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Effect of air quality improvement by urban parks on mitigating $PM_{2.5}$ and its associated heavy metals: A mobile-monitoring field study

Tzu-Hao Su [a], Chin-Sheng Lin [b], Shiang-Yue Lu [a], Jiunn-Cheng Lin [a], Hsiang-Hua Wang [a], Chiung-Pin Liu [c,*]

[a] Taiwan Forestry Research Institute, Taipei, 100051, Taiwan
[b] Agricultural Engineering Research Center, Zhongli 320, Taiwan
[c] Department of Forestry, National Chung Hsing University, 145, Xingda Rd., South Dist., Taichung, 402202, Taiwan

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ABSTRACT

Field mobile monitoring of $PM_{2.5}$, equipped with a highly accurate device, was performed for two types of urban parks in Taiwan. Measurements were taken in the morning and evening rush hours, on certain weekdays and weekends, every month over a year. We designed six calculation schemes of the rate of $PM_{2.5}$ mitigation by urban parks to comprehensively compare the average and maximum mitigation effects in relation to the vegetation barriers. The mitigation rate, from the lowest (2.51%) to the highest (35.57%) depended on the calculation schemes. The Taipei Botanical Garden (TBG) with a dense, multilevel forest has a stable $PM_{2.5}$ mitigation effect and strong ability to improve air quality inside the park under severe $PM_{2.5}$ pollution. In contrast, Zhonghe No.4 Park (ZHP), an open park with mostly a single-storied stand, has variable $PM_{2.5}$ mitigation effect, leading to either quick dissipation or accumulation of $PM_{2.5}$ inside the park. Furthermore, the dry deposition of PM and the associated heavy metals were investigated using camphor trees as bioaccumulators. Dry deposition flux of the leaf-deposited $PM_{2.5}$ exhibited similar results in ZHP; whereas, noticeable higher results were observed inside TBG. In addition, most of the $PM_{2.5}$ deposition flux from field estimations were similar to those in *i-Tree Eco* when considering the loss of mass due to the dissolution through water filtration, indicating that *i-Tree Eco* may be reliable to model the removal of $PM_{2.5}$ in the parks in Taiwan. Moreover, we examined nine heavy metals' content in the deposited PM, and most of the detectable elements were significantly higher outside both parks, demonstrating the mitigation effects of urban parks in reducing not only the $PM_{2.5}$ concentration but also the toxicity of the pollutant. This study provides direct evidence of the important ecosystem services, namely air quality improvement and biomonitoring effect, derived from urban parks.

1. Introduction

Air pollutants, especially fine particulate matter ($PM_{2.5}$), have become a global concern as they adversely affect human health, causing serious respiratory and lung diseases (Schlesinger, 2007), premature mortality (Maji et al., 2018), and hospital admissions for cardiovascular diseases (Amsalu et al., 2019). $PM_{2.5}$ also led to the induction and exacerbation of diabetes, inflammation and immunity disorder, and mutagenicity and genotoxicity (Feng et al., 2016). Heavy metals are among the major toxic components of $PM_{2.5}$ and are regarded responsible for health risks (Wang et al., 2018; Zheng et al., 2016). In East Asian cities, vehicle exhaust originating from fossil fuel combustion is an important source of $PM_{2.5}$ (Giang and Oanh, 2014; Liang et al., 2013; Liu et al., 2017) and contains heavy metals such as lead (Jeong et al., 2017). Vegetation is commonly believed to reduce ambient $PM_{2.5}$ via adsorption on vegetation surfaces (Han et al., 2020). However, the capacity of trees to capture $PM_{2.5}$ can vary in different leaf traits (Leonard et al., 2016; Zhang et al., 2020a, 2021) and tree species (Chen et al., 2016; Sæbø et al., 2012). In addition, the chances of particles depositing onto trees, and thus, being removed from the atmosphere are greater when the leaf area index (LAI) or leaf area density (LAD) is large (Janhäll, 2015). The PM deposition effect was determined by many field experiment studies using the wet extraction method (Hofman et al., 2014; Sgrigna et al., 2015; Xu et al., 2018). Moreover, *i-Tree Eco* is now the most commonly used model for PM deposition in urban forestry (Lin et al., 2019). Many studies used this model to evaluate ecosystem

* Corresponding author.
E-mail address: cpliu@nchu.edu.tw (C.-P. Liu).

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services, including removal of air pollutants, for urban forests (de Jalón et al., 2019; Riley et al., 2018; Selmi et al., 2016; Wu et al., 2019). This model has also been used recently to clarify the ecosystem services of urban parks (Fares et al., 2020; Pace et al., 2021).

However, trees are considered to mitigate turbulent flow and pollutant dispersion, and the ambient $PM_{2.5}$ concentration may be dominated by the aerodynamics effect instead of the deposition effect (Jeanjean et al., 2016). For example, studies have shown that in street canyons, trees can lead to an increased PM concentration at the leeward wall under perpendicular wind, and this phenomenon is usually described as the trapping effect (Buccolieri et al., 2018; Gromke and Ruck, 2007; Jeanjean et al., 2017). The results can also vary remarkably depending on the tree species, planting composition, vegetation density or porosity, and wind speed and direction (Han et al., 2020; Janhäll, 2015). Furthermore, *i-Tree Eco*, based on the wind tunnel parameters, can calculate the $PM_{2.5}$ removal by the urban forests by regarding them as a big leaf, and then, estimate the concentration reduction over the whole boundary layer. For example, Nowak et al. (2013) indicated that across 10 major U.S. cities, the average annual reduction in hourly $PM_{2.5}$ concentration was 0.006–0.030 $\mu g\ m^{-3}$; however, this reduction corresponded to only 0.05%–0.24% of the average improvement of air quality. It is noteworthy that although tree leaves can be good PM bioaccumulators, the mitigation of ambient PM concentration may be slight when only considering the deposition effect. Thus, when determining the effect of urban trees in influencing the ambient $PM_{2.5}$ levels, in addition to the PM removal by vegetation, the actual impact on ambient PM concentration should be examined in the urban environment, especially at the street and/or neighborhood scales.

Nevertheless, vegetation is considered as a barrier that can improve local air quality away from the pollution source by altering dispersion and capturing air pollutants. This positive effect that reduces ambient air pollutants downwind of the barriers, which is also considered as air purification effect, is well-documented (Chen et al., 2016; Hagler et al., 2012; Jia et al., 2021; Tong et al., 2016). However, the purification results can vary greatly and regionally. Lin et al. (2016) found a reduction of 37.7%–63.6% for ultrafine particle (UFP; aerodynamic diameter $\leq$0.1 μm) number concentrations by vegetation barriers, but no significant reduction in UFP concentration during periods without foliage. In contrast, negative pollutant reduction effects were observed in calm wind conditions (<0.6 $m\ s^{-1}$) (Ranasinghe et al., 2019). An urban park, with ample vegetation, is frequently visited by the local residents, who expect better air quality inside. However, it is difficult to quantify the real air purification effect of urban parks at a neighborhood scale due to the uneven distribution of micrometeorological conditions or the calm wind conditions in the park area. Thus, related studies generally discussed the air quality improvement at different points of the park to reflect how much the park can purify the air. For example, Silli et al. (2015) found only 2.56% of average $PM_{2.5}$ difference between the vegetated and non-vegetated sites in an urban park (160 ha). Moreover, Gómez-Moreno et al. (2019) selected two monitoring sites inside and outside the park (125 ha), and the result showed that no obvious discrepancies occurred for the $PM_{2.5}$ or PM_1 concentrations. In comparison, a larger area could lead to more obvious PM concentration differences between different sites in the park. For example, Zhu et al. (2019) found a ~48% difference of $PM_{2.5}$ in an urban park (713 ha) located in Wuhan. Furthermore, Grzędzicka (2019) also found that concentrations of $PM_{2.5}$ decreased ~22% for a distance to the road from 100 to 1,000 m (640 ha). However, the air quality at a few points in the park is difficult to represent the situation of the entire urban park. In addition, the large parks in the aforementioned studies are very rare in the prosperous cities in Taiwan. Thus, proper evaluations of the $PM_{2.5}$ mitigation by small urban parks (~10 ha) are necessary.

Because of the complexity of the real situation, the model result of $PM_{2.5}$ for the urban park may lack reliability. The computational fluid dynamic (CFD) model such as ENVI-met can provide $PM_{2.5}$ spatial distributions of the small parks, but the validation still requires field observations (Xing and Brimblecombe, 2020). Furthermore, considering the cost limitation, high-accuracy $PM_{2.5}$ monitoring devices cannot be deployed in many places simultaneously. In comparison, low-cost $PM_{2.5}$ monitoring sensors have been widely used in urban environments (Badura et al., 2020; Brzozowski et al., 2020; Cao et al., 2020). However, the influence of humidity could still be a major problem for low-cost sensors causing overestimations of the particle mass concentration (Jayaratne et al., 2018; Kosmopoulos et al., 2020). Hence, mobile monitoring offers a convenient method of obtaining information about the PM mitigation by urban parks with a high spatial resolution. Many studies use this technique to determine the air quality levels at pedestrian height along the road (Cummings et al., 2021; Shakya et al., 2019; Steinle et al., 2015; SM et al., 2019). In particular, mobile monitoring, equipped with a highly accurate $PM_{2.5}$ monitoring device, combines the advantages of obtaining reliable monitoring values and high spatial resolution. Moreover, when investigating the $PM_{2.5}$ levels in the small urban parks, the discrepancy at different sites may be slight, leading to a requirement of devices with greater accuracy.

