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Traces of urban forest in temperature and CO_2 signals in monsoon East Asia

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Abstract. Cities represent a key space for our sustainable trajectory in a changing environment, and our society is steadily embracing urban green space for its role in mitigating heatwaves and anthropogenic CO_2 emissions. This study reports two-year surface fluxes of energy and CO_2 measured via the eddy covariance method in an artificially constructed urban forest to examine the impact of urban forests on air temperature and net CO_2 exchange. The urban forest site shows typical seasonal patterns of forest canopies with the seasonal march of the East Asian summer monsoon. Our analysis indicates that the urban forest reduces both the warming trend and urban heat island intensity compared to the adjacent high-rise urban areas and that photosynthetic carbon uptake is large despite relatively small tree density and leaf area index. During the significant drought period in the second year, gross primary production and evapotranspiration decreased, but their reduction was not as significant as those in natural forest canopies. We speculate that forest management practices, such as artificial irrigation and fertilization, enhance vegetation activity. We also stipulate that ecosystem respiration in urban forests is more pronounced than typical natural forests in a similar climate zone. This can be attributed to the substantial amount of soil organic carbon available due to intensive historical soil use and soil transplantation during forest construction, as well as relatively warmer temperatures in urban heat domes. Our observational study also indicates the need for caution in soil management for less CO_2 emissions in urban areas.

1 Introduction

Cities inhabit only 2% of the Earth's land surface but hold more than 55% of the world's population. With the unprecedented rapid urbanization in the last century, our life trajectory heavily depends on urban structures and functions, and it is expected that the urban population will increase by up to 68% by 2050 (UN, 2019). Our current concern is regarding the disastrous impacts of climatic events (e.g., heatwaves, flooding, and drought) and environmental changes (e.g., air pollution and land degradation) on our socioeconomic system in a changing climate (McCarthy et al., 2010; Rahmstorf and Coumou, 2011). Accordingly, it remains an urgent issue to implement integrated policies for climate change mitigation and adaption toward sustainable cities against global warming and related natural disasters. In particular, urban green infrastructures, such as urban forest, have been recognized as a key solution toward alleviating climatic and environmental disasters (e.g., Oke et al., 2017; Chiesura, 2004; Haaland and van den Bosch, 2015; Kroeger et al., 2019). Green spaces in cities, as opposed to gray spaces, are exposed to wide ranges of environmental and climatic conditions across geographical locations.

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Especially when green spaces replace gray infrastructures during the urban redevelopment, it remains unclear whether their benefits emerge in real conditions and thereby outperforming their maintenance cost and other harmful effects. To leverage their full potential benefits, it is necessary to assess the biophysical effects of urban forests based on direct long-term monitoring in urban areas.

Urban forests are a key part of green infrastructures in a city, and two of their benefits, which have been mainly addressed in previous studies, are thermal mitigation and carbon uptake (Roy et al., 2012; Oke et al., 2017). Firstly, urban forests mitigate direct sunlight and diminish the incoming radiant energy on the land surface, thereby reducing surface temperature. Additionally, urban forests supply water to the atmosphere through transpiration and retain water for longer times than the impervious surfaces of urban structures. These processes contribute toward reducing air temperature by partitioning more available energy to latent heat flux (Q_E) than sensible heat flux (Q_H), thus creating favorable conditions for mitigating heatwaves and related health problems (e.g., Oke, 1982; Hong et al., 2019a). Eventually, this cooling effect reduces the electrical energy load of buildings, as well as greenhouse gas emissions. Previous studies have reported cooling effects of urban forests from street trees to parks scales (Oke et al., 1989; Bowler et al., 2010; Norton et al., 2015; Shashua-Bar and Hoffman, 2000). Such cooling effects depend not only on tree species and structures (Feyisa et al., 2014) but also on the size and vegetation density of urban green areas (Yu and Hien, 2006; Chang et al., 2007; Hamada and Ohta, 2010; Feyisa et al., 2014). However, despite the strong temperature-controlling factors of evapotranspiration (ET) and direct heat fluxes over urban forest canopies, only a few studies have reported surface energy balance (SEB) in urban forests in relation to thermal mitigation based on direct measurements (e.g., Oke et al., 1989; Spronken-Smith et al., 2000; Coutts et al., 2007a; Ballinas and Barradas, 2015; Hong and Hong, 2016;). Moreover, it is noticeable that forest cooling intensity depends on geography and even forests can produce a warming trend with the decreased albedo (Bonan, 2008; Wang et al., 2018). The lack of direct urban forest measurements hinders proper assessment of their influences on the climate and environment.

Furthermore, urban forests mitigate anthropogenic carbon emissions by photosynthetic CO_2 uptake. Traditionally, carbon uptake by urban forests has been estimated by empirical relationships (e.g., biomass allometric equation) or short-term inventory of biomass data and vegetation growth rates, which have limitations of spatiotemporal coverage (Rowntree and Nowak, 1991; Nowak, 1993; Nowak et al., 2008; Weissert et al., 2014). Currently, the eddy covariance (EC) method is being applied in various ecosystems from grasslands and natural forests to urban areas because it provides continuous net CO_2 flux measurements at the neighborhood scale every half hour (Christen 2014). From this perspective, the EC method is useful for studying the net CO_2 exchange (F_C) from diurnal to interannual variations, with its simultaneous measurement of surface energy fluxes. Recently, direct F_C measurements have been performed using the EC method in urban green spaces to examine SEB and carbon exchange (Coutts et al., 2007a, 2007b; Awal et al., 2010; Bergeron and Strachan, 2011; Crawford et al., 2011; Kordowski and Kuttler, 2010; Peters and McFadden, 2012; Velasco et al., 2013; Ward et al., 2013; Ueyama and Ando, 2016; Hong et al., 2019b; Hong et al., 2020). However, the EC method provides only the net effects of CO_2 exchange from various carbon sources and sinks, which limits the physical interpretation and assessment of the benefits and costs of urban forests.

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Flux partitioning into photosynthesis and ecosystem respiration from the EC measured F_C requires additional information and data processing (Stoy et al., 2006). It is more challenging to partition F_C into individual sources and sinks, particularly in urban areas because of the complex contributions from biogenic (e.g., vegetation photosynthesis, respiration of vegetation, soil, and humans) and extra anthropogenic (e.g., fossil fuel combustion by transportation or in households and commercial buildings) processes (Pataki et al., 2003). Stochastic F_C partitioning methods were recently applied by reprocessing EC observation data with auxiliary data and provided useful knowledge on urban carbon cycle (Hiller et al., 2011; Crawford and Christen, 2015; Menzer and McFadden, 2017; Stagakis et al., 2019).

With this background, the objectives of this study include: 1) reporting temporal changes in air temperature after the artificial construction of an urban forest park in the Seoul metropolitan area where a hot and humid summer season affects and shows steep global warming trends (Hong and Hong, 2016) and 2) quantifying the carbon uptake of urban forests based on the F_C partitioning through the data observed by the EC method and meteorological data (Lee et al., 2021). Here, we highlight the biotic and abiotic factors controlling the carbon cycle in urban forests and the impact of urban forests on the thermal environment after forest park construction.

2 Materials and Methods

2.1 Urban surface energy and CO_2 balances

The SEB is expressed as:

$$Q^* + Q_F = Q_H + Q_E + \Delta Q_S + \Delta Q_A \qquad (1)$$

where Q^* is the net all-wave radiation of the sum of outgoing and incoming short- and long-wave radiative fluxes, Q_F is the anthropogenic heat flux, Q_H is the turbulent sensible heat flux, Q_E is the latent heat flux, ΔQ_S is the net storage heat flux, and ΔQ_A is the net heat advection (Definitions of variables in Appendix A).

The surface CO_2 budget in an urban forest is formulated as follows:

$$F_C = E_R + E_B + RE - GPP \equiv E_R + E_B + NBE \qquad (2)$$

where F_C is the net CO_2 exchange at the city-atmosphere interface, E_R and E_B are the anthropogenic CO_2 emissions from fossil fuel combustion by vehicles and heating in a building, respectively. GPP and RE are biotic contributions to F_C; GPP is the gross primary production as a result of photosynthetic CO_2 uptake, and RE is the ecosystem respiration from soil and vegetation. Urban ecosystem respiration considers not only the autotrophic and heterotrophic respirations of vegetation and soil but also human respiration (Moriwaki and Kanda, 2004; Velasco and Roth, 2010; Ward et al., 2013, 2015; Hong et al., 2020). Human respiration is negligible in this study because there is no residential population in the park. Vegetation in urban areas includes trees and lawns in urban forests, as well as gardens and roadsides, and it offsets CO_2 emissions through CO_2 assimilation by photosynthesis as the only carbon sink.

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Additionally, *NBE* is the net biome CO_2 exchange and is typically defined as the net ecosystem exchange by *RE* – *GPP* for natural vegetation. Put differently, *NBE* refers to carbon losses in heterotrophic respiration minus the net primary production on natural vegetative surfaces; thus, negative *NBE* indicates the net carbon uptake by the natural ecosystem (Kirschbaum et al., 2001; Randerson et al., 2002). Unlike natural ecosystems, the F_C between an urban forest and atmosphere is a complex mixture of biogenic (i.e., *GPP* and *RE*) and anthropogenic (i.e., E_R and E_B) processes across various spatial and temporal scales. In urban environments, anthropogenic emissions depend on the local characteristics (e.g., climate, population density, levels of industrial activity, and existing carbon intensity of electricity supply) of the city and locations of the eddy covariance system (Feigenwinter et al., 2012; Kennedy et al., 2014; Lietzke et al., 2015; Stagakis et al., 2019).

