Research Paper

Spatial and temporal variations in soil respiration among different land cover types under wet and dry years in an urban park

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GRAPHICAL ABSTRACT
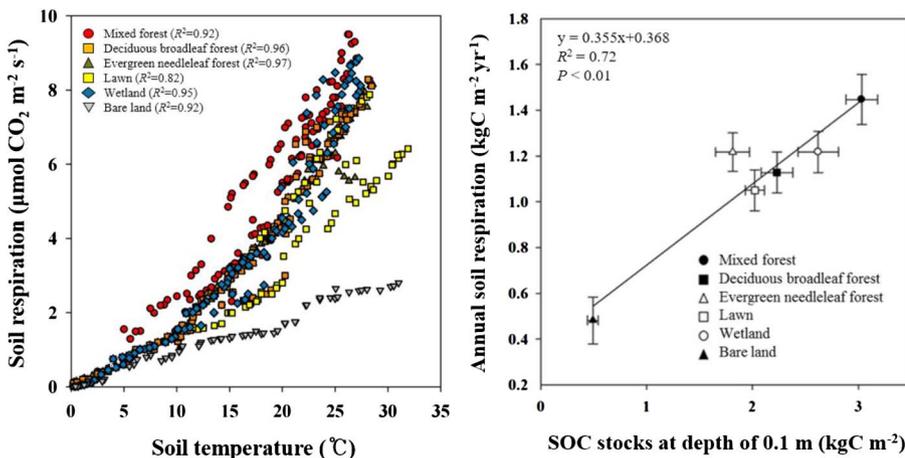
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ABSTRACT

Soil respiration (R_s) determines land surface carbon balance; however, there have been few studies that measured R_s in heterogeneous urban landscapes. Here, we investigated the spatial and temporal variations in R_s in six land cover types (mixed forest, deciduous broadleaf forest, evergreen needleleaf forest, lawn, wetland, and bare land) in Seoul Forest Park, Republic of Korea, between March 2013 and September 2014, which included a wet (2013) and an extremely dry (2014) summer. Spatially, there was a three-fold difference (0.48–1.45 kgC m^{-2}) in annual R_s among the six land cover types. The soil organic carbon stock at a depth of 0.1 m explained 72% of the spatial variation in the annual R_s across the land cover types. During the entire study period, the soil temperature explained 82–97% of the temporal variation in R_s among different land cover types. Comparing the two summers, the 2014 drought only resulted in a decrease in R_s in the lawn plots (25%), which was driven by a reduction in the leaf area index and the fine root density. The temperature sensitivity of R_s in 2014 (dry summer) compared to 2013 (wet summer) was significantly lower in mixed forest, deciduous

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broadleaf forest, and lawn, and did not change in evergreen needleleaf forest, wetland, or bare land. The differences in R_s in these drought responses highlight the importance of the careful selection of land cover type during park planning to better manage carbon cycles.

1. Introduction

Soil respiration (R_s) is one of the largest terrestrial carbon fluxes and determines carbon sink strength (Bond-Lamberty & Thomson, 2010; Valentini et al., 2000). With expected increases in both urban population and urban area (Seto, Guneralp, & Hutyra, 2012), the need for practical carbon management to achieve low-carbon societies has triggered research on R_s in urban ecosystems (Beesley, 2014; Jo & McPherson, 1995; Pouyat, Groffman, Yesilonis, & Hernandez, 2002). For instance, Decina et al. (2016) found that urban R_s accounted for 72% of fossil fuel carbon emissions during the vegetation growing season. Moreover, the annual R_s budget in an urban forest accounted for 33% of annual carbon dioxide emissions in a high-density urban area (Joo, Park, & Park, 2012; Park, Joo, & Park, 2014). Beyond aesthetic and recreational uses, well-managed urban parks can be useful for mitigating and adapting to climate change in urban ecosystems (Bae & Ryu, 2015; Millward & Sabir, 2011). In response to the Paris Agreement and global climate mitigation efforts, a number of cities have implemented strategies to enhance carbon dioxide sequestration in urban parks (Gratani, Varone, & Bonito, 2016; Velasco, Roth, Norford, & Molina, 2016). Thus, accurate monitoring and assessments of changes in R_s in urban parks are of great interest to scientists, park managers, and policymakers (Chen et al., 2013; Kordowski & Kuttler, 2010).

Land cover heterogeneity is an important consideration when quantifying R_s in urban parks. To meet the various needs of citizens, most constructed urban parks include different land cover types. Spatial variations in R_s are affected by both abiotic (e.g., soil temperature and moisture) and biotic (e.g., litterfall and root biomass) factors in different urban forest types (Chen et al., 2013; Wu, Yuan, Ma, Feng, & Zhang, 2015). Furthermore, differences among land cover types and their management strategies alter temporal variations in R_s in urban ecosystems (Jo & McPherson, 1995; Zhang, Tian, Pan, Lockaby, & Chappelka, 2014).

Understanding the effects of extreme drought events on R_s in urban parks is crucial to support carbon management in urban ecosystems. Recent studies have reported that climate change is likely to alter the frequency and intensity of climate extremes in urban ecosystems (Hanson et al., 2011; Stone, Hess, & Frumkin, 2010). For instance, the frequency and severity of summer extremes (e.g., floods and droughts) have increased in multiple cities in South Korea during recent decades (Jung, Choi, & Oh, 2002; Jung, Bae, & Kim, 2011). Borken, Savage, Davidson, and Trumbore (2006) reported that frequent drying of soils strongly affects temporal variations in R_s. In addition, a summary by Kim, Vargas, Bond-Lamberty, and Turetsky (2012) of R_s data from 338 plot-level studies highlighted the significant effects of rewetting on R_s in various terrestrial ecosystems after extreme drought events. Leon et al. (2014) quantified a sudden increase in R_s of up to 522% during rewetting of dry soil following a summer drought in a water-limited ecosystem. Thus, drought effects are directly linked to changes in environmental factors that control the spatial and temporal variations in R_s (Davidson, Belk, & Boone, 1998), and lead to uncertainty when estimating carbon fluxes in terrestrial ecosystems (Schindlbacher et al., 2012).

In our study site, there was more than a two-fold difference in summer (June–August) precipitation between 2013 and 2014. The summer of 2013 experienced the longest consecutive rainfall event recorded since 1960 in Seoul, Korea, resulting in 676 mm of precipitation in July 2013. In contrast, during the same period in 2014, the study site suffered an extreme drought, with precipitation in July

totaling only 208 mm. In comparison, the 30-year (1981–2010) mean precipitation in July is 394 mm. The marked difference in precipitation between 2013 and 2014 provided a unique opportunity to study the responses of diverse land cover types to extreme climate conditions and gain insights into the potential effects of future climate extremes on urban R_s estimates.