In the present study, we performed field measurements in urban parks in Taipei and New Taipei City, Taiwan. Two types of parks, one with dense vegetation and the other regarded an open park, were selected. A reliable mobile-monitoring device was used to measure the $PM_{2.5}$ concentration inside and outside the parks at well-distributed points every month over a year. The monitoring was performed during the morning and evening rush hours on both weekdays and weekends to obtain representative information. Six $PM_{2.5}$ mitigation calculation schemes were used to estimate the effect of air quality improvement in the parks by calculating the discrepancy between the unmitigated outside points and mitigated inside or geographic center points. Furthermore, we used leaves from the camphor trees as bioaccumulators to collect $PM_{2.5}$ and total suspended particulates (TSP) inside and outside the parks, and investigated the associated heavy metals/trace elements (Fe, Mn, Cu, Zn, Ni, Cr, Pb, Cd, and Al) under a dry deposition period of ~10 days in seasons of greater air pollution. We also calculated the dry deposition flux of the leaf-deposited $PM_{2.5}$ to compare with the simulations of *i-Tree Eco* and clarified the model applicability in the research parks. The study aims to provide a comprehensive evaluation and quantification of the PM mitigation effect of the two types of urban parks and can be divided into the following four research topics: (1) evaluate the $PM_{2.5}$ concentration mitigation from outside to inside the parks; (2) determine the difference of PM-associated heavy-metal contents between the internal and external sites of the parks; (3) calculate the high resolution of $PM_{2.5}$ spatial distribution in the parks; (4) estimate the $PM_{2.5}$ dry deposition effect inside and outside the parks and compare the $PM_{2.5}$ dry deposition flux on leaf surfaces from the field and model results.

2. Materials and methods

2.1. Study area and urban parks

The study was conducted at two sites in Taipei and New Taipei City, located in the northern part of Taiwan. These two cities, separated by Xindian River, are most prosperous cities in Taiwan. Taipei City is the capital of Taiwan, and New Taipei City is the most populous city with a population of more than 4 million. We selected two types of urban parks as the $PM_{2.5}$ monitoring sites (Fig. 1). The Taipei Botanical Garden (TBG), established in 1921, is spread over an area of 8.2 ha (25°01′56″ N, 121°30′35″ E) and houses more than 1,500 plant species. TBG is important for academic research and natural sciences education and also has great recreational value as an urban park. Zhonghe No.4 Park (ZHP) with an area of 10.8 ha is the largest and major park in the local district (25°00′04″ N, 121°30′48″ E). ZHP provides an essential space for social and recreational activities for the residents. Vegetation in both parks is mostly evergreen; however, the forest structures in these parks are quite different. The canopy coverage in TBG and ZHP is ~90% and ~55%,

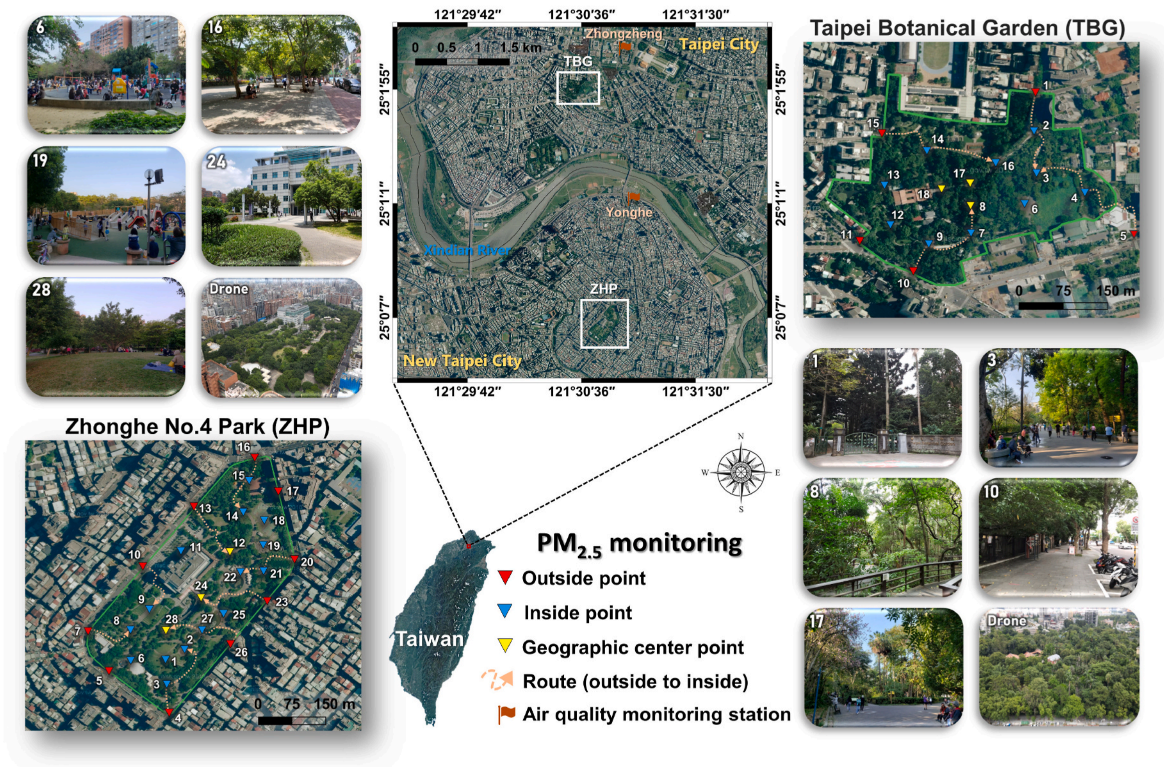

Fig. 1. Locations of the study area, monitoring parks, and sampling points.

respectively. More than 80% of the tree covered area in TBG constitutes of multilevel forest. In contrast, less than only 10% is multilevel in ZHP. The vegetation barriers in TBG are obvious and the inside and outside of the park can be easily distinguished (see point 10, Fig. 1), whereas large gaps in vegetation exist in the ZHP (see point 16, Fig. 1). Especially at the height of breathing zone (1.5 m), the vegetation canopy in TBG entirely blocks the inner view of the park from the outside (except for the entrances). In contrast, no vegetation canopy exists at breathing zone in ZHP, and thus, we can directly see the interior of the park from the outside. Both parks are located in districts with population densities greater than 20,000 per km^2, and the size of the selected parks are fairly common in Taiwan. Thus, it is important to validate the existence of the $PM_{2.5}$ mitigation effect of these parks as valuable information for the residents. Furthermore, the traffic flow of the main road near the two parks was similar based on the open data from Taipei and New Taipei City Bureau of Transport (weekday in 2019). The traffic flow of the road next to outside-point 10 in TBG was 938 and 1160 pcu h^{-1} in the morning (7:30–8:30) and evening (17:30–18:30), respectively. Moreover, the traffic flow of the road next to outside-point 16 in ZHP was 1040 and 749 pcu h^{-1} in the morning (7:30–8:30) and evening (17:30–18:30), respectively. In this study, the ambient $PM_{2.5}$ concentrations throughout the year were monitored to clarify the air quality around the parks.

2.2. $PM_{2.5}$ concentration monitoring

$PM_{2.5}$ concentrations were monitored at a height of 1.5 m above the ground (breathing zone) at constant points using pDR-1500 Aerosol Monitor devices (Thermo Fisher Scientific, Inc, Rockford, IL, USA) equipped with a $PM_{2.5}$ cyclone separator (ACGIH-traceable) and a built-in pump and with a resolution of 0.1 μg m^{-3} and measurement precision of 0.5 μg m^{-3} for 60-s averaging time. The pDR-1500 is a fully integrated sampling instrument incorporating a temperature and relative humidity (RH) sensor, which can mitigate the positive bias with increased ambient RH. Moreover, a barometric pressure sensor inside the device enables the flow control to provide accurate volumetric data. The device was calibrated by the manufacturer with a gravimetric standard traceable to the National Institute of Standards and Testing. It has been tested to meet the criterion of the United States Environmental Protection Agency (US EPA) for many kinds of aerosols (Sousan et al., 2016). After monitoring in an optical dust chamber, the particles are loaded on the instrument filter, thus enabling self-recalibration. The direct reading measurements are known to match well the filter gravimetric measurements (Wang et al., 2016). We selected 18 and 28 monitoring points for TBG and ZHP, respectively, and the order of $PM_{2.5}$ monitoring followed the order of point numbers. The monitoring time was at least 1 min (read per second) for each point to obtain the average value, but could be longer until the monitoring data were stable. We ensured that the overall walking monitoring time for each park was not more than 1 h to avoid significant changes in environmental conditions. Meanwhile, the design of the monitoring was such that measurements were made while moving inside and outside the park continuously in a short time. This design helped avoid the influence of changes in background conditions on the determined $PM_{2.5}$ mitigation effect. Field monitoring was performed simultaneously for both parks every month from April 2020 to March 2021, and two weekdays and one day on the weekends were selected in a week without rainfall in each month. For each day, monitoring was performed twice during rush hours (7–8 am and 5–6 pm). Each monitoring period consisted of ~45 min (for TBG) to 1 h (for ZHP). To summarize, $PM_{2.5}$ monitoring was conducted simultaneously in both parks 72 times (2 periods × 3 days × 12 months). The monitoring times were denoted as DM and DE for weekday mornings and evenings, respectively, and EM and EE for weekend mornings and evenings, respectively. As many people exercise in the parks during these periods, the associated air qualities are extremely important. We counted the number of visitors around each point during monitoring to identify the hotspots for the crowd. It is worth noting that, owing to the effective epidemic prevention and control measures by the government, Taiwan was not significantly affected by Coronavirus disease (COVID-19) pandemic during our monitoring period.