2.2 Site description

2.2.1 Seoul Forest Park

Micrometeorological measurements were taken at the Seoul Forest Park (SFP) in the Seoul metropolitan area, Korea (37.5446°N, 127.0379°E). SFP is the third largest park in Seoul with an area of 1.16 km^2 (Fig. 1a). This area had been used as a horse racetrack and a golf course inside the track since 1950 and was surrounded by cement factories to the west (Fig. 1b). The local government initially planned this area as a commercial district with a high-rise multi-purpose building complex but changed its plan to redevelop the area as a green space in late 1990s. The construction of the SFP began in December 2003, and it was opened to the public in June 2005 (Fig. 1c).

The dominant land cover within a 300-m radius of the measurement system is a deciduous forest with irrigated grass lawns (*Zoysia*), oak (*Quercus acutissima*), ginkgo (*Ginkgo biloba*), and ash trees (*Fraxinus rhynchophylla*), which correspond to the Local Climate Zone (LCZ) 'A', dense trees (Stewart and Oke, 2012). The maximum leaf area index (LAI) of 300 × 300 m^2 around the SFP tower is approximately 1.6 (Copernicus Service information, 2020). On the east side (0–120°), there are trees (approximately 230 stems ha^{-1}) with a small artificial lake and grasslands beyond it. Trees mainly occupy the southern and western directions of a tower (120–330°) within a 100-m radius area (approximately 540 stems ha^{-1}) and traffic roads lie outside of the dense vegetation. The mean tree height (h_c) is approximately 7.5 m and ranges between 5.8–9.5 m. The mean roughness length (z_0) and zero-plane displacement height (z_d) are estimated by the tree height-based approach within 100 m radius and they range between 0.3–0.6 and 4.1–8.2 m, respectively (Raupach et al., 1991). z_0 and z_d have seasonal and directional variations depending on the variability of the leaves on the vegetation (Lee, 2015; Kent et al., 2018). z_0 and z_d change from approximately 0.6 and 5.0 m during leaf-on period (June–August) to 1.2 and 3.0 m during the leaf-off periods (December–February) by the Macdonald method (Macdonald et al., 1998). Approximately 80% of the footprint area of the SFP tower is within 250 m (Fig. 1e).

The traffic roads consist of eight and ten lanes carrying heavy traffic throughout the day (~100,000 vehicles day^{-1}) to the south and west of the tower (Fig. 1c). Hourly traffic volume, which is used for surface flux partitioning, is evaluated on the road adjacent to the SFP tower every year by the Seoul Metropolitan Government

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(https://topis.seoul.go.kr). Across the road on the western side of the tower, a cement factory still exists, although its size is smaller than it used to be in the past (Fig. 1b and 1c).

2.2.2 Climate conditions

Climatic condition shows a distinct seasonal variation with the seasonal march of the East Asian summer monsoon. The mean climatological values (1981-2010) of the screen-level air temperature (T_{air}) and precipitation were 12.5°C and 1450 mm year^{-1}, respectively. During the study period (June 2013–May 2015), the observed T_{air} was higher than the climatological mean. Higher temperatures lasted longer in the summer of 2013 with the stagnation of the migratory anticyclones (June) and North Pacific anticyclone (July–August). There were strong heatwaves in the spring seasons of 2014 and 2015 (Hong et al., 2019a).

Notably, seasonal precipitation shows a contrasting pattern between two consecutive years (Fig. 2d). In the first year (June 2013–May 2014), annual precipitation was 1256 mm, which corresponded to approximately 90% of the climatological mean. In addition, approximately 50% of the annual rainfall was concentrated in the summer with an estimated 650 mm occurring only in July 2013; however, in the second year the annual rainfall was 932 mm (i.e., 67% of the climatological mean). The monthly precipitation values in the July and August of 2014 were 198 and 169 mm, respectively, which represented only approximately 35% of the climate mean. Accordingly, the vapor pressure deficit (*VPD*) and downward shortwave radiation ($K_{\downarrow}$) in July 2013 were relatively smaller than those in July 2014 (Fig. 2b and 2c).

2.2.3 Observations in the Seoul Metropolitan Area

In this study, meteorological data from six stations (one eddy covariance station, one aerodrome meteorological observation station, and four automatic weather stations) in the Seoul Metropolitan Area are additionally analyzed to examine the heat mitigation and CO_2 reduction effects of urban vegetation in the SFP (Table 1 and Fig. 1a). The Eunpyeong eddy covariance site (EP, 37.6350°N, 126.9287°E) is for surface flux observations in the northwest of Seoul, where there was a recent urban redevelopment to high-rise and high-population residential areas from low-rise areas (Hong and Hong, 2016; Hong et al., 2019b). Flux observations at the site have been conducted since 2012, and they show the SEB and turbulence characteristics of a typical urban residential area. Because the area around the SFP was originally planned to be redeveloped to high-rise high-population residential buildings, EP is selected for comparative analysis as a hypothetical place for the SFP region because they are close to each other and so have the similar synoptic conditions.

The Gimpo Airport Observatory (GP, 37.5722°N, 126.7751°E) is located on the western boundary of Seoul, and it is surrounded by grasslands and croplands, which corresponds to LCZ 'D'. As the dominant wind comes from the west, the GP site is generally affected by the same synoptic weather conditions as Seoul. The GP station represents the rural environment of the Seoul Metropolitan Area because urban development is restricted around the airport (Hong et al., 2019a). In this study, we select the GP site as a reference point and calculate the urban heat island intensity (UHIi) as the synchronous difference in T_{air} between the urban and rural areas accordingly (Stewart, 2011).

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The Seongdong Observatory (SD, 37.5472°N, 127.0389°E), the closest station to the SFP, is located approximately 300 m north of the SFP tower (Fig. 1c). Since the station began observations in August 2000, the meteorological data at SD are useful for analyzing temperature changes before and after the construction of the SFP. Accordingly, it is used to analyze local climatic changes caused by the SFP. Moreover, SD provides auxiliary weather variables (e.g., precipitation) that are not observed in SFP station and reference data for surface flux gap filling. The Gangnam, Seocho, and Songpa observatories (hereafter denoted as AVG) are located in Seoul's central business district, which corresponds to LCZ 1 or 2. These sites are also close to the SFP (~ 5 km); thus, temperatures in these regions can be assumed to be exposed to the same synoptic condition. These regions show greater UHIi than other parts of Seoul because of dense skyscrapers, according to the analysis of the spatial distribution of UHIi in Seoul (Hong et al., 2013). The average temperature of these three automatic weather stations is used to evaluate the temperature and UHIi reduction effects of the SFP construction. All meteorological data from the automatic weather station and aerodrome meteorological observation station are observed every minute, and they are averaged for 1 h for UHIi analysis. All the meteorological data are processed for quality control on the National Climate Data Portal of the Korea Meteorological Administration (http://data.kma.go.kr).

2.3 Instrumentation and data processing

The measurement system was installed on the rooftop of the SFP facility building (Fig. 1d). A three-dimensional sonic anemometer (CSAT3A, Campbell Scientific, USA) and enclosed infrared gas analyzer (EC155, Campbell Scientific, USA) were mounted 12.2 m above the ground level (2.8 m above the roof of an 8.4 m high building) for 2 years (Fig. 1d). The eddy covariance data were recorded using the data logger (CR3000, Campbell Scientific, USA) with a 10-Hz sampling rate and a 30-min averaging time. The gas analyzer was calibrated with standard CO_2 gas every three months. The main footprint covered the forest canopies, and the measurement height (z_m) satisfied the tower height requirement over forested or more structurally complex ecosystems (i.e., $z_m \cong z_d + 4(h_c - z_d)$) (Munger et al., 2012). Two radiometers (NR Lite2 and CMP3, Kipp&Zonen, Netherlands) were used to measure the radiative fluxes. An auxiliary measurement included a humidity and temperature probe (HMP155A, Vaisala, Finland).

The 30-min flux is computed using EddyPro (6.2.0 version, LI-COR), with the applications of the double rotation, time lag compensation using covariance maximization, spike removal and quality test (Vickers and Mahrt, 1997), spectral corrections for low-frequency (Moncrieff et al., 2004) and high-frequency (Fratini et al., 2012), as well as vertical sensor separation correction (Horst and Lenschow, 2009). We apply the following post processes for quality control: 1) plausible value check, 2) spike removal, and 3) discarding the negative F_C flux during the nighttime (Hong et al., 2020). The total study period from installation (31 May, 2013) to termination (03 June, 2015) is approximately 2 years (35,174 potential 30-min data), and the total available data are approximately 90.1%, 88.3%, and 85.4% (n = 31709, 31064, 30028) for Q_H, Q_E, and F_C after the processes, respectively.

It is important to partition the F_C into four contributing components (i.e., RE, GPP, E_R, and E_B in Eq. 2) to investigate their biotic and abiotic controlling factors in an artificially constructed park. This study applies for a

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statistical partitioning method described in Lee et al. (2021). More information and relevant figures on the flux partitioning are available in Lee et al. (2021).

