



Estimates of energy partitioning, evapotranspiration, and net ecosystem exchange of CO₂ for an urban lawn and a tallgrass prairie in the Denver metropolitan area under contrasting conditions

Thomas S. Thienelt¹ · Dean E. Anderson²

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Abstract

Lawns as a landcover change substantially alter evapotranspiration, CO₂, and energy exchanges and are of rising importance considering their spatial extent. We contrast eddy covariance (EC) flux measurements collected in the Denver, Colorado, USA metropolitan area in 2011 and 2012 over a lawn and a xeric tallgrass prairie. Close linkages between seasonal vegetation development, energy fluxes, and net ecosystem exchange (*NEE*) of CO₂ were found. Irrigation of the lawn modified energy and CO₂ fluxes and greatly contributed to differences observed between sites. Due to greater water inputs (precipitation + irrigation) at the lawn in this semi-arid climate, energy partitioning at the lawn was dominated by latent heat (*LE*) flux. As a result, evapotranspiration (*ET*) of the lawn was more than double that of tallgrass prairie (2011: 639(±17) mm vs. 302(±9) mm; 2012: 584(±15) mm vs. 265(±7) mm). *NEE* for the lawn was characterized by a longer growing season, higher daily net uptake of CO₂, and growing season *NEE* that was also more than twice that of the prairie (2011: -173(±23) g C m⁻² vs. -81(±10) g C m⁻²; 2012: -73(±22) g C m⁻² vs. -21(±8) g C m⁻²). During the drought year (2012), temperature and water stress greatly influenced the direction and magnitude of CO₂ flux at both sites. The results suggest that lawns in Denver can function as carbon sinks and conditionally contribute to the mitigation of carbon emissions - directly by CO₂ uptake and indirectly through effects of evaporative cooling on microclimate and energy use.

Keywords Urban lawns · Tallgrass prairie · Net ecosystem exchange of CO₂ · Evapotranspiration · Eddy-covariance method · Drought impact

Introduction

The exchange of energy, water, and greenhouse gases of terrestrial ecosystems with the atmosphere and the resulting feedbacks on climate and biogeochemical cycles has put them in the focus of science (Heimann and Reichstein 2008; Arneth et al. 2010). Despite extensive data on carbon dioxide (CO₂) emissions and accumulation rates in the atmosphere, some uncertainty remains regarding size, spatial distribution, and influencing parameters of the terrestrial sink. While the main anthropogenic sources of CO₂ are reasonably well quantified, the more diverse natural sinks of CO₂, including terrestrial

vegetation, need more study (Pacala, 2001; Clement 2004; Canadell et al., 2007a, b; Scholes et al. 2009; Warren et al. 2011). Photosynthesis and transpiration by vegetation are key processes regulating CO₂ exchange and can strongly affect the partitioning of available energy into latent (*LE*) and sensible heat (*H*). The uptake of CO₂ by vegetation varies spatially and temporally and is strongly influenced by management (Oncley et al. 1997; Katul et al. 2001; Leuning et al. 2004; Ma et al. 2007; Baldocchi 2008; Davis et al. 2010; Eugster et al. 2010). Grassland ecosystems are of great significance in this context because they cover approximately 25% of the terrestrial surface, contain about 20% of the global carbon stocks and have the potential for further carbon sequestration (Conant 2010). The prairies of the Great Plains, as the dominant grasslands of North America, have been dramatically impacted by land use change, mostly for agricultural production (Suttie et al. 2005). Another anthropogenic impact is the conversion to urban land use. Although urban ecosystems currently cover a relatively small fraction of the land surface in the United States (US) (3–5%), they are expanding rapidly

✉ Thomas S. Thienelt
thomas.thienelt@geo.uni-halle.de

¹ Institute of Geosciences and Geography, Martin-Luther-University Halle-Wittenberg, Halle, Germany

² U.S. Geological Survey, Denver, CO, USA

(Alig et al. 2004; Imhoff et al. 2004; USDA 2020; United Nations 2012). A prime example of this is the Denver, Colorado, USA metropolitan area, which has been, and continues to be, one of the fastest growing urban areas in the US (US Census 2010). The combination of this growth and the intensity of anthropogenic activities within urban areas contribute substantially to the rising concentrations of greenhouse gases in the atmosphere (Pataki et al. 2006). But the heterogeneity of urban ecosystems makes it difficult to investigate basic ecosystem functions and to quantify greenhouse gas budgets, which could result in the exclusion of important sinks and sources necessary for accurate biogeochemical modeling (Kaye et al. 2004; Pickett et al. 2011; Bulkeley 2013).

An important component of the urban ecosystems in the US are urban lawns, which are ubiquitous and typically occur as monocultures in residential, recreational, and industrial settings. Estimates found that lawns are the largest irrigated crop of the continental US, covering an area of approximately 1.9% (Milesi et al. 2005). Assuming that most of this lawn area is located within the US urban land cover fraction of 3–5% (see above), lawns could occupy between 38 and 63% of US cities; the percentage of land covered by lawns varies with population and geographic region (Steele and Wolz 2019). For the Denver metropolitan region, it was estimated that up to 40% of the land surface was covered by lawns (Thienelt and Frühauf 2007). Urban lawns are often subjected to management practices like fertilization and irrigation which may alter carbon cycling (Kaye et al. 2004). As natural grassland to urban land use change continues, there is the potential for considerable changes in regional carbon and water cycles and the need for a better understanding of the effects and impacts of these changes. Therefore, the main goals of this study are: 1) the analysis of diurnal fluxes of CO₂, water vapor, and energy (sensible heat) between urban lawn and prairie, and the atmosphere, with regard to bio-environmental controls and temporal variability; 2) the analysis of seasonal differences in energy partitioning, daily and seasonal evapotranspiration and net CO₂ uptake, and 3) the estimation of annual carbon budgets including direct and indirect emissions from management.

Methods

Study sites

The urban lawn site is located within the boundaries of the 86-ha Fort Logan (FL) National Cemetery in Denver, Colorado (39.647° N, 105.039° W, 1640 m a.s.l.). Fort Logan is in the southwestern portion of Denver, approximately 11 km from downtown, surrounded by mostly single-family residences with a 1900 people/km² population density (US Census 2010) (Fig. 1). The eddy covariance station was set up in a quiet, managed area of the cemetery situated on flat terrain. Foot and vehicle

traffic was light as there were no graves, roads or pathways at this site. The nearest graves and access road were 50 and 35 m distant, respectively. Fetch in the dominant wind direction (180–230°) was between 150 and 240 m. Species composition of vegetation was dominated by Kentucky bluegrass (*P. pratensis*), tall fescue (*F. arundinacea*), and perennial ryegrass (*L. perenne*). The lawn area was minimally managed until 2005 when application of irrigation and fertilizer began to sustain the vegetation (C. Hutchinson, Fort Logan National Cemetery, personal communication). Irrigation was active between May 9 and October 6 in 2011 and between April 23 and October 18 in 2012. Fertilizer was applied twice in 2011 (June 15, October 6) and once in 2012 (July 17) and amounted to approximately 49 kg N/ha per application. During the growing season (April–October), grass was mown to a height of about 5 cm once a week and clippings were left at the site. Because permanent (year-round) measurements were not permitted at Fort Logan, data are limited to March 16–November 18 2011 and March 12–December 12 2012. Despite this limitation, the Fort Logan site provided an adequately sized area of homogeneous turfgrass to conduct EC measurements, documented management as well as a level of protection from vandalism. Data for this site used in this study are available in Anderson and Thienelt (2020a).

The prairie site, Rocky Flats (RF), is part of the Rocky Flats National Wildlife Refuge (39.875° N, 105.218° W, 1860 m a.s.l.), approximately 25 km northwest of downtown Denver. There is over 400 m of fetch in any direction from the point of measurement. Due to restricted public access during the past decades, the area retained a diverse natural habitat including the rare xeric tallgrass prairie. Dominant species included big bluestem (*A. gerardi*), switchgrass (*P. virgatum*), and blue grama (*B. gracilis*) (USFWS, 2005). Available data covers the period from January 1 2011 to December 31 2012 (Anderson and Thienelt 2020b).

Flux and ancillary measurements

The instrument array at the urban lawn site consisted of a sonic anemometer (CSAT-3, Campbell Scientific, USA) and an open-path infrared gas analyzer (IRGA; LI-7500, LI-COR Inc., USA) mounted on a tripod 1.85 m above the ground, two soil heat flux plates at 5 cm soil depth (HFP-01, Campbell Scientific), a soil thermocouple located above one of the soil heat flux plates, a soil moisture probe measuring a 0–15 cm depth profile (CS-616, Campbell Scientific) 2.45 m south of the tripod center, a precipitation gage (Weathertronics, model 6010), a net radiometer (Q7.1 REBS, Campbell Scientific), a pyranometer (LI-200, LI-COR Inc.), and a temperature and humidity probe (HMP-45C, Campbell Scientific). Eddy covariance data were recorded at 10 Hz (sonic anemometer and IRGA) on a datalogger (CR-1000, Campbell Scientific) along with 30-min averages from all other instruments.

