kill insects such as bees and butterflies but also impact the respiratory and immune systems of humans (Zhang et al. 2019; Zhou et al. 2020).

In this study, atmospheric $PM_{2.5}$ samples were collected from the urban area of Wuhan between 2019 and 2021. The study focused on analyzing the concentration levels and distribution characteristics of 10 NNIs. Furthermore, the study conducted a preliminary assessment of the respiratory exposure levels and potential health risks associated with these NNIs in the study area, considering population exposure parameters. These findings offered valuable data to support air quality assessment and the control of emerging organic pollutants.

Materials and methods

Standards and reagents

Standards of NNIs, including ACE, CLO, DIN, IMI, NIT, THM, FLO (Flonicamid), N-ACE (Acetamiprid-N-desmethyl), 5-OH-IMI (5-Hydro-Imidacloprid), and 6-ClNA (6-Chloronicotinic acid), were acquired from Aladdin, America. The corresponding isotope labelling products were purchased from Anpel Laboratory Technologies (Shanghai) Inc. The above standard solutions were stored at -20 °C, shielded from light. Acetonitrile, methanol, n-hexane, and dichloromethane (LC–MS grade) were obtained from Fisher Chemical (Thermo Fisher Scientific, Inc., Waltham, MA, USA). A 2 mL automatic sampling bottle, cap and PTFE silicone gasket were purchased from Agilent Technology.

Sample collection

Outdoor $PM_{2.5}$ samples were collected using a TH-150 C medium-volume air sampler, with a continuous sampling duration of 24 h on a 90 mm diameter quartz filter membrane. The sampling flow rate was set at 100 L/min. We conducted sampling continuously for a minimum of one week during each of the three seasons spanning from 2019 to 2021, resulting in the collection of a total of 101 samples. The sampling site was located on the rooftop of Building 2, School of Public Health, Tongji Medical College (30°35′N, 114°15′E), approximately 12 m above ground level, with no industrial emission sources in the vicinity. Before sampling, the sampling filters were precleaned, and the instrument's gas flow rate was calibrated using a flow metre. Meteorological data such as atmospheric pressure, temperature, and humidity were recorded during the sampling process. After sampling, the filter membranes containing the collected air particulate matter were folded, placed in sealed bags and stored at -20 °C for subsequent analysis.

Sample preparation

A quarter section of the $PM_{2.5}$-loaded filter membrane was cut and transferred to a glass centrifuge tube. Then, 20 µL of a mixed internal standard solution was added. Then, 3 mL of dichloromethane/hexane (3/1, v/v) was added, followed by ultrasonic extraction at room temperature for 30 min. The mixture was then centrifuged for 5 min at 3000 rpm. After centrifugation, the supernatant was transferred to a new glass test tube. Subsequently, another 3 mL of dichloromethane/hexane was added, followed by a repeat of ultrasonic extraction. The second extraction was combined with the first extraction. Next, 3 mL of acetonitrile was added to the filter membrane sample, and the same ultrasonic extraction process was repeated twice. Finally, the total extracted solution from the four transfers was dried using a 40 °C water bath nitrogen evaporator, after which 200 µL of methanol was added for reconstitution. After vortexing at 2500 rpm for 1 min, 100 µL of the reconstituted solution was filtered through a 0.22 µm organic membrane and transferred to the injection vials with 200ul cannulae. The resulting sample was subjected to LC–MS/MS analysis.

Instrumental analysis

Ultra-performance liquid chromatography-tandem mass spectrometry (UPLC–MS/MS, Waters ACQUITY UPLC I-Class coupled with Xevo TQS, Waters Corporation, Milford, MA, USA) was performed in electrospray ionization positive mode and multiple reaction monitoring (MRM) mode. The MRM transitions and collision energies of the target NNIs are presented in Table S1. Chromatographic separation was achieved on an ACQUITY UPLC BEH C18 column (2.1×100 mm, 1.7 μm). The column temperature was maintained at 40 °C. The mobile phase consisted of water containing 0.1% formic acid (mobile phase A) and methanol (mobile phase B). Linear gradient elution was carried out at a flow rate of 0.3 mL/min. The total program time was 10 min, and the injection volume was 5 μL. The analyte liquid chromatography elution procedures are shown in Table S2.

Quality assurance and quality control

In this study, analytes were quantified using an internal standard calibration method. Twelve calibration levels (0.1 ng/mL-1000 ng/mL) were used for instrument calibration. The linearity (r^2) of all the calibration curves was >0.990. The limits of detection for the ten analytes ranged from 0.01 to 0.07 ng/mL. The recoveries and relative standard deviations of the spiked samples were calculated to assess the accuracy and precision, respectively. The results showed that the recoveries of the spiked analytes were in the range of 85.96-107.53%, and the relative standard deviations of the parallel measurements were less than 15%, which met the requirements of quantitative analysis.

Exposure assessment in the populations

Calculation of the relative potency factor (RPF)

The IMI was chosen as a reference NNI due to its extensive use and research (C.-H. Chang et al. 2018). As shown in the following equation, the RPF method was used to normalize each NNI to the IMI and integrate the cumulative exposure and risk of single NNI. The following formulae were obtained as follows:

$$\begin{aligned}&\mathrm{IMIeq}: \mathrm{imidacloprid}-\mathrm{equivalent\ total\ neonicotinoids, IMIeq}\\&=\mathrm{IMI}\ (\mathrm{NNI_i}\times\mathrm{RPF_i})=\mathrm{CLO}\times 5.82+\mathrm{IMI}\times 1\\&+\mathrm{DIN}\times 2.85+\mathrm{ACE}\times 0.80+\mathrm{THM}\times 9.50.\end{aligned}$$

The calculation of RPF was based on the chronic reference dose (cRfD) of IMI (C.-H. Chang et al. 2018; Lu et al. 2018) as shown in Table S3.