3 Results and discussion

3.1 Surface energy balance

The SEB at the SFP shows typical seasonal variations over natural forest canopies with the seasonal march of the East Asian monsoon (Fig. 4) (Hong and Kim, 2011; Hong et al., 2019b; Hong et al., 2020). In summer, there are lengthy rainy spells and large temporal variabilities of meteorological conditions with the impacts of the East Asian summer monsoon (Fig. 2d). This heavy rainfall causes substantial decreases in $K_{\downarrow}$, and thus Q^*, with large temporal variations, thereby leading to the mid-summer depression of surface fluxes (Fig. 2c and 4). Q^* also reaches its maximum in spring rather than in summer and decreases gradually from spring to winter (Fig. 4). More than half of Q^* is partitioned to Q_E, and Q_H is minimum in summer owing to the ample water supply from the summer rainfall. However, Q_H is maximum in spring and even larger in winter, despite the relatively smaller Q^*, because of the cold and dry climatic conditions induced by the winter monsoon. Accordingly, the seasonal mean Bowen ratio ($\beta = \sum Q_H / \sum Q_E$) ranges from near zero (summer) to approximately 4 (winter) with its daily maximum around 9 in early January 2015 (Fig. 5). Notably, β in the SFP is consistently lower than the high-rise, high-density residential area (i.e., the EP site) because of the ET from the vegetative canopies and the unpaved surfaces in the urban forest. This difference between the two distinct sites confirms that urban forests are responsible for substantial changes in the thermal environment in terms of Q_H and Q_E, as well as their related air and surface temperatures because of more evaporative cooling in green spaces compared to impervious surfaces such as roads and buildings in urban areas (Oke et al., 2017).

The SEB also shows interannual variabilities over forest canopies influenced by the timing of the onset and duration of the summer monsoon (Hong and Kim, 2011) (Fig. 6). As discussed in Section 2.1.2, annual precipitation is much larger in the first year than in the second year because of the interannual variations in the East Asian monsoon activity, thereby making substantial differences in surface radiative fluxes. Furthermore, Q_E shows the difference between the first and second years of the observation, particularly by responding to such interannual variability of radiation. In the first year of the observation, Q_E is more than 300 W m^{-2} and has a relatively larger temporal variability because of the frequent rainfall events in summer, compared to the second year.

Evapotranspiration rate ranges from 5 mm month^{-1} in January 2015 to 74 mm month^{-1} in August 2013, and the annual ET values are 367 and 320 mm year^{-1} in the first and second years, respectively (Fig. 5 and Table. 2). The ET values correspond to 29.3% and 34.3% of the annual precipitations (Fig. 2d). The difference in ET between the two consecutive years (i.e., 48 mm) mainly occurred in summer (42 mm), especially in August (30 mm). It has been reported that approximately 55% of the net radiation is partitioned to latent heat flux in forest canopies globally (Falge et al., 2001; Suyker and Verma, 2008). The annual ET to net radiation from the urban forest is smaller than this global average and it is also smaller than that of forests at similar latitudes in the East Asia

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(Khatun et al., 2011). The annual ET in the second year is smaller than that in the first year with extensive drought in the second year. However, it is notable that the ET in the second year shows only an approximately 12% decrease compared to the first year, despite a substantial decrease in precipitation (26% decrease) and similar net radiation (Table 2). Although the summer monsoon provides ample water to the ecosystem, its delay and weakness result in severe drought and stress to the ecosystem in this region (Hong and Kim, 2011); however, such ecosystem stress, such as the shrinking of ET and carbon uptake, is inexplicit for the urban forest. We speculate that artificial irrigation by a sprinkler mitigated ecosystem stress to a certain degree in the urban forest.

3.2 Air temperature

Figure 6 shows the mean diurnal pattern of the air temperature difference between the AVG and SD near the SFP ($\Delta T_{air} \equiv T_{air\_AVG} - T_{air\_SD}$ hereafter) before and after the park construction in summer. Notably, ΔT_{air} is always positive during the entire summer season (i.e., AVG is warmer than SD) and shows distinct impacts in terms of magnitude and diurnal variations after the park construction. The warming trend is evident at the AVG ($p<0.015$), wherein there were no changes in the urban structure and function around them. The warming rate at the AVG is 3.0 °C century^{-1}, which corresponds to the warming rate reported in the high-rise urban area in Seoul (Hong et al., 2019a). However, the warming rate around the SFP is approximately 1.6 °C century^{-1}, which is smaller than that of the AVG and other urban areas in Seoul and is comparable to the global mean warming rate of 0.9 °C century^{-1} (Hansen et al., 2010; Hong et al., 2019a).

Notably, such a lower warming trend around the SFP mainly occurs in the afternoon when ET is dominant. This difference will be larger if we consider that the measurement height at the AVG is higher than that at the SD (Table. 1). The maximum ΔT_{air} is approximately 0.3 °C around 10:00 before the park construction (Fig. 6a) and increases up to approximately 0.5 °C with its peak occurrence shifting from the morning to the afternoon (i.e., around 14:00) after the construction (Fig. 6b). This peak time in the afternoon is coincident with the time when photosynthesis is the highest in the vegetation; thus, Q_E increases in summer. Our results indicate that the thermal mitigation of the urban forest is important as a result of increases in ET, especially if we consider that the SFP area was originally planned to be developed as a high-population multipurpose building complex.

3.3 Urban heat island intensity

The influence of urban forests on summer temperature produces also evident traces in UHIi. Figure 7 shows the mean diurnal variation of UHIi at the SFP and AVG during summer. Apparently, the UHIi of the SFP ($UHIi^S$ hereafter) and AVG ($UHIi^A$ hereafter) gradually increases after mid-afternoon and is the largest at night. This diurnal pattern is consistent with previous reports in cities exposed to different geographical and climatic conditions because rural areas cool faster than urban areas **(**Oke et al., 2017**)**. Additionally, $UHIi^A$ is positive throughout all days ranging from 0.2–2.2 °C (i.e., warmer than rural area, GP) and is greater than $UHIi^S$ by 0–1.5 °C. A possible reason for this stronger $UHIi^A$ is that the AVG stations are located in the central business district; thus, the densities of buildings surrounding these stations are much higher than those surrounding the SFP station. At night (19:00–06:00), UHIi^A and UHI^S are approximately 1.8 °C and 1.4 °C, respectively. The maximum UHIi difference between the AVG and SFP was 0.7 °C in 2013 and 0.5 °C in 2014.

Around sunrise, sharp declines in the UHIi are observed because the air temperature near the urban area increases relatively slowly as urban fabrics, such as asphalt, brick, and concrete, have larger heat capacities and less sky view factors than the rural areas (Oke et al., 2017). Eventually, this slow increase in the air temperature reduces the differences in T_{air} among the stations, thereby reducing the UHIi. The minimum UHIi^A values were 0.3 °C (2013) at 09:30 and 0.2 °C (2014) at 08:30, while the minimum UHIi^S occurs at 10:30 with values of –0.1 °C (2013) and 0.0 °C (2014). This implies that the timing of the minimum UHIi is delayed in the SFP compared to the AVG. Our findings indicate that the urban forest has a similar air temperature in the daytime as compared to the rural area (i.e., GP) where has a lower thermal admittance because of its location within the airport. Especially when there is strong ET and more time is required to warm the SFP surface, the urban-rural difference in thermal admittance becomes relatively small. This can be attributed to the higher thermal capacity of the wetter soil of the SFP as a result of artificial irrigation and the absence of impervious surfaces (Oke et al., 1991).

The diurnal variations in UHIi^S also show the interannual variability in both amplitude and steepness over the two consecutive years. Despite the similar summertime UHIi^A for both years, the daytime UHIi^S in 2013 was approximately 0.2 °C lower than that in 2014. Notably, the summer Q_E was greater in 2013 than in 2014, and this observed summertime asymmetric difference between the SFP and AVG stations was not found in the winter when ET was negligible (not shown here).

Our results suggest that urban forests can play a significant role in mitigating the thermal environment because of the wetter soil surface of the park and subsequent increases in Q_E, compared to the impervious surfaces in urban areas. In particular, our findings indicate that the heat mitigation of the urban forest depends on the ratio of Q_E to net radiation. Indeed, there is an evident negative relationship between daytime Q_E and air temperature differences between the SFP and AVG stations (Fig. 8). As $K_\downarrow$ is more partitioned to Q_E, T_{air} of the SFP decreases more than that of the AVG, and the maximum temperature difference is observed in the summer season. The SFP is cooler than the AVG by up to 0.6 °C, but the SFP is warmer than the AVG during the winter-dormant season when ET is small.

3.4 Temporal dynamics of net CO_2 exchange

Figure 9 shows the diurnal evolution of F_C and footprint-weighted road fraction (λ). Overall, the mean daytime F_C is negative (i.e., carbon uptake) in the summer (June–August), indicating that photosynthesis, the only carbon sink, is dominant. This carbon uptake period is coincident with the active vegetation manifested by increases in EVI (not shown here). Summertime photosynthetic carbon uptake (*GPP*) has a daily average of 7.6 μmol m^{-2} s^{-1} with a maximum of 18.9 μmol m^{-2} s^{-1} around 12:30 (Fig. 8a in Lee et al., 2021). A daily minimum F_C also occurs around 12:30 with the maximum photosynthetic carbon uptake during this time accordingly. The vegetation around the SFP absorbs more CO_2 than carbon sources and F_C becomes negative only during the summer daytime. However, because of substantial amounts of anthropogenic emissions and ecosystem respiration, F_C changes from

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negative (i.e., carbon sink) to positive values (i.e., carbon source) even around 16:30 in summer unlike in natural ecosystems, despite the substantial downward shortwave.