2.3. Air quality monitoring stations

Considering the limited frequency of field measurement, the representativeness of monitoring data from air quality stations with regard to the real situation in the parks is extremely important. For each park, we compared the $PM_{2.5}$ concentration data from the neighboring air quality station with the average field result of all points in the same periods. The Zhongzheng Station near the TBG performed measurements at the top of a building (14.6 m above the ground) and was 10 m away from the road. The Yonghe Station close to ZHP monitored $PM_{2.5}$ at a height of 3–4 m next to the road (Fig. 1). After determining the linear correlations between field measurements and monitoring data from air quality stations for each park (excluding 10% outliers), the continuous monitoring data from stations were converted into field results. Thus, the representative temporal distribution of $PM_{2.5}$ concentration in these parks was verified. In addition, the air quality monitoring stations provided hourly wind speed and wind direction information during our monitoring periods for further evaluation.

2.4. Determination of PM deposited on leaf surfaces

To estimate the $PM_{2.5}$ dry deposition flux and the associated heavy-metal content around the parks, we selected the leaves of camphor trees (*Cinnamomum camphora*) as natural deposition surfaces for $PM_{2.5}$. Leaves were collected from outside and inside the TBG and ZHP (outside-point 10 and inside-point 3 for the TBG; outside-point 23 and inside-point 24 for the ZHP). The sampling date was March 17, 2021, and the dry deposition period was 245 h since the last daily rainfall of both parks (over 35 mm). The sampling date was selected because a heavy $PM_{2.5}$ episode was recorded by mobile monitoring in the evening on March 15, 2021 ($PM_{2.5}$ at most points in both parks were higher than 100 μg m^{-3}). Approximately 80–120 healthy leaves were taken as a leaf sample. Four replicate samples were collected randomly from the tree at a height of 3–5 m at each point. Leaf samples were carefully placed in plastic bags to avoid contamination, transported to the laboratory immediately, and stored at 4 °C until further analysis. The deposited particles from each leaf sample were washed off with 300 mL ultrapure water and passed through a 100 μm stainless steel sieve to remove particles larger than 100 μm. Then, 10 mL liquid samples were subjected to heavy-metal analyses (as metals in TSP or PM_{100}). Subsequently, the remaining filtered samples were passed through 2.5 μm pore size cellulose filter (type 42, Whatman Corporation, Piscataway, NJ, USA) to remove particles larger than 2.5 μm, followed by 0.45 μm pore size mixed cellulose ester filter (GN-6, Pall Corporation, East Hills, NY, USA). The $PM_{2.5}$ (diameters: 0.45–2.5 μm in this study) are intercepted by the last filter, and the TSP (diameters: 0.45–100 μm in this study) are intercepted by both filters. The area of the washed leaf surfaces can be obtained from scanned images of leaves using ImageJ software (National Institutes of Health, Bethesda, MD, USA). More PM water filtration details are available in our previous report (Su et al., 2021).

2.5. PM-associated heavy-metal analyses

The aforementioned $PM_{2.5}$ filter sample and TSP liquid sample were subjected to heavy-metal analyses by the aqua regia digestion method of Taiwan EPA (NIEA A305.10C). The filter was put into a digestion flask with 10 mL diluted aqua regia (55.5 mL HNO_3 and 167.5 mL HCl were added to 500 mL ultrapure water, and a volume of up to 1 L was made up with ultrapure water) and then heated on a hotplate at 180°C–200 °C for at least 2 h. After the filter was completely dissolved, the digestion solution (2–3 mL left due to volatilization) was made to the final volume of 20 mL with ultrapure water. The TSP liquid sample was analyzed using the same method as $PM_{2.5}$ except that the aqua regia was undiluted. The heavy-metal content of all the aforementioned digested samples were tested by inductively coupled plasma–optical emission spectrometry (ICP-OES; Profile Plus, Leeman Labs, Lowell, MA, USA). We determined heavy metals and trace elements, including Fe, Mn, Cu, Zn, Ni, Cr, Pb, Cd, and Al. Notably, because of the limitation of the water filtration method; the heavy metals from leaf-deposited $PM_{2.5}$ were specifically related to the insoluble $PM_{2.5}$ components.

2.6. $PM_{2.5}$ dry deposition flux

After determining the leaf-deposited $PM_{2.5}$ mass and the leaf area, the $PM_{2.5}$ deposition flux can be obtained by dividing the mass by the area and the deposition period. Moreover, the $PM_{2.5}$ deposition flux from the model (*i-Tree Eco*) results (*Model F*) can be calculated using Eq. (1).

$$Model\ F = \overline{i - Tree\ Eco\ Vd\ (hourly) \times C\ (hourly)} \quad (1)$$

where F is the dry deposition flux of $PM_{2.5}$ (μg m^{-2} h^{-1}), C (hourly) is the hourly ambient $PM_{2.5}$ concentration from the neighboring station (μg m^{-3}), and $i - Tree\ Eco\ Vd$ (hourly) is the $PM_{2.5}$ deposition velocity from hourly parameters in *i-Tree Eco* (m h^{-1}). According to Hirabayashi et al. (2015), for example, the parameters of the mean dry deposition velocity of $PM_{2.5}$ on the leaf surfaces used in *i-Tree Eco* under wind speeds of 1, 2, and 3 m s^{-1} were 0.03, 0.09, and 0.15 cm s^{-1}, respectively (corresponded to 1.5%, 3%, and 4.5% of particle resuspension). Thus, the hourly wind speed data from each air quality monitoring station were converted into hourly $PM_{2.5}$ dry deposition velocity (using linear interpolation). Afterwards, the hourly simulated deposition flux can be calculated by multiplying $i - Tree\ Eco\ Vd$ (hourly) and C (hourly). Finally, the average result of all the hourly flux simulations in the dry deposition period (*Model F*) was then compared to the field results for our monitoring parks.

Table 1
Descriptions of the calculation schemes of $PM_{2.5}$ mitigation rate.

Calculation scheme	Equation	Implication
A	$\frac{\text{outside mean} - \text{inside mean}}{\text{outside mean}}$	The mitigation effect of the entire park
B	$\frac{\text{outside mean} - \text{center mean}}{\text{outside mean}}$	The mitigation effect at the geographic center of the park
C	$\frac{\text{outside max} - \text{inside mean}}{\text{outside max}}$	The mitigation effect of the entire park to pollution hotspot
D	$\frac{\text{outside max} - \text{inside min}}{\text{outside max}}$	The maximum mitigation effect contributed by the park
E	Route $\left(\frac{\text{outside} - \text{inside}}{\text{outside}}\right)$ mean	The average mitigation effect by entering the park directly
F	Route $\left(\frac{\text{outside} - \text{inside}}{\text{outside}}\right)$ max	The maximum mitigation effect by entering the park directly

(Note: The routes passing through inside and outside in Schemes E and F indicate the head and tail of the arrows, as represented in Fig. 1.).

2.7. Data analyses

We evaluated the $PM_{2.5}$ mitigation effects of urban parks using different calculation schemes (Table 1). These schemes present different viewpoints regarding the mitigation effect. For example, Schemes A and B provide the average effect of $PM_{2.5}$ concentration reduction, whereas Schemes C and D focus on the extreme difference between the internal and external parts of the park. Schemes E and F were used because they represent the rapid mitigation of $PM_{2.5}$ when entering the park. The adjacent monitoring order in Schemes E and F also helps avoid the effect of changes in the background condition. When compared to that of Grzędzicka (2019), who spent 7.5–8 h collecting PM concentrations with mobile monitoring at 40 points in a large park, the monitoring time interval between the two sampling points (head and tail of the arrow in Fig. 1) in Schemes E and F is ~5–7 min. Hence, the estimated mitigation rate may be accurate. All schemes were calculated for each monitoring time, and the $PM_{2.5}$ mitigation rate of each scheme for different time scales were determined for the middle 90% data. In addition, point 19 in ZHP was excluded from mitigation calculation because a sandpit was located there, causing unpredictable interference. To clarify the $PM_{2.5}$ mitigation effect of the parks, we classified the average $PM_{2.5}$ field observations inside and outside the park into five categories: 0–10, 10–20, 20–30, 30–50, and 80–120 μg m^{-3}. We examined the reduction percentage of ambient concentration and that of concentration itself among different schemes and parks under these categories. In addition, linear regression between field measurements and monitoring data from air quality stations was conducted, as mentioned in Section 2.3. Thus, the original and converted $PM_{2.5}$ temporal distribution over the monitoring year can be described. We also used the Spearman correlation analysis to verify the relationship between the field observations of $PM_{2.5}$ and the hourly wind speed data from the station. Furthermore, the difference in $PM_{2.5}$ deposition fluxes on leaves and the associated heavy-metal contents among different sites in the parks were analyzed using a one-way analysis of variance with replications, followed by an application of Fisher's least significant difference test (α = 0.05). To intuitively perform the spatial distribution of $PM_{2.5}$ concentration of the parks, the inverse distance weighting interpolation method was applied with QGIS software (Quantum GIS Development, V3.18, Open Source Geospatial Foundation Project). All statistical analyses were performed and graphs were plotted using SigmaPlot 14.0 (Systat Software Inc, San Jose, CA, USA).