The calculation of estimated daily intake (EDI)

The primary route of entry for hazardous substances in atmospheric particulate matter into the human body are through inhalation via the respiratory system. The estimated daily intake (EDI) is computed based on the following formulaic model (U.S. Environmental Protection Agency.1992. 2014):

$$\mathrm{EDI(inh.)}_a=\frac{\mathrm{C_a}\times \mathrm{IR}\times \mathrm{RR}\times \mathrm{ABS_a}\times \mathrm{ET}\times \mathrm{EF}\times \mathrm{ED}}{\mathrm{BW}\times \mathrm{AT}}$$

The equation above represents the estimation model for calculating the estimated daily intake (EDI) of each target substance (denoted as $\mathrm{EDI(inh.)_a}$) via inhalation, expressed in picograms per kilogram of body weight per day (pg/kg·day). In the equation, C_a represents the concentration of the target substance a in the air (pg/m^3), IR and BW denote the inhalation rate (m^3/day) and body weight (kg), respectively, RR refers to the respiratory retention rate (%), ABS_a represents the percentage of the target substance a absorbed into the bloodstream (%), ET represents the duration of exposure (h/day), EF refers to the exposure frequency (days/years), and ED denotes the duration of exposure (years). AT represents the average exposure time (in days). For noncarcinogenic effects, AT corresponds to the number of days of exposure in EDI. Other exposure parameters can be found in Table S4.

Statistical analyses

The data analysis was completed with SPSS 26.0 software. All figures were created by origin2022. The concentration levels less than the detection limit were replaced by detection limit/√2. Differences between groups were analysed by one-way ANOVA when the two sets of data were normally distributed; otherwise, the Mann–Whitney U test was used. Spearman correlation analysis were performed to explore the relationships between each NNI. $P<0.05$ was considered statistically significant.

Results and discussion

Overall detection of NNIs in $PM_{2.5}$ samples from Wuhan

Seven NNIs, namely, CLO, ACE, NIT, DIN, FLO, IMI, and THM, as well as three metabolites, 5-OH-IMI, N-ACE, and 6-CINA, were analysed in 101 $PM_{2.5}$ samples. Table 1 summarized the detection frequency (DF), detection range, geometric mean (GM), median and mean of these analytes. Among the $PM_{2.5}$ samples collected, the detection frequencies of all 10 substances reached more than 60%. IMI and DIN reached 100%, followed by 5-OH-IMI (98%), N-ACE (98%), and ACE (97%), and 6-CINA had the lowest detection rate (64.45%). In terms of the median concentration, DIN (52.4 pg/m^3) and IMI (43.0 pg/m^3) had higher concentrations than the other substances. Moreover, the

Table 1 Concentration (pg/m^3) and detection frequencies of NNIs in PM.5 samples from 2019 to 2021 in Wuhan

NNIs	DF	GM	Median	Average	Range
CLO	75.2%	4.4	8.3	8.0	< LOD-24.3
ACE	97.0%	2.7	3.3	5.0	< LOD-110.6
NIT	88.1%	1.7	2.2	10.4	< LOD-267.7
DIN	100.0%	52.4	53.3	80.0	7.8-286.1
FLO	94.1%	2.9	2.8	3.5	< LOD-10.4
IMI	100.0%	43.0	42.8	70.6	5.0-268.9
THM	77.2%	1.3	1.1	2.6	< LOD-20.6
5-OH-IMI	98.0%	11.4	10.6	16.1	< LOD-103.9
N-ACE	98.0%	5.3	5.6	6.2	< LOD-22.8
6-CINA	64.45%	0.4	0.6	2.8	< LOD-85.0
IMIeq	/	374.1	256.1	374.1	/

average concentrations of these two were also relatively high, reaching 80 pg/m^3 and 70.6 pg/m^3.

Overall, a correlation was found between the frequency and concentration of substances identified in $PM_{2.5}$ samples and their levels of production and usage in real-world settings. For instance, the high detection frequency of IMI (100%) further supported its widespread use (Xu et al. 2021). Additionally, the total productivity of ACE is 8000 tons per year in China (Shao et al. 2013). Notably, ACE was also detected at a higher frequency and concentration in this study. Wang et al (Wang et al. 2019) collected dust from three cities, Taiyuan, Wuhan and Shenzhen. It was found that NNIs and their metabolites were widely present in dust, with ACE and IMI exhibiting the highest detection rates. The metabolite of ACE, known as N-ACE, was detected in samples almost as frequently (98%) as ACE (97%). Furthermore, the median and geometric mean of N-ACE were found to be higher than ACE. In addition, 5-OH-IMI also exhibited a high detection frequency and concentration. This suggested that NNIs were readily metabolized to specific metabolites through biotransformation (demethylation, hydroxylation, and nitro reduction) (Casida 2011; Taira et al. 2013). Different metabolic pathways result in different metabolites, leading to significant differences in toxicity among NNIs metabolites (Marfo et al. 2015). The acute and chronic thresholds of 5-OH-IMI have been reported to be significantly greater than IMI (Suchail et al. 2004). Therefore, in-depth studies of the metabolites of NNIs are also necessary.