CO_2 uptake is highest in June, with a maximum of approximately 13 μmol m^{-2} s^{-1} (Fig. 9a). In the middle of summer (4th and 31st two-week data in Fig. 9a), CO_2 uptake decreases significantly because photosynthesis is limited because of the reduced $K_{\downarrow}$ by cloud and rainfall with the onset of the summer monsoon (Fig. 2c). This mid-summer depression of carbon uptake has been reported in the Asian natural vegetations (e.g., Kwon et al., 2009; Hong and Kim, 2011; Hong et al., 2014). Higher reduction in CO_2 uptake observed in 2013 than in 2014 was attributed to a longer monsoon period in 2013. Indeed, from 8 to 21 July 2013 (4th two-week data in Fig. 9a), the accumulated precipitation was approximately 400 mm for two weeks, and the daily averaged $K_{\downarrow}$ was only 70 W m^{-2}.

As photosynthesis decreases, F_C changed to positive values from November. During the non-growing season (i.e., late autumn, winter, and early spring), anthropogenic emissions were also dominant because photosynthesis and ecosystem respiration decrease with smaller $K_{\downarrow}$ and lower temperatures. During these periods, F_C had minimum values at 04:00–05:00 and increases until 15:00–16:00. Therefore, the diurnal variations in F_C mainly followed the traffic volume (Fig. 4a in Lee et al., 2021), and there also is a clear positive relationship between F_C and λ (23rd, 45th, and 47th two-week data in Fig. 9). It is also noteworthy that the peak time of F_C (16:00) is earlier than the peak time of λ (18:00) from December to early March because E_B is the largest at around 15:00–16:00, indicating that E_R and E_B are the controlling factor of F_C in this period.

With such apparent seasonal F_C variation, it is notable that its variability depends on the spatio-temporal distribution of CO_2 sources and flux source area because the latter covers various land use with changes in wind direction and atmospheric stability (Fig. 10). In autumn, the main wind direction changed to the north as the synoptic conditions change particularly (Fig. 3); therefore, λ is smaller in autumn compared to other seasons (Fig. 9b). For example, the road fraction was smallest at < 1% from midnight to midday and < 3% during the afternoon in October and November (11th, 12th, 36th, and 37th two-week data in Fig. 9b). In these periods, the nighttime F_C showed the lowest value of approximately 2.9 μmol m^{-2} s^{-1}, which was attributable to the smallest road fraction, lower respiration, and minimal heating usage.

In early spring, λ was generally larger; thus, E_R played a significant role in F_C, and E_B also remained non-zero until early April, thereby resulting in the largest F_C in this period. With a shutdown of the heating system (i.e., zero E_B) and the sprouting of leaves in April, there was a sharp decrease in F_C (Fig. 10b and 10c). From December to March, CO_2 emissions increased up to 30 μmol m^{-2} s^{-1} with larger variability in the south–west direction because of intermittent anthropogenic emissions from the park facility (due to space heating and boiling water), as well as the relatively increased contribution of vehicles on the road in the western part of the site.

Although the positive F_C in the winter decreased in spring, its magnitude showed directional differences (Fig. 10c). On the eastern side, the mean F_C showed a negative value in May, whereas it remained positive on the western side (210–270°) until May. Therefore, these findings further indicating the different contributions of

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various carbon sources and sinks among the different wind directions. For the wind directions from the north to the east (0–120°), F_C showed a relatively weaker carbon sink than other directions because of the relatively low tree fraction in this direction (Fig. 10a and 10c). On the southern side (150–180°) having the highest tree cover fraction, a maximum carbon uptake about 15 μmol m^{-2} s^{-1} on average was found in June. However, despite the dense vegetation on the south and west side (120–330°), the F_C magnitude was much smaller than that of other natural forests. This is related to the anthropogenic emissions from vehicles on the roads which is discussed in section 3.6.

3.5 Light use efficiency of biogenic CO_2 components

F_C at the SFP shows a typical light response to the photosynthetically active radiation (PAR) in a way similar to natural ecosystems in spite of anthropogenic CO_2 sources from vehicles (Fig. 11a and 11b). However, this light response in the urban forest is a distinct contrast with the non-dependent F_C in high-rise high-population residential areas in Seoul under the same climatic conditions (EP station). Importantly, *GPP, NBE,* and F_C show different trends on PAR depending on the direction. As stated in Section 2.1.1 and 3.4, the western side has a higher density of trees as against more grass on the eastern side, and biotic CO_2 uptake from the western side is substantially larger than that on the eastern side. Accordingly, the slope of the light response curve for PAR on the western side is steeper than on the eastern side. F_C at zero PAR (F_{C_0}) is larger on the western side (9.7 μmol m^{-2} s^{-1}) than on the eastern side (5.1 μmol m^{-2} s^{-1}) because of a contribution of E_R from roads on the western side of the tower.

NBE shows a comparable light response to natural vegetation (e.g., Schmid et al., 2003). A rectangular hyperbolic equation has been used to examine the light response of *NBE* and elucidate the directional differences in carbon uptake:

$$NBE = -GPP + RE = -\frac{\alpha \cdot GPP_{sat} \cdot PAR}{GPP_{sat} + \alpha \cdot PAR} + RE \qquad (3)$$

where α is the quantum yield efficiency (the initial slope of the light-response curve), GPP_{sat} is the potential rate of the ecosystem CO_2 uptake. α is approximately 0.0651 and 0.0558 μmol CO_2 (μmol photon)$^{-1}$ on the western and eastern sides, respectively. Notably, α on the western side is comparable to the high initial quantum yield in crops and subtropical forests in East Asia (Hong et al., 2019b; Emmel et al., 2020). Additionally, GPP_{sat} is 30.9 and 12.7 μmol m^{-2} s^{-1} on the western and eastern sides, respectively. In addition, the light saturation points are at a PAR of 1500 μmol m^{-2} s^{-1} on the eastern side, which occur at a relatively lower PAR than on the western side. Daytime respiration estimated from equation (3) is 6.7 and 6.3 μmol m^{-2} s^{-1} on the western and eastern sides, respectively. Because *GPP* is related to PAR, the difference in monthly cumulative *GPP* between the two years shows a close relationship with the difference in the monthly sunshine duration (r^2 = 0.75, not shown here), thereby suggesting a possible impact of change in the onset of the summer monsoon on urban forests.

The magnitude of *NBE* from the western side is larger than that from the suburban area having about 50% vegetative fraction in Montreal, Canada (Fig. 7b in Bergeron and Strachan, 2011) and F_C from a highly vegetated

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environment of about 67% vegetative fraction in Baltimore, USA (Crawford et al., 2011). Also, *GPP* from the western side is comparable to the dense forest canopies in subtropical forests in Korea (Hong et al., 2019b), deciduous forest ecosystems (Goulden et al., 1996), and a mixed hardwood forest ecosystem (Schmid et al., 2000). However, *NBE* from the eastern side is similar to F_C from the suburban areas of about 44%, 50%, and 64% vegetative fraction in Swindon, UK (Ward et al., 2013) and Montreal, Canada (Bergeron and Strachan, 2011), and Ochang, Korea in the same climate zone (Hong et al., 2019b), respectively.

3.6 Annual budget of CO_2 sources and sink

The annual budget of the F_C and its components is summarized in Table 3. The annual sums of the *GPP* and *RE* in the SFP are 4.6 kg CO_2 m^{-2} year^{-1} (1244 gC m^{-2} year^{-1}) and 5.1 kg CO_2 m^{-2} year^{-1} (1378 gC m^{-2} year^{-1}), respectively. This photosynthetic carbon uptake is smaller than its global mean *GPP* in natural deciduous broadleaf forests with similar annual precipitation and annual mean air temperature (total 8 years of data from 4 sites of FLUXNET2015 dataset reported in Pastorello et al., 2020) and similar to that of deciduous broadleaf forests in East Asia (Awal et al., 2010; Kwon et al., 2010) (Table 4). Our speculation is, however, that this GPP is relatively larger if we consider the low vegetation fraction and leaf area index (LAI) at our urban park. Indeed, *GPP* is comparable to values reported in other urban sites if it is scaled with the vegetation cover fraction. Previous studies have shown that the *GPP* of urban vegetation is scaled with vegetation cover fraction with an increase of about 0.7 kg CO_2 m^{-2} year^{-1} per 10% increase in vegetation cover fraction (Awal et al., 2010; Crawford and Christen, 2015; Velasco et al., 2016; Menzer and McFadden, 2017). GPP at the SFP with a 46.6% vegetation cover fraction is approximately 1.5 kg CO_2 m^{-2} year^{-1} larger than this scale (Fig. 12a).