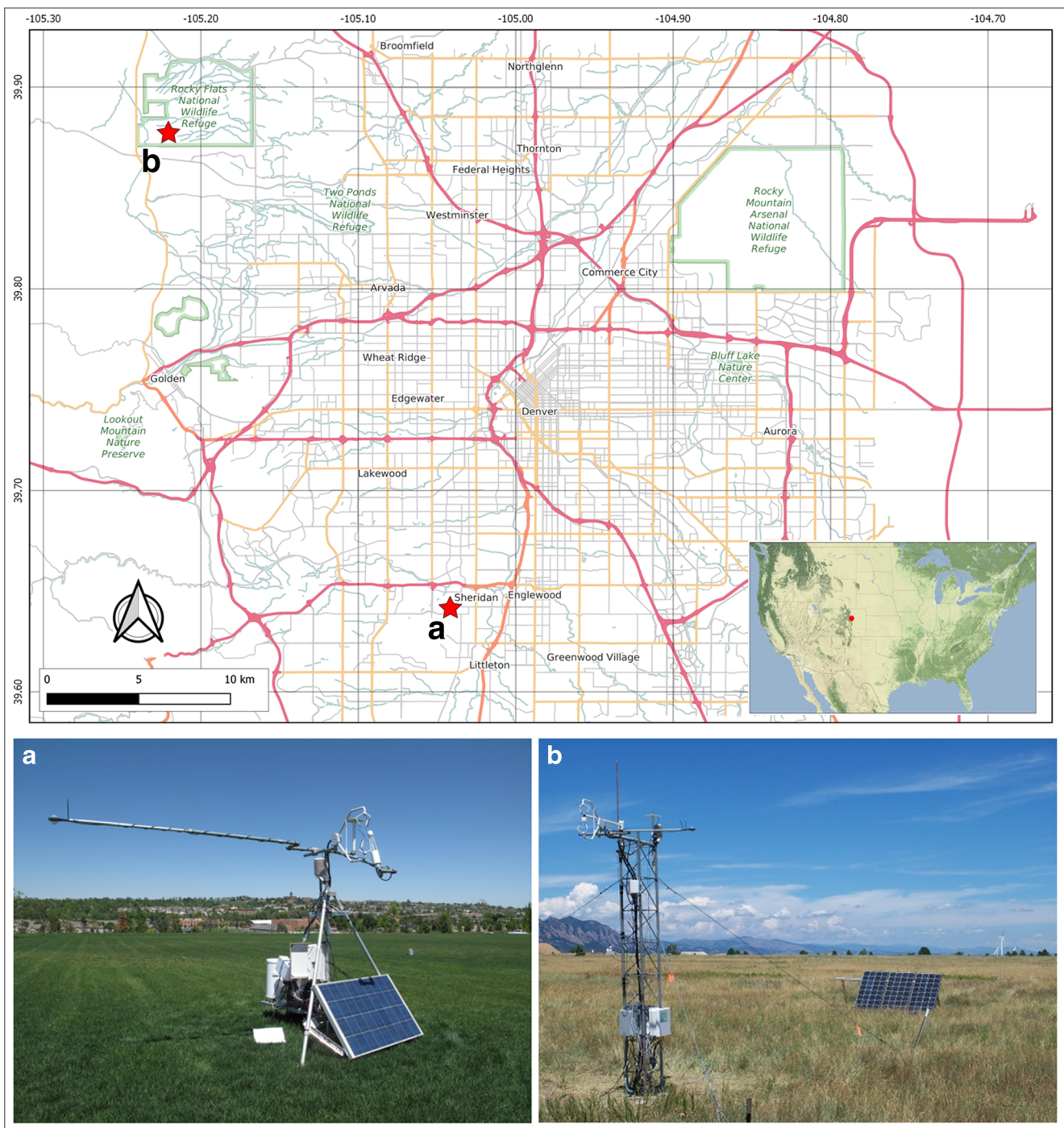


Fig. 1 Location of the study sites in the Denver region: **(a)** Fort Logan and **(b)** Rocky Flats (map data source: © OpenStreetMap contributors, Stamen Design, photographs by the U.S. Geological Survey)

Irrigation amounts were determined by subtracting total water input measured by the gage at the urban lawn from precipitation data reported from a nearby U.S. Geological Survey (USGS) operated precipitation gage, located approximately 2.7 km E-NE of Fort Logan (SDT-South Denver Tower: 39.659° N, 105.013° W, 1620 m a.s.l.).

Flux measuring equipment at the prairie site consisted of a sonic anemometer (CSAT-3, Campbell Scientific) and a closed-path IRGA (LI-7200, LI-COR Inc.) mounted on a tower 3 m above the ground. Sample air flowing through the IRGA was drawn through a 1 m tube (inside diameter = 7.7 mm) at 19.5 LPM (actual) using a flow module (LI-7550, LI-COR). The setup also included two soil heat flux

plates at 5 cm soil depth (HFP-01, Campbell Scientific), two soil thermocouples located above each of the soil heat flux plates, and two soil moisture probes measuring a 0–15 cm depth profile (CS-616, Campbell Scientific) located 9.5 m southwest and 6.0 m east of the tower, a precipitation gage (Weathertronics, model 6010), a net radiometer (Q7.1 REBS, Campbell Scientific), a pyranometer (LI-200, LI-COR Inc.), a quantum sensor (LI-190, LI-COR Inc.), and a temperature and humidity probe (HMP-45C, Campbell Scientific). Eddy covariance data were recorded on a datalogger (CR-3000, Campbell Scientific) at 10 Hz (sonic anemometer and IRGA) along with 30-min averages from all other instruments.

Leaf area index (LAI) at both sites was measured at 3 locations within the flux footprint of the instrument tower and analyzed using a leaf area meter (LI-3100, LI-COR Inc.). LAI values were then averaged for each sampling date. Sampling locations were randomly chosen for each sampling day.

Soil heat flux at the surface (G) was derived from values measured at 5 cm soil depth, corrected for changes in soil heat storage in the soil volume above the heat flux plates (Campbell Scientific, Inc. 2012).

Data processing for calculation of turbulent fluxes

Fluxes of latent heat, sensible heat, and carbon dioxide were calculated using EddyPro software (version 4.2; LI-COR, Inc.). Raw data (vertical wind component, CO₂, water vapor, sonic temperature) were statistically screened following Vickers and Mahrt (1997).

Quality control of the output data included plausibility tests of the measured parameters and checks for instrument diagnostics given by the IRGA and anemometer. If the count of diagnostic flags exceeded 10% (i.e., 1800 out of 18,000) for any half-hour record, the record was deleted. Processed data were also filtered for periods of insufficient turbulence, i.e., low friction velocity (u_*). Records when u_* was less than 0.05 m s⁻¹ at the urban lawn site or less than 0.11 m s⁻¹ at the prairie site were removed. Remaining data for the energy and CO₂ fluxes ranged from 64 to 80% of the total period, varying with year and site.

Gaps were filled using an algorithm suggested by Reichstein et al. (2005), which considers the covariation of the calculated fluxes with meteorological parameters (incoming shortwave radiation, air temperature, vapor pressure deficit (VPD)) and the temporal auto-correlation of these fluxes. Implementation of this gap-fill algorithm was aided by the MPI-BGC online tool (MPI 2013). Vehicle and foot traffic was minimal within the urban lawn flux footprint. While these potential impacts were not specifically included in gap-filling, it is likely that periods of measurable impact would be removed by data quality screening.

Calculation of *annual* net ecosystem exchange (NEE) of CO₂ for the urban lawn required additional gap-filling between and outside the measuring period in 2011 and 2012. Missing days in March 2011/2012, November 2011, and December 2012 were filled using the method by Reichstein et al. (2005). The remaining missing months (i.e., January and February 2011/2012; December 2011) which lie outside of the growing season were filled using a soil temperature/soil respiration regression (Fig. 2a) assuming that CO₂-uptake during these months was negligible. Missing soil temperature data at the urban lawn were derived using a regression between soil temperature at the prairie site and soil temperature at the lawn (Fig. 2b).

Estimates of carbon emissions related to lawn management are based on results published in Townsend-Small and Czimczik (2010). Monthly emissions due to fossil fuel use were adjusted for the aerial extent of the urban lawn and only considered for months of active irrigation. For the prairie site, annual NEE was calculated based on the gap-filled data for 2011 and 2012.

Uncertainties for cumulative sums of NEE and evapotranspiration (ET) were estimated from the 30-min-average datasets (including gap-filled data) using a bootstrap approach (Efron and Tibshirani 1986). For each of the 1000 replicates, the cumulative sum was calculated and 95% confidence intervals were derived from the bootstrap distribution of cumulative sums.

Results

Environmental conditions

Denver's high elevation and interior continental position results in a semi-arid climate with distinct seasons. Average annual temperature in Denver is 10.4 °C with the warmest month being July (23.4 °C) and the coldest December (−0.9 °C) (1981–2010). As a result of Denver's distance from major sources of moisture (e.g., the Pacific Ocean) and the predominant westerly flows creating a rain shadow on the eastern slope of the Rocky Mountains, precipitation is generally light and relative humidity low. Summer months are typically characterized by increased shower and thunderstorm frequency providing a large portion of the year's precipitation. Average annual precipitation in Denver is 381 mm (1981–2010) (Doesken et al. 2003; Paschke 2011; NOAA 2013a, b).

Measured temperature at both sites (Fig. 3) showed noticeable differences between the 2011 and 2012 seasons and when compared to Denver's long-term average (1981–2010). In 2011, average monthly air temperatures were below average, particularly in May, from the long-term mean (−2.8 °C for FL, −4.0 °C for RF). Seasonal averages (April–October) were 16.3 °C for the urban lawn site and 15.7 °C for the prairie site (Denver 1981–2010: 16.6 °C). In contrast, monthly mean temperatures for the 2012 season were typically above-average, most notably in April and June (3.1 °C and 2.6 °C for FL;

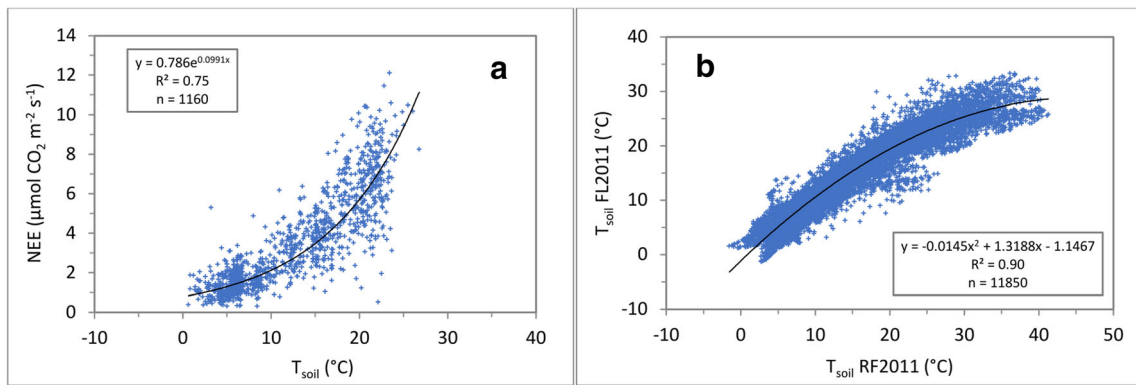


Fig. 2 (a) Relation between soil temperature and nighttime *NEE* at the urban lawn site (data: nighttime *NEE* when $R_s < 10 \text{ W m}^{-2}$; Foken Quality Control level = 1 (Foken et al., 2004)) and (b) Relation between soil temperature at the prairie (RF) and soil temperature at the urban lawn (FL)

2.1 °C and 2.9 °C for RF). Seasonal averages were also higher than the mean (FL: 17.4 °C; RF: 17.3 °C).

Precipitation at the lawn during the 2011 season was above average in May (+31 mm, 155% of normal) and July (+38 mm, 167% of normal) but below average in August (−41 mm, 6% of normal). Total seasonal precipitation (April–October) at the urban lawn was 331 mm in 2011, exceeding Denver’s long-term mean (299 mm). During the 2012 season, precipitation was below average in each month except for September (+26 mm, 205% of normal). Total seasonal precipitation was 226 mm and considerably lower compared to the 2011 season and Denver’s long-term mean. Total water input, *P* (precipitation and irrigation), at the lawn was 1059 mm and 1107 mm in 2011 and 2012, respectively, clearly illustrating the contribution of irrigation to available water.

