Carbon dioxide and water vapor fluxes in winter wheat and tallgrass prairie in central Oklahoma

Rajen Bajgain[a], Xiangming Xiao[a*], Jeffrey Basara[b, c], Pradeep Wagle[d], Yuting Zhou[e], Hayden Mahan[b], Prasanna Gowda [d], Heather R. McCarthy[a], Brian Northup[d], Jim Neel[d] and Jean Steiner [d]

[a]Department of Microbiology and Plant Biology, Center for Spatial Analysis, University of Oklahoma, Norman, Oklahoma, USA

[b]School of Meteorology, University of Oklahoma, Norman, Oklahoma, USA

[c]Oklahoma Climate Survey, Norman, Oklahoma, USA

USDA-ARS Grazinglands Research Laboratory, El Reno, Oklahoma, USA

Department of Plant and Soil Sciences, Oklahoma State University, Oklahoma, USA

*Corresponding author

Xiangming Xiao

101 David L. Boren Blvd, Norman, OK73019, USA

Phone: 405-325-8941

Fax: 405-325-3442

Email: xiangming.xiao@ou.edu

Abstract

Winter wheat (*Triticum aestivum* L.) and tallgrass prairie are common land cover types in the Southern Plains of the United States. During the last century, agricultural expansion into native grasslands was extensive, particularly managed pasture or winter wheat. In this study, we measured carbon dioxide (CO_2) and water vapor (H_2O) fluxes from winter wheat and tallgrass prairie sites in Central Oklahoma using the eddy covariance in 2015 and 2016. The objective of this study was to contrast CO_2 and H_2O fluxes between these two ecosystems to provide insights on the impacts of conversion of tallgrass prairie to winter wheat on carbon and water budgets. Daily net ecosystem CO_2 exchange (NEE) reached seasonal peaks of - 9.4 and -8.8 g C m^{-2} in 2015 and - 6.2 and -7.5 g C m^{-2} in 2016 at winter wheat and tall grass prairie sites, respectively. Both sites were net sink of carbon during their growing seasons. At the annual scale, the winter wheat site was a net source of carbon (56 ± 13 and 33 ± 9 g C m^{-2} yr^{-1} in 2015 and 2016, respectively). In contrast, the tallgrass prairie site was a net sink of carbon (-128 ± 69 and -119 ± 53 g C m^{-2} yr^{-1} in 2015 and 2016, respectively). Daily ET reached seasonal maximums of 6.0 and 5.3 mm day^{-1} in 2015, and 7.2 and 8.2 mm day^{-1} in 2016 at the winter wheat and tallgrass prairie sites, respectively. Although ecosystem water use efficiency (EWUE) was higher in winter wheat than in tallgrass prairie at the seasonal scale, summer fallow contributed higher water loss from the wheat site per unit of carbon fixed, resulting into lower EWUE at the annual scale. Results indicate that the differences in magnitudes and patterns of fluxes between the two ecosystems can influence carbon and water budgets.

Keywords: carbon sink, evapotranspiration, land use change, net ecosystem exchange, ecosystem water use efficiency

1. Introduction

Land use has changed rapidly across much of North America during the last century, mainly due to intensification and expansion of agricultural cultivation in many central and western states (Turner and Meyer, 1994; Wright and Wimberly, 2013). In the Great Plains region, agricultural expansion into native grasslands has been extensive, as particularly either managed pasture or dryland crops such as wheat (winter and spring, *Triticum aestivum* L.) and sorghum (*Sorghum bicolor* L.) (Lark et al., 2015; Riebsame, 1990). The savannas and tallgrass prairie have been replaced by cultivated crops and only about 4 % of tall grass prairie which once covered a large portion of the central US remains today (Claassen et al., 2011; Fischer et al., 2007).

About 77% (2.3 million hectares) of new croplands in the US from 2008-2012 were originally grassland (Lark et al., 2015). Out of the converted land, 26% was planted to corn (*Zea mays L.*), followed by wheat (25%). The expansion of wheat was more common across the central plains, with spring wheat in the north and winter wheat in the south (Lark et al., 2015). Land cover remote sensing datasets from the Cropland Data Layer (CDL) produced by United States Department of Agriculture (USDA) National Agricultural Statistics Service (NASS) also showed about 11,000 km^2 of grassland was converted to winter wheat from 2008 to 2015 in the Southern Plains (Oklahoma, Texas) (Fig. S1a). The area converted from tallgrass prairie to winter wheat was highest in 2008 and lowest in 2013 (Fig.S1a inset bar graph). To understand the broader implications of such land use change in the region, a comparative analysis of carbon dioxide (CO_2) and water vapor (H_2O) fluxes of cultivated systems (e.g., winter wheat) and their native counterparts (tallgrass prairie) is useful which can provide insights into the resulting changes in carbon and water budgets.