Eventually, this large *GPP* results in a substantial decrease in F_C when they are scaled by vegetation fraction. Hong et al. (2019b) reported a linear decrease in *Fc* of approximately 3.0 kg CO_2 m^{-2} year^{-1} per 10% increase in vegetation cover fraction based on the observed F_C across an urbanization gradient in Korea (Fig. 12b). The annual F_C in the SFP of 7.1 kg CO_2 m^{-2} is 1.2 kg CO_2 m^{-2} year^{-1} smaller than this scaled relationship (i.e., more carbon uptake). In particular, F_C in the SFP is approximately 3.0 kg CO_2 m^{-2} year^{-1} less than that in recently developed high-rise high-population urban areas in Seoul. Our results suggest that efficient management of urban forests, such as regular irrigation and fertilization, can be an efficient way to adapt and mitigate climate change by increasing CO_2 uptake in artificial forest constructions in East Asia.

Meanwhile, *RE* at our site is much larger than that in temperate deciduous forests in East Asia (Takanashi et al., 2005; Kwon et al., 2010) and similar to that in the urban forest in East Asia (Awal et al., 2010), as well as to the global mean *RE* over forests with similar annual precipitation and annual mean air temperatures (Pastorello et al., 2020). Put differently, the urban forest considered in our study is an outlier compared to other natural forest canopies and urban forests because *RE/GPP* > 1 (Table 4). Autotrophic respiration is considered to be approximately half of GPP as a rule of thumb (Piao et al., 2010), which corresponds to approximately 45% of the RE at our site, thereby indicating a large contribution of heterotrophic respiration to *RE*. Indeed, it was reported that soil respiration at the same site was approximately 4 kg CO_2 m^{-2} year^{-1} (Bae and Ryu, 2017). The reason for the large soil organic carbon was mainly because rice cultivation was carried out in this region before the 1950s,

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and organic carbon-rich soil was transplanted during the SFP construction, and fertilizers were applied regularly. It has also been reported that *RE* is enhanced in urban areas because of the relatively warmer temperature in urban regions (i.e., UHI) (Awal et al., 2010). Notably, Q_{10} (the rate by which respiration is multiplied when temperature increases by 10 °C) is about 1.9 at the site and matches the Q_{10} value for ecosystem respiration (2.2 ± 0.7) calculated for natural forests across 42 FLUXNET sites (Mahecha et al., 2010). Further analysis based on the observed Q_{10} and the UHIi at the SFP indicates that UHI leads to an approximately 5% increase in *RE*.

Figure 13 shows the monthly cumulative sum of the F_C and its partitioned components. Seasonal variations in the strength of carbon sources and sink as well as F_C are mainly regulated by the biogenic component in summer and the anthropogenic component in winter. Furthermore, F_C is minimum in June, despite the similar *GPP* from June to August because of the relatively smaller *RE* during the summer season. Even in summer, photosynthetic carbon uptake is balanced with ecosystem respiration and does not offset all biotic and anthropogenic emissions, thus resulting in positive F_C values throughout the year. In winter, E_B is dominant with negligible *GPP* and *RE* due to cold temperatures, and E_R also becomes larger than *RE* from November. E_R shows apparent seasonal variation in wind direction and atmospheric stability. Its magnitude is about 0.0666 μmol m^{-2} veh^{-1} h s^{-1} in neutral condition and consistent with the value in the inventory data (Lee et al., 2021). The average monthly traffic speed for the road in front of the SFP is 50–60 km h^{-1} (based on the January 2014 data from the Seoul Metropolitan Government Traffic Speed Report), and the CO_2 emission rate is approximately 150 g CO_2 km^{-1} veh^{-1} based on the emission data at this speed (Kim et al., 2011). With the width of the ten-lane road (25–30 m), the inventory-based slope (i.e., CO_2 emission rate per vehicle per area per half-hour) is approximately in the range of 0.0631–0.0757 μmol m^{-2} veh^{-1} half-hour s^{-1} (≅ 150 gCO_2 km^{-1} veh^{-1} × 1/30 or 1/25 m^{-1} × 1/44 mol gCO_2^{-1}× 10^{-3} km m^{-1} × 10^6 μmol mol^{-1} × 1/1800 half-hour s^{-1}).

There is an evident yearly difference in individual carbon sources and sink in two consecutive years. E_B is mainly caused by heating buildings and hot water in park facilities using natural gas. Notably, E_B is also smaller in the first year because of the relatively smaller number of park visitors and consequently smaller gas consumption, compared to the second year. Indeed, E_B is highly correlated with gas consumption in SFP during winter on a monthly basis (R^2 = 0.94; Fig. 6 in Lee et al., 2021). Eventually, these annual differences lead to a smaller annual mean total F_C in the first year than in the second year (Table 3). However, *RE* is maximum in the August of the first year, while it is highest in July of the second year because of the interannual variations in air temperature with changes in the timing and duration of the East Asian summer monsoon, of which impacts have also been reported in natural vegetation in the same region (Hong and Kim, 2011; Hong et al., 2019b). In other words, the monthly mean air temperature is highest in August of the first and July of the second year because of the short East Asian monsoon period and drought in July of the second year. However, the *GPP* in summer is relatively smaller in the first year by the mid-summer depression of solar radiation because of the elongated monsoon period (Fig. 2). However, *GPP* does not shrink in the second year of significant drought because there is ample water supply by a sprinkler. Our results emphasize the important role of forest management in enhancing carbon uptake and evaporative cooling despite the low vegetation fraction.

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4 Summary and conclusions

This study reported two-year surface fluxes of energy and CO_2 measured by the eddy covariance method while also examining the role of artificially generated urban forests in mitigating air temperature and anthropogenic CO_2 emissions. The study area is located in the East Asian monsoon region, characterized by a lengthy summer rainy season. During the measurement period, the second year was contrasted with the first year because of the drought compared to the normal climate condition in the first year. The study region is a park with an artificially planted forest in the Seoul Metropolitan Area. The urban forest had a heavy traffic volume around it and was redeveloped from a racetrack and factory in the mid-2000s. To examine the mitigation of air temperature, this study compared meteorological conditions in the urban forest with the surrounding high-rise high-population urban areas. This study also proposed a statistical CO_2 flux partitioning method based on temporal subsets of flux data and high-resolution footprint-weighted land use data to understand the abiotic and biotic contributions to F_C.

Surface energy balance in the SFP is influenced by the summer monsoon, and more energy is distributed to Q_E than Q_H in the summer when vegetation is active, similar to natural forests in this climate zone. Therefore, the Bowen ratio in this urban forest ranges from near 0 (summer) to about 4 (winter), which is lower throughout the year than that of high-rise and high-density residential areas in Seoul. This suggests that the vegetation and unpaved surfaces of urban forests facilitate more evaporative cooling compared to the impervious surfaces in urban areas. Furthermore, ET decreased in the second year when there was a drought, but this drop was not as much as the reduced precipitation if we consider the substantial changes in precipitation and radiative forcing in two consecutive years.

It is also evident that the urban forest reduced the warming trend and UHIi around the study area. Air temperature in the SFP was lower than the surrounding area, but this coolness was reinforced after the park was created. The warming trend diminished after the construction of the park and was smaller than that in other urban regions in the Seoul Metropolitan Area. In addition, the construction of the park delayed the timing of the maximum temperature difference between the urban forest and high-rise commercial from the morning to the afternoon, coinciding with the timing of the maximum Q_E. The SFP shows a general diurnal UHIi variation pattern, which has a higher temperature at night than in rural areas. However, the UHIi in SFP is lower by 0.6 °C in summer compared to the surrounding urban area, and the minimum peak time is delayed, possibly because vegetation and permeable soils in SFP have a larger thermal capacity. Notably, UHIi decreased more in the partitioning of incoming energy into latent heat fluxes. As a rule of thumb, there was cooling by 0.2 °C compared to the surrounding urban area if $Q_E/K_\downarrow$ increased by 10%.

Net CO_2 exchange at the urban forest showed typical temporal variations in natural forest canopies influenced by the East Asian summer monsoon (Hong and Kim, 2011; Hong et al., 2019b). A mid-summer depression of carbon uptake was observed with the onset of the summer monsoon, like vegetation in the East Asian monsoon region. The *GPP* was estimated by the statistical partitioning method, and the non-zero *GPP* period was coincident with the active vegetation of the significant vegetation index. Summertime photosynthetic carbon uptake had a daily average of 7.6 μmol m^{-2} s^{-1} with a maximum of 18.9 μmol m^{-2} s^{-1} around 12:30. However, even during the growing

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season, vegetative carbon uptake was insufficient to offset anthropogenic CO_2 emissions and ecosystem respiration on a time scale of > 1 day. Our estimations of anthropogenic CO_2 emissions from vehicles and buildings agreed with the estimations based on inventory data such as CO_2 emission rate of vehicles and monthly gas consumption, and their annual budgets each had a comparable magnitude to *GPP*.