Monthly precipitation sums at the prairie site (where there is no irrigation) in 2011 showed similar deviations as the urban

lawn had from Denver’s 1981–2010 mean, e.g., in May (+57 mm, 202% of normal), July (+41 mm, 173% of normal), and August (−34 mm, 22% of normal). Seasonal precipitation (April–October) totaled 411 mm with the wettest month being May at 113 mm. In 2012, seasonal precipitation summed to 225 mm, 25% below Denver’s climatic mean for those months.

Denver’s official climate record for 2011 (NOAA 2012) showed monthly deviations that agreed well with variations observed at the urban and prairie site. Regarding precipitation, 2011 was a wet year (+59 mm, 115%), mainly due to above-average precipitation in May, June, and July. In contrast, 2012 was characterized by severe drought conditions. Average air temperature in Denver was nearly 2 °C above the long-term mean and intensifying drought conditions during the summer months of June, July and August limited total precipitation to 257 mm, 32% below average (NOAA 2013c).

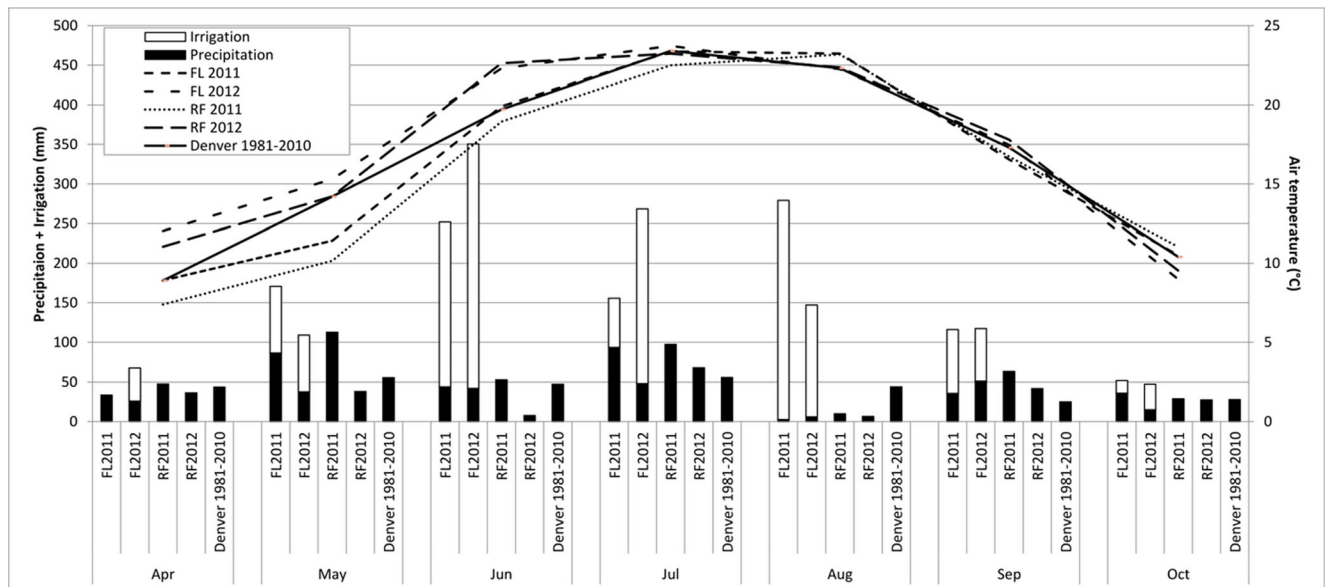


Fig. 3 Monthly averages of air temperature and total monthly precipitation (+ irrigation) at the urban lawn (FL) and the prairie site (RF) in comparison to Denver’s climate normal (1981–2010)

Energy partitioning and energy balance closure

For the following discussion, we consider this form of the surface energy balance equation:

$$R_n - G - H - LE = A$$

where the energy flux terms are net radiation (R_n), soil heat (G), sensible heat (H), and latent heat (LE). We consider radiative fluxes toward the surface positive, while non-radiative energy fluxes away from the surface are positive and vice versa. The residual term (A), an indicator of lack of energy balance closure, is 0 under ideal conditions with the assumption of no measurement errors and that other biophysical fluxes, including storage fluxes, are negligible.

Analysis of energy partitioning between April and October using average daytime (06:00–18:00 MST) energy flux values indicated that net radiation (R_n) over the lawn was primarily converted into latent heat (LE). The share of sensible heat (H) was comparable to LE in April of 2011 (35% and 40%) (Fig. 4) but by June in both years, LE clearly dominated as daytime values amounted to 64% and 71% of R_n in 2011 and 2012, respectively. While LE accounted for 76% of R_n in August 2011, this share was down to 57% in August 2012, coinciding with severe drought conditions and intermittent irrigation. The residual term (A) ranged between 6 and 27% of R_n in 2011 and between 17 and 36% in 2012.

The prairie site generally showed a more balanced distribution between H and LE . Midday values for H in April and May of 2011 amounted to 47% and 42% of R_n , respectively, compared to 17% and 18% for LE . The relative share of LE peaked in July, clearly exceeding H (47% vs. 22%). Similar to the urban lawn, drought conditions in 2012 affected energy partitioning, mainly during the summer: a very dry August resulted in relative shares of H and LE of 47% and 18%, respectively. The residual term (A) ranged between 18 and 27% of R_n in 2011 and between 22 and 32% in 2012.

The analysis of energy balance closure (using ordinary least squares linear regression – non-gapfill, quality-controlled data only) indicated a general lack of closure. Greatest discrepancies in energy balance (absolute values) at both sites were usually found between midday and early afternoon, the time of greatest magnitude with regard to relevant fluxes. Including storage change in G generally increased slopes of the derived regression lines, whereas R^2 remained nearly unchanged.

Regression analysis using 30-min average data (data not shown) yielded a slope, intercept, and R^2 of 0.84, -2.3 , and 0.90, respectively, for the urban lawn in 2011. The energy balance ratio ($EBR = \Sigma(LE + H) / \Sigma(R_n - G)$) amounted to 0.82. In 2012, the regression coefficients for slope and intercept were 0.73 and 3.1, while R^2 equaled 0.94. The EBR was calculated to be 0.76. Regression analysis for the prairie resulted in very similar results for 2011 and 2012. For 2011, slope, intercept, and R^2 were 0.79, -11.4 , and 0.94, respectively. In 2012, the coefficients showed little change with a slope, intercept, and R^2 of 0.77, -10.9 , and 0.95, respectively. The EBRs were 0.69 in 2011 and 0.67 in 2012. Using 24-h averages for analysis improved energy balance closure slightly, primarily for the lawn site (Fig. 5). Poorer energy balance closure in 2012 at both sites may have been due to impacts of drought on footprint vegetation.

Evapotranspiration

As indicated by the energy partitioning data, ET showed distinct seasonal courses. April daily ET at the urban lawn averaged $1.6(\pm 0.6)$ mm d^{-1} in 2011 and $2.1(\pm 0.8)$ mm d^{-1} in 2012. In comparison, prairie ET average daily sums, as well as monthly sums, were approximately 50% lower (Figs. 6 and 7). Higher ET rates in the spring of 2012 compared to 2011 likely resulted from above-average temperatures and higher monthly averages of VPD, and, in the case of the urban lawn, an earlier start of irrigation.

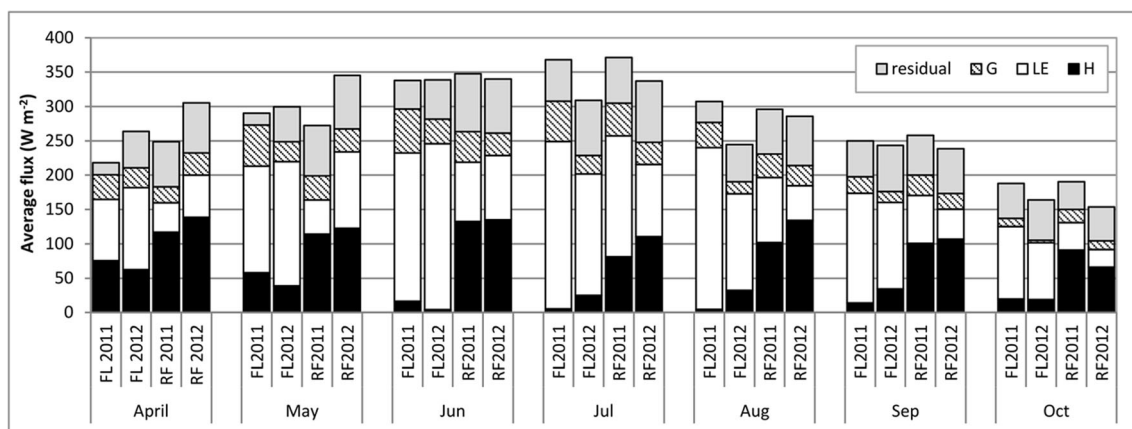
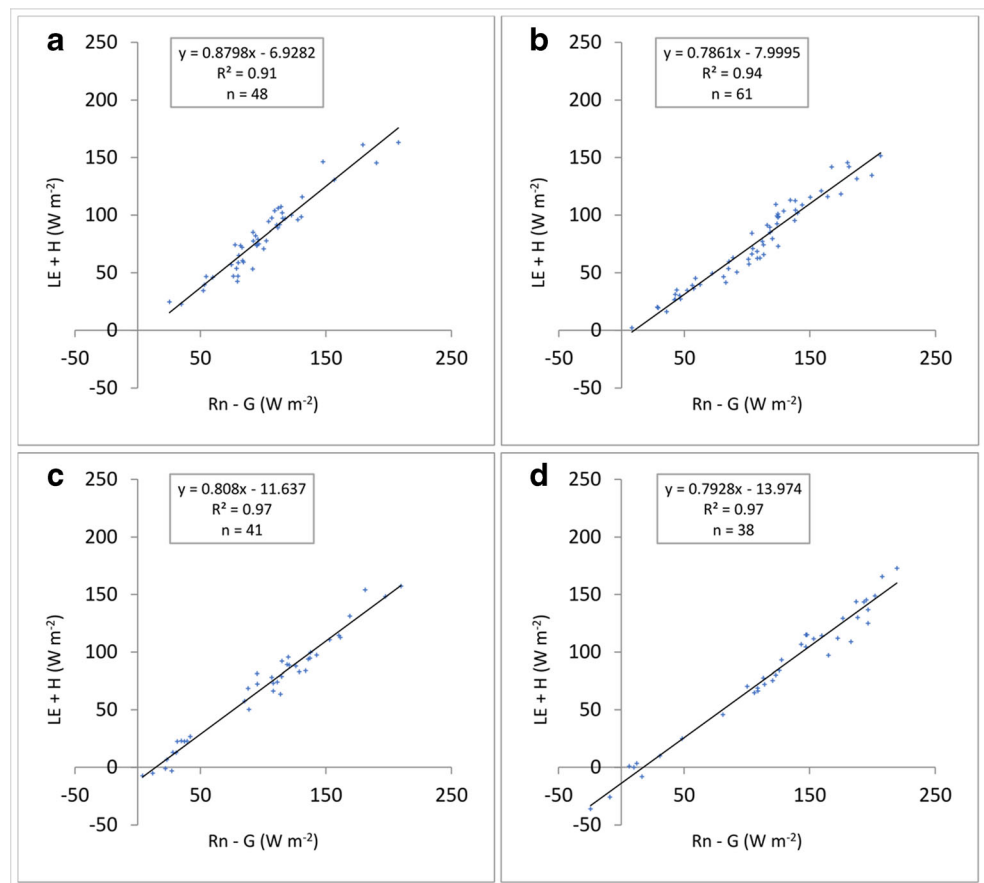