Correlation analysis between the NNIs detected in the $PM_{2.5}$ samples

Spearman correlation analysis of the 10 analytes detected in $PM_{2.5}$ samples indicated positive correlations between CLO and IMI, 5-OH-IMI, and N-ACE ($P<0.05$), as well as a negative correlation between CLO and THM ($P<0.05$). ACE showed positive associations with NIT, DIN, IMI, THM, N-ACE, and 6-CINA ($P<0.05$). A positive correlation was observed between the NIT and DIN ($P<0.01$). DIN was positively associated with 5-OH-IMI and N-ACE ($P<0.05$). In addition, FLO was positively correlated with IMI, 5-OH-IMI and N-ACE ($P<0.05$). IMI was positively correlated with N-ACE ($P<0.001$), and 5-OH-IMI was positively correlated with N-ACE ($P<0.001$) (Fig. 1). Most of the 10 analytes were positively correlated, suggesting that they might share a common source of environmental contamination, and the negative correlations between several substances suggested that they may be substitutes for each other in practical applications. The average concentration of CLO (8.0 pg/m^3) exceeded that of THM (2.6 pg/m^3), and a negative correlation was observed between the two, with a correlation coefficient of 0.2 ($P<0.05$). This phenomenon may be explained by the fact that, besides the presence of CLO itself, THM in the environment has the potential to undergo transformation into CLO (Simon-Delso et al. 2015). We also observed that the correlation between IMI and 5-OH-IMI did not reach statistical significance, possibly due to the limited sample size and the singular environmental medium from which the samples were obtained. More comprehensive studies are needed to explore their correlations in the future.

The distribution characteristics of the 10 NNIs in the $PM_{2.5}$ samples in different seasons

The collected $PM_{2.5}$ samples were categorized by seasons to investigate variations in the concentrations of NNIs across different seasons. The results of the analysis were shown in Figs. 2 and 3, details are in Table S6. The 101 $PM_{2.5}$ samples were categorized into spring ($n=27$), summer ($n=15$), autumn ($n=33$), and winter ($n=26$) according to the time of collection. The concentration distributions of ACE, DIN, IMI, THM and 5-OH-IMI were statistically significant among the four seasons ($P<0.05$). Concentration levels of IMI and ACE were lower in autumn compared to the other seasons ($P<0.05$). The distributions of NNIs in the four seasons were similar. These compounds primarily consisted of IMI, DIN, 5-OH-IMI, and CLO. IMI emerged as the predominant NNI during the summer and spring, constituting 71% and 47% of the total composition, respectively. DIN, on the other hand, dominated in autumn and winter, accounting for 55% and 46%, respectively.

In general, the seasons with elevated pesticide application are typically spring and summer. Warmer temperatures and higher humidity during these seasons promote the proliferation and dissemination of pests. Additionally, crop growth peaks during this period,