In this study, we quantified the spatio-temporal variations in R_s among different land cover types in an urban park with the following objectives: (1) to quantify the annual R_s of six different land cover types; (2) to investigate the factors that control the spatial and temporal variations in R_s; and (3) to understand the influence of extreme drought on R_s across diverse land cover types in an urban park.

2. Methods

2.1. Site description and plot design

The study was conducted in Seoul Forest Park (37.5450°N, 127.0382°E), the third largest urban park in Seoul (116 ha), South Korea. The site is located in a cool temperate zone under the influence of the Asian monsoon climate, and is characterized by dry springs, hot and humid summers, and cold and snowy winters. The mean annual temperature at the site is 12.5 °C, and the mean annual precipitation is 1450 mm, 70% of which falls from June to August (Seoul Station, Korean Meteorological Administration). The two summers in 2013 and 2014 experience the opposite extreme rainfalls. Rainfall records in July are 676 mm and 208 mm for 2013 and 2014, consecutively. The 30-year (1981–2010) mean rainfall in July is 394 mm. The parent material is underlain by Daebo granite from the Quaternary period. According to the Soil and Environmental Information System of Korea (http://soil.rda.go.kr), the soils in the park are classified as Entisols. The site is flat, and the mean altitude is approximately 20 m above sea level (Google Earth ver. 7.1.2.2041).

We randomly selected six plots (10 × 10 m quadrats) in each land cover type. All plots were a minimum of 2 m apart from edges to avoid edge effects (Han, Huang, Liu, Zhou, & Xiao, 2015; Langton, 1990). The land cover types included lawn (n = 6), mixed forest (n = 6), evergreen needleleaf forest (n = 6), deciduous broadleaf forest (n = 6), wetland (n = 6), and bare land (n = 6). The proportions of the land cover types were 15% lawn, 27% mixed forest, 10% evergreen needleleaf forest, 18% deciduous broadleaf forest, 5% wetland, and 2% bare land (Hwang, 2012). The dominant plant species included: *Zoysia japonica* in lawn; *Quercus acutissima*, *Q. mongolica*, and *Pinus rigida* in mixed forest; *P. rigida* and *P. strobus* in evergreen needleleaf forest; *Q. acutissima*, *Q. mongolica*, and *Q. serrata* in deciduous broadleaf forest; and *Phragmites japonica*, *Phragmites communis*, and *Miscanthus sacchariflorus* in wetland. Bare land refers to areas without vegetation cover.

2.2. Data collection

The R_s measurements were conducted from March 2013 to September 2014 with three or four sampling collections per month. The sampling points were permanently deployed with a polyvinyl chloride soil collar (length, 2 cm; inner diameter, 10 cm). Active cooperation with park managers and use of collars similar to the PVC pipes in park facilities enabled us to leave collars in the field. All observations were performed during the daytime, from 09:00 h to 16:00 h (local time). R_s was measured by attaching a LI-COR 6400-09 soil chamber with an area of 71.6 cm^2 to a LI-COR 6400 portable photosynthesis system (LI-COR, Environmental Division, Lincoln, NE, USA). All living vegetation inside

the chamber was removed using tweezers to exclude above-ground plant respiration. The soil chamber was inserted 2 cm into the soil surface. Three measurements were performed for each chamber, from which the mean and 95% confidence interval (CI) were derived for subsequent calculations. The LI-COR 6400 system was calibrated annually by LI-COR technical support services.

We collected soil temperature (T_s) and soil moisture (volumetric water content, VWC) data manually and automatically. Before the R_s observations, we manually measured T_s and VWC in the upper topsoil layer at a depth of 20 cm using a portable soil thermometer (MIC99300; Meter Industrial Company, Inc., Taiwan) and portable HydroSense II with CS658 (20 cm rod) probes (Campbell Scientific, Inc., USA). Continuous T_s and VWC measurement data (GP1 and SM300; Delta-T Devices, Ltd., Cambridge, UK) were collected at a depth of 20 cm from both a deciduous broadleaf forest plot and a lawn plot in the central area of the park. The T_s and VWC sensors were cross-calibrated before and after the experiment.

We measured the living fine root mass density (FRMD) in July 2013 and 2014 in each land cover type. Soil samples were strained through a 2 mm standard testing sieve (Chung Gye Sang Gong Sa, Seoul, Korea) to remove stones. We separated dead roots from living roots with tweezers, washed the fine roots (diameter, < 2 mm) to remove soil, and dried them in an oven at 70 °C for 2 d to determine their dry weight (Olsthoorn, 1991). We directly measured the leaf area index (LAI) in the six lawn plots (0.5 × 0.5 m quadrat) from April to September in both 2013 and 2014. We clipped grass samples, scanned them using a standard desktop scanner (CanoScan LiDE 110; Canon USA, Inc., New York, USA), and measured the leaf areas using MATLAB (Ryu, Lee, Jeon, Song, & Kimm, 2014).

To investigate the drought sensitivity of the different land cover types, we used normalized difference vegetation index (NDVI) maps from Landsat 8 from similar dates during the wet (2013, day of the year (DOY) 259) and dry (2014, DOY 262) years. We downloaded atmospherically corrected, georeferenced spectral reflectance images of Landsat 8 from the United States Geological Survey (https://earthexplorer.usgs.gov). The NDVI was computed as the ratio of (NIR − RED) to (NIR + RED) (Tucker, 1979), where NIR and RED are spectral bands 5 and 4, respectively.

2.3. Data processing

To quantify the annual R_s estimates from the manual observations in each land cover type, we developed regression models for R_s using T_s and VWC. The relationships between R_s and T_s (Lloyd & Taylor, 1994) (Eq. (1)) and between R_s and T_s and VWC (Gaumont-Guay et al., 2006) (Eq. (2)) were evaluated as follows:

$$R_s = ae^{bT_s} \tag{1}$$

$$R_s = [(a + cVWC + d/VWC)][e^{bT_s}] \tag{2}$$

where *a, b, c,* and *d* are the fitted parameters for each equation, R_s is soil respiration (μmol CO_2 m^{-2} s^{-1}), T_s is the soil temperature at a depth of 20 cm (°C), and VWC is the volumetric water content (m^3 m^{-3}). To obtain half-hourly continuous R_s estimates for each land cover type from the models (Eqs. (1) and (2)), we generated continuous T_s and VWC data for each land cover type with linear regression models between the continuous T_s and VWC datasets and the manually observed T_s and VWC data at each plot. To build the linear regression models, we used the continuously measured data from the lawn plots for lawn and the deciduous broadleaf forest plot data for the other land cover types. The coefficients of determination (R^2) of the linear regression models were 0.93–0.97 and 0.83–0.95 for T_s and VWC, respectively. We made continuous R_s estimates for each land cover type by combining the modeled continuous T_s and VWC data with the R_s models (Eqs. (1) and (2)). We used a Q_{10} model (defined as the factor by which the rate of R_s

increases when T_s increases by 10 °C) for the different land cover types to quantify the dependence of R_s on T_s at a depth of 20 cm on both annual and seasonal timescales. The Q_{10} values were calculated using the following equation (Davidson et al., 1998):

$$Q_{10} = e^{10b} \tag{3}$$

where *b* is taken from the regression models of each land cover type (Eq. (1)).