Fig. 4 Average daytime (06:00–18:00 MST) fluxes of sensible heat (H), latent heat (LE), soil heat (G) and residual at the lawn (FL) and the prairie site (RF) (2011 and 2012) ($W m^{-2}$). The sums of all components in each bar equal R_n

Fig. 5 Energy balance closure ($R_n - G = H + LE$; 24-h-averages) for the urban lawn site (FL) in 2011 (a) and 2012 (b) and the prairie site (RF) in 2011 (c) and 2012 (d). Black lines represent regression lines (OLS)



In June, average daily ET for the lawn had increased to $4.1(\pm 1.2)$ mm d^{-1} in 2011 and $4.5(\pm 1.3)$ mm d^{-1} in 2012, coinciding with high water input from irrigation and high VPD. Average total ET per day remained near 4.0 mm d^{-1} throughout July and August in 2011, while in 2012 the respective monthly averages decreased to $3.2(\pm 0.8)$ mm d^{-1} and $2.6(\pm 0.7)$ mm d^{-1} . This decrease was paralleled by a decline in soil moisture and LAI (Fig. 11) until mid-August as well as generally lower, intermittent irrigation in comparison to the previous year. Shortly after irrigation had stopped in October of both years, average daily ET dropped to about 1 mm d^{-1} or less.

Similar to the lawn, ET rates at the prairie increased through the spring and early summer of 2011, peaking in July when daily ET rates averaged $3.0(\pm 0.7)$ mm d^{-1} . In contrast, ET rates in 2012 reached a first peak in May before declining again in June, paralleled by gradually decreasing soil moisture due to a lack of precipitation (Fig. 11). This trend continued into the first week of July when daily ET dropped to less than 1 mm d^{-1} . Following strong precipitation events, daily ET sums temporarily recovered to 2 – 3 mm d^{-1} , coinciding with some of the highest seasonal values for LAI (Fig. 9). In August, daily ET averaged $1.7(\pm 0.4)$ mm d^{-1} in 2011 and $0.9(\pm 0.3)$ mm d^{-1} in 2012 which were about half that of July's values due to lower monthly precipitation and soil moisture, and high VPD.

Overall, seasonal ET (April–October) at the lawn reached $639(\pm 17)$ mm in 2011 with 331 mm precipitation and 728 mm irrigation ($ET/P = 0.60$) while in 2012 the totals were $584(\pm 15)$ mm of ET with 226 mm precipitation, and 881 mm of irrigation ($ET/P = 0.53$). The amount of water utilized by ET of the lawn was more than twice that at the prairie site. Here, the respective values for ET in 2011 and 2012 were $302(\pm 9)$ mm (precipitation: 412 mm , $ET/P = 0.73$) and $265(\pm 7)$ mm (precipitation: 225 mm , $ET/P = 1.18$).

Net ecosystem exchange of CO_2

Daily sums and cumulative trends for NEE of CO_2 varied between years and sites, reflecting differing climate conditions and management. For the lawn, daily sums of NEE in April 2011 (Fig. 8) were mainly positive (average: $+0.3(\pm 0.4)$ $\text{g C m}^{-2} \text{d}^{-1}$). This indicated a net release of CO_2 as turbulent fluxes away from the surface towards the atmosphere were considered positive and vice versa. In early May there was a distinct increase in positive daily sums following a marked peak in soil temperature, the onset of irrigation and coinciding with several lows of the daily light integral (DLI) (Fig. 11). During this period, NEE approached and exceeded at times $+2$ $\text{g C m}^{-2} \text{d}^{-1}$. Shortly after fertilization on June 15, net uptake of CO_2 started to dominate (June average: -0.7

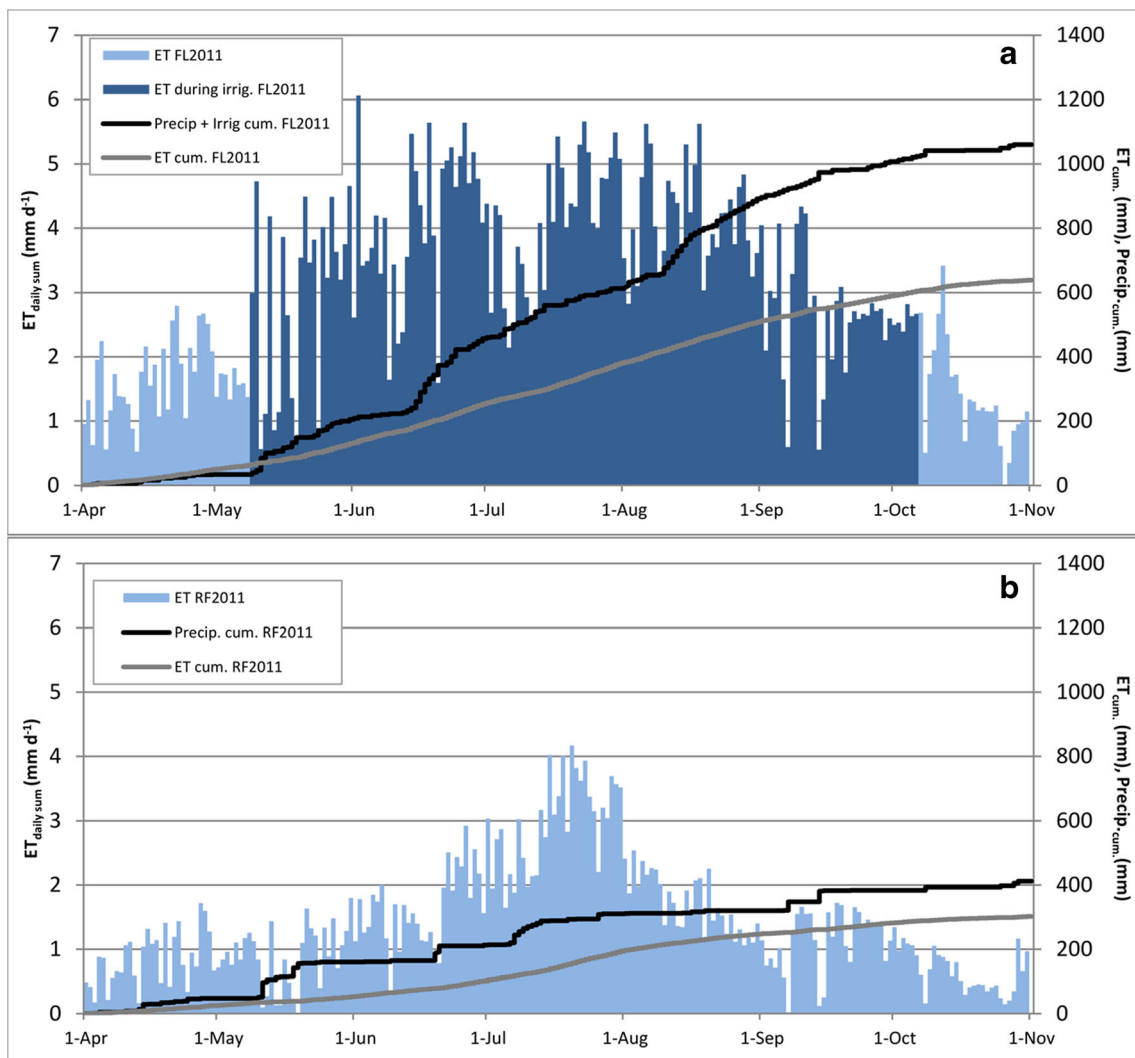


Fig. 6 Growing season cumulative and daily sums of evapotranspiration (ET) and precipitation (including lawn irrigation) at the urban lawn site (FL) (a) and the prairie site (RF) (b) in 2011

(± 0.9) $\text{g C m}^{-2} \text{d}^{-1}$) and continued in July, reaching a first seasonal peak near the middle of the month when values were close to $-3 \text{ g C m}^{-2} \text{d}^{-1}$ (average: $-1.6 (\pm 0.9) \text{ g C m}^{-2} \text{d}^{-1}$). Net CO_2 uptake slowed in August (average: $-0.9 (\pm 0.8) \text{ g C m}^{-2} \text{d}^{-1}$) which also had the highest monthly average VPD. Net CO_2 uptake increased again in September and remained high throughout the month (average: $-2.1 (\pm 1.0) \text{ g C m}^{-2} \text{d}^{-1}$), coinciding with the season's highest values for LAI (Fig. 11). Net daily uptake of CO_2 prevailed until the middle of October.

In contrast to 2011, the urban lawn in April 2012 (Fig. 8) already showed discernible net uptake (average: $-0.4 (\pm 0.9) \text{ g C m}^{-2} \text{d}^{-1}$). Paralleled by increasing LAI, net uptake continued to grow stronger in May (average: $-1.9 (\pm 1.7) \text{ g C m}^{-2} \text{d}^{-1}$), but during the last week of the month, NEE became positive, following and overlapping with a period of practically no water input and declining soil moisture. In the first half of July following a spike in water input (Fig. 7), NEE became predominantly negative, indicating uptake. The remaining month was characterized

by positive NEE , i.e., an overall loss of carbon. This period coincided with intermittent irrigation, declining soil moisture, high soil temperature, and a noticeable drop in LAI (Fig. 11), all of which continued into early August. Until mid-August, daily NEE sums were all positive, at times losses exceeded $4 \text{ g C m}^{-2} \text{d}^{-1}$ (August average: $+1.3 (\pm 1.7) \text{ g C m}^{-2} \text{d}^{-1}$). Within these three weeks of severe drought conditions, approximately 50 g C m^{-2} were released before irrigation resumed in early August. Irrigation was again reduced at the end of August and led to a loss of carbon lasting into early September. With heavier precipitation events occurring around mid-September, soil moisture and LAI (Fig. 11) recovered, net uptake of CO_2 started to dominate again and continued throughout October (October average: $-0.9 (\pm 1.0) \text{ g C m}^{-2} \text{d}^{-1}$).