Annual *GPP* of the urban forest was relatively smaller than that of the forest in East Asia exposed to similar climatic conditions because of the relatively smaller vegetation cover fraction and LAI. However, it was larger than the *GPP* expected from the relationship from previous urban studies if it was normalized by the vegetation cover fraction. *RE* is, however, much larger than that in the temperate East Asian forests and similar to the urban forest in East Asia. We speculate that soil respiration enhanced such large ecosystem respiration by relatively warmer temperatures in a city and rich soil organic carbon in the SFP. Eventually, the annual mean total F_C is 7.1 kg CO_2 m^{-2} $year^{-1}$, which is smaller than the estimate from the scaling between annual total F_C and vegetation fraction (Hong et al., 2019b). Because of the spatial heterogeneity, F_C and its components showed directional changes. *NBE* from the eastern side is similar to F_C of suburban areas with approximately 44%, 50%, and 64% vegetative fraction in Swindon, UK (Ward et al., 2013) and Montreal, Canada (Bergeron and Strachan, 2011), and Ochang, Korea in the same climate zone (Hong et al., 2019b), respectively. However, the *NBE* and *GPP* from the western side are comparable to dense forest canopies in subtropical forests in Korea (Hong et al., 2019b), deciduous forest ecosystems (Goulden et al., 1996), and a mixed hardwood forest ecosystem (Schmid et al., 2000).

Our study reveals that urban forests make significant traces of air temperature and CO_2 fluxes despite their relatively small area. Our key findings are that urban forests in East Asia are highly influenced by the East Asian monsoon like natural forests in this region, but such influence is mitigated by artificial irrigation and fertilization in urban forests. In particular, our results emphasize the importance of forest management for efficient carbon uptake and evaporative cooling despite the low vegetation fraction. Furthermore, our observation study also indicates that caution in soil management is necessary to reduce CO_2 emissions in urban forests, mainly resulting from large soil organic carbon. We also highlight that our statistical CO_2 flux partitioning is a promising method to improve our understanding of the carbon cycle in urban and suburban areas, and a more extensive study is required for validation in another geographical zone.

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Table 1. Details of the stations used in this study.

Sites	Location	LCZ	Height [m]	Used variables
Eddy covariance station				
SFP (Seoul Forest Park)	37.5446°N, 127.0379°E	LCZ_A	12.2	
EP (Eunpyeong)	37.6350°N, 126.9287°E	LCZ_1	30	*Flux*
Automatic weather station				
SD (Seongdong)	37.5472°N, 127.0389°E	LCZ_{5B}	25	T_{air}, *RH, WS, WD, Precipitation*
AVG				
(Gangnam)	37.5134°N, 127.0467°E	LCZ_{21}	59	
(Seocho)	37.4889°N, 127.0156°E	LCZ_{21}	35.5	T_{air}
(Songpa)	37.5115°N, 127.0967°E	LCZ_{15}	58.2	
Aerodrome meteorological observation station				
GP (Gimpo)	37.5722°N, 126.7751°E	LCZ_D	11.4	T_{air}

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Table 2. Gap-filled annual budgets for surface energy fluxes and precipitation (P).

Sites	*ET* (mm)	Q_H (MJ m^{-2})	Q_E (MJ m^{-2})	Q^* (MJ m^{-2})	P (mm)
1st year (2013.06 – 2014.05)	367	726	896	1797	1256
2nd year (2014.06 – 2015.05)	320	867	781	1848	932
Mean annual sum of two-year	344	797	839	1823	1094

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Table 3. Gap-filled annual budgets for F_C (observed by EC measurement) and its components, indicating ecosystem respiration (*RE*), photosynthetic uptake by vegetation (*GPP*), vehicle emissions (E_R), and building emissions (E_B). All fluxes are in kg CO_2 m^{-2} $year^{-1}$.

Sites	F_C	*RE*	*GPP*	E_R	E_B
1st year (2013.06 – 2014.05)	6.6	5.1 (71%)	4.7 (64%)	3.3 (76%)	1.0 (20%)
2nd year (2014.06 – 2015.05)	7.6	5.0 (77%)	4.5 (70%)	3.2 (81%)	1.9 (15%)
Mean annual sum of two-year	7.1	5.1 (65%)	4.6 (59%)	3.3 (71%)	1.5 (25%)

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Table 4. Annual budgets of biogenic F_C components and ratios in deciduous broadleaf forests in similar climatic conditions reported in previous studies. All fluxes are in kg CO_2 m^{-2} $year^{-1}$.

Sites name	Reference	MAT (°C)	MAP (mm)	maximum LAI	*RE*	*GPP*	*NBE*	*RE/GPP*
Seoul Forest Park	This study	13.9	1094	1.6	5.1	4.6	+0.5	1.11
Nagoya urban forest	Awal et al. (2010)	15.9	1680	5.5	4.9	6.2	–1.3	0.74
Toyota rural forest		14.5	1518	4.5	2.6	4.6	–2.0	0.56
Gwangneung deciduous forest	Kwon et al. (2010)	12.8	1487	5	3.8	4.1	–0.3	0.93
Kiryu Experimental Watershed	Takanashi et al. (2005)	14.1	1309	5.5	3.9	5.6	–1.7	0.70
FLUXNET2015 dataset*	Pastorello et al. (2020)	14.5	1113		4.1	6.0	–1.9	0.68

*Average value of 8-year data from 4 sites having mean annual temperature (MAT) of 12-16°C, mean annual precipitation (MAP) of 900-2000 mm.

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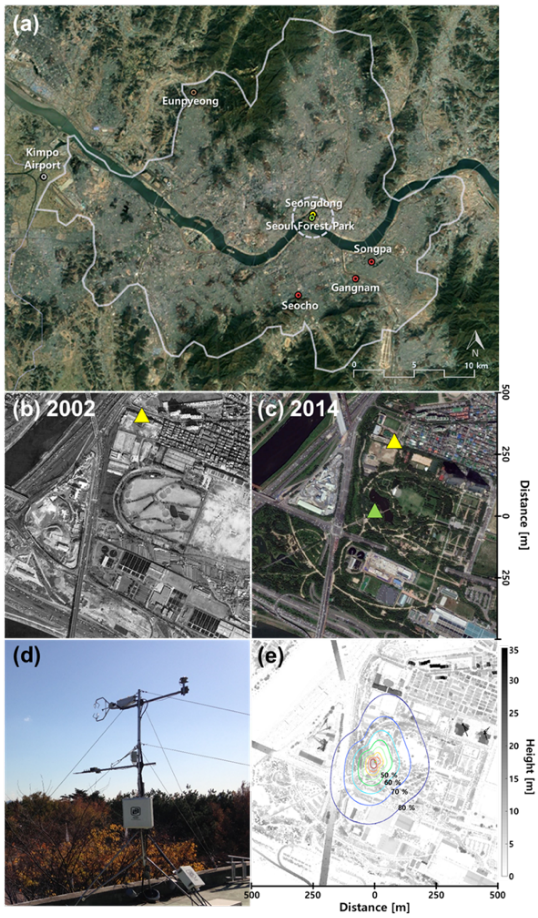

Figure 1. Site descriptions. (a) Location of the stations in Seoul (modified from map data © Google Earth 2019), (b) aerial photographs around Seoul Forest Park (SFP) in 2002 before the creation of the park and (c) in 2014 during the observation period (SFP; *green triangle*, SD; *yellow triangle*), (d) photograph of the SFP station, and (e) footprint climatology (Hsieh et al., 2000) with the height of surrounding obstacles around the SFP station.

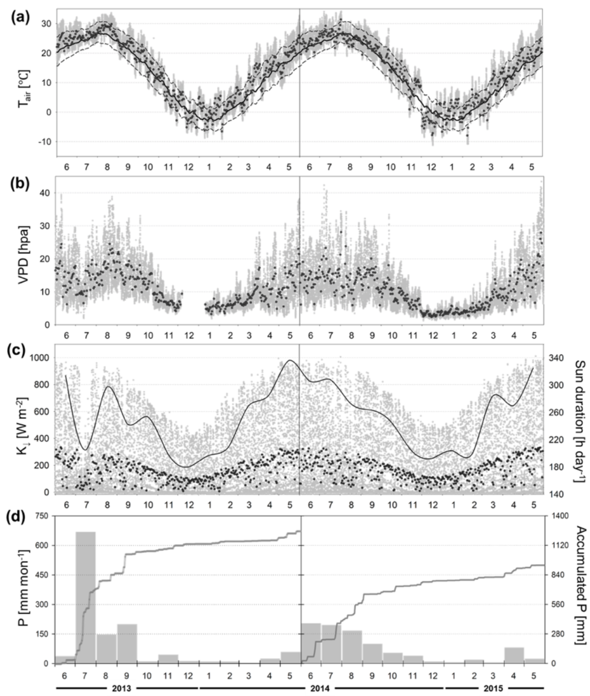

Figure 2. Climatic conditions of the SFP for two years from June 2013 to May 2015: 30-min (*gray dots)* and daily mean (*black dots*) (a) air temperature with 30-year normal values of Seoul (daily mean; *solid line*, min and max; *dashed lines*), (b) vapor pressure deficit (VPD) and missing data existing on December 2013, (c) downward shortwave radiation ($K_{\downarrow}$) and monthly averaged sunshine duration per day (*black line*), (d) monthly precipitation (*gray bars*) and yearly accumulated precipitation (*solid line*).