2.4. Statistical analyses

All statistical analyses were performed using SigmaPlot 12.0 (Systat Software, Inc., Chicago, IL, USA). Analysis of variance (ANOVA) followed by Tukey's post hoc test was used to test differences in the monthly or annual mean R_s among the six land cover types. We used the Student's *t*-test to compare the FRMD and NDVI values of the land cover types and assess differences in the monthly lawn LAI (April–September) between 2013 and 2014. All data are presented as the means ± 95% CI, unless otherwise specified.

3. Results

3.1. Spatial variations in annual R_s

The mean annual estimates (March 2013–February 2014) of R_s for the mixed forest, deciduous broadleaf forest, evergreen needleleaf forest, lawn, wetland, and bare land stands were 1.45 ± 0.11, 1.13 ± 0.09, 1.22 ± 0.08, 1.05 ± 0.09, 1.22 ± 0.09, and 0.48 ± 0.10 kgC m^{-2}, respectively (Fig. 1). The annual R_s was significantly higher in mixed forest (1.45 ± 0.11 kgC m^{-2}) than in the other land cover types (ANOVA, P < 0.01). There was no significant difference in the annual R_s among deciduous broadleaf forest, evergreen needleleaf forest, lawn, and wetland (ANOVA, P > 0.05). Bare
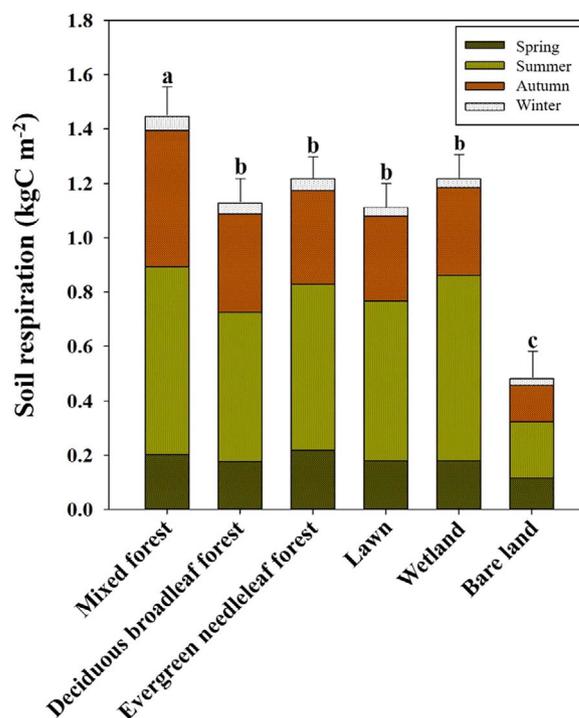

Fig. 1. Estimated seasonal and soil respiration (kgC m^{-2}) from March 2013 to February 2014 in six land cover types. Sum of seasonal values give an annual estimate. The different letters indicate significant differences in annual soil respiration among the land cover types (ANOVA, P < 0.01). The seasonal timescales are as follows: spring (March–May), summer (June–August), autumn (September–November), and winter (December–February). Error bars indicate 95% CI.

land had the lowest annual R_s (0.48 ± 0.10 kgC m^{-2}). The mean R_s in summer (June–August) accounted for 43–56% of the annual R_s among the land cover types. The basal R_s rates at 10 °C (R_{10}), a factor that often used to compare the spatial variation in R_s among ecosystems (Bahn et al., 2008; Li, Yan, Yue, & Wang, 2008), were 2.16, 1.62, 1.89, 1.93, 1.63, and 0.91 μmol CO$_2$ m^{-2} s^{-1} for mixed forest, deciduous broadleaf forest, evergreen needleleaf forest, lawn, wetland, and bare land, respectively. The annual R_s estimates from the Q_{10} models (Eq. (1)) and bivariate models (Eq. (2)) did not differ significantly (t-test, P > 0.05) (data not shown). Across the whole park, annual total R_s estimate (2013 March to 2014 February) was 1087 tC yr^{-1}, which was upscaled from land cover type dependent annual R_s estimates multiplied by each land cover areas.

3.2. Temporal variations in the temperature sensitivity of soil respiration (Q_{10})

T_s explained 82–97% of the temporal variation in R_s among the land cover types during the entire study period (March 2013–September 2014) (Fig. 2). We estimated the temporal variations in the temperature sensitivity of soil respiration (Q_{10}) among the land cover types (Table 1). The mean Q_{10} values on an annual timescale (March 2013–February 2014) for mixed forest, deciduous broadleaf forest, evergreen needleleaf forest, lawn, wetland, and bare land were 2.46 ± 0.10, 2.64 ± 0.08, 2.46 ± 0.05, 2.59 ± 0.08, 2.86 ± 0.11, and 1.82 ± 0.10, respectively. Q_{10} explained > 91% of the annual variation (March 2013–February 2014) in R_s for all land cover types. When analyzed on a seasonal timescale, Q_{10} explained > 75% of the seasonal variation in R_s in each land cover type, except for lawn (47–98%). In all land cover types, the Q_{10} values were higher during winter than in all other seasons. There were significant decreases in Q_{10} during the 2014 summer in all land cover types compared to summer 2013. Comparing the two summers, the reduction of Q_{10} for mixed forest, deciduous broadleaf forest, evergreen needleleaf forest, lawn, wetland, and bare land were −27%, −20%, −4%, −48%, −2%, and −3%, respectively.

3.3. Effects of extreme drought on R_s

During the summer 2014 drought, the R_s rates did not increase with T_s in mixed forest, evergreen needleleaf forest, or lawn (Fig. 3). At the first rainfall after the drought (around DOY 200), we observed an abrupt increase in R_s following rewetting in mixed forest (28%) and evergreen needleleaf forest (42%), but this pattern did not appear in lawn, deciduous broadleaf forest, or wetland.

The living FRMD values in lawn and evergreen forest were significantly lower in July 2014 (dry summer) than in July 2013 (wet summer) (t-test, P < 0.05) (Fig. 4). However, there was no difference in the living FRMD between these two periods in mixed forest, deciduous broadleaf forest, or wetland.