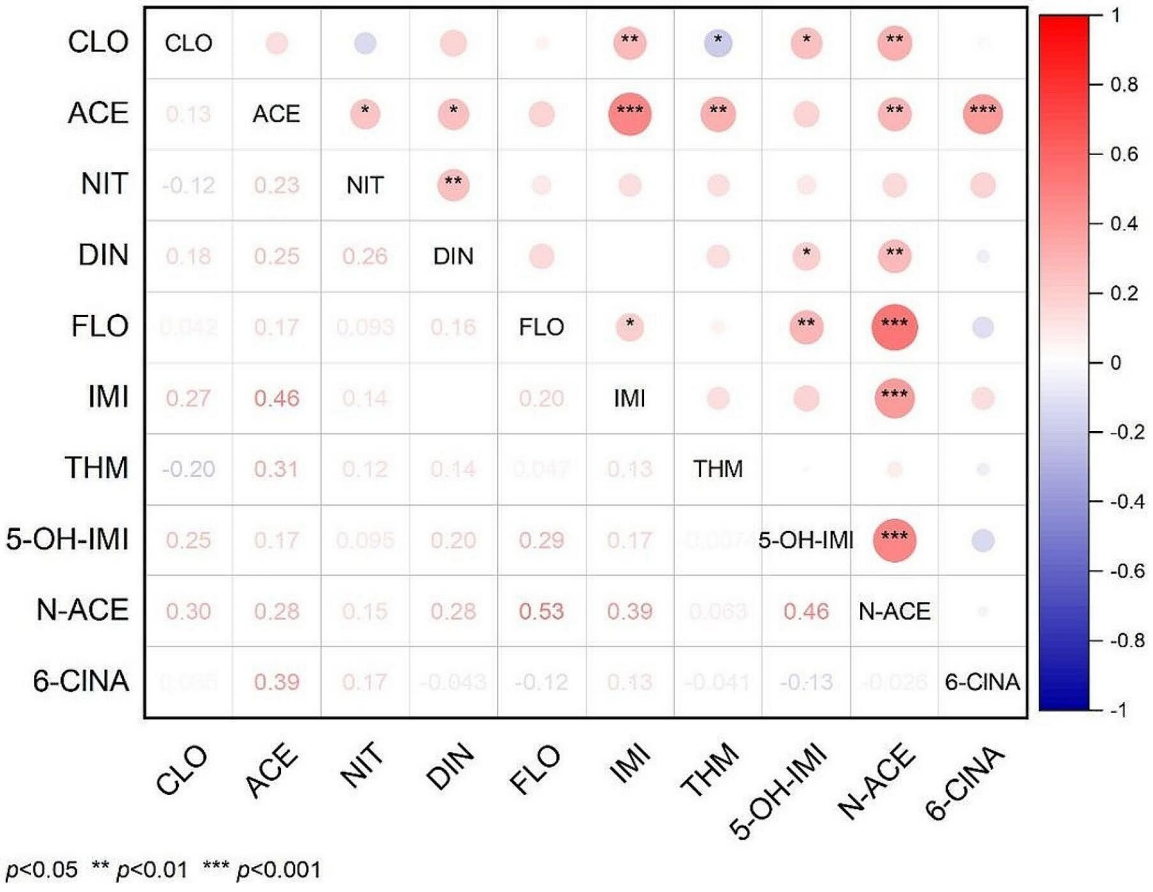

Fig. 1 Correlation analysis between the 10 NNIs detected in the PM2.5 samples

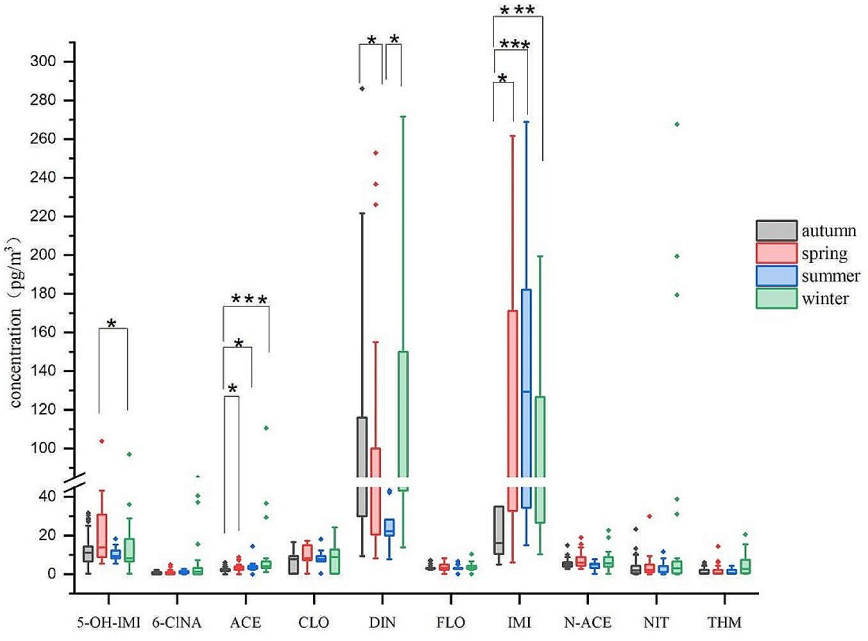

Fig. 2 The concentration distribution and differential analysis of NNIs across different seasons

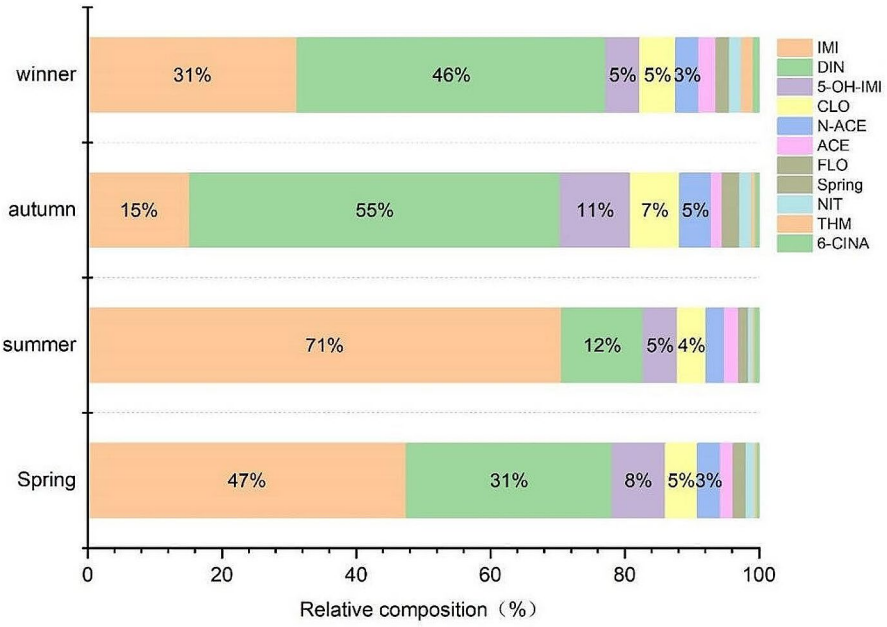

Fig. 3 Distribution of NNIs composition across different seasons

heightening vulnerability to pest attacks. Consequently, the NNIs analyzed in our study consistently exhibited higher concentrations during the spring and summer. However, our findings suggested that there were also some NNIs that had higher levels in the winter, such as NIT (3.06 pg/m^3) and DIN (81.39 pg/m^3). We hypothesize that this variation may be attributed to the relative atmospheric stability near the surface during the fall and winter. During this time, particulate matter in the air tends to accumulate, potentially promoting the formation of aerosols containing organic compounds and other pollutants, or their complete attachment to suspended particulate matter in the atmosphere (Achar et al. 2020). In addition, the distribution characteristics of NNIs may also be controlled by chemical processes and regional sources. NNIs have small molecular weights and strong water solubility, which make them highly mobile in various environmental media (Dores et al. 2008). NNIs have relatively low water solubility because they are polar insecticides. Therefore, NNIs are generally found at low concentrations in the atmosphere while are more abundant in soil and water environments. The water solubility of NNIs are also related to temperature (Yi et al. 2019). We speculate that when the water solubility of NNIs decrease, they may more frequently reach environmental media other than water, such as the atmosphere. Thus, the distribution characteristics of NNIs in atmospheric particulate matter vary under different seasonal conditions and with changes in temperature. In terms of regional distribution, a study of drinking water in eight cities across the country showed that CLO, IMI, and THM are the predominant NNIs in South China, Central China, and Southwest China, while ACE, IMI, and THM are the major NNIs in East China and Northeast China. ACE and IMI are the most significant NNIs in Northwest China and North China (Mahai et al. 2021). Different provinces and regions grow different types of crops, which may result in different types and doses of NNIs being used. During the widespread application of NNIs, only approximately 5% of the active ingredient is absorbed by crops, 90% of the active ingredient enters the soil, and the remainder is dispersed into water and the atmosphere (Giorio et al. 2021; Zhang et al. 2018). Some NNIs in soil are enriched in soil partly by adsorption, partly by desorption into farm water, and migrate to rivers, lakes and wetlands through irrigation leakage. In contrast, NNIs in the atmosphere are produced mainly by particulate matter formed during the sowing of coated seeds and during pesticide spraying. Pesticide spraying, in turn, is related to the type of crop grown. According to the website of the China Pesticide Information Network, NNIs registered in China are mainly used for rice and wheat (National Bureau of Statistics of China. 2022), with relatively few used for other crops, such as corn, fruit trees, and vegetables. As a result, in provinces or regions where rice and wheat are predominantly grown, environmental media tend to have higher levels of NNIs.

