Tree species richness and diversity predicts the magnitude of urban heat island mitigation effects of greenspaces

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Abstract

The Urban Heat Island Effect (UHIE) is a widely recognised phenomenon that profoundly affects the quality of life for urban citizens. Urban greenspace can help mitigate the UHIE, but the characteristics that determine the extent to which any given greenspace can cool an urban area are not well understood. A key characteristic is likely to be the properties of trees that are found in a greenspace. Here, we explore the sensitivity of the strength of the cooling effect to tree community structure for greenspaces in Changzhou, China. Land surface temperatures were retrieved from Landsat 7 ETM+ and Landsat 8 TIRS and were used to evaluate the temperature drop amplitude (TDA) and cooling range (CR) of 15 greenspaces across each of the four seasons. Tree community structure of the greenspaces was investigated using 156 sample plots across the 15 greenspaces. We found that a number of plant community structure indicators of greenspaces have a significant impact on the strength of the cooling effect. The Shannon-Wiener diversity index, tree species richness and tree canopy coverage of greenspaces are all positively correlated with the magnitude of the temperature drop amplitude, with the strength of their influence varying seasonally. We also find that mean crown width is positively correlated with cooling range in summer and autumn, whilst greenspace tree density is negatively correlated with cooling range in winter. Our findings improve understanding of the relationship between plant community structure and the cooling effect of greenspaces. In particular, we highlight the important role that tree species diversity provides for mitigating the UHIE, and suggest that if planners wish to improve the role of urban greenspaces in cooling cities,

they should include a higher diversity of trees species.

Keywords: cooling effect, plant community structure, land surface temperature, Shannon-Wiener diversity index, species richness, ecosystem services, tree coverage

Abbreviations: UHIE, Urban heat island effect; LST, Land surface temperature; DBH, Diameter at breast height; TDA, Temperature drop amplitude; CR, Cooling range

1. Introduction

The proportion of the global human population living in towns and cities is projected to reach 66% by 2050, with the majority of this growth occurring in Asia and Africa (United Nations, 2014). The environmental impacts of rapid urbanization have received greater attention and include increased, and more extreme temperatures as part of the Urban Heat Island Effect (UHIE) (Buyantuyev and Wu, 2010; Liu et al., 2009), poor air quality (Fu and Chen, 2017), increased water and energy consumption and poorer health and wellbeing of urban residents (Morris et al., 2017; Santamouris, 2020; Yang et al., 2020b).

In parallel with this rapid urbanisation, global mean surface temperatures are projected to rise throughout the century, to an extent that depends upon future anthropogenic emissions (IPCC, 2018). The emergence of urban heat islands is one of the key environmental issues of the 21st century (Oke, 1973), not least because rising temperatures associated with global climate change may aggravate the impacts of the UHIE that are already common in urban areas (Luber and McGeehin, 2008), leading to more extreme heat stress, especially in megacities (Argüeso et al., 2015; IPCC, 2018).

Urbanization changes the reflection and absorption of solar radiation by the Earth's surface, which leads to a higher temperature in urban areas than suburban and rural areas. This difference in temperature has been termed the UHIE (Bowler et al., 2010). The UHIE can profoundly affect the quality of life for urban citizens, with higher temperatures, water use increases, for instance when irrigation is required (Guhathakurta and Gober, 2007; McDonald et al., 2011). Similarly, the demand for living and office spaces to be cooled, through the provision of air conditioning can require more energy (Salvati et al., 2017; Yang et al., 2020b), which leads to higher levels of air pollution. Finally, raised temperatures in cities are associated with an increase in mortality and heat-related health conditions (Boumans et al., 2014; Morris et al., 2017). Mitigating the UHIE has therefore become a core concern for urban sustainability.

Urban greenspaces have increasingly been recognized as providing multiple benefits for improving urban sustainability and liveability (Grilo et al., 2020; Shekhar and Aryal, 2019; Verdú-Vázquez et al., 2017). Greenspaces include parks and reserves, sports fields, community gardens, street trees, and nature conservation areas, as well as less conventional spaces such as green walls and green alleyways (Wolch et al., 2014). In terms of the UHIE, greenspaces provide a cooling effect thereby mitigating higher temperatures especially in summer (Bernatzky, 1982; Cao et al., 2010; Chang and Li, 2014; Du et al., 2017; Qiu and Jia, 2020) while a lack of greenspaces is associated with higher temperatures (Luber and McGeehin, 2008). Greenspaces do this in two ways. First, trees and other vegetation can directly shade surfaces and thereby reduce the amount of incident solar radiation (Dimoudi and Nikolopoulou, 2003). Second, greenspaces enable the conversion of incident radiation to latent rather than sensible heat, through transpiration by vegetation (Moss et al., 2019). These effects combine to lower temperatures within greenspaces, compared to the surrounding built-up areas. The cooling effect of greenspaces also extends beyond the edge of the greenspace itself into surrounding streets (Lin et al., 2015).

The physical structure of greenspaces can alter their ability to mitigate the UHIE. Size, impervious surface coverage, vegetation coverage and the presence of water have all been shown to correlate with the magnitude of the cooling effect (Du et al., 2017; Li et al., 2011; Lu et al., 2017; Srivanit and Hokao, 2013). The size of greenspaces is positively correlated with the strength of the cooling effect (Vaz Monteiro et al., 2016; Yang et al., 2020a). Further research has focused on threshold-size of greenspaces that is helpful to optimize greenspace's cooling effects (Yu et al., 2020; Yu et al., 2017). However, the requirement to provide more greenspaces is often difficult in cities due to the intense pressure on land for development and construction (Jim, 1998). An alternative approach could be to optimize plant community diversity and structure within greenspaces to maximize any cooling effect and UHIE mitigation (Tang et al., 2017).