In lawn, the remote sensing-derived NDVI was significantly lower (~25%) after the 2014 drought compared to the same period in summer 2013 (Fig. 5a). The other land cover types did not show changes in NDVI between the two summers. In addition, the 2014 drought significantly reduced the LAI in lawn compared to the same period in 2013 (t-test, P < 0.05) (Fig. 5b).

4. Discussion

4.1. Controlling factors of spatio-temporal variations in R_s among land cover types

The annual R_s budgets in the vegetated urban park (1.13–1.45 kgC m^{-2}) in this study were within the top 3% of those in a meta-analysis of 439 studies in temperate climates (Bond-Lamberty & Thomson, 2010). We assumed that the high annual R_s

values in Seoul Forest Park were related to increases in soil organic carbon (SOC) concentrations (256%) and NDVI (60%) over the past decade (Bae & Ryu, 2015), indicative of higher carbon inputs into the soil. The annual R_s and mean R_{10} values of each land cover type were within the high end of the ranges of published values for similar mid-latitude mixed forests (Davidson et al., 1998; Raich & Schlesinger, 1992) and deciduous forests (Curtis et al., 2002; Mo et al., 2005; Zhou, Guo, & Meng, 2013). Among urban parks, the annual vegetated R_s estimates in this study were similar to those of a deciduous forest (1.26 kgC m^{-2}) in Mt. Nam-san Park, Seoul, Korea (Joo et al., 2012) and substantially higher than those in a mixed forest in Beijing Olympic Park, China (0.47 kgC m^{-2}) (Chen et al., 2013).

The spatial distribution of SOC stocks was positively correlated with the spatial variation in annual R_s (Fig. 6). The largest SOC stock appeared in the mixed forest site, probably due to its relatively well-preserved soil surface layers and high quantity of litter. It is widely accepted that SOC in urban parks is significantly altered by litter management strategies (Takahashi, Amano, Kuchimura, & Kobayashi, 2008; Weissert, Salmond, & Schwendenmann, 2014). In Seoul Forest Park, most litterfall is removed by park managers for the safety of pedestrians and is used to make compost, which is added back into planted areas. However, the mixed forest is located far from paths and recreation areas, reducing direct disturbances to topsoil such as litter removal and soil compaction (Bae & Ryu, 2015). Moreover, mixed forest had a significantly higher NDVI than the other land cover types (Fig. 5a). The seasonal LAI data, determined from digital cover photography (Ryu et al., 2012; Hwang et al., 2016), revealed that mixed forest had the highest maximum LAI (4.4), whereas all other land cover types had LAIs < 3.2 (Kimm & Ryu, 2015). Thus, the higher LAI and NDVI values in mixed forest were indicative of higher carbon inputs into the soil (Ladd et al., 2014), supporting the occurrence of its highest annual R_s.

T_s was the primary factor driving the temporal variations in R_s in the urban park (Fig. 2). The T_s-based Q_{10} models in all land cover types performed well on an annual scale, although the Q_{10} model in the lawn area did not capture well the seasonal dynamics in R_s in both spring seasons. During the spring seasons, the Q_{10} model explained above 83% of the temporal variations in R_s in all land cover types, except for lawn area (R^2 = 0.55 and 0.60 for 2013 and 2014, respectively; Table 1). In lawn, the R_s rates in spring 2013 were 63% lower than those in autumn
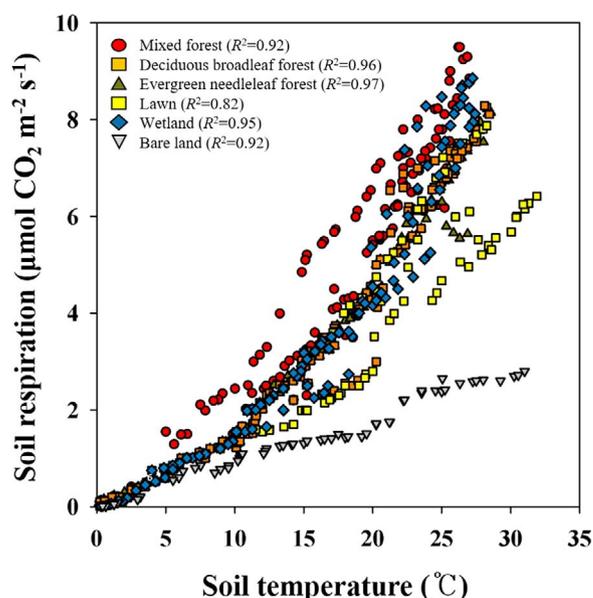

Fig. 2. Relationship between soil respiration (μmol CO$_2$ m^{-2} s^{-1}) and soil temperature (°C) at a depth of 20 cm during the entire study period (March 2013–September 2014) in six land cover types.

Table 1
Temporal variations in the temperature sensitivity of soil respiration (Q_{10}) among six land cover types from March 2013 to August 2014. The annual timescale is March 2013 to February 2014. The seasonal timescales are as follows: spring (March–May), summer (June–August), autumn (September–November), and winter (December–February). Summer 2014 including a drought period. ± represents standard deviation.

Land cover type [Annual Q_{10} (R^2)]	Year	Temperature sensitivity of soil respiration (Q_{10})			
		Spring (R^2)	Summer (R^2)	Autumn (R^2)	Winter (R^2)
Mixed forest	2013	3.42 ± 0.10 (0.83)	2.89 ± 0.10 (0.91)	2.11 ± 0.04 (0.87)	17.90 ± 0.33 (0.88)
[2.46 ± 0.10 (0.91)]	2014	3.79 ± 0.12 (0.85)	2.11 ± 0.07 (0.84)	–	–
Deciduous broadleaf forest	2013	1.79 ± 0.07 (0.90)	2.51 ± 0.10 (0.98)	2.41 ± 0.03 (0.98)	12.06 ± 0.20 (0.91)
[2.64 ± 0.08 (0.94)]	2014	2.25 ± 0.11 (0.89)	2.01 ± 0.07 (0.87)	–	–
Evergreen needleleaf forest	2013	3.00 ± 0.06 (0.98)	2.01 ± 0.03 (0.97)	2.56 ± 0.03 (0.98)	11.02 ± 0.17 (0.93)
[2.46 ± 0.05 (0.97)]	2014	3.21 ± 0.09 (0.97)	1.93 ± 0.08 (0.75)	–	–
Lawn	2013	2.33 ± 0.13 (0.55)	2.41 ± 0.11 (0.87)	2.66 ± 0.03 (0.98)	20.29 ± 0.55 (0.87)
[2.59 ± 0.08 (0.95)]	2014	2.29 ± 0.14 (0.60)	1.25 ± 0.06 (0.47)	–	–
Wetland	2013	2.21 ± 0.07 (0.92)	2.12 ± 0.11 (0.83)	2.36 ± 0.04 (0.94)	22.20 ± 0.56 (0.87)
[2.86 ± 0.11 (0.94)]	2014	3.13 ± 0.12 (0.90)	2.08 ± 0.11 (0.77)	–	–
Bare land	2013	1.65 ± 0.07 (0.88)	1.55 ± 0.05 (0.92)	1.62 ± 0.06 (0.90)	8.36 ± 0.12 (0.87)
[1.82 ± 0.10 (0.95)]	2014	1.54 ± 0.06 (0.90)	1.50 ± 0.08 (0.88)	–	–

