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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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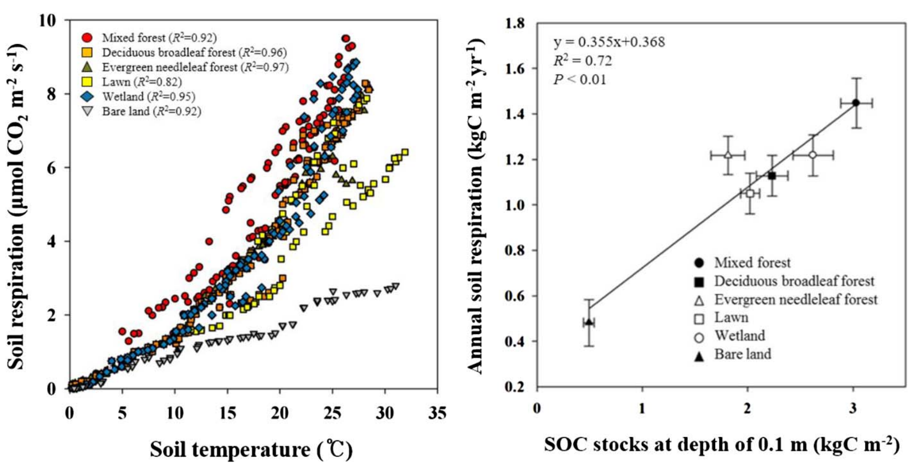
GRAPHICAL ABSTRACT



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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