Plant community structure, such as the number of trees, species composition, tree size, tree canopy cover, tree height, tree location and tree health, all influence ecosystem service provision from greenspaces (Chen et al., 2020; Moser et al., 2015; Nowak, 2008; Zhang et al., 2013). However, not all urban vegetation is equally effective in reducing temperatures. Indeed, tree canopies differ in their ability to intercept and absorb radiation and to transpire (Kelliher et al., 1993). The foliage characteristics and tree's mature shape (for example, canopy geometry and total height) can affect the thermal performance and shade effectiveness (Leuzinger et al., 2010; Shahidan et al., 2010). The particular type of treed habitat has also been shown to have an influence on the cooling effect. For instance, in Singapore, secondary forest was the most efficient in providing cooling. Vegetation in secondary forest was denser than elsewhere, thus preventing more radiation from reaching the forest floor and simultaneously increasing evaporation cooling (Richards et al., 2020). The "forest-like" characteristics of secondary forest, with its complex vertical structure and, by extension, varied species composition might be important. Further, for a range of tree and bamboo species, tree foliage density makes the greatest contribution to the cooling effect, followed by leaf thickness, leaf texture and leaf colour lightness (Bau-Show and Yann-Jou, 2010). Thermal satellite data indicated that every unit increase in leaf area index decreased land surface temperature by 1.2°C (Hardin and Jensen, 2007). Canopy density, leaf area index and tree height were the most significant drivers of the magnitude of the cooling effect in Changchun, China (Tang et al., 2017). Multilayer plant communities were the most effective in terms of the cooling effect, and bamboo groves were the least effective (Zhang et al., 2013). Plant community structure has a significant impact on the cooling effect of greenspaces (Petri et al., 2019; Tang et al., 2017; Zhang et al., 2013). Despite this research, thus far the particular contribution of tree diversity to the cooling effect remains poorly explored.

The cooling effect of urban greenspaces will vary with the season as vegetation cover, soil moisture, impervious surfaces, air humidity and albedo vary seasonally (Haashemi et al., 2016; Jonsson, 2004). Seasonal variation in correlations between vegetation and LST suggests much stronger coupling during warmer seasons than cooler seasons (Buyantuyev and Wu, 2010). Most of the previous studies have studied the thermal environment relief function of the plant community structure at a specific time, especially in summer (Du et al., 2017; Perini and Magliocco, 2014; Yang et al., 2017). We therefore know relatively little about the seasonal variations of cooling effects in relation to plant community structure.

Here we address an important knowledge gap regarding the influence of tree diversity in UHIE mitigation. We use two Landsat datasets (Landsat 8 TIRS and Landsat 7 ETM+) and ground-based tree plots across the rapidly urbanizing city of Changzhou, China. We answer the following three questions that have not previously been adequately addressed: (1) How do the cooling effects of urban greenspaces vary seasonally? (2) What role does tree species diversity play in underpinning the cooling effect?

2. Materials and methods

2.1 Study area

Covering 4385 km^2 Changzhou city, Jiangsu Province, China, is urbanizing rapidly. In 2018, the city had 4,729,000 residents, living at a density of 1079 inhabitants/ km^2 (Changzhou Statistics Bureau, 2019). Changzhou lies in the Yangtze River Delta in the northern subtropical humid area (Zheng et al., 2010), and the seasons are divided into spring, summer, autumn and winter from March to May, June to August, September to November and December to February (Yu et al., 2014). Mean seasonal temperatures for the city between 1952 and 2006 were 14.7℃, 26.7℃, 17.5℃ and 4.4℃ for spring, summer, autumn and winter respectively (Qin et al., 2008). Around 57% of the city is covered in impervious surfaces such as buildings and roads (Changzhou Statistics Bureau, 2019).

Based on the 2011-2020 Changzhou City Master Plan (Figure 1, a), the central area is the construction concentration area of Changzhou, which includes the most densely populated and built up region of the city, but also includes greenspaces and infrastructure, as well as canals and rivers. Fifteen publicly accessible greenspaces in the core area of the central city were selected (Figure 1, b). They varied in size, tree canopy cover and tree community composition. Most of these greenspaces were instigated in the last 40 years and are therefore characterised by relatively small trees compared to more established public parks in Europe and North America (Figure 1, c). As public spaces, most of the greenspaces is composed of impervious surfaces, characterised by grassed areas and mixed plantings of deciduous and evergreen tree species. Water bodies are also commonly present.

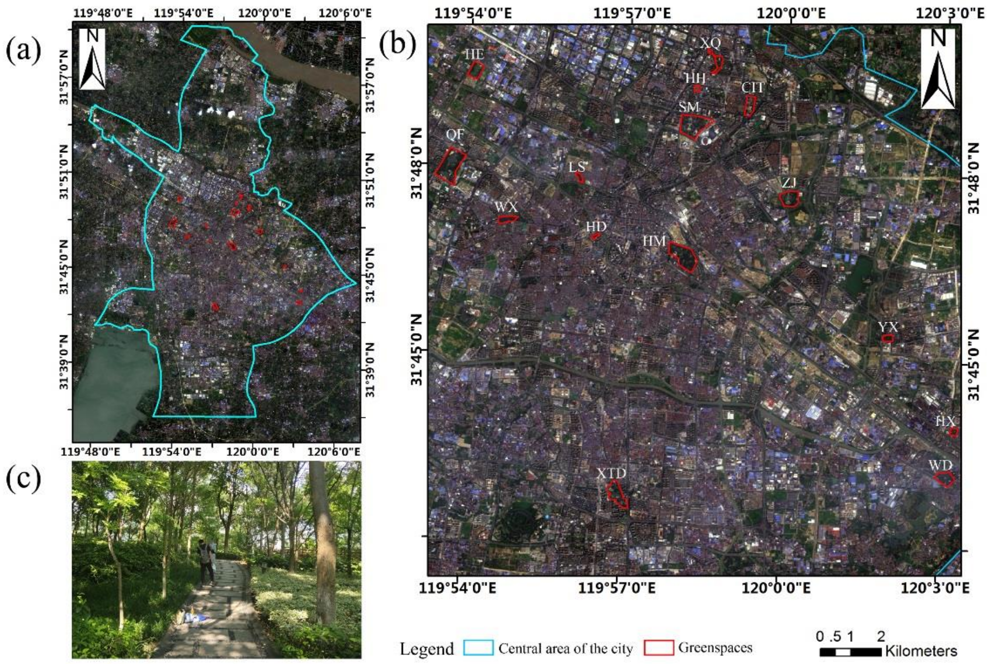

Figure 1 (a) The central area of the city. (b) The location of 15 greenspaces (outlined in red) in the core of centre area of Changzhou, China. (c) The typical plant community in greenspaces of Changzhou. Letters give abbreviated greenspaces names: He=He park, XQ=Xinqu park, HH=Hehai residential area, SM=Shiming square, CIT=Campus of Changzhou institute of technology, QF=Qingfeng Park, WX=Wuxing Park, LS=Lushu park, HD=Huaide square, HM=Hongmei park, ZJ=Zijing Park, YX=Yuxiu park, XTD=Xintiandi park, HX=Huaxi park, WD=Weidun park.