2013, which had a similar T_s range (10–20 °C). This difference pattern of R_s so-called "hysteresis effect" (Vargas & Allen, 2008a) suggests that factors other than T_s regulate the seasonal variation in R_s in lawn. Chen et al. (2013) found a similar seasonal hysteresis pattern in R_s in a similar mid-latitude urban forest park in Beijing, China. The hysteresis pattern of R_s might be explained by the soil water deficit within a site-specific threshold value (Davidson et al., 1998). In lawn, the seasonal difference in VWC between spring 2013 (0.11 ± 0.04 m^3 m^{-3}) and autumn 2014 (0.20 ± 0.02 m^3 m^{-3}) was more significant than those of the other land cover types (Fig. 3). Moreover, during the dry summer in the lawn area, T_s (Eq. (1)) only explained 47% of the temporal variation, whereas T_s and VWC (Eq. (2)) explained 64% of the variation. Given that soil moisture influences soil thermal and gas diffusivity (Riveros-Iregui et al., 2007; Vargas & Allen, 2008b), the pronounced seasonal variations in VWC in the lawn area might be related to the decoupling of R_s from T_s. These results are in agreement with previous findings that the difference in soil moisture between spring and autumn is the most important explanatory factor of seasonal variations in R_s in a general monsoon climate (Kominami et al., 2012; Nikolova et al., 2009).

4.2. Effects of extreme drought on spatio-temporal variations in R_s

The summer drought in 2014 influenced the seasonal and spatial variations in R_s, although the effects of the summer drought on R_s were non-uniform across land cover types. Forests and wetland with high living FRMD in the subsoil (depth, 0.3–1.0 m) maintained high R_s rates during the drought or recovered R_s rates quickly after the drought. In the three forest types, Q_{10} varied from 2.01 to 2.89 in the wet summer, but converged around 2 in the dry summer (Table 1). In other words, the temperature sensitivity of R_s in forested plots was heterogeneous under conditions of sufficient soil moisture (2013, wet summer), but it became homogeneous under drought conditions (2014, dry summer). The summer R_s budget in lawn decreased by 25% in 2014, despite the fact that the mean T_s was 3.2 °C higher than that in summer 2013. The Q_{10} in lawn during the dry summer (1.25 ± 0.06) was significantly lower than that during the wet summer (2.41 ± 0.11) (Table 1), implying that the drought strongly altered the temperature sensitivity of R_s in shallow-rooted species.

Park management practices might explain the different drought effects on R_s. A park staff member (Ok-han Bae, pers. comm.) reported
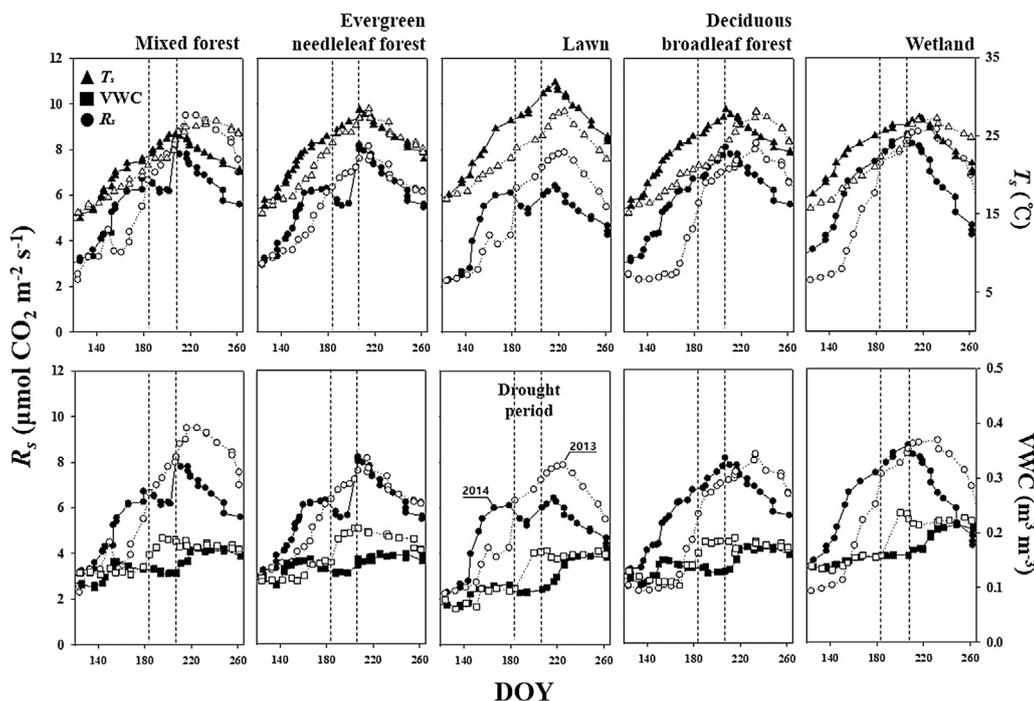