3. Results

3.1. $PM_{2.5}$ mitigation rate of different schemes

Based on all the monitoring days, the mean percentage difference of the average $PM_{2.5}$ concentration from outside points between the two parks was ~13 ± 10%. Overall, the $PM_{2.5}$ pollution outside both parks showed similar results. The $PM_{2.5}$ mitigation rates of different calculation schemes are listed in Table 2. We excluded the mitigation results with an average $PM_{2.5}$ concentration lower than 5 μg m^{-3} because the mitigation effect in these situations was relatively meaningless, and a slight $PM_{2.5}$ change can lead to a large mitigation rate (~80% of the total monitoring data was still retained). The lowest mitigation rate occurred in EM in ZHP with an average of only 2.51% (Scheme A). The maximum $PM_{2.5}$ concentration removal was 24.29% in EM in TBG and 35.57% in DE in ZHP in Scheme D. From the average results of all times, ZHP exhibited a higher $PM_{2.5}$ mitigation rate in schemes calculated with the maximum value, such as Schemes C, D, and F; on the other hand, TBG exhibited higher results for schemes calculated with average values, such as A, B, and E. The mitigation rates in TBG at different times were similar in each scheme, whereas in ZHP, the rates were mostly higher in evenings or on weekdays. Furthermore, Scheme B revealed that the $PM_{2.5}$ mitigation rate increased by 1.75% and 1.24% of the average result in TBG and ZHP, respectively, when traveling into the geographic center of the park compared to the interior of the whole park (Scheme A). In both parks, Schemes E and F exhibited results similar to those of Schemes A and C, respectively. Overall, Schemes A and E represented as the average mitigation effect, exhibited similar results for both parks (~5%), and the average value of all the monitoring times was slightly higher in TBG than in ZHP. However, notably, the standard deviations of most results for all schemes were smaller for the TBG than for ZHP.

3.2. Mitigation effect under different $PM_{2.5}$ concentration categories

The $PM_{2.5}$ mitigation rates for different schemes generally decreased when the ambient concentration increased, except that an expectation occurred in the $PM_{2.5}$ category of 20–30 μg m^{-3} in ZHP (Fig. 2A). However, only in TBG, the mitigation rate rose to severe $PM_{2.5}$ levels (80–120 μg m^{-3}). The average and maximum rate based on Scheme A and D was 2.9% and 15.0%, respectively, in ZHP, whereas they remained around 6.6% and 21.0%, respectively, in TBG. The mitigation rate as per Schemes A, B, and E for TBG revealed similar results in terms of the lowest and the highest $PM_{2.5}$ ambient concentration categories (~1.5% difference of mitigation rate); in contrast, in ZHP, the range of mitigation rate dropped from 8.9%–11.8% to 1.6%–3.3% between these

Table 2
$PM_{2.5}$ mitigation results from different calculation schemes for Taipei Botanical Garden (TBG) and Zhonghe No. 4 Park (ZHP).

Calculation scheme	$PM_{2.5}$ mitigation rate (%)									
	TBG					ZHP				
	DM	DE	EM	EE	Ave.	DM	DE	EM	EE	Ave.
A	5.34 ±3.03	4.91 ±2.90	5.89 ±3.19	6.86 ±1.83	5.58 ±2.81	5.54 ±4.91	6.98 ±5.86	2.51 ±2.95	4.33 ±3.75	5.10 ±4.62
B	7.25 ±4.18	7.16 ±4.16	7.35 ±4.07	7.59 ±2.86	7.33 ±3.80	6.30 ±5.63	8.09 ±6.93	4.02 ±4.51	3.97 ±5.02	6.34 ±5.48
C	13.12 ±5.04	15.40 ±6.73	17.04 ±9.17	16.54 ±4.98	15.08 ±6.26	17.91 ±9.25	23.78 ±13.77	13.83 ±6.99	17.20 ±8.18	19.31 ±10.26
D	22.12 ±7.09	23.67 ±7.88	24.29 ±9.99	24.02 ±5.22	23.32 ±7.39	25.84 ±11.08	35.57 ±19.32	23.38 ±10.24	26.59 ±10.59	28.97 ±13.36
E	4.34 ±3.36	4.37 ±3.33	6.08 ±3.92	7.65 ±3.18	5.20 ±3.51	6.11 ±5.50	5.26 ±4.84	2.58 ±3.99	4.33 ±4.15	4.81 ±4.53
F	12.97 ±6.76	15.49 ±8.27	16.45 ±8.26	17.31 ±6.34	15.00 ±7.23	17.26 ±9.42	23.65 ±13.90	14.30 ±8.55	15.79 ±8.14	18.70 ±10.28
Sample size	20	19	10	9	58	19	18	11	10	58

(Note: DM, weekday morning; DE, weekday evening; EM, weekend morning; EE, weekend evening; Ave., average result of all monitoring times; data for each time scale represented as the mean value of the middle 90% observations and the standard deviation.).

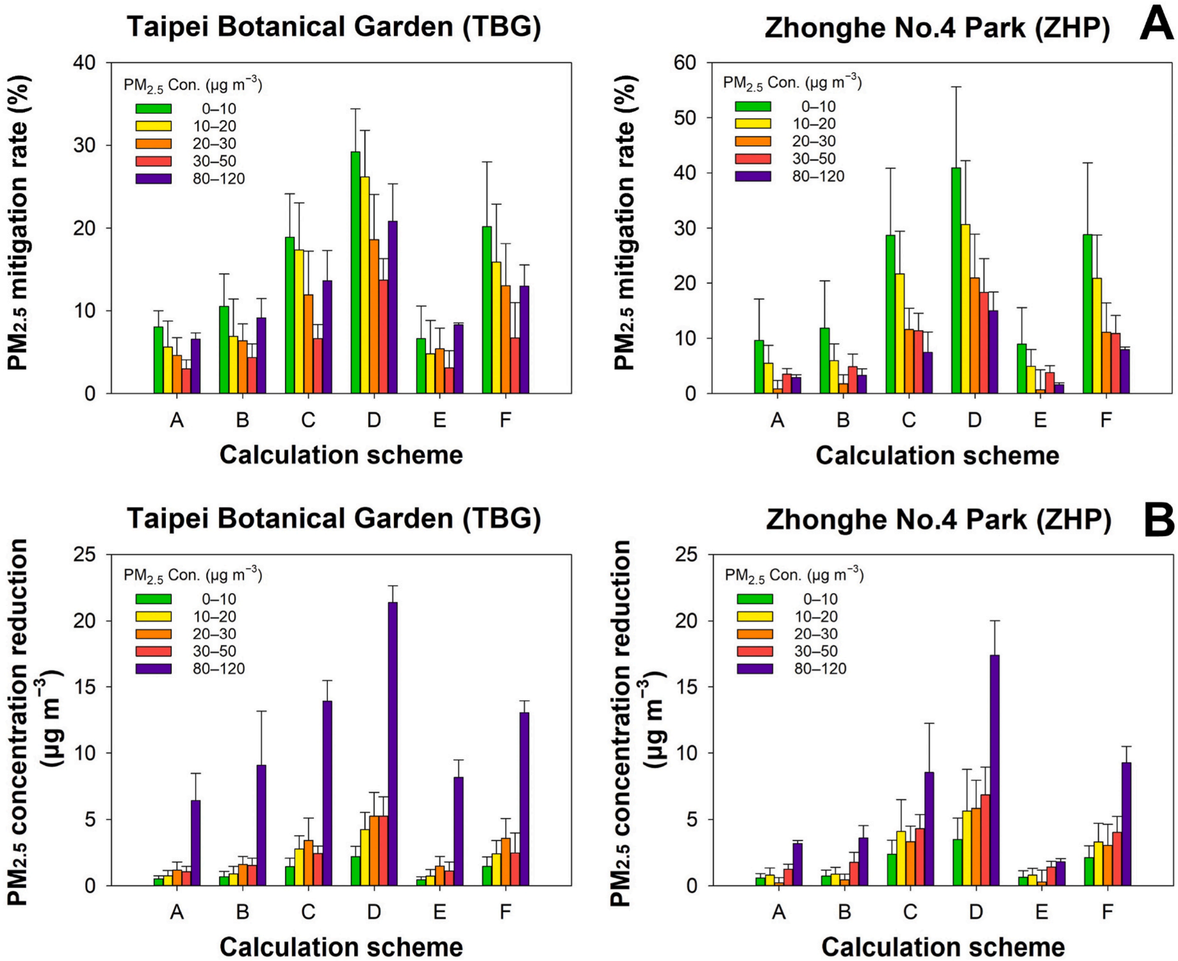

Fig. 2. Mitigation rate (A) and concentration reduction (B) of $PM_{2.5}$ from different calculation schemes under 5 $PM_{2.5}$ ambient concentration categories (thin vertical bars represent the standard deviation).

two categories. Furthermore, when considering the reduction in the $PM_{2.5}$ concentration, the results of all schemes in the two parks increased with the ambient concentration; however, the first 4 $PM_{2.5}$ categories differed only slightly (Fig. 2B). In contrast, the $PM_{2.5}$ concentration clearly decreased from 6.4 μg m^{-3} (Scheme A) to 21.4 μg m^{-3} (Scheme D) for the highest $PM_{2.5}$ level in TBG.