2.2 Data sources and processing

2.2.1 Data sources

Land surface temperatures collected from remote sensing thermal infrared images are positively correlated with air temperatures (Ren et al., 2016). In this study, we therefore used Landsat images of the study area to measure temperature across all four seasons in 2017-2018. To achieve better results for the retrieved land surface temperatures, four cloudless remote images were selected from Landsat datasets at Row/Path: 38/119. Two Landsat 8 TIRS images covered winter (21 December 2017 GMT: 02:31:23) and spring (27 March 2018, GMT: 02:30:40). Two Landsat 7 ETM+ images were used for summer (07 June 2018, GMT: 02:31:02) and autumn (29 October 2018, GMT: 02:28:05). In all cases cloud cover was ＜10%, images covered daytime and were download from https://glovis.usgs.gov/. The lowest and highest air temperature corresponding to the day of remote sensing images are 13℃~25℃ (27 March 2018) as spring, 22℃~31℃ (7 June 2018) as summer, 11℃~23℃ (29 October 2018) as autumn and 1℃~11℃ 21 December 2017）as winter.

2.2.2 Estimating land surface temperatures

We calculated land surface temperature (LST) based on a previously validated radiative transfer equation (Du et al., 2017; Masoudi and Tan, 2019; Qiu and Jia, 2020; Wang et al., 2018). Land surface emissivity (ε) is an essential parameter for retrieving LST from thermal infrared remote sensing data. The land surface can be viewed as composed of three land cover patterns: vegetation, bare soil and water (Qin et al., 2004). Land surface emissivity (ε) is estimated by using the values of NDVI and green cover ratio (P_v):

$$P_v = [(NDVI - NDVI_{soil})/(NDVI_{veg} - NDVI_{soil})] \quad (1)$$

Where NDVI is normalized difference vegetation index, $NDVI_{soil}$ and $NDVI_{veg}$

are NDVI in bare land and vegetation area, set as 0.05 and 0.7, respectively.

Based on the land cover patterns, ε_{water} (the Land surface emissivity of water) is 0.995.

$$\varepsilon_{vegetation} = 0.9625+0.0614P_v-0.0461P_v^2 \quad (2)$$

$$\varepsilon_{building} = 0.9589+0.086P_v-0.0671P_v^2 \quad (3)$$

where $\varepsilon_{vegetation}$ is the emissivity of the natural surface, $\varepsilon_{building}$ is the emissivity of the built surface (Qin et al., 2004; Shi and Zhang, 2018).

$$B(T_s) = \left[L_\lambda - L^{\uparrow} - \tau(1-\varepsilon)L^{\downarrow}\right]/\tau\varepsilon \quad (4)$$

L_λ is the radiance registered by the sensor, B(Loreau et al.) is the blackbody radiance related to the surface temperature by Planck's law, $L^{\uparrow}$ and $L^{\downarrow}$ are the upward and downward atmospheric radiance, respectively, τ is the atmospheric transmission.

$$T_s = K_2/\ln[K_1/B(T_s\ \ ) \ + 1] \quad (5)$$

T_s is the LST, calculated by Planck formula, for band 10 of TIRS, K_1= 774.89 W/(m^2·μm·sr)， K_2 = 1321.08 K; for band 6 of ETM+, K_1= 666.09 W/(m^2·μm· sr)， K_2 = 1282.71 K (Yu et al., 2017). Atmospheric profile parameters ($L^{\uparrow}$, $L^{\downarrow}$, and τ) can be obtained by entering the imaging time and the centre coordinate by latitude: 31.48, longitude: 119.53 on the website provided by NASA (http://atmcorr.gsfc.nasa.gov/). These data allow us to calculate the LST for each greenspace and each season, as well as the mean LST for the central area of the city (Fig. 2).

2.2.3 Metrics of cooling effects

The cooling effect of urban greenspaces is evaluated by quantifying the difference in LST between a greenspace and the surrounding built area. There are several metrics which can be used for this (Du et al., 2017; Sun et al., 2012; Yang et al., 2020a). Here we used the cooling range and temperature drop amplitude. The cooling range is defined as the distance from the greenspace boundary at which the first turning point in LST occurs (Du et al., 2017; Qiu and Jia, 2020). The turning point of an LST curve was identified by piecewise regression as the point where the slope of LST curve changes sharply. To estimate the cooling range, we calculated the mean LST within 15 m buffers around each greenspace. TDA is the temperature difference between the mean LST of the greenspace and the mean LST at the cooling range (Figure 3).

2.2.4 Urban greenspace tree community structure

We followed the i-Tree Eco protocol (i-Tree, 2020), which generates sample plot locations across a survey area in order to achieve a standard error of 12% for estimates of the total number of trees present (Martin et al., 2013). In addition to the i-Tree Eco protocol, to ensure we captured variation in tree community structure, we constrained plot site selection so that no portion of any plots fell on water or sealed surface, and that plots were not clustered in small areas of larger greenspaces. Each plot was approximately 400 m^2 (a circle with an 11.34 m radius). The plot selection process resulted in 156 sample plots within our study greenspaces, which sampled 62400 m^2 from the total unsealed surface land area of the 15 greenspaces of 1300341 m^2 (Appendix 1). Within each plot we identified each tree to species, recorded its height, diameter at breast height (1.37 m above ground, DBH), crown width, crown height, crown health, crown base height, as well as the total number of trees in each plot. Every tree with diameter greater than 4 cm was measured (Table 2). Data were collected between June 2017 and February 2018. We further calculated the tree coverage of each greenspace by using i-Tree Canopy (i-Tree Canopy, 2020) in which tree coverage is assessed by a random sample of Google Maps aerial photography. In i-Tree Canopy, sample points are generated randomly, and pre-defined cover types can be chosen for the sample points. We evaluated between 300 and 1000 sample points in each greenspace, with the number of points proportional to greenspace area.

We calculated two metrics of tree diversity, namely species richness (i.e., the number of species) and the Shannon-Wiener diversity index which combines species richness and the relative frequency of these species (evenness), as per equation (6).

$$H' = -\sum_{i=1}^{s} p_i \ln(p_i) \qquad (6)$$

Where p_i is the relative frequency of species *i* in the community and S is the number of species. Species richness is the number of different species in a community. In this study, large-sized greenspaces have more sample plots than small-sized greenspaces, so we calculate *richness per plot* as an indicator of the richness to account for the relationships that exist between sampled area and species richness. We wished to understand the strength of the relationship between tree diversity and the cooling effects of urban greenspaces, taking into account the role of tree community structure. We did this in two stages. Firstly, we used correlations (Pearson's or Spearman's correlation coefficient depending on whether specific data were normally distributed) to identify aspects of tree diversity and community structure that had a significant association with TDA or the cooling range (Schober et al., 2018). Then, variables that had a significant association with cooling effects were included in multiple regression models. Analyses were repeated for each season, all analyses were carried out in SPSS 22.