Fig. 3. Temporal variations in soil temperature (T_s), volumetric water content (VWC) at a depth of 20 cm, and soil respiration (R_s) during the growing season in 2013 (empty symbols) and 2014 (filled symbols) among land cover types. The vertical dotted lines indicate the drought period in 2014.
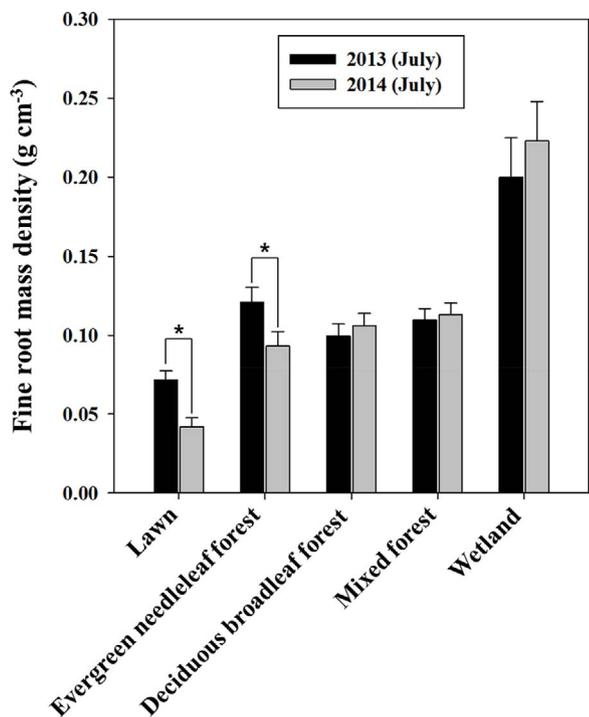
Fig. 4. Comparison of living fine root mass density (FRMD) in July 2013 and July 2014 among land cover types. Asterisks indicate statistically significant differences in FRMD (*t*-test; $P < 0.05$). Error bars indicate 95% CI.
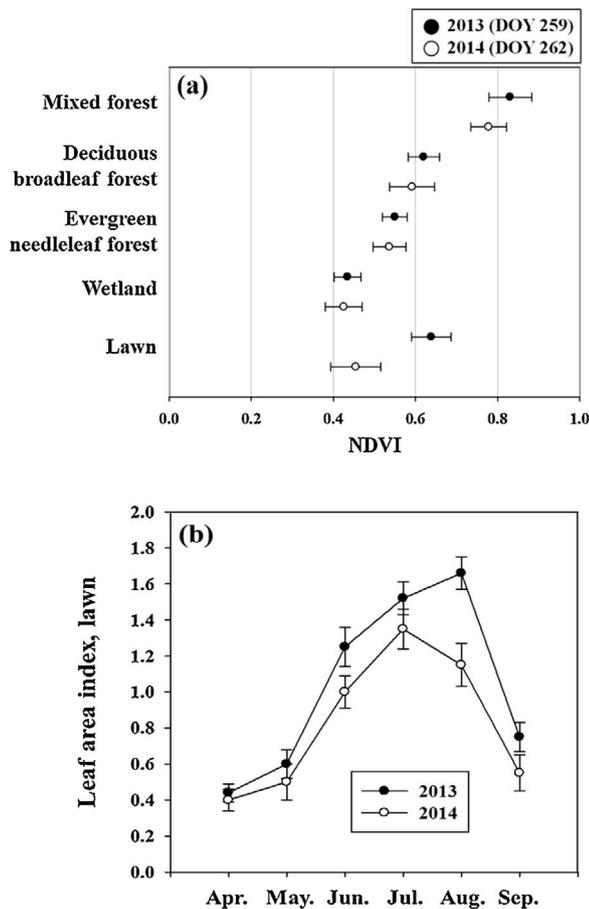

Fig. 5. (a) Comparison of Landsat-derived NDVI after the drought (2014, DOY 262) and in a non-drought year (2013, DOY 259) among land cover types. Error bars indicate 95% CI. (b) Comparison of measured LAI in the lawn area during the growing season (April–September) between 2013 and 2014. Error bars indicate 95% CI.
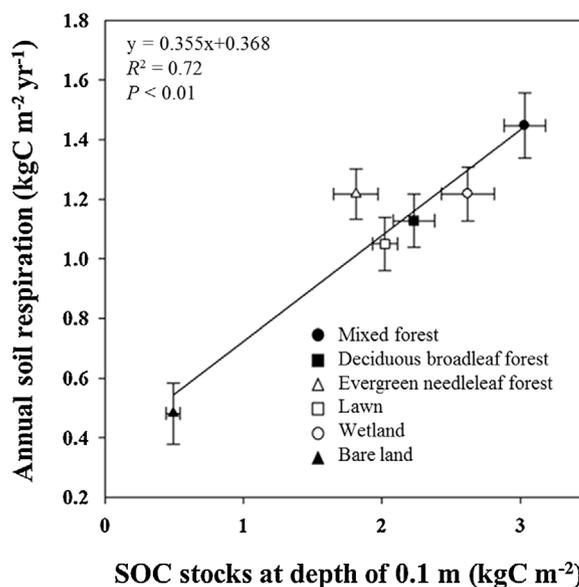

Fig. 6. Relationship between soil organic carbon (SOC) stocks (kgC m^{-2}) at a 0.1 m depth and annual soil respiration (kgC m^{-2} yr^{-1}) among land cover types. Vertical and horizontal bars indicate 95% CI.

that the water supply in Seoul Forest Park in 2014 was 54,011 t, which was 1.56 times higher than that in 2013. The estimated irrigation budgets in summer 2014 for the forested land cover types and lawn were 46.56 kg m^{-2} and 69.84 kg m^{-2}, respectively, which were comparable to the annual irrigation budget in a normal year (Cho, Kim, & Lee, 2005). Therefore, excessive irrigation during the dry summer might have helped minimize the effects of extreme drought in the forested sites, but it was insufficient in the lawn area. Maintaining urban lawns in the study site could require substantial amounts of water resources in the future due to continuous soil warming trends in Korean cities (Cheon, Ham, Lee, Park, & Lee, 2014). The metabolic activity of urban lawns (e.g., stomata closure and transpiration) is highly sensitive to drought, even in the summer wet monsoon period (Lee, Ryu, & Jiang, 2015). In contrast, the wetland area in the park relied on a water-level control system; therefore, the drought had minimal effects on R_s in the wetland. If the water management systems (e.g., irrigation and water level control system) were stopped, the overall carbon cycle in the urban park could change substantially.

The major limitation of this study is that it does not include long-term data, which is essential for testing the robustness of the Q_{10} model. In the limited dataset, we found that the conventional Q_{10} model effectively predicted the annual R_s in the urban park, whereas summer drought had a significant effect on the seasonal variation in R_s (Table 1). In the lawn area in this study, the climate extreme event in 2014 greatly reduced the summer R_s and Q_{10} values, which may have led to different annual Q_{10} values in the lawn area compared to those of 2013 (wet summer). However, this was not tested due to a lack of autumn and winter data in 2014. To better understand the interaction between climate extremes and variability in R_s in urban ecosystems under a warming climate, long-term monitoring systems across diverse land cover types will be essential.

Our findings provide broader implications in carbon managements in urban parks. Park planners and designers must be careful in choosing land cover types by considering large spatial and temporal variations in R_s across different land cover types (Figs. 1 and 2). In particular, R_s in shallow rooting species such as lawn will be very sensitive to drought events. We suggest park managers to collect T_s data for representative land cover types which will be instrumental in estimating R_s across different land cover types (Table 1).