3.3. Influence of wind speed

The median wind speeds during the monitoring times in TBG and ZHP were 2.0 and 1.2 m s^{-1}, respectively. The relationship between the wind speeds from air quality stations and the average $PM_{2.5}$ concentrations of the parks from field measurement was analyzed. The Spearman correlation coefficient (ρ) was −0.248, and the relationship was significant ($p = 0.005$) (Fig. S1). Because Yonghe Station near ZHP performed near-surface monitoring, the results from this station can better represent the real situation near the ground. All the monitoring wind speeds in ZHP were below 3 m s^{-1}, and more than 70% of them were below 1.5 m s^{-1}. The Spearman correlation between wind speeds and $PM_{2.5}$ concentrations was insignificant ($\rho = -0.211$; $p = 0.093$) (Fig. S1). Although the aforementioned correlations were weak, we still observed that the severe $PM_{2.5}$ pollution occurred when the winds were almost calm. Moreover, we analyzed the relationship between the wind speed and $PM_{2.5}$ mitigation rate; however, insignificant results were obtained for each park ($\rho < 0.2$; $p > 0.05$). Even under higher wind speed conditions (>4 m s^{-1}), the $PM_{2.5}$ mitigation rate was not particularly high or low under any $PM_{2.5}$ concentration category. Furthermore, the wind direction did not exhibit any obvious impact on $PM_{2.5}$ mitigation; therefore, the associated analyses were not performed in this study.

3.4. $PM_{2.5}$ spatial distributions

We exhibited the spatial distribution of the last 2 $PM_{2.5}$ categories because of their relatively higher health risk. Major $PM_{2.5}$ pollution occurred outside the two parks. Under the moderate $PM_{2.5}$ pollution category (30–50 μg m^{-3}), the most polluted site was to the east (point 5) of TBG and the east (point 23) and north (point 16) of ZHP. Under the severe $PM_{2.5}$ pollution category (80–120 μg m^{-3}), pollution mainly occurred to the west (point 15) of TBG (Fig. 3); with regard to ZHP, point 19 exhibited extremely high $PM_{2.5}$ concentration because of the particles released from the sandpit. Nevertheless, lower $PM_{2.5}$ concentration was found inside the parks, near the centers (e.g., point 17 and 24 for TBG and ZHP, respectively). The visitor hotspots, crowded with local residents who exercised there, were at points 3 and 7 in TBG and point 6 in ZHP.

3.5. $PM_{2.5}$ temporal distributions

A significant linear correlation of $PM_{2.5}$ concentration exists between the monitoring data from station and field measurement ($R^2 = 0.8603$ and 0.8552 for TBG and ZHP, respectively; data excluded 10% outliers) (Fig. 4A). The regression lines shifted to the left for both parks; thus, the actual monitoring values were mostly higher than the results from

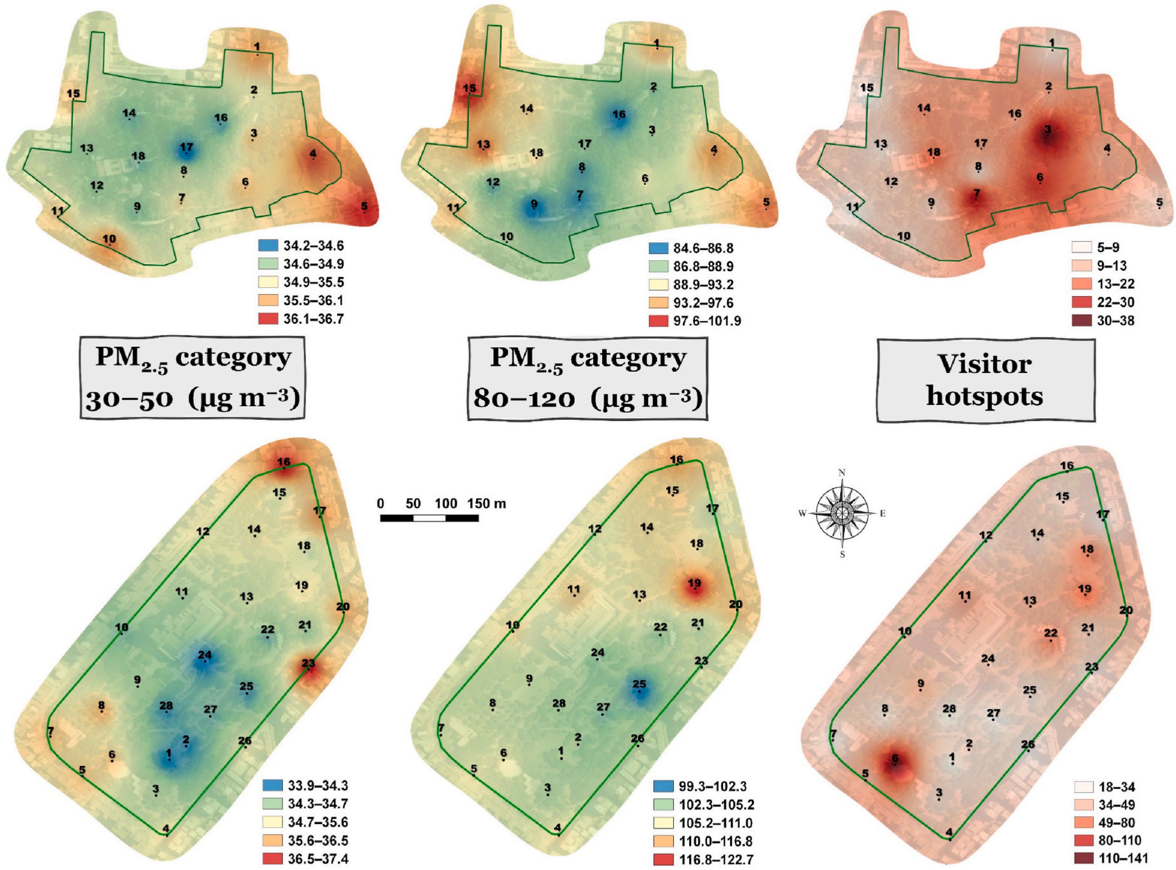

Fig. 3. Spatial distribution map of two high $PM_{2.5}$ concentration categories and visitor hotspots in TBG (upper) and ZHP (lower).

monitoring stations. Differences arose between the results from the station and field measurement when the $PM_{2.5}$ concentration was approximately higher than 10–20 μg m^{-3}. Furthermore, the linear regression equations were used to convert the hourly $PM_{2.5}$ monitoring data from stations into the field results over the monitoring year (Fig. 4B). The daily and annual $PM_{2.5}$ standards in Taiwan, developed in 2012, are 35 and 15 μg m^{-3}, respectively. The number of days on which the daily standard was exceeded increased from 2 to 36 in TBG and 6 to 42 in ZHP. The annual $PM_{2.5}$ concentration increased from 10.26 to 14.29 μg m^{-3} for TBG and from 12.74 to 13.29 μg m^{-3} for ZHP (excluding 5% outliers), and both meet the annual standard.

3.6. Flux of dry deposition of $PM_{2.5}$ on leaves

The dry deposition fluxes of $PM_{2.5}$ on leaves were 63.46 ± 16.89 and 20.14 ± 3.24 μg m^{-2} h^{-1} inside and outside the TBG, respectively. In contrast, the deposition fluxes were 12.13 ± 1.87 and 13.47 ± 2.72 μg m^{-2} h^{-1} inside and outside the ZHP, respectively. Furthermore, we calculated the $PM_{2.5}$ deposition flux in the dry deposition period using *i-Tree Eco*. The simulated dry deposition flux of $PM_{2.5}$ on leaves in TBG and ZHP was 57.71 and 48.63 μg m^{-2} h^{-1}, respectively (Fig. 5).

3.7. Heavy-metals analyses

Because of its limitation, our extraction method could detect only four of the nine detected elements in $PM_{2.5}$ (Fe, Al, Cr, and Ni) (Fig. 6). Among these metals, Cd was not detected in any of the PM samples deposited on the leaves. In $PM_{2.5}$ and TSP, Fe and Al had the highest content results among all elements, and their contents were mostly significantly higher outside the parks. A significant difference in Cr was observed only in the $PM_{2.5}$ in ZHP. Zn and Pb were detected only in TSP, and both contents were significantly higher outside the parks. Furthermore, for both PMs, Ni was detected only outside the parks.