3. Results

3.1 The cooling effect of the greenspaces

At the city scale, we found that the mean LST of the Changzhou central area was 26.8±2.4℃, 37.7±3.7℃, 25.0±2.3℃, 11.1±1.3℃ in spring, summer, autumn and winter respectively (Fig 2). Across the greenspaces, the mean LST was cooler throughout the year, at 25.3±1.4℃, 34.2±2.4℃, 23.4±1.3℃, and 10.0±0.8℃ respectively. Most of the greenspaces had a lower LST than the surrounding area throughout the year, except HH and CIT (greenspace abbreviations given in Figure 1 and Appendix 1). The temperature drop amplitude of HH was 0.0℃ in spring, and the temperature drop amplitude of CIT was 0.0℃ in spring, summer, and autumn.

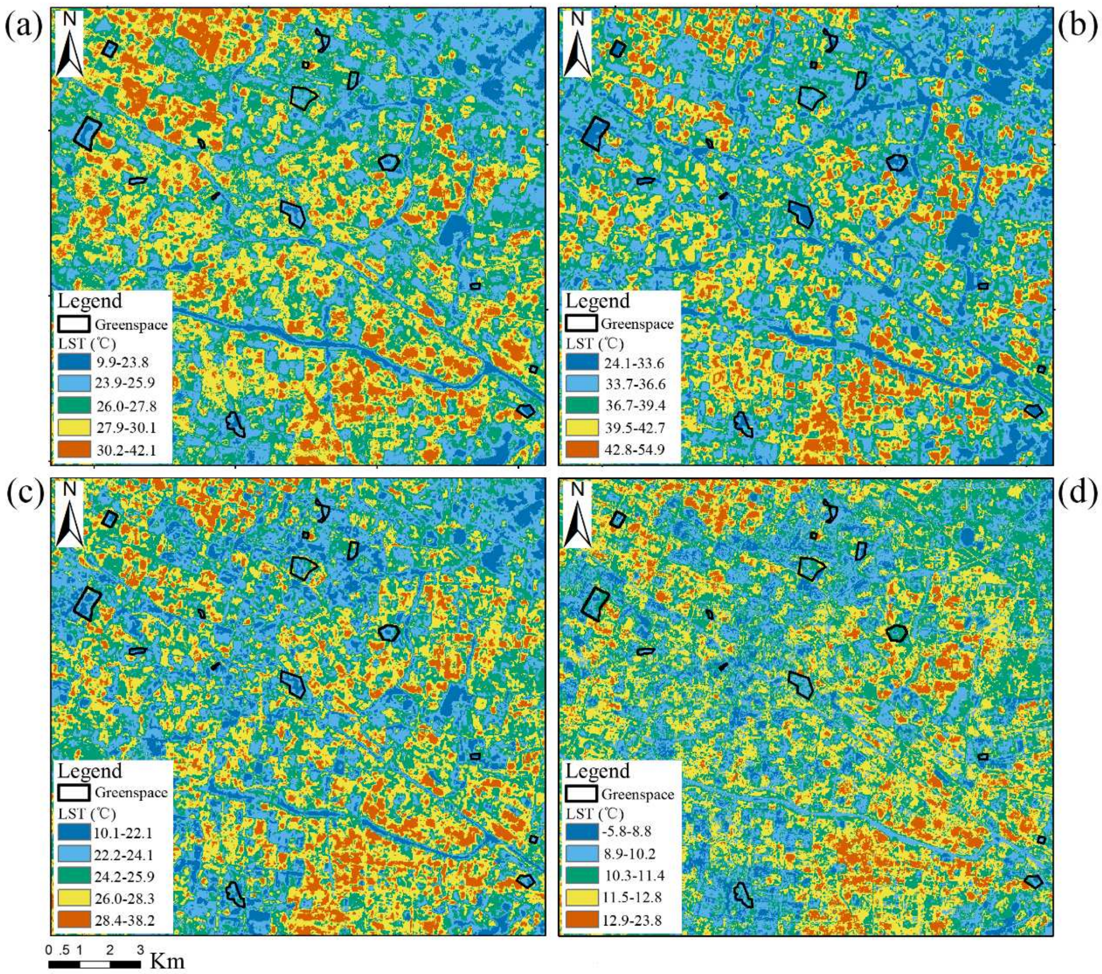

Figure 2 Land surface temperature for central Changzhou, China for (a) spring, (b) summer, (c) autumn and (d) winter. The 15 sampled greenspaces are outlined in black.

At the neighbourhood scale, piecewise regressions were used to determine the cooling range outside each greenspace (Figure 3, for greenspace HM as an example) in different seasons. The results show that the median cooling range across the 15 greenspaces was 75 m (interquartile range (IQR) 45 m to 300 m), 75 m (IQR 30 m to 255 m), 60 m (IQR 30 m to 120 m) and 60 m (IQR 30 m to 90 m) in spring, summer, autumn and winter respectively (Figure 4a). The median cooling range value of greenspaces was the same in spring and summer, and the same in autumn and winter, although they had different interquartile ranges. The cooling range of each greenspace varied across seasons, with the longest cooling ranges in spring, and the shortest in winter. For one of our greenspaces (CIT; Figure 1), the cooling range was near zero except in winter. Similarly XTD had no cooling range in winter. Greenspace XQ had the largest cooling range in spring and summer, while LS has the largest cooling range in autumn and winter (Table 1).