5. Conclusion

There is little information on urban biotic carbon fluxes and their spatial and temporal variations, although they are crucial for informing sustainable urban planning in climate-smart cities. In this study, we investigated the spatial and temporal variations in R_s in Seoul Forest Park, Republic of Korea, between March 2013 and September 2014, which included a wet (2013) and an extremely dry (2014) summer. Spatially, we observed a three-fold difference (0.48–1.45 kgC m^{-2}) in the maximum and minimum annual R_s among the six land cover types assessed in this study. For the whole study period, T_s explained 82–97% of the temporal variations in R_s across the six land cover types, whereas soil moisture was important in explaining the seasonal variations in R_s during the dry period. Except for lawn, there was no significant decrease in R_s during the 2014 drought in any land cover type compared to summer 2013, perhaps due to irrigation practices. The temperature sensitivity of R_s between the two summers varied among land cover types, indicating that drought could have different consequences on R_s in different land cover types under a warming climate. These results contribute to improving our understanding of carbon cycles in different urban land cover types, and can be used to help inform the practices of park planners and managers.

Acknowledgments

References

Bae, J., & Ryu, Y. (2015). Land use and land cover changes explain spatial and temporal variations of the soil organic carbon stocks in a constructed urban park. *Landscape and Urban Planning, 136*, 57–67.

Bahn, M., Rodeghiero, M., Anderson-Dunn, M., Dore, S., Gimeno, C., Drosler, M., et al. (2008). Soil respiration in European grasslands in relation to climate and assimilate supply. *Ecosystems, 11*, 1352–1367.

Beesley, L. (2014). Respiration (CO₂ flux) from urban and pen-urban soils amended with green waste compost. *Geoderma, 223*, 68–72.

Bond-Lamberty, B., & Thomson, A. (2010). Temperature-associated increases in the global soil respiration record. *Nature, 464* [579-U132].

Borken, W., Savage, K., Davidson, E. A., & Trumbore, S. E. (2006). Effects of experimental drought on soil respiration and radiocarbon efflux from a temperate forest soil. *Global Change Biology, 12*, 177–193.

Chen, W. J., Jia, X., Zha, T. S., Wu, B., Zhang, Y. Q., Li, C. Y., et al. (2013). Soil respiration in a mixed urban forest in China in relation to soil temperature and water content. *European Journal of Soil Biology, 54*, 63–68.

Cheon, J. Y., Ham, B. S., Lee, J. Y., Park, Y., & Lee, K. K. (2014). Soil temperatures in four metropolitan cities of Korea from 1960 to 2010: Implications for climate change and urban heat. *Environmental Earth Sciences, 71*, 5215–5230.

Cho, Y., Kim, S., & Lee, J. (2005). In SD Institute (Ed.), *A study on the maintenance of rainwater management facilities: Focused on Mt. Mangwoosan and Seoul Forest Park* (pp. 146–). Seoul: Seoul Development Institute.

Curtis, P. S., Hanson, P. J., Bolstad, P., Barford, C., Randolph, J. C., Schmid, H. P., et al. (2002). Biometric and eddy-covariance based estimates of annual carbon storage in five eastern North American deciduous forests. *Agricultural and Forest Meteorology, 113*, 3–19.

Davidson, E. A., Belk, E., & Boone, R. D. (1998). Soil water content and temperature as independent or confounded factors controlling soil respiration in a temperate mixed hardwood forest. *Global Change Biology, 4*, 217–227.

Decina, S. M., Hutyra, L. R., Gately, C. K., Getson, J. M., Reinmann, A. B., Gianotti, A. G. S., et al. (2016). Soil respiration contributes substantially to urban carbon fluxes in the greater Boston area. *Environmental Pollution, 212*, 433–439.

Gaumont-Guay, D., Black, T. A., Griffis, T. J., Barr, A. G., Jassal, R. S., & Nesic, Z. (2006). Interpreting the dependence of soil respiration on soil temperature and water content in a boreal aspen stand. *Agricultural and Forest Meteorology, 140*, 220–235.

Gratani, L., Varone, L., & Bonito, A. (2016). Carbon sequestration of four urban parks in Rome. *Urban Forestry & Urban Greening, 19*, 184–193.

Han, T. F., Huang, W. J., Liu, J. X., Zhou, G. Y., & Xiao, Y. (2015). Different soil respiration responses to litter manipulation in three subtropical successional forests. *Scientific Reports, 5*, 8.

Hanson, S., Nicholls, R., Ranger, N., Hallegatte, S., Corfee-Morlot, J., Herweijer, C., et al. (2011). A global ranking of port cities with high exposure to climate extremes. *Climatic Change, 104*, 89–111.

Hwang, Y., Ryu, Y., Kimm, H., Jiang, C., Lang, M., Macfarlane, C., et al. (2016).

Correction for light scattering combined with sub-pixel classification improves estimation of gap fraction from digital cover photography. *Agricultural and Forest Meteorology, 222*, 32–44.

Hwang, J. (2012). *A study on the effectiveness of carbon dioxide sequestration of urban park – The case study of Seoul Forest Park*. Graduate School of Engineering, Hanyang University.

Jo, H. K., & McPherson, E. G. (1995). Carbon storage and flux in urban residential greenspace. *Journal of Environmental Management, 45*, 109–133.

Joo, S. J., Park, S. U., Park, M. S., & Lee, C. S. (2012). Estimation of soil respiration using automated chamber systems in an oak (Quercus mongolica) forest at the Nam-San site in Seoul, Korea. *Science of the Total Environment, 416*, 400–409.

Jung, H. S., Choi, Y., Oh, J. H., & Lim, G. H. (2002). Recent trends in temperature and precipitation over South Korea. *International Journal of Climatology, 22*, 1327–1337.

Jung, I. W., Bae, D. H., & Kim, G. (2011). Recent trends of mean and extreme precipitation in Korea. *International Journal of Climatology, 31*, 359–370.

Kim, D. G., Vargas, R., Bond-Lamberty, B., & Turetsky, M. R. (2012). Effects of soil rewetting and thawing on soil gas fluxes: A review of current literature and suggestions for future research. *Biogeosciences, 9*, 2459–2483.

Kimm, H., & Ryu, Y. (2015). Seasonal variations in photosynthetic parameters and leaf area index in an urban park. *Urban Forestry & Urban Greening, 14*, 1059–1067.

Kominami, Y., Jomura, M., Ataka, M., Tamai, K., Miyama, T., Dannoura, M., et al. (2012). Heterotrophic respiration causes seasonal hysteresis in soil respiration in a warm-temperate forest. *Journal of Forest Research, 17*, 296–304.