NEE of CO_2 at the prairie site in April 2011 (Fig. 9) was weak, but became increasingly negative in May (average: $-0.2 (\pm 0.4) \text{ g C m}^{-2} \text{d}^{-1}$), especially during the second half of the month. This trend continued into and past June

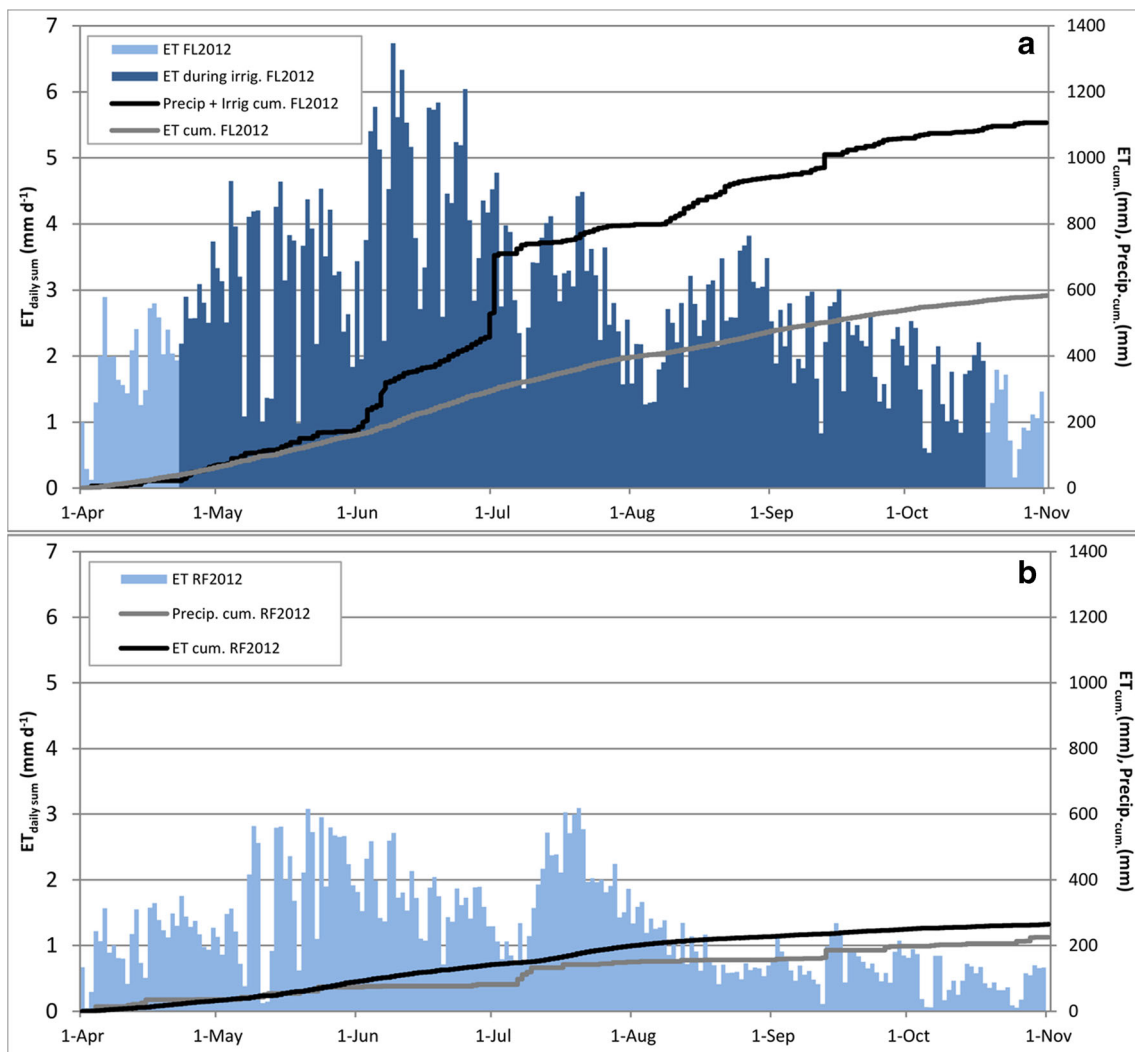


Fig. 7 Growing season cumulative and daily sums of evapotranspiration (ET) and precipitation (including lawn irrigation) at the urban lawn site (FL) (a) and the prairie site (RF) (b) in 2012

(average: $-0.8(\pm 0.5)$ g C $m^{-2} d^{-1}$) with the highest daily accumulation of carbon observed in late July when daily sums for *NEE* were near -2.5 g C $m^{-2} d^{-1}$. The summer months, from June to early August, were generally characterized by net uptake, paralleled by increasing LAI values (Fig. 11). During the second half of August, uptake slowed as VPD peaked, LAI declined, and precipitation was minimal. Weak uptake continued until early October. Occasionally, uptake during the season was interrupted after heavy precipitation events, followed by short periods of net release of CO₂.

Similar to the lawn in 2012, net uptake dominated at the prairie site by mid-April 2012 (Fig. 9) and became increasingly stronger in May (average: $-0.6(\pm 0.6)$ g C $m^{-2} d^{-1}$) coinciding with increasing values of LAI (Fig. 11) and almost regular precipitation. Late May showed the highest daily uptake sums which approached -1.8 g C $m^{-2} d^{-1}$. Stronger net uptake continued into June (average: $-0.7(\pm 0.6)$ g C $m^{-2} d^{-1}$) but became weaker as the month progressed while VPD

increased and soil moisture declined (Fig. 11). Similar to 2011, heavy precipitation events occasionally interrupted net uptake, especially in early July. Uptake lasted until late August but daily sums were small (average: $-0.1(\pm 0.2)$ g C $m^{-2} d^{-1}$). Net loss of CO₂ characterized September and October (average: $+0.3(\pm 0.2)$ g C $m^{-2} d^{-1}$).

Overall, cumulative *NEE* between April and October for the lawn was $-173(\pm 23)$ g C m^{-2} in 2011 and $-73(\pm 22)$ g C m^{-2} in 2012. The respective sums for the prairie were $-81(\pm 10)$ g C m^{-2} and $-21(\pm 8)$ g C m^{-2} .

Comparison of annual carbon budgets

Data for annual *NEE* (including modeled *NEE* for the lawn for months outside the measurement period) indicated that both the lawn and tallgrass prairie sites were net sinks for CO₂ in 2011 and 2012 (Table 1, Fig. 10). In both years, the CO₂ sink strength of the lawn was twice that of the prairie. Prolonged regional

Table 1 Estimates for NEE at the lawn (FL) and prairie sites (RF). Negative numbers indicate net carbon uptake and superscripts indicate method of computation: a - data modelled using soil temperature/night NEE regression; b - gap-filled data following Reichstein et al. (2005); c - gap-filled data including data derived from soil temperature/NEE regression and gap-filling after Reichstein et al. (2005)

Cumulative NEE (g C m ⁻²)				
Period	FL 2011	FL 2012	RF 2011	RF 2012
Jan - Feb	+38 ^a	+40 ^a	+11 ^b	+7 ^b
Mar – Nov	-190 ^b	-78 ^b	-76 ^b	-19 ^b
Dec	+21 ^a	+20 ^b	+4 ^b	+3 ^b
Growing Season (Apr-Oct)	-173(±23) ^b	-73(±22) ^b	-81(±10) ^b	-21(±8) ^b
Annual	-131(±24) ^c	-18(±22) ^c	-61(±10) ^b	-9(±8) ^b

drought conditions in 2012 coincided with reductions in sink strength of approximately 85% at both locations. Moreover, management activities for the lawn led to direct and indirect emissions of CO₂ which further impacted annual budget estimates. Applying the results of Townsend-Small and Czimczik (2010), including direct emissions from fossil fuel use (i.e., general maintenance, mowing, aeration, etc.) resulted in a monthly carbon offset of approximately +3 g C m⁻². Indirect emissions due to irrigation had a considerably stronger impact on the annual carbon budget as total management emissions increased by +53 g C m⁻² annually. Carbon emissions due to the production of fertilizer further added approximately +6 g C m⁻² per application. As a result, management-related CO₂ emissions for the

lawn in 2011 offset annual NEE by nearly two-thirds (-49 g C m⁻² a⁻¹), while in 2012, these emissions exceeded net uptake indicating this urban lawn was a net source of carbon (+60 g C m⁻² a⁻¹).

Discussion

Evapotranspiration

As shown in Figs. 5 and 6, the highest daily *ET* sums at the lawn and prairie sites occurred during the summer months when the impact of parameters including precipitation,

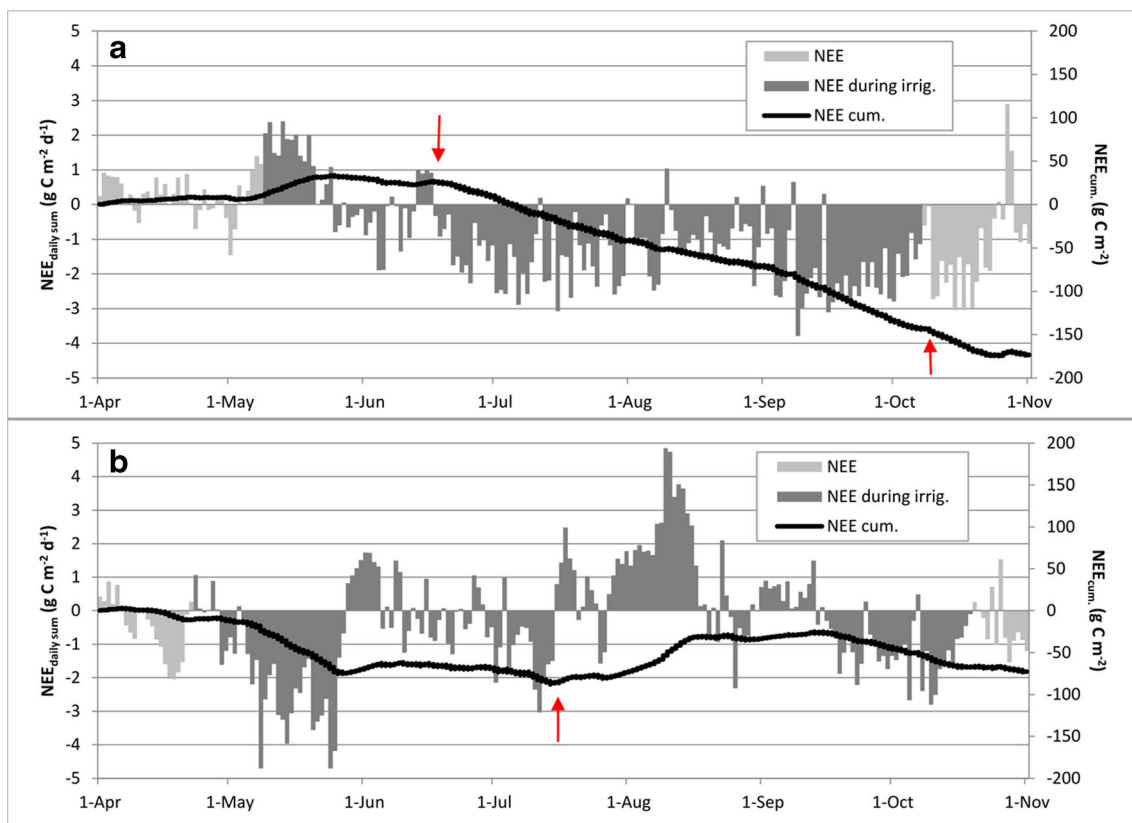


Fig. 8 Seasonal course of net ecosystem exchange of CO₂ (NEE, daily sums and cumulative) at the urban lawn site (FL) in 2011 (a) and 2012 (b) – red arrows indicate times of fertilizer application