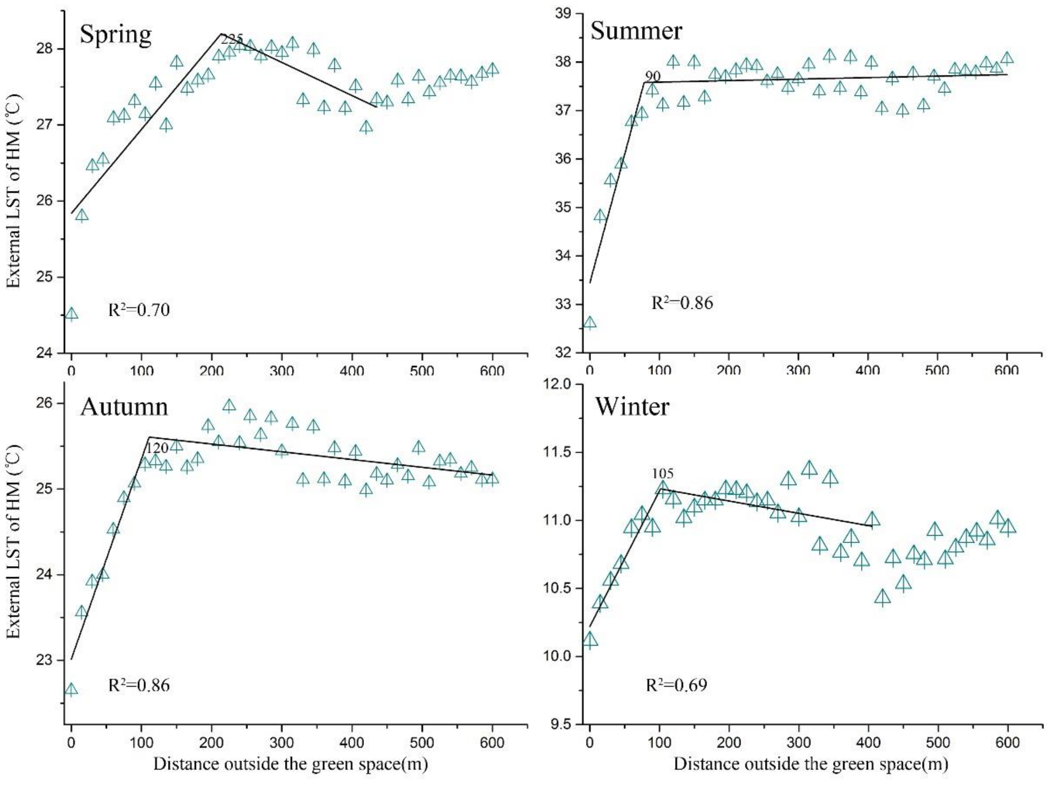

Figure 3 An example of cooling range and TDA for greenspaces “HM”. Across the four seasons, the cooling range was 225m, 90m, 120m and 105m, as estimated via the identification of turning points using piecewise analysis. TDA was 28.0℃-24.5℃=3.5℃, 37.4℃-32.6℃=4.8℃, 25.3℃-22.7℃=2.6℃, 11.2℃-10.1℃=1.1℃.

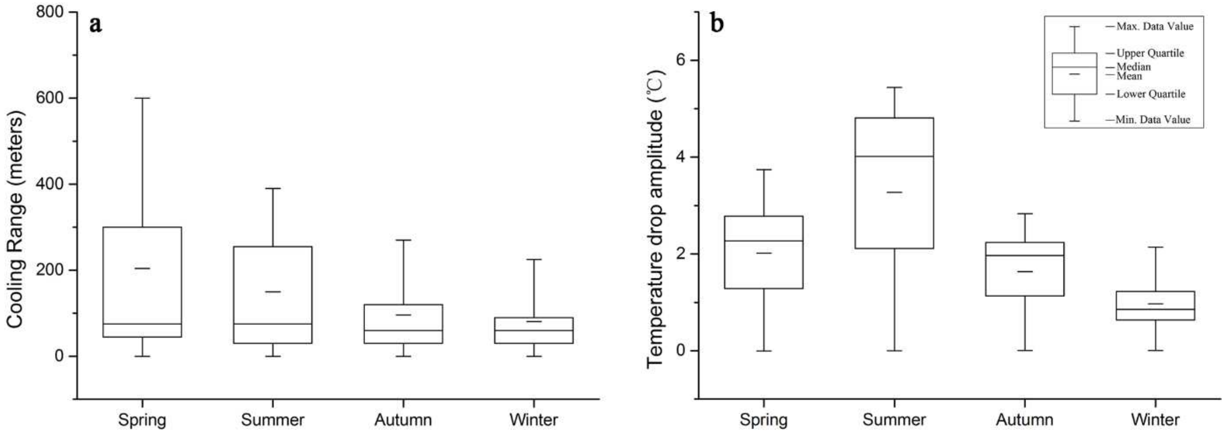

Figure 4 (a) The cooling range for 15 greenspaces in central Changzhou, China, across four seasons in 2017-2018. (b) The temperature drop amplitude of 15 green spaces across four seasons.

The TDA of each greenspace was calculated after the cooling range was determined by piecewise regression analysis. The TDA of greenspaces varied seasonally (Figure 4b); the median TDA of the 15 greenspaces was 2.3°C (IQR 1.3°C to 2.8°C), 4.0°C (IQR 2.1 to 4.8°C), 2.0°C (IQR 1.1°C to 2.2°C), and 0.9°C (IQR 0.6°C to 1.2°C) in spring, summer, autumn and winter. The maximum TDA was, therefore, in summer and the minimum in winter. The TDA of each greenspace also varied seasonally (Table 1). The only exception to the general seasonal pattern was for greenspace CIT which had a near zero TDA in spring, summer and autumn, and a maximum, although still small, TDA in winter.

3.2 Plant community structure of the greenspaces

There was considerable variation in the number of trees present in each greenspace. This was partly a consequence of the different proportion of green land, water and impervious surface area in each greenspaces, and partly driven by the history of the spaces. As Changzhou has urbanised, more greenspaces have been constructed. The majority of these have been built within the last 20 years. Of those we sampled, HM was the oldest. Built in 1960 it was redesigned in 2005. Given the short history of greenspaces in the city, there are few mature trees. For instance, the median DBH was 19.67 cm (IQR 17.30 cm to 21.54 cm), the 75% of trees have a DBH smaller than 21.54 cm. This indicates that there are many young trees, as DBH is a proxy of tree age. Similarly, median tree height was 7.1 m (IQR 6.9 m to 7.8 m), and 25% of trees were between 7.8 m and 15 m tall. Around half of all trees were shorter than 7.1 m. These trees can, therefore, act as a lower canopy layer and help to form a more structured tree canopy, where taller trees are also present. Tree density varied across the greenspaces. The median tree density was 235 trees/hectare (IQR 143 trees/hectare to 360 trees/hectare). The median crown height was 4.6 m (IQR 4.3 m to 4.7 m), the median crown width was 5.0 m (IQR 4.3 m to 5.7 m), and the median tree canopy coverage was 52% (IQR 44% to 56%).