4. Discussion

4.1. Effects of urban parks on ambient $PM_{2.5}$ concentration

Traffic emission is one of the major sources of PM in urban areas. Parks with ample trees are regarded a suitable solution to reduce traffic-related air pollution (Hernandez et al., 2019). Our mitigation results indicate that $PM_{2.5}$ concentrations inside both parks were lower than the external concentrations, establishing the existence of the barrier effect of vegetation and implying that air quality is better inside parks. Generally, vegetation with higher LAI (Lin et al., 2020a), crown diameter (Gromke and Ruck, 2007), or LAD (Zhang et al., 2020b) can reduce air flow and trap more PM near the road, and consequently mitigate the PM concentration inside the park. However, the difference in average $PM_{2.5}$ mitigation rates (Scheme A) between these two parks was slight; the rate in TBG was higher by only 0.48% (Table 2). This phenomenon indicated that the barrier effect of vegetation at a neighborhood scale is limited based on the overall observations; this result agrees with that of Viippola et al. (2018), who pointed out that canopy cover and tree density did not explain differences in ambient air pollution levels. From this perspective, more vegetation covers or barriers seemed to have little impact on mitigating the ambient $PM_{2.5}$ concentration, which exhibited similar results compared to those of studies based on deposition effect only (e.g., Nowak et al., 2013).

Turbulent mixing and dry deposition were the two major atmospheric transport processes of PM near roadways (Steffens et al., 2012). Research on vegetation effect in street canyons mainly focused on perpendicular wind (Abhijith et al., 2017). However, under low wind,

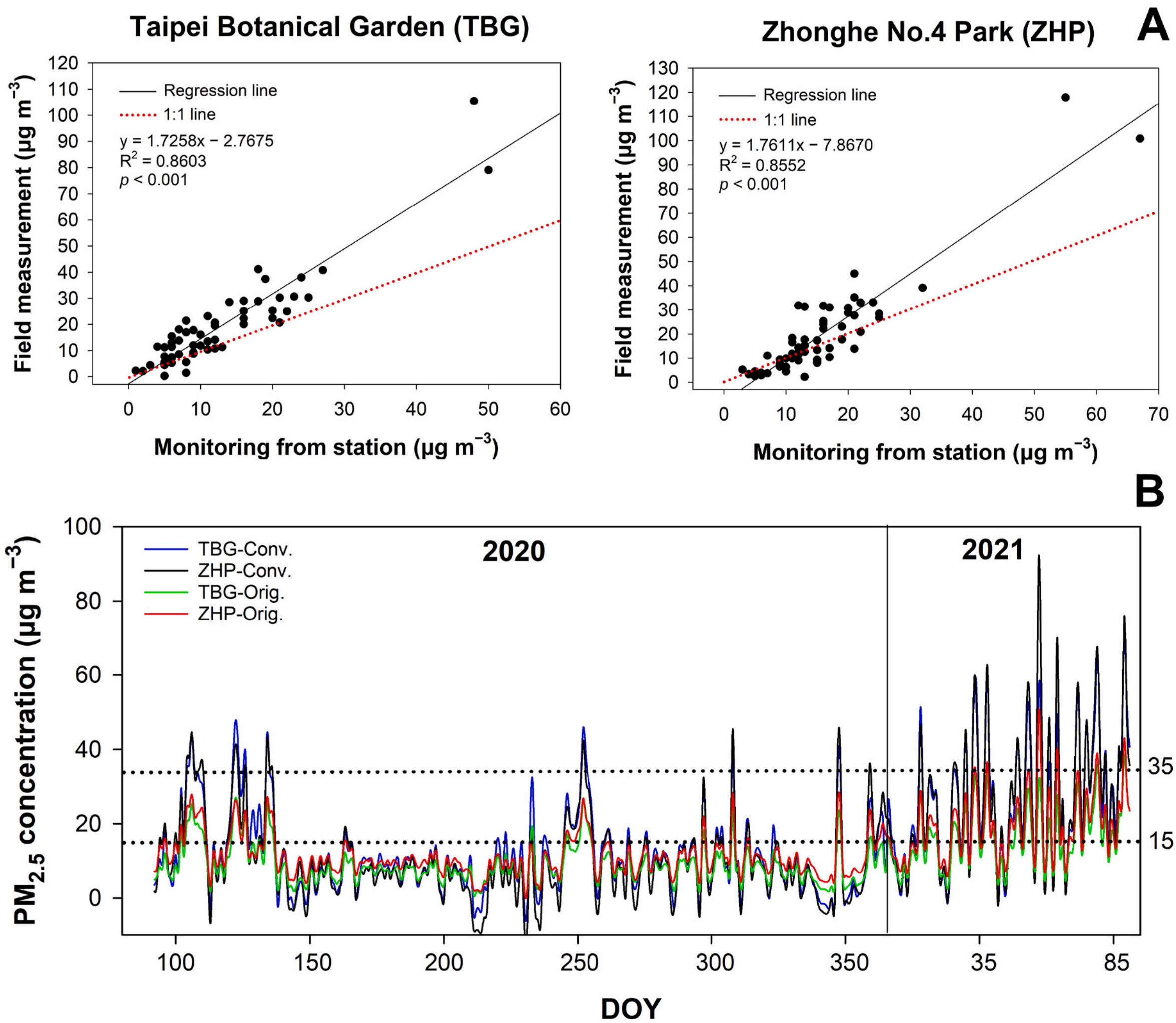

Fig. 4. Linear correlation between the monitoring results from the station and field measurement (A), and temporal distribution of the original daily average $PM_{2.5}$ concentration from the station and the converted value obtained using the linear equation (B) (the lower black dotted lines corresponding to values of 35 and 15 μg m^{-3} represent the daily and annual $PM_{2.5}$ standards in Taiwan, respectively).

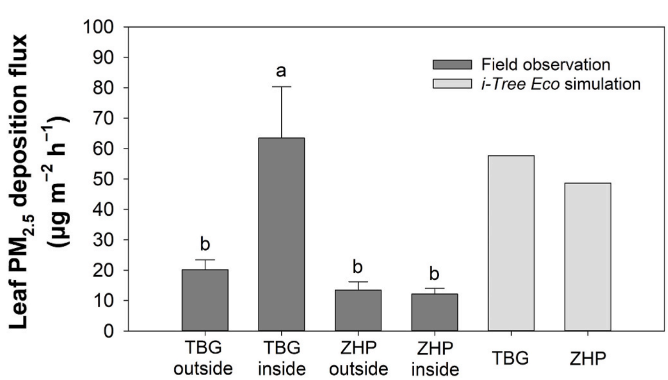

Fig. 5. Dry deposition flux of $PM_{2.5}$ deposited on the leaves from the field and model results (the letters represent significant differences at the 0.05 level; thin vertical bars represent the standard deviation).

particle deposition may be rare because the fine particles are continuously suspended in air. Moreover, dispersion too may be rare because of the laminar flow (Jeanjean et al., 2016). The monitoring results from Yonghe Station exhibited that wind speeds near roadside were very low and stable (Fig. S1). Considering the monitoring height, this result may be more representative of the near-surface condition in TBG and ZHP. Meanwhile, because of the low wind speeds, the associated wind directions were meaningless, and thus, no obvious relationships between wind directions and $PM_{2.5}$ mitigation rates were observed in this study. This result agrees with that of Hagler et al. (2012), who reported that the local wind trend near roadside were frequently low speed and variable in direction, limiting to assess real downwind conditions of the vegetation barriers. A similar result was reported by Viippola et al. (2018): shifting wind direction did not affect the pollution patterns, and the higher peaks were observed nearer the roadside.