The concentration levels of the 10 NNIs in the $PM_{2.5}$ samples in different years

We categorised the collected $PM_{2.5}$ samples according to time of sampling: 2019 ($n=35$), 2020 ($n=27$), 2021 ($n=39$). The results are shown in Fig. 4 and Table S7. The distribution of concentrations of CLO, ACE, IMI and 6-CINA was statistically significant ($P<0.05$) across the three groups. The results after multiple comparisons indicated that IMI concentration levels in 2021 were higher than 2020 and 2019 ($P<0.05$), and ACE showed the same trend. The concentration level of CLO in 2020 are lower than 2019 and 2021 ($P<0.05$). Overall, the concentration levels of most NNIs are lower in 2020 than in 2019 and 2021 with the exception of DIN. In terms of year distribution, the citywide annual average $PM_{2.5}$ concentrations in Wuhan from 2019 to 2021 are 45 μg/m^3, 37 μg/m^3 and 40 μg/m^3, respectively (Wuhan Ecological Environment Bureau. 2022). It is conceivable that variations in atmospheric $PM_{2.5}$ mass concentrations across different years could have influenced this outcome. Elevated levels of $PM_{2.5}$ are likely to lead to the accumulation of greater amounts of NNIs. Furthermore, the outdoor concentrations of NNIs are expected to decrease in 2020 due to reduced time spent by individuals engaging in various daily activities outside their residences.

Inhalation exposure in different populations

Humans can be exposed to NNI through inhalation, potentially leading to health risks. Based on the concentrations obtaining at the monitoring sites, the geometric mean concentrations were used to calculate the EDI for the low-exposure scenario, and the 95th percentile was used to calculate the EDI for the high-exposure scenario. The EDIs for different populations exposed to NNIs are shown in Table S8. Figure 5 showed that ACE, CLO, IMI and DIN are the main NNIs for population exposure. The daily exposure dose per unit body weight to NNIs in the population was much greater in the high-exposure scenario than in the low-exposure scenario. The daily exposure dose per unit body weight for children in both exposure scenarios was 1.3 times that of adult residents and 2.8 times that of adult workers. IMI and DIN had high EDIs in both exposure scenarios.

Humans are exposed to NNIs mainly through diet, drinking water, soil/dust and respiratory intake. Diet and drinking water intake may be the main ways of human exposure to NNIs. However, respiratory intake is also a route that should not be underestimated. NNIs can be sprayed directly on crop surfaces or taken up by plants after application and distributed in tissues and organs, where pollen can be dispersed into the ambient air by windborne vectors or pollinators. A test of NNIs in pollen samples collected from 35 hives in