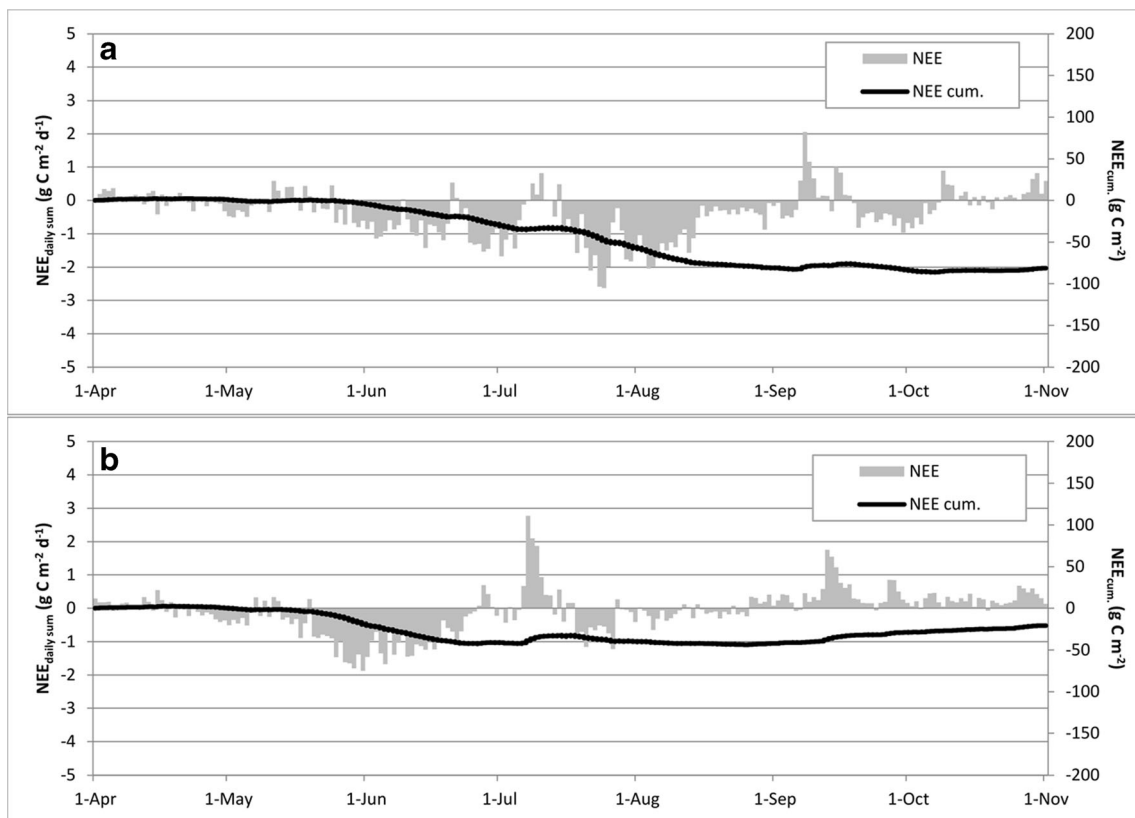


Fig. 9 Seasonal course of net ecosystem exchange of CO_2 (NEE, daily sums and cumulative) at the prairie site (RF) in 2011 (a) and 2012 (b)

irrigation, R_n , VPD, and LAI was generally greatest (Figs. 3, 4, 11). For the lawn, this finding agrees well with Peters et al. (2011) who reported highest average daily sums near 4 mm d^{-1} during June for an unirrigated turfgrass site in Minneapolis, Minnesota. Higher daily ET rates for irrigated lawns (up to 10 mm d^{-1}) were found by Feldhake et al. (1983) for sites in Colorado and by Litvak et al. (2013) in California. Romero and Dukes (2009) reported a span for ET of $3\text{--}12 \text{ mm d}^{-1}$ for different turfgrass species (e.g., fescue, Kentucky bluegrass).

In comparison to the lawn, the mix of warm-season grasses at the prairie site displayed a similar timing but lower absolute values ($<3 \text{ mm d}^{-1}$) for ET maxima. Here, highest ET rates were also lower than the average (5 mm d^{-1}) reported by Bremer et al. (2001) or the range ($3.5\text{--}5 \text{ mm d}^{-1}$) by Burba and Verma (2005) for tallgrass prairie sites in Kansas and Oklahoma, respectively. However, average ET rates ($1.2\text{--}2.1 \text{ mm d}^{-1}$) between May and October at a shortgrass prairie site in NE Colorado (Ferretti et al. 2003) compared well to the values at this study's prairie site ($1.3\text{--}1.5 \text{ mm d}^{-1}$), likely due to both prairies experiencing a similar climate.

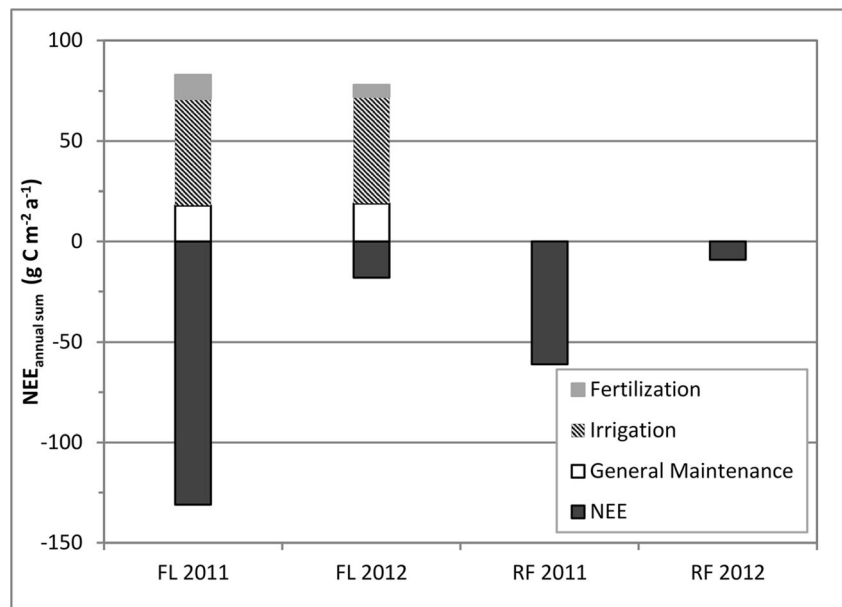
The observed differences in management (irrigation) and environmental conditions (e.g., soil moisture, type/development of vegetation) between sites also affected total ET . For the lawn, ET sums between April and October were about 25–36% higher in comparison to the annual ET reported

by Peters et al. (2011) for a suburban Minneapolis site dominated by turfgrass (74% of land cover). Annual ET at the prairie site (2011: $349(\pm 9) \text{ mm}$; 2012: $310(\pm 7) \text{ mm}$) was within the span found in the literature for grasslands in North America (e.g., Krishnan et al. (2012) reported $196\text{--}284 \text{ mm}$ and Burba and Verma (2005) found $637\text{--}807 \text{ mm}$).

Urban lawn and tallgrass prairie also differed with regard to ET/P , an indicator for how efficiently ecosystems utilize available water. Urban ecosystems are typically characterized by low ET/P because impervious surfaces and sewer systems increase runoff (Moriwaki and Kanda 2004). For urban areas containing large plots of green space, this may also be a sign of excess water use for irrigation. For the lawn, ET/P showed little difference between seasons (0.60 vs. 0.53) as total ET and water input over the investigated periods were very similar. Peters et al. (2011) in their study of suburban ET found ratios of $0.42\text{--}0.61$ for a residential area with about one-third turfgrass cover and of $0.62\text{--}0.85$ for a recreational area with nearly three-quarters covered by turfgrass.

In comparison to the urban lawn, ET/P for the prairie site was higher and differed notably between years, likely a result of thicker and generally drier soils. The ET/P value in 2011 was 0.73 but in the drought year 2012, ET/P exceeded unity (1.18). This may have been due to a higher water-use efficiency of prairie vegetation and/or its ability to access water deeper within the soil that was stored during the previous,

Fig. 10 Estimated annual sums ($\text{g C m}^{-2} \text{ a}^{-1}$) of *NEE* and carbon emission offsets due to management at the lawn (FL) and the prairie site (RF) (2011 and 2012)



wetter year. Similar patterns of increasing *ET/P* during drought years have been found for a semi-arid grassland in Arizona where values ranged between 0.74–0.84 but peaked at 1.20 during the driest year (Scott 2010). Other studies have also reported increased *ET/P* for grasslands during drought years, such as Aires et al. (2008) and Hussain et al. (2011).