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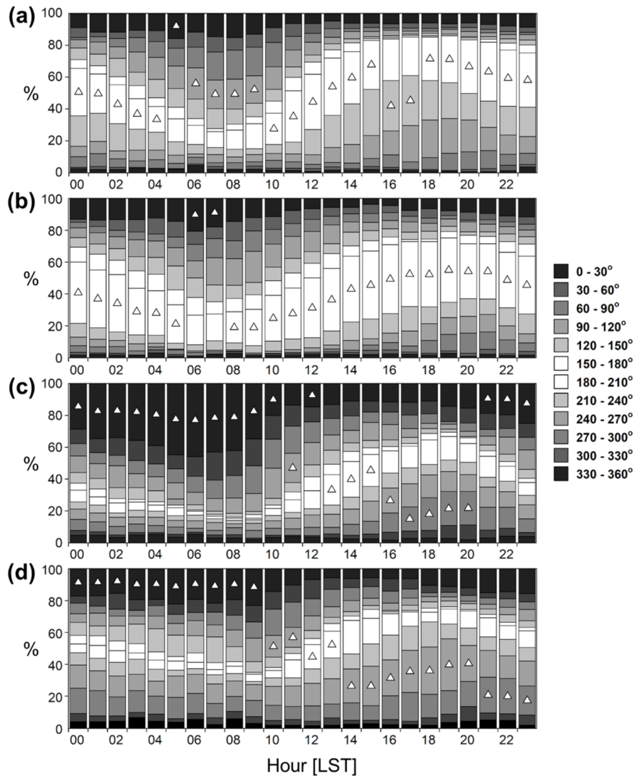

Figure 3. Seasonal mean diurnal courses of wind direction for 30° intervals. Each column represents the hourly ratio of the wind direction of the season: (a) spring (b) summer (c) autumn (d) winter. The white triangle indicates the dominant wind direction during that hour.

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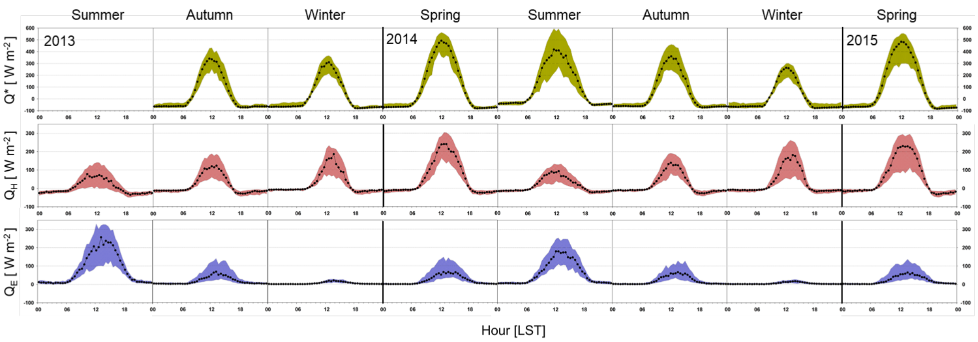

Figure 4. Diurnal variations of surface energy fluxes. Seasonal median diurnal variations (*points*) and interquartile ranges (*shaded*) of 30-min sensible heat flux (Q_H), latent heat flux (Q_E), and net radiation ($Q*$) for two years. Since the net radiation system was installed in September 2013, there was no $Q*$ value in the first summer.

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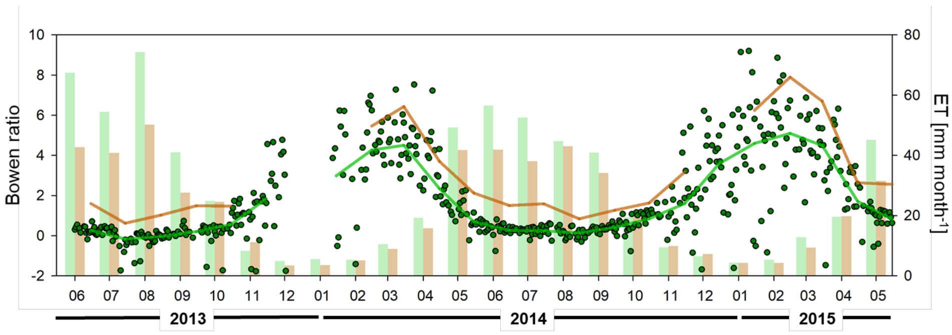

Figure 5. Daily Bowen ratio (β=$\sum Q_H/\sum Q_E$; *dots*), monthly Bowen ratio *(lines*), and gap-filled monthly evapotranspiration (ET; *bars*) for two years (SFP; *green*, EP; *brown*).

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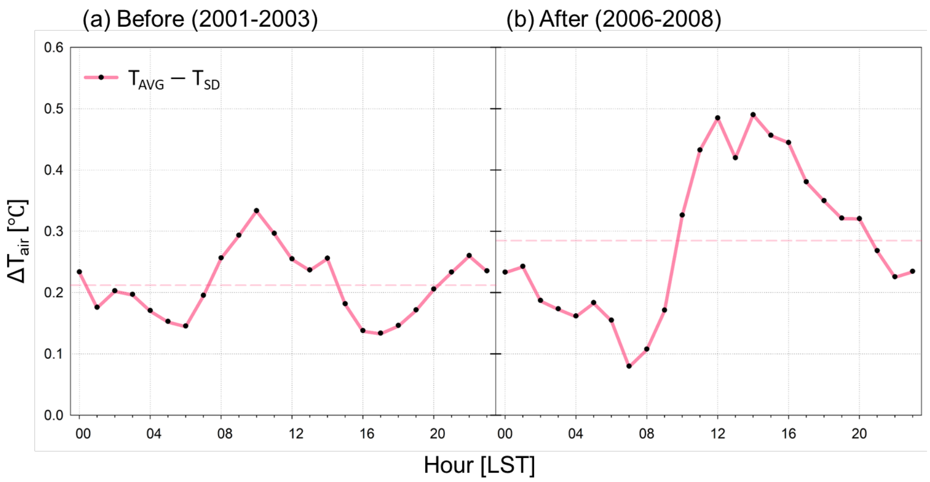

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809 Figure 6. Mean diurnal pattern of air temperature difference (ΔT_{air}) between AVG and SD (a) before and (b) after
810 the construction of the park in summer. AVG indicates an average of three automatic weather stations (Gangnam,
811 Seocho, Songpa) in Seoul. The red dash line indicates the mean ΔT_{air} before and after the construction of the park.

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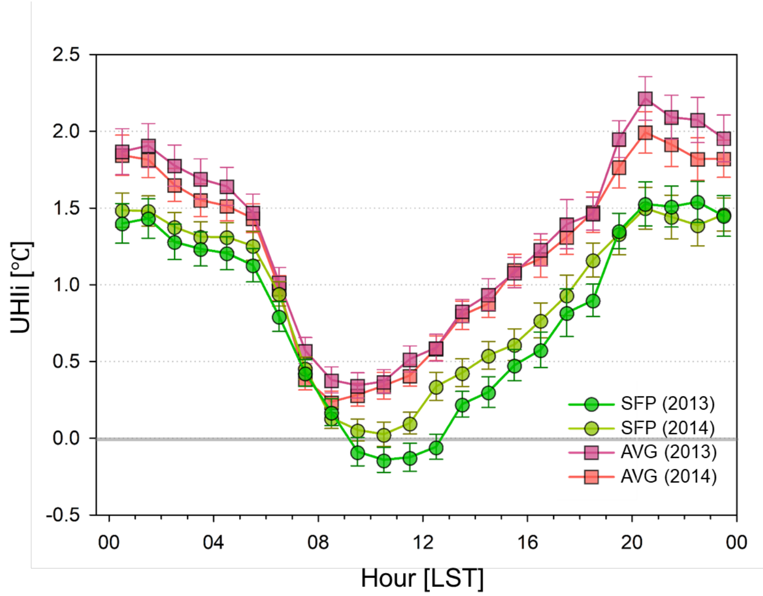

Figure 7. Hourly mean diurnal variation of the urban heat island intensity (UHIi) of the SFP and AVG in the summer of 2013 and 2014. The error bars represent standard errors.

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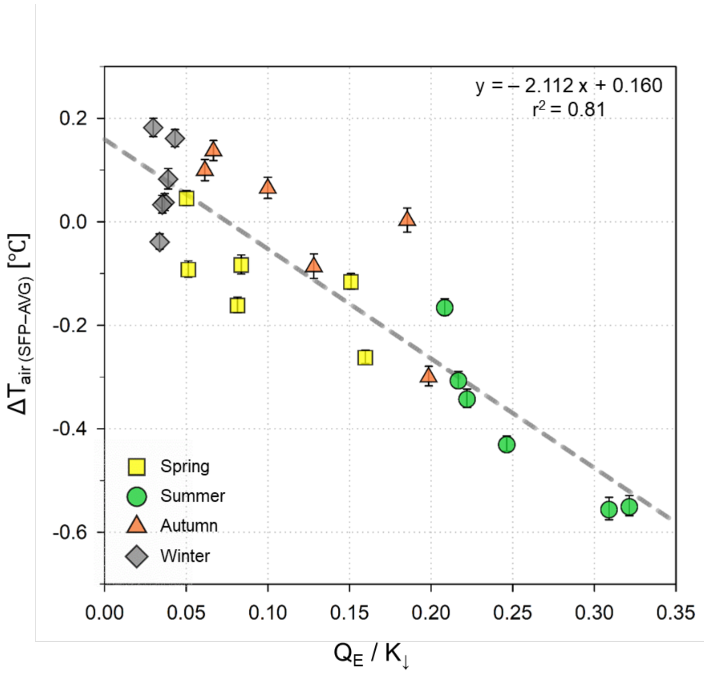

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818 Figure 8. Relationship between the ratio of monthly Q_E to $K_\downarrow$ and mean air temperature difference between SFP
819 and AVG during the daytime ($K_\downarrow$ > 120 W m^{-2}) for two years. The error bars represent standard errors.