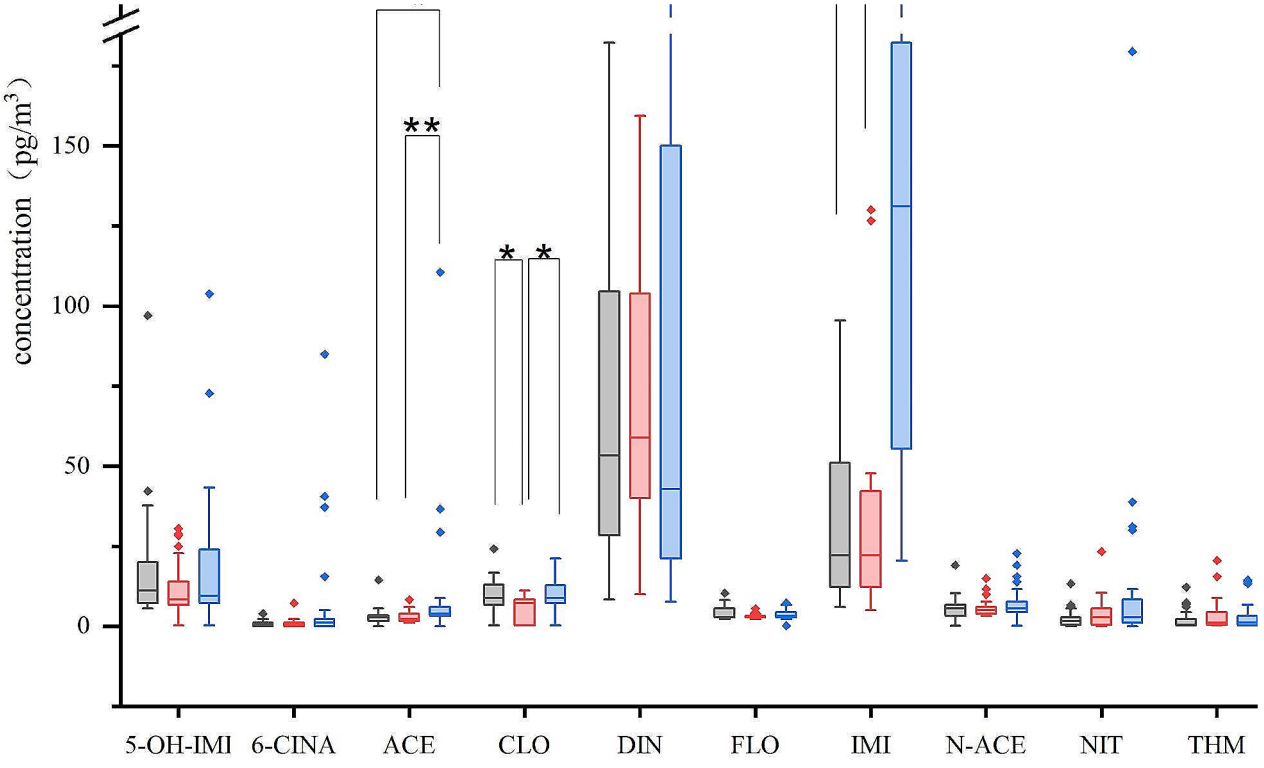

Fig. 4 The concentration distribution and differential analysis of NNIs across different years

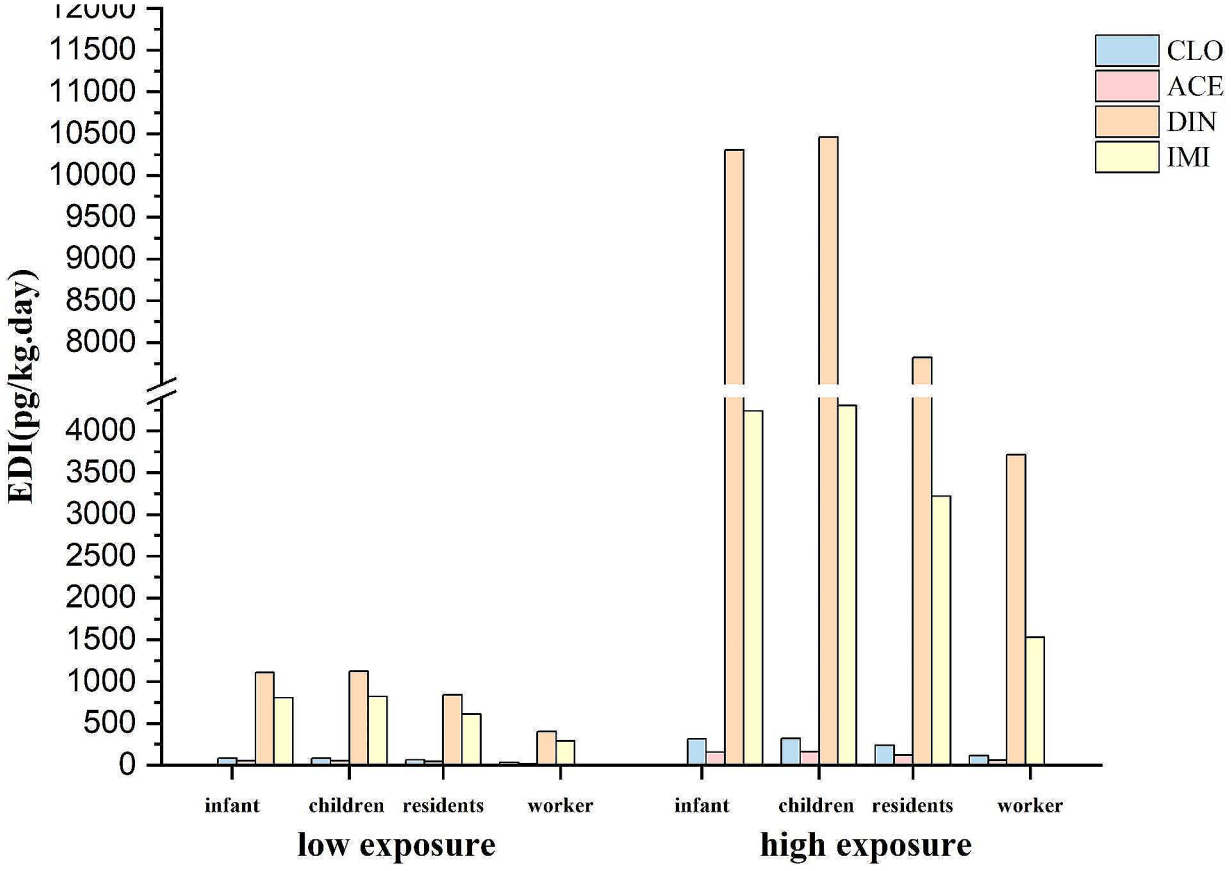

Fig. 5 EDI for different age groups exposed to major NNIs

North America revealed THM in 19 samples, ACE in 11 samples, and IMI in 10 samples (Mullin et al. 2010). NNIs are commonly found in pollen, and airborne pollen can be ingested by humans through respiration, and once inhaled, it may be adsorbed in the human respiratory tract and lungs due to its high water solubility. In Hernandez et al.'s study, respiratory functions were measured and compared between 89 pesticide sprayers and 25 non-spraying control farmers in southeastern Spain. The results suggested a relationship between NNIs application and lung dysfunction (lower total lung capacity, residual volume and functional residual capacity) (Hernández et al. 2008). Therefore, research on the risks of human health from respiratory exposure should be strengthened. Most of the previous studies have focused on the effects of the NNIs on the respiratory health of occupational populations during agricultural activities (Hernández et al. 2008). However, in our study, infants and young children, a sensitive population, were more susceptible to health effects due to NNIs exposure.