Dense forests were believed to inhibit the dispersion effect and create recirculating eddies, resulting in longer PM residence times (Janhäll, 2015; Tong et al., 2016). In this study, this phenomenon can be observed based on three aspects. First, the maximum $PM_{2.5}$ mitigation rates (Scheme D) were higher in ZHP than those in TBG (Table 2) because there may be stronger dispersion in ZHP, causing faster dilution of $PM_{2.5}$. Further, because of the larger area of ZHP, the chances for cleaner air at certain points are higher for this park. This result agrees with that of Xing and Brimblecombe (2020), who indicated that a park with larger area was conducive to providing cleaner air inside the park. Second, the mitigation rates in TBG exhibited lower differences among different times, whereas ZHP exhibited clearly higher results in the evenings and weekdays. This phenomenon proved that despite differences in meteorological conditions at different times or days, the $PM_{2.5}$ mitigation rates in TBG remained relatively stable. Third, the standard deviation of the

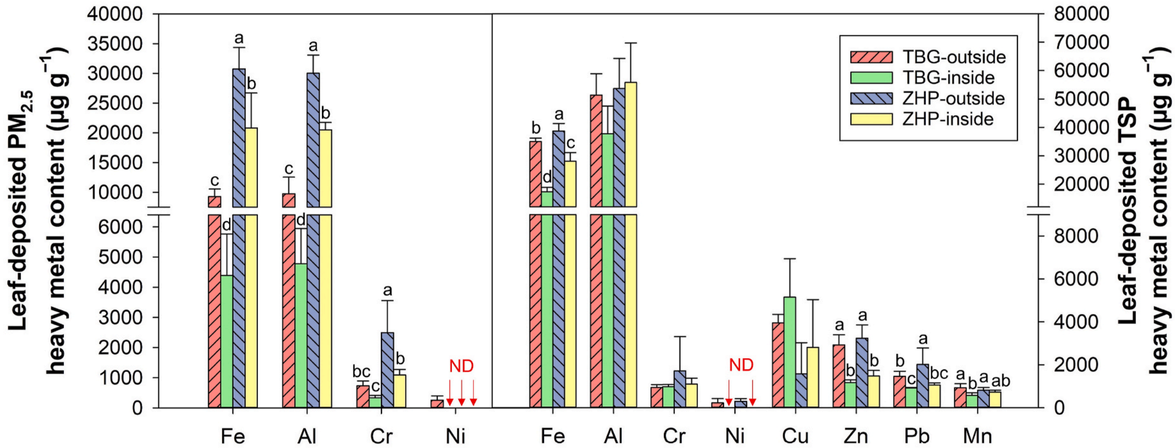

Fig. 6. Heavy-metal content in $PM_{2.5}$ and total suspended particulates deposited on the leaves (the letters represent significant differences at the 0.05 level; thin vertical bars represent the standard deviation; ND, not detected).

mitigation rates among different schemes was lower in TBG than in ZHP. The results from some schemes for ZHP had higher standard deviation than the mean because the $PM_{2.5}$ concentrations at certain times were higher inside the park. This phenomenon suggests that though the open field in ZHP may be conducive for the diffusion of $PM_{2.5}$, particles can easily enter the park, resulting in accumulation. Similarly, increased concentration inside a park in Beijing has been reported (Chen et al., 2015). Moreover, Grzędzicka (2019) observed a significant positive correlation between $PM_{2.5}$ concentration and vegetation porosity. In contrast, although the dense vegetation in TBG may not drastically reduce the $PM_{2.5}$ concentration because of low dispersion, stable mitigation effect was observed. These findings agree with previous studies in which vegetation decreased the variation in $PM_{2.5}$ concentration instead of drastically reducing the concentration itself in comparison with open fields (Tong et al., 2015; Viippola et al., 2018).

4.2. *Effects of vegetation barriers on air quality improvement*

The mitigation effect was more meaningful under higher polluted conditions. The mitigation rates and associated concentration reductions under different $PM_{2.5}$ ambient concentration categories were examined. The $PM_{2.5}$ concentration reduction increased slightly with increasing ambient concentration, causing a noticeable decrease in the mitigation rate (Fig. 2). This phenomenon revealed that the efficiency of urban parks in mitigating $PM_{2.5}$ concentration remains stable in most situations. However, interestingly, we observed that TBG with more vegetation had greater $PM_{2.5}$ concentration reduction under severe pollution, with the average and maximum mitigation rates (6.56% and 20.85%, respectively) that were ~2.3 and 1.4 times those in ZHP. This finding suggested that the vegetation barrier effect of urban parks is strong for severe PM pollution under low wind conditions (<1 m s^{-1}). The highest $PM_{2.5}$ ambient concentration occurred under stagnant wind field. Also, during particulate matter episodes in northern Taiwan, vehicular emissions and road dust can contribute 78% of the ambient PM (Liang et al., 2013). In this situation, dense vegetation in TBG may lead to a great and clear trapping effect on traffic pollutants, leading to drastic differences between the inside and outside of a park. Furthermore, as shown by Jeanjean et al. (2016), in street canyons under a wind speed of 1 m s^{-1}, dispersion by trees increased $PM_{2.5}$ concentration by 8% compared to that in the city as a whole, whereas deposition was negligible (less than 1%); that is, the negative aerodynamics effect of trees prevailed over deposition. However, on a neighborhood scale of an urban park and surrounding roads, the trapping effect of the vegetation barrier may lead to better air quality inside the park. Generally, the overall PM mitigation effect of urban parks can be very complex because (1) various sources of PM exist around the park (traffic, industries, human activities, and biological sources); (2) complex landscape structures lead to different micrometeorological conditions; and (3) the aerodynamics and deposition effect are unevenly distributed in a park. Thus, long-term spatial and temporal observations are necessary.

Field mobile monitoring of $PM_{2.5}$ is suitable for a neighborhood scale because of its rapid and simple monitoring with high spatial resolution. However, we should be aware of the effectiveness of the monitoring data. Hence, (1) the monitoring device should be highly accurate and capable of self-recalibration; (2) the readings should be stabilized in a minute or longer if needed; and (3) the overall monitoring time should be as low as possible to avoid changes in background concentration, especially at rush hours. Therefore, when investigating the PM mitigation effect of an urban park, monitoring inside and outside the park alternately is necessary. Thus, mobile monitoring is more appropriate at small urban scale. Studies using mobile monitoring in a large park can monitor better in different transects (e.g., Gómez-Moreno et al., 2019) or focus on the effect of distance from the road (e.g., Grzędzicka, 2019). Furthermore, we designed Schemes E and F because they can almost exclude the influence of changes over time. The results of Schemes E and F are similar to those of Schemes A and C, respectively, indicating that our monitoring design is feasible and proving the existence of the PM mitigation effect when walking into the park. Our results indicate that $PM_{2.5}$ concentration decreases from an average of 4.8% to a maximum of 18.7% for a distance of 100–150 m from the road (Table 2). Many studies reported the mitigation effects of vegetation barriers, showing reduction in the PM concentration with increasing distance from the road (Mukherjee et al., 2020; Tong et al., 2015, 2016; Viippola et al., 2018). Nevertheless, the studies usually focused on the effect of air pollutants mitigation at points or along lines at different distances from the main road, especially under perpendicular wind conditions. However, the air quality in an urban park should be considered for multiple points on a plane, allowing people to understand the overall effect of air quality improvement upon entering the park.

4.3. *Spatiotemporal $PM_{2.5}$ distributions in the parks*

The spatial distribution of $PM_{2.5}$ concentration of the two parks under moderate $PM_{2.5}$ pollution (30–50 μg m^{-3}) shows that the values were higher outside the park (points 5 and 10 in TBG and 16 and 23 in ZHP); whereas, the air quality was better closer to the center of the parks (Fig. 3). These results intuitively indicate the effect of vegetation barriers in the urban parks for reducing ambient $PM_{2.5}$ from the exterior to the interior side of the park without the prevailing wind direction. Similar spatial distribution could be found in previous studies using the CFD model (Xing and Brimblecombe, 2020) or field measurements (Zhu et al., 2019). Notably, this phenomenon also demonstrated that the barrier effect of the park, which led to the lower $PM_{2.5}$ near the geographic center, prevails over the difference between the different types of landscapes at such a neighborhood scale. Moreover, under severe $PM_{2.5}$ monitoring condition, point 15 outside the TBG exhibited the highest concentration, indicating that the source direction of pollutant was not regular in this area. Notably, point 19 with a sandpit inside the ZHP had a remarkably high $PM_{2.5}$ only under severe $PM_{2.5}$ category. The particles from the sandpit were large and deposited easily. However, under calm wind conditions with high PM pollution, the particles generated by children playing in the sandpit were suspended for longer times, adversely affecting the air quality greatly. Previous studies have shown that the sandpits and sandboxes could be contaminated with hazardous heavy metals from the neighboring road traffic (Aleksander-Kwaterczak and Rajca, 2015) or industrial emissions (Zupančič et al., 2021). Hence, under such calm wind conditions, children's activities in the sandpit should be avoided. Moreover, the visitor hotspots inside the two parks were far from the polluted points, indicating many people benefited from the functional effects of air quality improvement in these parks. This finding also suggests that when planning and designing a park, it is important to consider the locations where people will gather and ensure these places are far from the air pollution hotspots. Moreover, human activities, such as moving or smoking, can also be the emission sources of ambient $PM_{2.5}$ (Zhou et al., 2016). However, except for the result in the sandpit, we did not find obvious increase of $PM_{2.5}$ level at visitor hotspots. As smoking in most urban parks in Taiwan is illegal, the results revealed that general recreational activities in our study parks did not cause noticeable impacts on ambient $PM_{2.5}$.