Kordowski, K., & Kuttler, W. (2010). Carbon dioxide fluxes over an urban park area. *Atmospheric Environment, 44*, 2722–2730.

Ladd, B., Peri, P. L., Pepper, D. A., Silva, L. C. R., Sheil, D., Bonser, S. P., et al. (2014). Carbon isotopic signatures of soil organic matter correlate with leaf area index across woody biomes. *Journal of Ecology, 102*, 1606–1611.

Langton, S. (1990). Avoiding edge effects in agroforestry experiments − The use of neighbor-balanced designs and guard areas. *Agroforestry Systems, 12*, 173–185.

Lee, S., Ryu, Y., & Jiang, C. (2015). Urban heat mitigation by roof surface materials during the East Asian summer monsoon. *Environmental Research Letters, 10*(12), 124012.

Leon, E., Vargas, R., Bullock, S., Lopez, E., Panosso, A. R., & La Scala, N. (2014). Hot spots, hot moments, and spatio-temporal controls on soil CO₂ efflux in a water-limited ecosystem. *Soil Biology & Biochemistry, 77*, 12–21.

Li, H. J., Yan, J. X., Yue, X. F., & Wang, M. B. (2008). Significance of soil temperature and moisture for soil respiration in a Chinese mountain area. *Agricultural and Forest Meteorology, 148*, 490–503.

Lloyd, J., & Taylor, J. A. (1994). On the temperature dependence of soil respiration. *Functional Ecology, 8*, 315–323.

Millward, A. A., & Sabir, S. (2011). Benefits of a forested urban park: What is the value of Allan Gardens to the city of Toronto, Canada? *Landscape and Urban Planning, 100*, 177–188.

Mo, W., Lee, M. S., Uchida, M., Inatomi, M., Saigusa, N., Mariko, S., et al. (2005). Seasonal and annual variations in soil respiration in a cool-temperate deciduous broad-leaved forest in Japan. *Agricultural and Forest Meteorology, 134*, 81–94.

Nikolova, P. S., Raspe, S., Andersen, C. P., Mainiero, R., Blaschke, H., Matyssek, R., et al. (2009). Effects of the extreme drought in 2003 on soil respiration in a mixed forest. *European Journal of Forest Research, 128*, 87–98.

Olsthoorn, A. F. M. (1991). Fine root density and root biomass of two Douglas-fir stands on sandy soils in the Netherlands. 1. Root biomass in early summer. *Netherlands Journal of Agricultural Science, 39*, 49–60.

Park, M. S., Joo, S. J., & Park, S. U. (2014). Carbon dioxide concentration and flux in an urban residential area in Seoul, Korea. *Advances in Atmospheric Sciences, 31*, 1101–1112.

Pouyat, R., Groffman, P., Yesilonis, I., & Hernandez, L. (2002). Soil carbon pools and fluxes in urban ecosystems. *Environmental Pollution, 116*, 107–118.

Raich, J. W., & Schlesinger, W. H. (1992). The global carbon dioxide flux in soil respiration and its relationship to vegetation and climate. *Tellus, 44*, 81–99.

Riveros-Iregui, D. A., Emanuel, R. E., Muth, D. J., McGlynn, B. L., Epstein, H. E., Welsch, D. L., et al. (2007). Diurnal hysteresis between soil CO₂ and soil temperature is controlled by soil water content. *Geophysical Research Letters, 34*, 5.

Ryu, Y., Verfaillie, J., Macfarlane, C., Kobayashi, H., Sonnentag, O., Vargas, R., et al. (2012). Continuous observation of tree leaf area index at ecosystem scale using upward-pointing digital cameras. *Remote Sensing of Environment, 126*, 116–125.

Ryu, Y., Lee, G., Jeon, S., Song, Y., & Kimm, H. (2014). Monitoring multi-layer canopy spring phenology of temperate deciduous and evergreen forests using low-cost spectral sensors. *Remote Sensing of Environment, 149*, 227–238.

Schindlbacher, A., Wunderlich, S., Borken, W., Kitzler, B., Zechmeister-Boltenstern, S., & Jandl, R. (2012). Soil respiration under climate change: Prolonged summer drought offsets soil warming effects. *Global Change Biology, 18*, 2270–2279.

Seto, K. C., Guneralp, B., & Hutyra, L. R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proceedings of the National Academy of Sciences of the United States of America, 109*, 16083–16088.

Stone, B., Hess, J. J., & Frumkin, H. (2010). Urban form and extreme heat events: Are sprawling cities more vulnerable to climate change than compact cities? *Environmental Health Perspectives, 118*, 1425–1428.

Takahashi, T., Amano, Y., Kuchimura, K., & Kobayashi, T. (2008). Carbon content of soil in urban parks in Tokyo, Japan. *Landscape and Ecological Engineering, 4*, 139–142.

Tucker, C. J. (1979). Red and photographic infrared linear combinations for monitoring vegetation. *Remote Sensing of Environment, 8*, 127–150.

Valentini, R., Matteucci, G., Dolman, A. J., Schulze, E. D., Rebmann, C., Moors, E. J., et al. (2000). Respiration as the main determinant of carbon balance in European forests. *Nature, 404*, 861–865.

Vargas, R., & Allen, M. F. (2008a). Diel patterns of soil respiration in a tropical forest after

Hurricane Wilma. *Journal of Geophysical Research-Biogeosciences*, 113.

Vargas, R., & Allen, M. F. (2008b). Environmental controls and the influence of vegetation type, fine roots and rhizomorphs on diel and seasonal variation in soil respiration. *New Phytologist, 179*, 460–471.

Velasco, E., Roth, M., Norford, L., & Molina, L. T. (2016). Does urban vegetation enhance carbon sequestration? *Landscape and Urban Planning, 148*, 99–107.

Weissert, L. F., Salmond, J. A., & Schwendenmann, L. (2014). A review of the current progress in quantifying the potential of urban forests to mitigate urban CO_2

emissions. *Urban Climate, 8*, 100–125.

Wu, X., Yuan, J., Ma, S., Feng, S., Zhang, X., & Hu, D. (2015). Seasonal spatial pattern of soil respiration in a temperate urban forest in Beijing. *Urban Forestry & Urban Greening, 14*, 1122–1130.

Zhang, C., Tian, H., Pan, S., Lockaby, G., & Chappelka, A. (2014). Multi-factor controls on terrestrial carbon dynamics in urbanized areas. *Biogeosciences, 11*, 7107–7124.

Zhou, Z. Y., Guo, C., & Meng, H. (2013). Temperature sensitivity and basal rate of soil respiration and their determinants in temperate forests of North China. *PLos One, 8*, 9.