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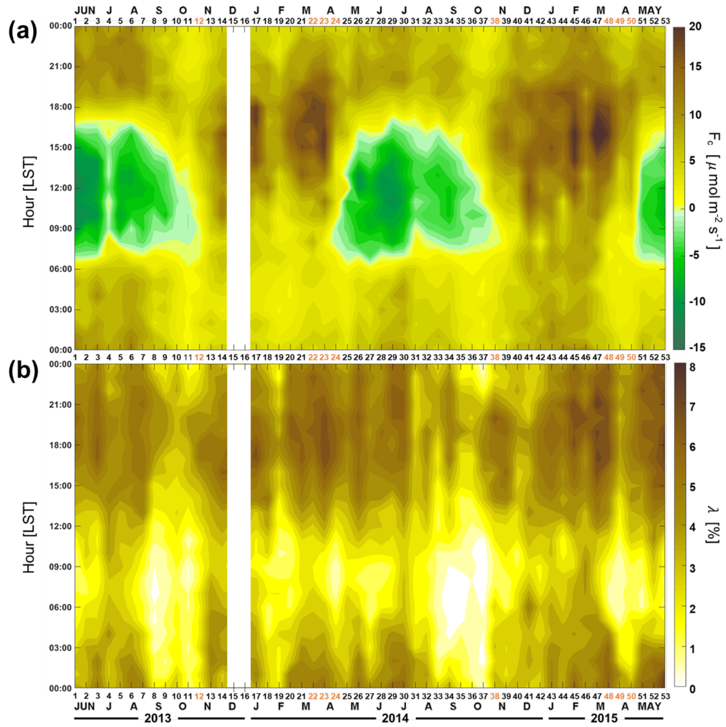

Figure 9. (a) Temporal variation of hourly averaged F_C and (b) source area weighted road fraction (λ) for every two-week. The horizontal axis indicates the order of every two-week for two years, and the vertical axis is the time of day. In December 2013, there was a gap for approximately 4 weeks due to the power system failure. The yellow numbers indicate the two-week (12th, 22nd–24th, 38th, and 48th –50th) having the transition period when the observed F_C is primarily attributable to traffic emissions (E_R).

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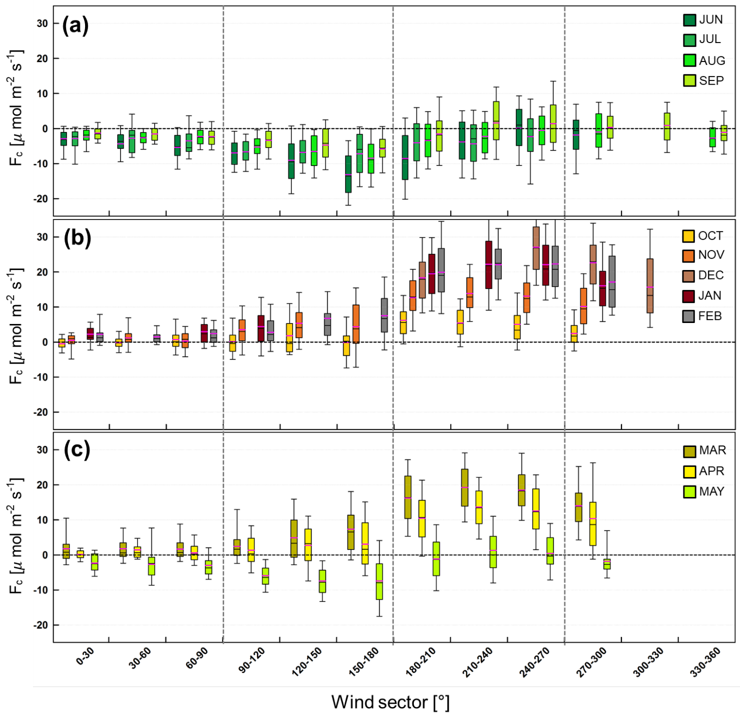

Figure 10. Monthly boxplots of daytime ($K_{\downarrow} > 120$ W m^{-2}) F_C by wind direction. Boxes have a minimum of 20 samples. Box limits are upper and lower quartiles, and whiskers are distances of 1.5 times the interquartile range from each quartile. Median and mean values are indicated by the black and pink horizontal lines.

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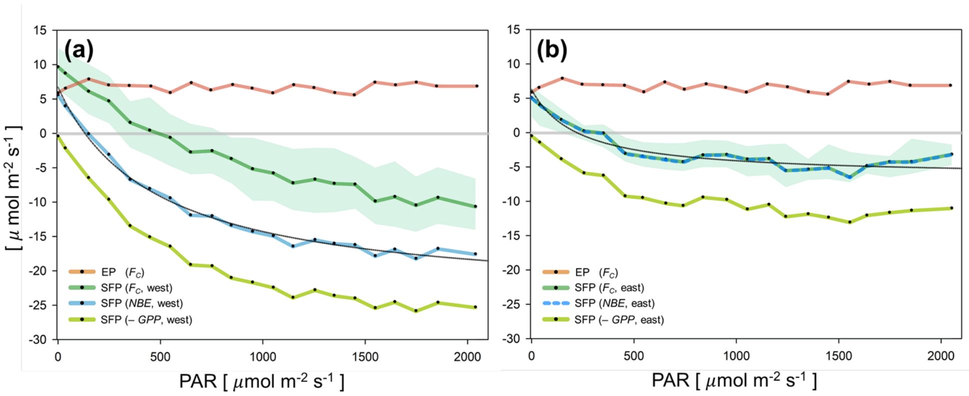

Figure 11. During the growing season (June–August 2013, 2014), light-response curves as a function of photosynthetically active radiation (PAR, in bins of 100 μmol m^{-2} s^{-1}): (a) for the western sectors (150° < Φ < 300°) and (b) for the eastern sectors (30° < Φ < 90°). Black line is a rectangular hyperbolic equation fitting net biome exchange (*NBE* = *RE* – *GPP* = F_C – E_R) to PAR, and EP (*brown line*) is a light-response curve for the high-rise high-population residential area in Seoul. The shaded areas indicate interquartile range.

https://doi.org/10.5194/acp-2021-354
Preprint. Discussion started: 21 May 2021

Atmospheric Chemistry and Physics
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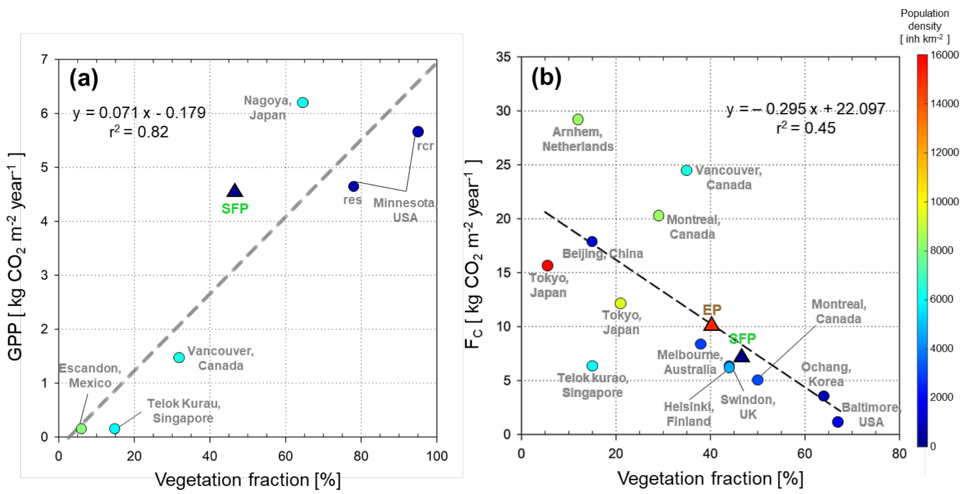

Figure 12. Relationship between vegetation fraction (a) annual *GPP* and (b) annual F_C in urban sites (Fig. 12a in Hong et al., 2019b).

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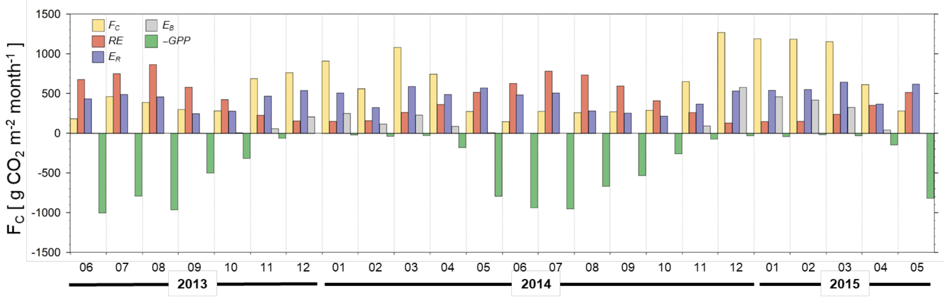

Figure 13. Monthly sums for gap-filled F_C (*yellow bar*) with *RE* (*red bar*), E_R (*blue bar*), E_B (*gray bar*), and –*GPP* (*green bar*)