Net ecosystem exchange of CO₂

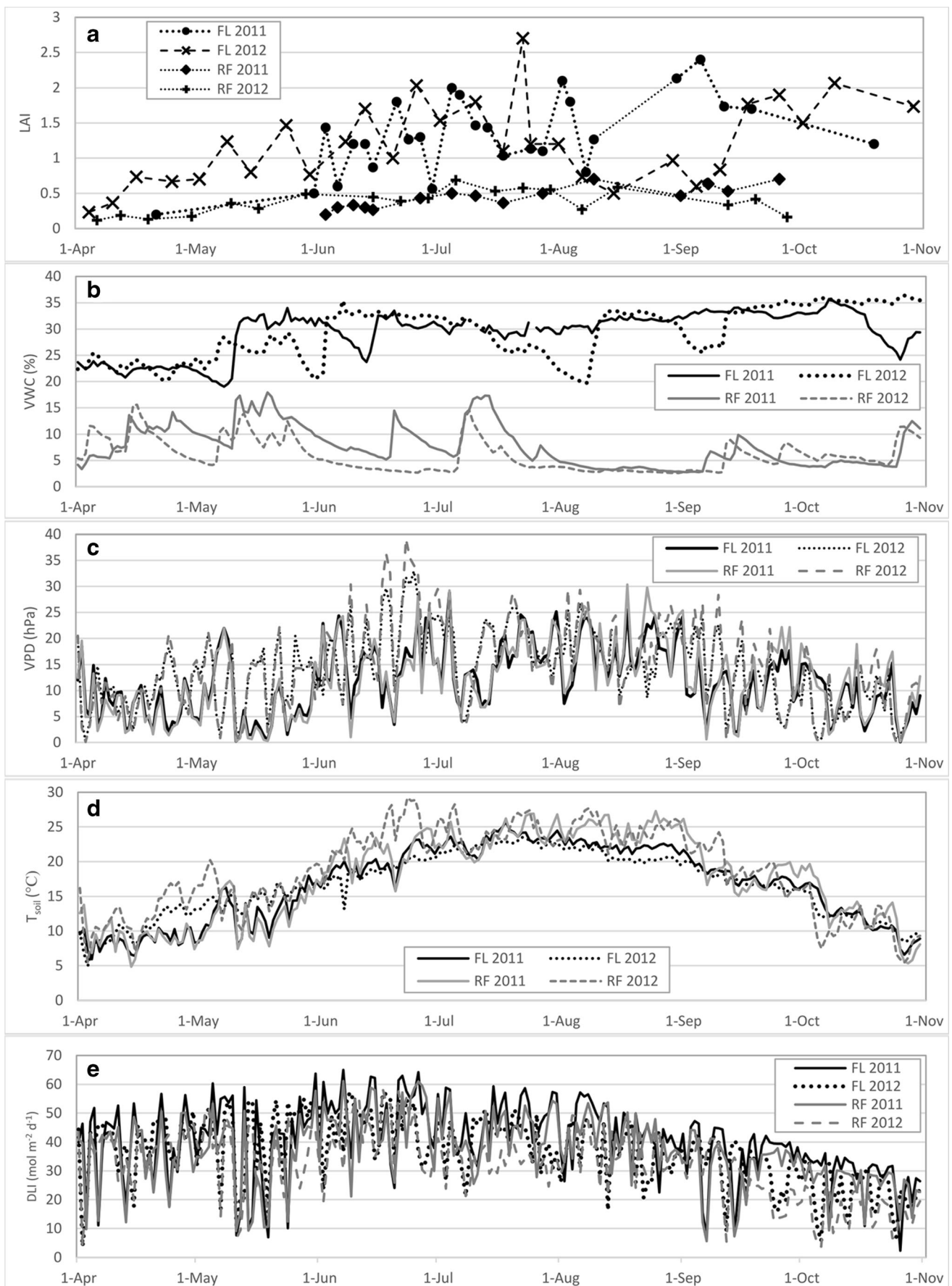
Urban lawn

During the 2011 season, *NEE* was dominated by net uptake of CO₂ as more than 70% of daily *NEE* sums indicated net uptake. Brief periods of net emission occurred in response to specific environmental conditions and management. For example, a distinct period of net emissions in mid-May 2011 coincided with a rapid increase in soil temperature (daily averages increased 8.0°C within 7 days) and the start of irrigation. The influence of soil temperature on soil respiration is well established and remains a subject of research (Fang and Moncrieff 2001; Risk et al. 2002; Hibbard et al. 2005; Davidson et al. 2006; Graf et al. 2008; Karhu et al. 2014). But soil respiration in most terrestrial ecosystems is also influenced by soil moisture (Risch and Frank 2007; Balogh et al. 2011; Lellei-Kovács et al. 2011). The observed pulses of soil CO₂ efflux following the re-wetting of (dry) soils (also known as the “Birch effect” – see discussion below) have been reported in other studies (e.g., Lee et al. 2004; Jarvis et al. 2007; Chowdhury et al. 2011; Kim et al. 2012). The immediate response of *NEE* to precipitation/irrigation events, however, was difficult to estimate since data during these events was often incomplete due to the interference of water on the open-path IRGA instrument. Fertilization may also have led to short

periods of net emissions or clearly reduced uptake in both seasons as fertilizer application can considerably enhance soil respiration (Fierer et al. 2003; Verburg et al. 2004). For most of the remaining period in 2011, net uptake dominated but was not uniform in strength. *NEE* noticeably slowed in August 2011 despite both, high irrigation input and LAI. We observed small, positive daily sums of *NEE* (net emission) during some of the warmest days. High air temperatures and high VPD likely stressed vegetation, reducing photosynthetic activity and weakening uptake (Mathur et al. 2014).

NEE in 2012 appeared to be impacted by above average heat, especially drought conditions, and lack of sufficient irrigation. Above-average temperatures and earlier vegetation development (as indicated by LAI) in spring of 2012 led to increases in soil respiration and photosynthetic fluxes, resulting in greater diurnal amplitudes of *NEE* compared to 2011. Irrigation was a major influence on *NEE* through feedbacks via soil moisture and LAI. These factors primarily determined the seasonal course of *NEE* until early September, together with record high air temperatures and high VPD (Fig. 11). The effect of irrigation limits became apparent in late May and more so in late July/August. When irrigation was stopped for periods of a few days or more, soil moisture and LAI declined, photosynthetic flux was reduced and net losses of carbon followed. The resilience of the lawn preserved seasonal net uptake of CO₂ as LAI

Fig. 11 Panels from top down are: (a) Leaf area index sample average (LAI), (b) daily averages of volumetric water content (VWC), (c) vapor pressure deficit (VPD), (d) soil temperature (T_{soil}) and (e) daily light index (DLI) at the urban lawn site (FL) and prairie site (RF) in 2011 and 2012



repeatedly recovered once irrigation resumed, resulting in negative *NEE*, indicating uptake.

The effect of lawn-mowing on *NEE* could not be clearly identified as mowing varied temporally and spatially on the lawn. Reduced LAI may have contributed to decreased net uptake of CO₂, but data that coincided with or followed lawn-mowing, also showed other, potentially influencing factors (e.g., high temperature, low soil moisture, low photosynthetically active radiation (PAR), increased photosynthesis by previously shaded vegetation), confounding interpretation.

To evaluate *NEE* of CO₂ at the urban lawn, data drawn for comparison also included urban studies that contained a significant portion of lawns (e.g., suburban neighborhoods, park landscapes), since studies investigating *NEE* of lawns are still relatively rare. Bergeron and Strachan (2011) studied *NEE* at a suburban site in Montreal, Canada, where in spring (April–May) and fall (September–November) *NEE* was generally close to zero and in summer (Jun–Aug) showed an average midday uptake of $-7 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$. In comparison, diurnal averages observed at this study's urban lawn were usually stronger, reaching up to -10.6 (spring 2012), -11.8 (fall 2011), and $-13.3 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ (summer 2011). For other suburban sites, Kordowski and Kuttler (2010) reported average summer *NEE* maxima of up to $-10 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$, while Buckley et al. (2014) working in Syracuse, New York, found a midday average CO₂ flux of $-11 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ during the summer months. Average *NEE* during June–August in suburban Baltimore, Maryland, ranged between -14 and $+10 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ and diurnal amplitudes showed a sensitivity to PAR and soil temperature (Crawford et al. 2011). *NEE* measured at suburban/residential sites could also be influenced by anthropogenic emissions (e.g., traffic) but, in general, these results compare well to this study's data.

Studies that aim to quantify annual net carbon uptake of turfgrass are often based on the analysis of changes in soil carbon stocks as the carbon is mostly stored in the soil and not in shoots or roots (Guertal 2012). Qian and Follett (2002) investigated turfgrass sites at golf courses of different ages (mainly in Colorado) and found average net uptake rates of -90 to $-100 \text{ g C m}^{-2} \text{ a}^{-1}$ during the first 30 years following establishment while Qian et al. (2010) reported a narrower range of -34 to $-78 \text{ g C m}^{-2} \text{ a}^{-1}$. Milesi et al. (2005) estimated that turfgrass in the United States could uptake between -36 to $-100 \text{ g C m}^{-2} \text{ a}^{-1}$. Carbon uptake for ornamental lawns in Irvine, California, was $-140 \text{ g C m}^{-2} \text{ a}^{-1}$ (Townsend-Small and Czimczik 2010). These values compare well to the estimated annual uptake for the urban lawn in this study in 2011 ($-131(\pm 24) \text{ g C m}^{-2} \text{ a}^{-1}$) but also illustrated how severely summer drought conditions impacted carbon uptake potential in 2012 ($-18(\pm 22) \text{ g C m}^{-2} \text{ a}^{-1}$).

Tallgrass prairie

Like the urban lawn, the tallgrass prairie showed substantial *NEE* of CO₂ variability day-to-day, through the course of a season, and between years. Similar variability has been observed in various other (multi-year) studies in the Great Plains, including changes in ecosystem function from carbon sink to source (Frank and Dugas 2001; Sims and Bradford 2001; Polley et al. 2008; Parton et al. 2012). These differences in (annual) carbon budgets are often attributed to climatic variability which can directly and indirectly impact *NEE*, for example, by influencing aboveground-net primary productivity in grasslands (Knapp and Smith 2001; Flanagan et al. 2002; Xu and Baldocchi 2004).

NEE at the prairie site during the 2011 season followed a seasonal cycle strongly influenced by soil moisture and temperature conditions, solar radiation, VPD and feedbacks via LAI. Diurnal cycles of *NEE* in spring were weak but were evidence that rising soil temperatures stimulated soil respiration while early vegetative growth initiated uptake of CO₂, which strengthened further into the season. However, similar to the lawn, periods of net emission were also observed, e.g., in May (Fig. 9), following a rapid increase in soil temperature and significant precipitation but also low DLI. The individual impact of temperature and moisture on soil respiration is difficult to estimate as both parameters have been shown to influence soil CO₂ efflux in prairie ecosystems (Mielnick and Dugas 2000; Frank and Dugas 2001; Chimner and Welker 2005). Net CO₂ uptake dominated until late July/early August with rising LAI and significant precipitation input, but slowed abruptly thereafter, likely due to stress conditions (high temperatures and VPD, depleted soil moisture) impacting photosynthetic activity. Cooling temperatures, declining VPD, and moderate precipitation in early September led to a partial recovery of net uptake, but senescence of vegetation in early October ended net uptake.