Of the 2082 trees that were sampled, 1253 (60.2%) were deciduous and the remainder evergreen (Appendix 1). There were 49 deciduous tree species and 29 evergreen tree species. The five most abundant deciduous tree species were *Acer palmatum* Thunb. (195, 9.4%), *Metasequoia glyptostroboides* Hu et W. C. Cheng (140, 6.7%), *Cerasus serrulata* (Lindl.) London var. lannesiana (Carri.) Makino (78, 3.8%), *Lagerstroemia indica* Linn. (65, 3.1%), *Celtis sinensis* Pers. (63, 3%). The top five evergreen tree species are *Cinnamomum camphora* (L.) Presl (223, 10.7%), *Osmanthus fragrans* (Thunb.) Lour. (175, 8,4%), *Magnolia grandiflora* Linn. (83, 4%), *Trachycarpus fortunei* (Hook.) H. Wendl. (59, 2.8%), *Photinia serrulata* Lindl. (46, 2.2%). The median Shannon-Wiener diversity index was 2.48 (IQR 2.07 to 2.72), and the median richness per plot was 1.92 (IQR 1.37 to 2.67).

Table 1. The cooling effect of 15 greenspaces within Changzhou city centre, China, across different seasons. Greenspace abbreviations are given in Figure 1 and Appendix 1.

Greenspace	Green land Area (ha)	TDA-spring (℃)	TDA-summer (℃)	TDA-autumn (℃)	TDA-winter (℃)	CR-Spring (m)	CR-summer (m)	CR-autumn (m)	CR-winter (m)
HD	1.03	0.65	2.05	1.13	1.12	75	60	45	90
HH	0.45	0.00	0.97	1.22	0.24	0	75	75	75
LS	2.12	2.61	3.07	2.20	1.23	255	225	270	225
HX	2.05	3.45	5.35	2.19	1.54	300	270	60	90
YX	2.76	2.27	3.49	1.66	0.64	75	30	75	15
WX	4.54	2.02	4.29	2.25	0.83	60	165	165	60
XQ	5.25	2.46	4.15	0.54	0.21	600	390	210	30
CIT	4.73	0.00	0.00	0.00	1.31	0	0	0	180
HE	6.10	2.78	4.99	2.83	2.14	30	30	30	60

WD	7.16	3.74	5.44	1.97	1.01	75	40	60	30
ZJ	12.76	1.28	2.49	2.02	0.85	180	15	15	60
XTD	12.10	2.21	4.42	0.74	0.00	390	375	45	0
HM	22.66	3.44	4.81	2.68	1.11	210	90	120	105
SM	19.54	1.31	2.11	1.42	0.64	45	45	30	15
QF	26.80	2.58	4.02	2.24	0.79	375	255	105	45

Table 2 Tree community structure of 15 greenspaces within Changzhou city centre, China.

Greenspace	Mean DBH (cm)	Mean height (m)	Density (trees/ha)	Mean crown height (m)	Mean crown width (m)	Tree canopy coverage (%)	Shannon-Wiener diversity index	Species richness per plot
HD	23.37	8.3	81.00	4.6	5.7	59.60	2.07	1.33
HH	19.67	7.1	15.00	3.4	3.3	13.80	1.27	1.00
LS	21.54	8.8	288.00	6.1	5.6	63.60	2.04	2.67
HX	18.88	6.9	231.00	4.7	5.9	56.30	2.40	2.50
YX	16.93	6.8	361.00	4.7	5.0	56.10	2.75	3.38
WX	21.25	7.2	235.00	4.2	5.9	54.20	2.51	2.13
XQ	20.27	7.1	219.00	4.6	5.7	52.30	2.24	1.87
CIT	19.67	7.1	48.00	3.4	3.0	29.30	1.52	1.37
HE	17.39	7.5	360.00	5.0	5.0	47.30	2.72	2.45

WD	12.22	6.8	418.00	3.5	3.5	54.20	3.01	2.80
ZJ	19.01	7.7	305.00	5.1	4.3	48.70	2.60	1.79
XTD	18.41	7.8	326.00	4.6	4.8	40.30	2.48	1.54
HM	26.46	9.2	143.00	6.9	5.5	60.20	2.67	1.92
SM	21.67	6.9	175.00	4.2	4.7	44.20	2.18	0.76
QF	16.31	6.1	678.00	3.8	4.5	45.40	3.24	3.19

3.3 Associations between plant community structure and cooling effects

There were no significant correlations between mean DBH, tree height, crown height and the two measures of cooling effect across all seasons (Table 3a). However, significant correlations were found for the remaining plant community structure indicators and cooling effect. Firstly, in spring and summer, TDA was significantly correlated with tree density, tree coverage, Shannon-Wiener diversity index, and richness per plot. In autumn, TDA had a significant correlation with the Shannon-Wiener diversity index and tree richness. However, plant community structure had no significant correlation with TDA in winter. The correlation between CR and plant community structure indicators differed from those with TDA. In spring, indicators had no significant correlation with the CR, while in summer and autumn there was a significant correlation for mean crown width, and in winter with tree density (Table 3a).

Table 3 For 15 greenspaces in central Changzhou, China the (a) correlation between cooling effect and plant community structure variables by seasons; and (b) correlation between variables. ** Significant at P<0.01, * Significant at P<0.05

(a)

Cooling effect	Seasons	Mean DBH	Mean Height	Density	Mean Crown Height	Mean Crown width	Tree-coverage	Shannon-Wiener diversity index	Richness per plot
Temperature drop amplitude	Spring	-0.246	-0.063	0.583*	0.454	0.438	0.653**	0.754**	0.681**
	Summer	-0.267	-0.02	0.569*	0.358	0.505	0.581*	0.772**	0.598*
	Autumn	-0.016	0.104	0.447	0.5	0.371	0.485	0.598*	0.534*
	Winter	-0.003	0.232	0.048	0.261	0.131	0.328	0.154	0.292
Cooling range	Spring	-0.095	0.145	0.368	0.435	0.425	0.344	0.274	0.359
	Summer	0.125	0.131	0.03	0.118	0.556*	0.159	-0.07	0.113
	Autumn	0.191	0.092	0.09	0.172	0.543*	0.487	0.075	0.448
	Winter	0.473	0.469	-0.518*	0.188	0.11	0.348	-0.455	-0.072

(b)

Variables	Height	Density	Crown height	Crown width	Tree-coverage	Shannon-Wiener diversity index	Richness per plot
Mean DBH	0.624*	-0.627*	0.587*	0.471	0.182	-0.406	-0.514
Mean height		-0.341	0.578*	0.331	0.358	-0.316	-0.318
Density			-0.022	0.06	0.296	0.851**	0.764**
Mean crown height				0.619*	0.639*	0.218	0.158
Mean crown width					0.745**	0.271	0.215
Tree coverage						0.558*	0.507
Shannon-Wiener diversity index							0.704**

To quantify the relationship between cooling effect variables and plant community structure we carried out multiple regressions using those community structure variables that had significant correlations with the cooling effect as predictor variables. However, many of the plant community structure variables were also significantly inter-correlated (Table 3b). To avoid violating assumptions of multiple regression we therefore used the Shannon-Wiener diversity index (represented by 'H' in equation 6 to 16) and tree coverage (%, represented by 'T') only in our regression models. We selected the models from Table 4 with the highest adjusted R^2 value for the final model of TDA and CR in different seasons. Equations 6 to 8 describe the TDA prediction models for greenspaces community structure in spring, summer and autumn.