Limitations

This study also have several limitations. Firstly, the samples were collected exclusively from the outdoor atmosphere of the Wuhan urban area, leading to a restricted sample size that may not fully represent atmospheric NNIs pollution across the entire city. Moreover, the study only investigated limited NNIs and their metabolites, omitting information on the concentrations of other NNIs. Furthermore, focusing on outdoor samples overlooks the potential high concentrations of NNIs found in indoor dust samples. (Wang et al. 2019). In conclusion, a thorough evaluation of the human health impacts of NNIs necessitates a holistic assessment across various environmental mediums, with particular emphasis on the analysis of drinking water quality.

Conclusion

In this study, we measured the concentrations of NNIs in 101 outdoor $PM_{2.5}$ sources in urban areas of Wuhan from 2019 to 2021, analysed their distributions in different seasons, and evaluated the daily exposures of the population. Although the volatility of the NNI is low, its concentration in the atmosphere is relatively

high. With the widespread use of NNIs, population inhalation health risks may increase. In addition, the risk of exposure to NNIs is particularly high for sensitive populations such as infants and young children. In conclusion, sustained monitoring of the risk of NNIs for air is essential.

Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1186/s40068-024-00342-x.

Supplementary Material 1

Acknowledgements
This work was supported by the Contract grant sponsor: Science and Technology Program of Shenzhen, China (Contract grant number: JCYJ20180301170716407).

Author contributions
Du was responsible for the main manuscript text and the completion of the experiments, Zhu was involved in the production of the figures and tables, Yao was responsible for the planning of the experiments and the preliminary research, and all authors reviewed the manuscript.

Funding
This work was supported by the Funding source: Shenzhen Municipal Science and Technology Innovation Commission (Grant number: JCYJ20230807152459009).

Data availability
No datasets were generated or analysed during the current study.

Declarations

Competing interests
The authors declare no competing interests.