Anomalous climate conditions in spring and summer of 2012 impacted *NEE* at the prairie site. Amount and timing of precipitation appeared as important parameters affecting strength and direction of CO₂ flux (uptake/emission) and annual carbon balance, a finding consistent with other studies (e.g., Frank and Dugas 2001; Sims and Bradford 2001; Huxman et al. 2004; Harper et al. 2005; St. Clair et al. 2009). Similar to the lawn, above-average springtime temperatures led to an earlier and more rapid development of vegetation. CO₂ uptake sharply increased in mid-May, enhanced by nearly regular precipitation and a further increase in LAI. However, vegetation vitality was impacted towards the end of the month and into June by changing environmental conditions. Lowering soil moisture with air temperature and VPD

increasing towards annual maxima led to drastically declining net uptake. Uptake temporarily resumed as precipitation in early July replenished soil moisture, but as air temperatures and VPD remained high and soil moisture decreased, daily uptake sums gradually decreased. By the end of August, drought conditions resulted in net emissions and the prairie became a net source of CO₂ until the end of October.

NEE of CO₂ of the prairie displayed a distinct sensitivity in summer to heavy precipitation events after periods of low or no precipitation during both years. Net uptake ceased after precipitation pulses, followed by a peak of net release of CO₂ (up to 10 g C m⁻² within one week) before uptake usually resumed. Notable events occurred around the same time in both years, i.e., early July and early-mid September; although during the drought year 2012 resultant net emissions appeared stronger. The CO₂ emission spike in a semiarid ecosystem following a precipitation pulse after a dry period is known as the “*Birch effect*” (Birch 1958), a subject of various studies (e.g., Huxman et al. 2004; Parton et al. 2012) including those in shortgrass (Munson et al. 2010) and tallgrass prairies (Liu et al. 2002). Interest in these precipitation-induced carbon losses is due to the fact that these events can lead to large CO₂ effluxes, representing a considerable portion of annual respiration (Ma et al. 2012).

CO₂ emission pulses may be the result of multiple processes (Ma et al. 2012). Initially, the infiltration of rain into the soil causes the physical displacement of CO₂-rich air from soil pores. The CO₂ is also generated by the stimulation of microbial activity by a sudden increase in soil moisture. Microbial activity may be further enhanced by increased carbon and nutrient availability through photo-degradation of dead biomass. This latter process has been shown to cause direct CO₂ emissions from litter (Rutledge et al. 2010). Parameters such as timing and magnitude of precipitation pulses also seem to exert an influence on respiration response (Harper et al. 2005; Munson et al. 2010; Ma et al. 2012).

Comparing cumulative carbon uptake to other studies illustrates that the span in uptake observed between years at the prairie site is not atypical. For example, Frank and Dugas (2001), measuring *NEE* over 4 years at a mixed prairie site, found that cumulative net uptake ranged between -50 to -130 g C m⁻² (April–October; average: -95 g C m⁻²). They also noted that, similar to this study’s prairie site, seasonal variability in *NEE* was clearly related to LAI/biomass responding to moisture and temperature stress and that maximum CO₂ flux occurred at the time of maximum LAI. Sims and Bradford (2001) reported an average annual uptake of -70 g C m⁻² a⁻¹ for a prairie site, comparable to this study’s prairie in 2011 (-61(±10) g C m⁻² a⁻¹) and emphasized the importance of the timing of precipitation. Similar observations regarding the change of grasslands from carbon sink to source in response to precipitation patterns have been made by Meyers (2001), Ma et al. (2007) and Xu and Baldocchi (2004). Polley et al. (2008) found that interannual

variability of net uptake was clearly reflected in average daily *NEE* during the growing season which varied by more than a factor of 3 between years, comparable to the ratio found for the prairie site. Daily *NEE* sums for this study’s prairie site during the growing season displayed a range (May–July 2011: -2.6 to +2.1 g C m⁻² d⁻¹; July–September 2012: -1.9 to +2.8 g C m⁻² d⁻¹) similar to that found by Suyker and Verma (2001) (July–August: -1.8 to +2.2 g C m⁻² d⁻¹). For the same tallgrass site (Suyker et al. 2003), the *annual* uptake was larger (-274 g C m⁻² a⁻¹) in comparison to this study’s xeric tallgrass prairie but severe drought conditions also reduced annual *NEE* by more than 80%, similar to what the prairie site experienced in 2012.

Impact of lawns in urban ecosystems

Reported data showed that available energy was primarily partitioned into *LE* over lawns, leading to cumulative *ET* at the urban lawn exceeding *ET* at the prairie site by more than factor of 2 between April and October. Considering these differences and the estimates for the land cover fraction of lawn in Denver (~ 40%; Thienelt and Frühauf 2007), a substantial influence of watered lawns through evaporative cooling on urban microclimate can be assumed. The cooling effect of urban green space on air and surface temperatures has been measured in other urban studies (Taha et al., 1991; Ca et al. 1998; Bonan 2000; Kong et al. 2014) and may partially help alleviate the urban heat island effect. Cooling facilitated by vegetation may also lead to reduced carbon emissions with lower demand for air conditioning (Ca et al. 1998; Pataki et al. 2006; Salamanca et al. 2014). Considering these potentials may benefit urban planning when trying to adapt urban areas to climate change, i.e., rising temperatures (Kong et al. 2014). But urban vegetation, including lawns, receives additional energy and resource inputs associated with regular maintenance (mowing, aerating, thatching) as well as application of synthetic inputs (fertilizer/pesticides) which can equal or even surpass that of agricultural ecosystems on a per-area basis (Robbins and Birkenholtz 2003; Alumai et al. 2009). Frequent irrigation can also put enormous pressure on water resources, especially in arid and semi-arid climates (Milesi et al. 2005). At the urban lawn site, precipitation was only 31% (2011) and 20% (2012) of total water input between April and October and, thus, irrigation exceeded precipitation by a factor of more than 2 and nearly 4, respectively. Kjelgren et al. (2000) reported that irrigation can amount to nearly half of annual municipal water use. At the household level, irrigation was estimated to account for 40–70% of domestic water use depending on regional climate, often showing considerable potential for water conservation (Hilaire et al. 2008).

Although year-to-year climatic variability notably impacted net carbon uptake, urban lawns were stronger sinks for CO₂ than the xeric tallgrass prairie as seasonal *NEE* sums of the lawn exceeded those of the tallgrass prairie by factors of more

than 2 and 3 for 2011 and 2012, respectively. The observed net uptake by urban lawns is consistent with other findings of substantial carbon storage in urban vegetation and soils. Churkina et al. (2010), for example, estimated that urban ecosystems in the US contain up to 10% of total US land carbon with 64% of this share stored in soils and a further 20% in vegetation. Golubiewski (2006) found that urban lawns in the Colorado Front Range had more biomass and higher carbon storage on a per-area basis than native grassland or agricultural land. Pouyat et al. (2009) reported soil organic carbon values for lawns in Denver that in some cases were more than double in comparison to shortgrass prairie. Thus, urbanization can enhance carbon uptake and carbon storage, especially in drier climates where combined higher productivity of urban vegetation and management can boost carbon storage in soils compared to natural ecosystems (Pataki et al. 2006; Pouyat et al. 2006; Townsend-Small and Czimczik 2010).

Including direct and indirect carbon emissions in a more complete accounting of the carbon budget for an urban lawn, however, can lead to substantial offsets to measured net uptake (Townsend-Small and Czimczik 2010; Zirkle et al. 2011). Estimated annual *NEE* for the urban lawn in 2011 was reduced by more than 60% when accounting for maintenance emissions, indicating that the lawn had become a slightly weaker carbon sink that year than the prairie site. For the drought year 2012, maintenance emissions shifted the urban lawn from sink to source while the prairie site remained a (weak) net carbon sink. Besides emissions related to management, other sources of CO₂ and greenhouse gases in urban areas (e.g., traffic, industrial activities) can be assumed to further offset potential carbon gains by urban lawns. Without decreases in general energy use and carbon intensity, urban ecosystems are unlikely to become substantive net carbon sinks despite harboring large carbon pools (Golubiewski 2006; Pataki et al. 2006; Churkina et al. 2010; Hutyra et al., 2011).

Conclusion

This study demonstrated that an urban lawn and xeric tallgrass prairie differ notably with regard to *NEE* of CO₂, energy partitioning, and *ET*, a finding that appears most relevant considering the present rapid expansion of urban areas such as Denver and predicted future urban growth. Vegetation acted as an important modifier for energy partitioning as well as rates and direction of net carbon exchange. Close links were found between seasonal vegetation development (i.e., LAI), energy fluxes, and *NEE* of CO₂. In Denver's semi-arid climate, water availability further influenced carbon and energy exchange. Irrigation at the urban lawn site greatly contributed to the differences in energy partitioning observed between turfgrass and prairie as the urban lawn showed a distinct shift

from *H* to *LE*, resulting in *LE* consuming, on average, more than 50% of available energy between April and October during the day. Greater water availability also allowed for a longer period of high daily *ET* sums and resulted in considerably higher seasonal totals over the lawn, which exceeded prairie totals by more than a factor of 2 (2011: 639(±17) mm vs. 302(±9) mm; 2012: 584(±15) mm vs. 265(±7) mm). Higher productivity and vegetation density of the lawn as well as management (e.g., irrigation and fertilization) led to large differences regarding diurnal and seasonal carbon fluxes. *NEE* of CO₂ for the urban lawn was affected by a longer growing season with higher daily net uptake and, hence, higher seasonal uptake sums of CO₂, exceeding those of tallgrass prairie by a factor of 2–3. Above-average temperatures (+1.8 °C) and reduced precipitation (–30%) in 2012, however, influenced diurnal and seasonal *NEE* at both research sites (2011: –173(±23) g C m^{–2} vs. –81(±10) g C m^{–2}; 2012: –73(±22) g C m^{–2} vs. –21(±8) g C m^{–2}). Vegetation vitality was clearly affected by temperature and water stress during the summer months, influencing the direction and magnitude of CO₂ flux. If the net carbon uptake by the lawn we measured is typical and given the spatial extent of urban lawns in Denver, our findings suggest that lawns can function as important carbon sinks/carbon pools within urban ecosystems but require considerable amounts of irrigation, particularly in a semi-arid climate. The transformation of natural grasslands to urban land uses could therefore increase carbon storage on a per-area basis but simultaneously strain water resources. The establishment of urban vegetation may therefore contribute to the mitigation of carbon emissions in urban areas to a certain degree, but assessments of the magnitude of these effects regarding urban carbon budgets also need to integrate emissions from maintenance and costs of resource allocation.

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Availability of data and material Urban lawn: <https://doi.org/10.5066/P9B70DC6>

Tallgrass prairie: <https://doi.org/10.5066/P9J466JY>

Code availability (software application or custom code) Not applicable.

Authors' contributions All authors contributed to the study conception and design. Material preparation, data collection and analysis were performed by Thomas S. Thienelt and Dean E. Anderson. The first draft of the manuscript was written by Thomas S. Thienelt and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

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Declarations

Conflicts of interest The authors declare that they have no conflict of interest.

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