Furthermore, our $PM_{2.5}$ field measurements were higher than the results obtained from monitoring stations for both parks, especially under conditions of severe pollution (Fig. 4B). The $PM_{2.5}$ observations from our mobile monitoring have systematic biases compared to the results obtained from air quality monitoring stations. However, overestimations and underestimations could possibly occur in the monitoring results from our mobile devices and the stations, respectively. Although our monitoring samples of high $PM_{2.5}$ concentration were insufficient, the number of days on which the daily standard was exceeded increased to more than one month over the year, suggesting that the air quality of the parks might be worse than we expected. Interestingly, similar higher results were found in the low-cost sensor networks (labeled as the Airbox) in Taiwan (Li et al., 2020; Lin et al., 2020b), and their linear relationships between the results from the air quality monitoring stations and the Airboxes were quite close to ours. Although the overestimations of these low-cost sensors may sometimes occur, the consistent trend obtained from our mobile-monitoring results with highly accurate devices suggested a possibility that these low-cost sensors could detect air pollution near the ground to a certain extent. Previous studies in the cities of Taiwan have also shown that the ground-level $PM_{2.5}$ was higher, and the ratio of the monitoring concentration between low- (first to third floors) and mid-floors (fourth to sixth floors) at a vertical level was 1.13 (Ho et al., 2015) or 1.90 (Wu et al., 2014). Furthermore, the mobile-monitoring results in ZHP were closer to the 1:1 line compared to those in TBG (Fig. 4A), which may be due to the reason that the Yonghe Station monitored $PM_{2.5}$ closer to the ground. However, the discrepancy in ZHP still occurred when $PM_{2.5}$ was higher than ~15 μg m^{-3}. This phenomenon may be due to the longer distance between ZHP and Yonghe Station (~2 km). Nevertheless, more accurate devices, such as high volume $PM_{2.5}$ sampling system, should still be applied inside the parks in the future to clarify whether the air quality is worse than expected compared to the information from neighboring stations. Our finding demonstrates that accurate air quality information of urban parks is necessary and important for visitors. The representativeness of the monitoring results from stations to the actual air quality of the urban park must be confirmed.

4.4. PM interception and biomonitoring by trees

The dry deposition of $PM_{2.5}$ on leaf surfaces exhibited similar results both inside and outside ZHP, which indicates that open vegetation is unconducive for particle deposition. In contrast, inside TBG, the deposition was approximately thrice times that outside the park, revealing that dense vegetation possibly promoted deposition (Fig. 5). Our results are in contrast with those of previous studies in which mostly higher PM deposition was observed near the road or at the bottom of tree crowns, mainly owing to shorter distances from the road (Hofman et al., 2014; Mori et al., 2015; Sgrigna et al., 2015). Since the monitoring $PM_{2.5}$ concentrations generally exhibited relatively slight differences between the inside and outside of the parks, the greater difference of $PM_{2.5}$ deposition flux in TBG suggested the existence of higher deposition velocity inside TBG. This phenomenon can be attributed to the recirculating eddies and longer PM residence times, as we mentioned in Section 4.1. Furthermore, microclimate regulating functions of urban trees, such as increasing air humidity, can accelerate particle deposition (Ding et al., 2018). It also indicates that although dispersion may decrease inside TBG, higher deposition can still help remove $PM_{2.5}$, resulting in a mitigation effect. Furthermore, we obtained the $PM_{2.5}$ deposition flux of *i-Tree Eco* using concentration data from stations. The model generally overestimated the deposition in both parks (Fig. 5). Interestingly, if the wind speed information from Zhongzheng Station near TBG was used, the *i-Tree Eco* simulation was closer to the field results inside the TBG, leading to a significant overestimation near the roadside. In contrast, the lower wind speed information monitored from Yonghe Station near ZHP corresponded to a lower deposition velocity of $PM_{2.5}$ in *i-Tree Eco*, which led to the closer flux result to the field observations next to the street (outside the parks). Moreover, if ~67% of the total water-soluble part of $PM_{2.5}$ (Qi et al., 2019) was further considered ($PM_{2.5}$ mass may be lost in this study due to water filtration method), the originally overestimated results from *i-Tree Eco* may be consistent with the field results except those inside the TBG. Overall, these phenomena have three implications: (1) the representativeness of meteorological conditions and PM concentration from monitoring stations for the modeled area is very important, especially at a neighborhood scale; (2) the PM deposition in a dense forest can be very unevenly distributed; and (3) the field verification method of *i-Tree Eco* still requires improvement. Most of our results were also different from those of Pace et al. (2021), who found that the modeling results for $PM_{2.5}$ deposition were underestimated 6–20 times compared to field measurements by an extraction method similar to ours. Nevertheless, in this study, only one tree species, 3–5 m of crown height, and two sites of each park were considered, while the large difference in deposition results in TBG suggest that when using *i-Tree Eco* to model the $PM_{2.5}$ removal of an urban forest park, field examinations at a neighborhood scale are necessary.

Examination of the heavy metals and trace elements in the deposited PM in our study indicated that all the significantly higher results occurred outside both parks (Fig. 6). Heavy metals such as Zn and Pb, regarded to originate from traffic (Chen et al., 2010; Guney et al., 2010; Świetlik et al., 2013), are present in the deposited TSP with higher contents close to the roads. A significantly high Cr content occurred outside ZHP, revealing the discrepancy in air pollution between these two parks. Furthermore, although Ni contents were relatively low, they could still be found only outside the parks for both $PM_{2.5}$ and TSP. High contents elements, namely Fe and Al, were mainly attributed to Earth's crust or road dust (Liu et al., 2020, 2021; Schlosser et al., 2017). However, they also exhibited higher results outside the parks, indicating that traffic-related sources also led to the obvious emission of them. Notably, the content of these two metals exhibited larger difference between the inside and outside of TBG (~2 times) than those of ZHP (~1.5 times), indicating that greater barrier effect from the dense forest in TBG can better contribute to mitigate the harmful PM from road emission. Moreover, this result may also be attributed to more bio-source particles (such as pollen) with less heavy-metals inside TBG. Our results demonstrate that not only the PM concentration but also the associated heavy metals were mitigated by the urban parks. Since the $PM_{2.5}$ concentrations exhibited relatively slight differences between the inside and outside of the parks, the greater difference in heavy-metal contents may suggest that the mitigation effect of urban parks on improving air quality was manifested in less toxic ambient PM inside the parks. Moreover, these findings reveal the good biomonitoring effect of tree leaves on PM-associated heavy metals. Tree leaves as a bioaccumulator were reported in previous studies to determine the spatial or temporal distributions of heavy metals (Norouzi et al., 2015; Ugolini et al., 2013). Nevertheless, the $PM_{2.5}$ mitigation effect of urban parks should also consider the harmful components such as heavy metals, and the overall effectiveness of urban parks in providing cleaner air is promising.

Ecosystem services are commonly defined as the positive benefits that ecosystems provide to people. In urban area, purifying the air is assumed to be one of the ecosystem services derived from urban forest for a long time (Bolund and Hunhammar, 1999; Escobedo et al., 2011; Lin et al., 2019). Previous studies have quantified this function by focusing on the deposition effect verifying the amount of air pollutants that can be intercepted/filtered by urban trees. Based on the law of conservation of mass, the removal of air pollutants can be regarded as the effect of air purification. However, it is difficult to observe the real air purification especially at small scales. Generally, urban residents pay more attention on how the urban trees can improve the local air quality (Davies et al., 2017). Therefore, we consider that expect for the air pollutants removal, we should focus more on the mitigation of ambient concentration of the air pollutants by urban forest to respond to the expectations of local people. Furthermore, the biomonitoring effect, derived from urban trees, provides another effective monitoring tool in addition to the aforementioned environmental monitoring such as low-cost sensor, air quality monitoring station, or mobile monitoring. Although this function is difficult to be quantified, it should still be considered as an ecosystem service of urban trees in providing useful information in relation to air quality.

5. Conclusion

Based on the average results calculated throughout the monitoring year, the difference in average $PM_{2.5}$ mitigation rates between these two parks was negligible, indicating the effect of air quality improvement by vegetation in urban parks may be limited. However, the results from this study exhibited the different $PM_{2.5}$ mitigation types of urban parks. Dense vegetation in the park showed a stable $PM_{2.5}$ mitigation effect throughout the monitoring year and a strong ability to provide cleaner air under severe $PM_{2.5}$ pollution. In contrast, an open park with a single-storied stand has a variable mitigation effect, causing either quick dissipation or accumulation of $PM_{2.5}$ inside the park. We observed that under low wind conditions, the vegetation barrier effect may mitigate air pollution from the road especially under a severe $PM_{2.5}$ level, improving the air quality inside the urban park. Our results revealed that the impact of urban vegetation on improving local air quality should be discussed in finer scales along with long-term spatial and temporal observations. Furthermore, by analyzing the heavy metals from leaf-deposited PMs, we found the existence of a less toxic PM level inside the urban parks, indicating cleaner air inside the parks. These findings provide strong evidence that urban parks can improve the urban environmental quality at a neighborhood scale. Moreover, urban parks provide an essential space for social and recreational activities for the residents; therefore, the associated air quality is extremely important. We performed the spatial distribution of $PM_{2.5}$ concentration of the parks using mobile monitoring, which directly exhibited the barrier effect of urban parks; this information is valuable for local visitors. Nevertheless, further field studies should be conducted on the actual air quality inside the urban park and the role of vegetation in affecting air pollutants.

Credit author statement

Tzu-Hao Su: Conceptualization, Methodology, Investigation, Formal analysis, Visualization, Writing – original draft. **Chin-Sheng Lin** Methodology, Investigation, Writing – review & editing. **Shiang-Yue Lu**: Project administration. **Jiunn-Cheng Lin**: Resources. **Hsiang-Hua Wang**: Resources. **Chiung-Pin Liu**: Supervision, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2022.116283.

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