Table 4. Parameter estimates for the relationship between TDA and plant community structure across different seasons.

	Shannon-Wiener diversity index	Tree coverage	Intercept	Adjusted R^2	p
TDA Spring	1.29	3.07	-2.49	0.59	0.002
TDA Summer	2.03	2.74	-2.72	0.57	0.003
TDA Autumn	0.93		-0.55	0.31	0.018

$$\text{TDA Spring} = -2.49 + 3.07T + 1.29H \quad (R^2 = 0.59, P=0.002) \quad (6)$$

$$\text{TDA Summer} = -2.72 + 2.74T + 2.03H \quad (R^2 = 0.57, P=0.003) \quad (7)$$

$$\text{TDA Autumn} = -0.55 + 0.93H \quad (R^2 = 0.31, P=0.018) \quad (8)$$

For TDA, in spring, our regression model (Equation 6) explained 59% of the variation. With the Shannon-Wiener diversity index kept constant, TDA increases 0.31°C for every 10% increase in tree coverage. When tree coverage is kept constant, TDA increases by 0.13°C for every 0.1 increase in the Shannon-Wiener diversity index. In summer, our model (Equation 7) explained 57% of the variance. When the Shannon-Wiener diversity index is kept constant, TDA increases by 0.27°C for every 10% increase in tree coverage. When the tree coverage is kept constant, TDA increases 0.2°C for every 0.1 increase in Shannon-Wiener diversity index. In autumn, our model (Equation 8) explained 31% of the variance and TDA amplitude increases 0.09°C for every 0.1 increase in Shannon Wiener diversity index.

$$\text{CR Summer} = -178.4 + 65.4W \quad (R^2=0.15, P=0.082) \quad (9)$$

$$\text{CR Autumn} = -114.8 + 41.8W \quad (R^2=0.21, P=0.048) \quad (10)$$

$$\text{CR Winter} = -0.13D + 105.5 \quad (R^2=0.06, P=0.201) \quad (11)$$

In summer, autumn and winter, the cooling range (CR) is significantly correlated with both the mean crown width and tree density; we therefore included mean crown width in metres (‘W’ in Equations 9 and 10), and density (trees/hectare, represented as ‘D’ in Equations 11) in our regression models. In summer, our regression model (Equation 9) explained 15% of the variation. Cooling range increases 65.4 metres for every 1-metre increase in mean crown width. In autumn, our model (Equation 10) explains 21% of the variation. The cooling range increases by 41.8 metres for every 1-metre increase in mean crown width. In winter, our model (Equation 11) explains 6% of the variance and the cooling range decreases by 1.3 metres for every 10 trees per hectare increase.

4. Discussion

4.1 Influence of tree diversity on cooling effect

Biodiversity underpins the delivery of ecosystem function (Cardinale et al., 2011), and, therefore many ecosystem services. Indeed, tree species richness of forests shows a positive relationship with the provision of multiple ecosystem services (Gamfeldt et al., 2013). Despite this, to date, most research on the extent to which urban greenspaces might mitigate urban heat island effects has concentrated on quantifying the threshold size of greenspaces (Fan et al., 2019; Yu et al., 2020), greenspace configuration (Du et al., 2017; Lin et al., 2015), and particular structural characteristics such as canopy coverage (Giridharan et al., 2008; Petri et al., 2019; Yang et al., 2017) or tree height (Zhang et al., 2013), rather than exploring the role of tree diversity *per se*. Here we show that when the tree coverage keep constant the diversity of the tree community within greenspaces can have a significant effect on the magnitude of cooling effect in spring, summer and autumn, and maximum TDA benefit in summer. If cities, and their residents, are to make best use of greenspaces as one way to mitigate urban heat island effects, then ensuring that tree diversity is taken into account should be a central consideration.

There are substantial differences between tree species in terms of their ability to cool the surrounding air (Bau-Show and Yann-Jou, 2010; Rahman et al., 2015). Tree canopies with high leaf area and transpiration rates have been shown to be the most effective in terms of cooling (Rahman et al., 2018). Canopy shape and leaf colour of tree species may influence their cooling efficiency (Feyisa et al., 2014). Further, a plant community with multiple layers of trees, shrubs and herbs can decrease air temperature by 1°C on a sunny day, and 0.5 °C on a cloudy day in summer, compared with a simple biomass structure dominated by a tall canopy layer (Fung and Jim, 2019). A combination of tree species with high cooling efficiencies could improve the magnitude of cooling effects provided by greenspaces. Where space is limited, such a diversity of trees are likely to be more efficient at intercepting solar radiation before it reaches the surface, and also in providing a greater leaf area to enhance transpiration rates, further ameliorating cooling effects. Further, such a community structure is likely to result in greater levels of biomass production (Gamfeldt et al., 2013), which will, over time, increase the effectiveness of cooling, as larger and larger leaf surface areas will be produced.

4.2 Plant community structure and variation in cooling effects across seasons

Our work was carried out in a sub-tropical location, characterised by seasons delineated by changes in air temperature and solar radiation, we would therefore expect the cooling effects of greenspaces to vary seasonally. In line with our expectations, we find that the cooling effects are greatest in summer and least in winter (Jenerette et al., 2011; Mingjuan et al., 2019). Although we find some differences in terms of the magnitude of cooling effects in spring and autumn, our findings also align with research in Shenzheng, China, where the TDA of parks was greatest in in summer (4.6℃), followed by autumn (4.0℃) and spring (3.8℃), TDA was least in winter (2.9℃) (Zhang et al., 2013). Given the role that tree species diversity plays in explaining cooling effects, the cooling effect efficacy of the main species found in those cities, as well as climatic and seasonality variations, could explain the differences between the two studies.

We found that the importance of tree species diversity in explaining cooling effects varies seasonally. This might be due to the fact that the dominant tree species, which each constitute over 4% of the sampled tree numbers, namely *Acer palmatum*, *Metasequoia glyptostroboides*, *Cinnamomum camphora*, *Osmanthus fragrans*, *Magnolia grandiflora.,* exhibit changes in transpiration rates between seasons, from very high levels to in summer to very weak in winter (Mingjuan et al., 2019; Peters et al., 2010). Shading efficiency is also associated with the seasonal variation of leaf cover, resulting in deciduous tree species having very low shading function in winter, while evergreen tree species can provide shade throughout the year. Therefore, the cooling effect of a plant community is consistent with how the seasonal climate changes alters vegetation physiology, as well as the specific characteristics of individual tree species.