Received: 15 March 2024 / Accepted: 17 April 2024
Published online: 28 May 2024

References

Achar JC, Nam G, Jung J, Klammler H, Mohamed MM (2020) Microbubble Ozonation of the antioxidant butylated hydroxytoluene: degradation kinetics and toxicity reduction. Environ Res 186:109496
Bass C, Denholm I, Williamson MS, Nauen R (2015) The global status of Insect Resistance to Neonicotinoid Insecticides. Pestic Biochem Physiol 121:78–87
Casida JE (2011) Neonicotinoid metabolism: compounds, substituents, pathways, enzymes, organisms, and relevance. J Agric Food Chem 59(7):2923–2931
Chang C-H, Macintosh D, Lemos B, Zhang Q, Lu C (2018) Characterization of Daily Dietary Intake and the Health risk of Neonicotinoid insecticides for the U.S. Population. J Agric Food Chem 66(38):10097–10105
Chang Chi-Hsuan D, MacIntosh B, Lemos Q, Zhang, Lu C (2018) Characterization of Daily Dietary Intake and the Health risk of Neonicotinoid insecticides for the U.S. Population. J Agric Food Chem 66(38):10097–10105
Chen D, Zhang Y, Lv B, Liu Z, Han J, Li J, Zhao Y, Wu Y (2020) Dietary exposure to Neonicotinoid Insecticides and Health risks in the Chinese General Population through two consecutive total Diet studies. Environ Int 135:105399
Christen V, Mittner F, Fent K (2016) Molecular effects of neonicotinoids in Honey bees (Apis Mellifera). Environ Sci Technol 50(7):4071–4081
Coscollà C, Yusà V, P Martí, and, Pastor A (2008) Analysis of currently used pesticides in fine Airborne Particulate Matter (PM 2.5) by pressurized liquid extraction and liquid chromatography–Tandem Mass Spectrometry. J Chromatogr A 1200(2):100–107
Cox-Foster DL, Conlan S, Holmes EC, Palacios G, Evans JD, Moran NA, Quan P-L, Briese T, Hornig M, Geiser DM, Martinson V, vanEngelsdorp D, Kalkstein AL, Drysdale A, Hui J, Zhai J, Cui L, Hutchison SK et al (2007) A Metagenomic Survey of microbes in Honey Bee colony collapse disorder, vol 318. Science, pp 283–287. (New York, N.Y.)5848
Dores EFGC, Carbo L, Ribeiro ML, De-Lamonica-Freire EM (2008) Pesticide levels in Ground and Surface Waters of Primavera do Leste Region, Mato Grosso, Brazil. J Chromatogr Sci 46(7):585–590
Giorio C, Safer A, Sánchez-Bayo F, Tapparo A, Lentola A, Girolami V, van Lexmond MB, Bonmatin J-M (2021) An update of the Worldwide Integrated Assessment (WIA) on systemic insecticides. Part 1: New molecules, Metabolism, Fate, and transport. Environ Sci Pollut Res Int 28(10):11716–11748
Han W, Tian Y, Shen X (2018) Human exposure to Neonicotinoid Insecticides and the evaluation of their potential toxicity: an overview. Chemosphere 192:59–65
Hernández AF, Casado I, Pena G, Gil F, Villanueva E, Pla A (2008) Low level of exposure to Pesticides leads to lung dysfunction in occupationally exposed subjects. Inhalation Toxicol 20(9):839–849
Li Y, Miao R, Khanna M (2020) Neonicotinoids and decline in Bird Biodiversity in the United States. Nat Sustain 3(12):1027–1035
Lu C, Chang C-H, Palmer C, Zhao M, Zhang Q (2018) Neonicotinoid residues in fruits and vegetables: an Integrated Dietary exposure Assessment Approach. Environ Sci Technol 52(5):3175–3184
Mahai G, Wan Y, Xia W, Wang A, Shi L, Qian X, He Z, Xu S (2021) A nationwide study of occurrence and exposure Assessment of Neonicotinoid insecticides and their metabolites in drinking Water of China. Water Res 189:116630
Marfo JT, Fujioka K, Ikenaka Y, Nakayama SMM, Mizukawa H, Aoyama Y, Ishizuka M, Taira K (2015) Relationship between urinary N-Desmethyl-acetamiprid and typical symptoms including neurological findings: a prevalence case-control study. PLoS ONE 10(11):e0142172
Matsuda K, Buckingham SD, Kleier D, Rauh JJ, Grauso M, Sattelle DB (2001) Neonicotinoids: insecticides acting on Insect Nicotinic Acetylcholine receptors. Trends Pharmacol Sci 22(11):573–580
Montiel-León JM, Munoz G, Vo Duy S, Do DT, Vaudreuil M-A, Goeury K, Guillemette F, Amyot M, Sauvé S (2019) Widespread occurrence and spatial distribution of glyphosate, atrazine, and Neonicotinoids pesticides in the St. Lawrence and Tributary Rivers. Environ Pollution (Barking Essex: 1987) 250:29–39
Mullin CA, Frazier M, Frazier JL, Ashcraft S, Simonds R, D vanEngelsdorp, and, Pettis JS (2010) High levels of Miticides and agrochemicals in North American Apiaries: implications for Honey Bee Health ed. Frederic Marion-Poll. PLoS ONE 5(3):e9754
National Bureau of Statistics of China (2022) China Statistical Yearbook 2022. https://www.stats.gov.cn/sj/ndsj/2022/indexeh.htm
Shao X, Liu Z, Xu X, Li Z, Qian X (2013) Overall status of Neonicotinoid Insecticides in China: production, Application and Innovation. J Pesticide Sci 38(1):1–9
Simon-Delso N, Amaral-Rogers V, Belzunces L, Jm Bonmatin (2015). Systemic Insecticides (Neonicotinoids and Fipronil): Trends, Uses, Mode of Action and Metabolites. Environmental science and pollution research international 22(1)
Suchail S, Debrauwer L, Belzunces LP (2004) Metabolism of Imidacloprid in Apis Mellifera. Pest Manag Sci 60(3):291–296
Taira K, Fujioka K, Aoyama Y (2013) Qualitative profiling and quantification of Neonicotinoid metabolites in Human urine by Liquid Chromatography coupled with Mass Spectrometry. PLoS ONE 8(11)
U.S. Environmental Protection Agency.1992. O. (2014) Guidelines for Exposure Assessment. Guidelines for exposure assessment. https://www.epa.gov/risk/guidelines-exposure-assessment
Valavanidis A, Fiotakis K, Vlachogianni T (2008) Airborne Particulate Matter and Human Health: Toxicological Assessment and Importance of size and composition of particles for oxidative damage and carcinogenic mechanisms. J Environ Sci Health C 26(4):339–362
Vijver MG, van den Brink PJ (2014) Macro-invertebrate decline in Surface Water Polluted with Imidacloprid: a rebuttal and some new analyses. PLoS ONE 9(2):e89837
Wang A, Mahai G, Wan Y, Jiang Y, Meng Q, Xia W, He Z, Xu S (2019) Neonicotinoids and Carbendazim in indoor dust from three cities in China: spatial and temporal variations. Sci Total Environ 695:133790
Wuhan Ecological Environment Bureau Environmental Data. http://hbj.wuhan.gov.cn/hjsj/
Xiao X (2003) Uncertainties in estimates of Cropland Area in China: a comparison between an AVHRR-Derived dataset and a landsat TM-Derived dataset. Glob Planet Change 37:297–306
Xu M, Zhang Z, Li Z, Kan S, Liu Z, Wang D, Liu Q, Zhang H (2021) Profiles of Neonicotinoid insecticides and characteristic metabolites in paired urine and blood

samples: partitioning between urine and blood and implications for human exposure. Sci Total Environ 773:145582
Yi Y, Zhang C, Liu H, Wu R, Tian D, Ruan J, Zhang T, Huang M, Ying G (2019) Occurrence and distribution of Neonicotinoid Insecticides in Surface Water and Sediment of the Guangzhou section of the Pearl River, South China. Environ Pollution (Barking Essex: 1987) 251:892–900
Zhang P, Ren C, Sun H, Min L (2018) Sorption, Desorption and Degradation of Neonicotinoids in Four Agricultural soils and their effects on Soil microorganisms. Sci Total Environ 615:59–69
Zhang S, Huo X, Zhang Y, Huang Y, Zheng X, Xu X (2019) Ambient fine particulate matter inhibits Innate Airway Antimicrobial Activity in Preschool Children in E-Waste Areas. Environ Int 123:535–542
Zhou Y, Guo J, Wang Z, Zhang B, Sun Z, Yun X, Zhang J (2020) Levels and Inhalation Health risk of Neonicotinoid insecticides in Fine Particulate Matter ($PM_{2.5}$) in Urban and Rural areas of China. Environ Int 142:105822

Publisher's Note