The picture for the cooling range of our greenspaces was more complex. There are two sets of factors that will influence the cooling range of a greenspace, one is the characteristics of the greenspace itself such as the size (Zhang et al., 2009) and shape (Du et al., 2017; Lin et al., 2015). The second set of factors relate to the nature of the built up area surrounding the greenspace (Lin et al., 2015; Shiflett et al., 2017). In cities, heat is stored and re-radiated from massive and complex urban structures (Rizwan Ahmed Memon et al., 2008) and these heat sources outside the greenspaces bring uncertainty to the prediction of the cooling range, such as the uneven distribution of buildings and the heat released from air conditioner units in the summer. We found that the mean crown width and tree density of the greenspace can explain part of the variation in the cooling range. In terms of plant community structure, trees with the widest crowns will generate more shading and potentially more evapotranspiration in summer and autumn. However, the explanatory power of our models for the cooling range was substantially lower than those for temperature drop amplitude. CIT is an exception amongst the greenspaces in our study as it provides no cooling to the surrounding area in spring, summer and autumn. There are a number of possible reasons for this. Firstly, there are greater number of impervious surfaces in CIT than other greenspaces. Additionally, the green land that is present has some of the lowest tree coverage, tree density and tree diversity values of all the greenspaces we studied. It is also possible that the urban area surrounding CIT has a better cooling potential than the CIT greenspace itself; specifically there is a river to the east of CIT. Whilst previous research indicates that tree height may also influence temperature reduction (Zhang et al., 2013), We do not find a significant relationship between the mean tree height of a greenspace and its cooling effect (also see (Speak et al., 2020). This emphasizes the greater complexity involved in understanding how cooling effects from greenspaces propagate into the built-up areas that surround them.

More generally, urban greenspaces are composed of trees, shrubs, and grassland, and the components play a different cooling function in different seasons (Feyisa et al., 2014; Oliveira and Costa, 2012; Yu et al., 2020). Trees act as heat-sinks in all seasons, grassland serves as a heat-sink only in summer and spring-autumn, but it become a heat-source in winter as the increased amount of bare soil can change the thermal characteristics of this vegetation type (Yu et al., 2020). Trees maintain greater physiological activity in warm seasons with sufficient water to support them (Oliveira and Costa, 2012). More solar radiation energy can reach the ground surface and lower layers of greenspaces in winter after leaves fall from deciduous trees (Wang et al., 2016). The increasing solar radiation and weak shading and transpiration of plant community decreases the total cooling effect of greenspaces in winter, and may explain the negative correlation between tree density and cooling range. Despite this seasonal variation, enhancing tree diversity within greenspaces remains an important UHIE mitigation tool as cooling effects will be more beneficial to a city and its residents in warmer, summer months, rather than cooler, winter months.

4.3 Conclusions and implications for future study and greenspace design

Urban greenspace planning needs to accommodate a wide range of conflicting demands on space and resources (Littke, 2015). If urban planners wish to make the most use of greenspaces as an UHIE mitigation tool, it will be increasingly important to optimise tree community structure within those greenspaces. Shannon-Wiener diversity index can, therefore, be used as an indicator of the likely cooling effect of tree stands, woodlands and forests as this metric is positively associated with cooling effects in spring, summer and autumn. Given the importance of tree diversity for the delivery of several other ecosystem services (McCarthy et al., 2011), taking such an approach, rather than planting stands of single/few species, is also likely to deliver other benefits for city residents.

Despite the benefits that diverse tree communities can deliver, it is common that one or several tree species become dominant in urban greenspaces (Nagendra and Gopal, 2011). We also found the tree species are not distributed evenly in Changzhou's greenspaces, with a few species, such as *Cinnamomum camphora, Acer palmatum*, *and Osmanthus fragrans*, being substantially more abundant than others. Ensuring that a handful of species do not become dominant is one way that urban planners can raise the Shannon-Wiener diversity index, which takes into account species evenness as well as richness (Strong, 2016). For new greenspaces, landscape designers and city planners could, therefore, include a greater diversity of species, and plant those species evenly if they wish to maximise the Shannon-Wiener diversity index, and the resulting cooling effects. For the existing greenspaces, it may take longer to diversify the tree community structure. This could be done, however, by ensuring that a broader range of tree species are considered when replacing dead or diseased trees.

In addition to diversity, we recommend that canopy coverage, canopy width, and density are all prioritised when it comes to designing greenspaces to maximise their role in mitigating the UHIE. Increasing tree canopy coverage results in a greater cooling effect, as does ensuring that tree crowns are wide (Zhou et al., 2014; Ziter et al., 2019), something that is associated with higher light attenuation and, therefore a greater cooling effect (Speak et al., 2020). In contrast, we do not find a significant relationship between the mean DBH, tree height, and crown height of greenspace and its cooling effect, so these indicators should not be prioritised.

Our findings are based on observational data so are correlational only. Future research should be undertaken using more rigorous designs, such as 'before-after-control-intervention (BACI)' which would enable cooling effects to be more directly attributed to tree community composition metrics that we have identified here. Further potential research topics include: (1) identify how transpiration and shading efficiency varies between tree species. Here we found that tree diversity could enhance the cooling effect of plant community. It is likely that the cooling benefit could be further enhanced if the tree community was also composed of a mix of species that are known to be particularly efficient in providing shade and/or transpiration. However, we still know relatively little about these particular characteristics and how they might result in increased cooling potential (Rahman et al., 2015; Stratópoulos et al., 2018). What we do know, suggests that substantial improvements could be made. For instance in Manchester, UK *Pyrus calleryana* and *Crataegus laevigata* provided 3 to 4 times greater cooling than *Prunus 'Umineko'*, *Sorbus arnoldiana* or *Malus 'Rudolph'* (Rahman et al., 2015).(2) The mechanism that underpins how tree diversity underpins any cooling effect is still unclear. Further studies, including those of a more experimental nature, are required to explore and understand these mechanisms.

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.

CRediT Author Statement

Xinjun Wang: Methodology, Investigation, Writing- Original draft preparation. **Martin Dallimer:** Formal analysis, Writing- Reviewing and Editing. **Catherine E. Scott:** Formal analysis, Writing- Reviewing and Editing. **Weiting Shi:** Investigation**.** **Jixi Gao:** Conceptualization.

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