Major sections of the Southern Great Plains are dominated by winter wheat, a C3 species, generally planted in September/October and harvested in June/July of the following year. Traditionally, a 3-4 months fallow period from harvest to planting is considered an important component of the farming system to accumulate soil moisture for the next wheat crop cycle (Dhuyvetter et al., 1996; Lyon et al., 2007). In contrast, the growing season of tallgrass prairie (mixture of C3 and C4 species, dominantly warm season) starts between March and May depending on spring temperature and remains active until September/October (Cooley et al., 2005; Fischer et al., 2007). The dynamics of the land surface processes resulting from the combined interactions of climate, vegetation cover, and management practices are closely coupled with the dynamics of the lower atmosphere and are very significant in the mid-continental regions such as the Southern Plains. Thus, the change from prairie to winter wheat shifts the magnitude and seasonal timing of energy, momentum, CO_2, and H_2O fluxes between the atmosphere and ecosystem in this region (McPherson and Stensrud, 2005). Further, the change in vegetation over an extended region may induce changes in the circulation patterns resulting into changes in weather conditions over much larger regions (Cooley et al., 2005; Fischer et al., 2007; Song et al., 1997). Observational analysis using Oklahoma Mesonet data (McPherson et al., 2007) demonstrated that the winter wheat belt in Oklahoma significantly altered the mesoscale atmospheric environment (Haugland and Crawford, 2005; McPherson et al., 2004). Numerical modeling simulations, conducted by replacing native grassland vegetation with winter wheat in Oklahoma, showed weakened winds within the planetary boundary layer due to insufficient sensible heat, which impacted the mesoscale circulation (McPherson and Stensrud, 2005). Similarly, spatial heterogeneity caused by intermixing of winter wheat in Oklahoma into grasslands induced the vertical velocities of 1-2 ms^{-1}, which is linked to convective cloud formation (Weaver and Avissar, 2001). Various studies in the past have quantified within-season and inter-annual variations in CO_2, H_2O, and energy fluxes in tallgrass prairies and crop fields of the Southern Plains (Fischer et al., 2012; Meyers, 2001; Suyker and Verma, 2001; Suyker et al., 2003). However, this study focuses on contrasting CO_2 and H_2O fluxes of winter wheat and tallgrass prairie ecosystems within the context of land use change, a significant human intervention in the native prairies of Southern Plains over the past century and continuing today. Specifically, the following questions were addressed in this study: a) What were the magnitudes and seasonal patterns of CO_2 and ET fluxes in tallgrass prairie and winter wheat? b) What is the impact of management activities (fallow on winter wheat site and grazing on tallgrass prairie site) on the total annual carbon and water budgets of the two sites? and c) what are the differences in seasonal and annual dynamics of ecosystem water use efficiency (EWUE) between winter wheat and tallgrass prairie sites? This study has a great significance to understand the impacts of land conversion from grassland to winter wheat on carbon and water budgets since the Southern Plains of the United States has seen the dramatic land use change in the last decades.

2. Materials and methods

2.1. Study sites

The measurements were conducted in two sites: a) native tallgrass pastureland (0.64 km^2) and b) winter wheat cropland (0.11 km^2) spaced about 2.7 km from each other at the US Department of Agriculture- Agricultural Research Service (USDA- ARS), Grazinglands Research Laboratory (GRL, 35.561319, -98.035742, 428m), in El Reno, Oklahoma (Fig.S1b). The average slope of the landscapes within the flux foot print is 1-2 % at winter wheat site and

2% at tallgrass prairie site. El Reno has a temperate continental climate with an average air temperature of 14.9 °C and an average annual rainfall of 860 mm for the 1971-2000 period (Fischer et al., 2012).

Tallgrass prairie is predominantly warm season vegetation representing the native, mixed species grassland of Oklahoma. The site has big bluestem (*Andropogon gerardi* Vitman) and little bluestem (*Schizachyrium halapense* (Michx.) Nash.) as dominant species. The soil is classified as Norge loamy prairie (Fine, mixed, thermic Udertic Paleustalf) with a depth greater than 1 m, high water holding capacity, and slope averaging about 1%. Historical management of the native pasture has varied over time (Fisher et al., 2012). In 2012 through present, the native pasture was combined with four other pastures of rangeland of similar size into a year-round system of rotational grazing with a 50-head herd of mature cows with calves. Pastures are grazed for 30-day periods, interspersed between 90-day rest periods, with individual pastures receiving prescribed spring burns on a 4-year rotation; the pasture was burned 3/6/2013 as part of normal assigned management. The study site was grazed for nine months (Jan-Feb, Jun-Dec) in 2015 and for six months in 2016 (Jan, May-Jun, Aug-Oct) at different grazing intensities.

Winter wheat is a cool season crop representing the dominant cultivated ecosystem (converted from tallgrass prairie) of central Oklahoma. Soil of the wheat field is characterized as deep, well drained, loamy soils with clayey or loamy subsoil. Soil series mapped within the pasture include: Renfrow-Kirkland complex silt loams (Fine-mixed, thermic Udertic Paleustolls) 3 to 5% slopes; and Norge silt loams (Fine, mixed, thermic Udertic Paleustalf), 1 to 3%, 3 to 5%, and 5 to 8% slopes. These soils have depths to 1.0 m, and moderate water holding capacity. Historical management of this 11-ha pasture extends from the early 1950's through 2013. Predominant management over time was production of continuous winter wheat as part of a larger 66 ha unit, with the majority of wheat crops being grazed from November through April (graze-out) each cropping cycle by herds of yearling stocker cattle. Wheat crops were managed by various forms of conventional tillage for weed control and seedbed preparation for September planting, except for 2005 through 2008 when a minimum tillage approach was utilized.

For our study period, the first season of winter wheat was planted on the 29th of September 2014, on a tilled and fertilized field dominated by silt loam soil. The wheat was harvested on the 10th of June 2015, and the land was kept fallow during the summer months with weed control by tillage and herbicide application. The second season of winter wheat was planted on the 9th of September 2015, and harvested on the 10th of June 2016. The other details on management practices in both sites are presented on Table S1. More information on the management activities at study sites and climatic features of the study years are presented in Table S1 and Fig. 1.

2.2. Eddy covariance and other supplementary measurements

Eddy covariance (EC) towers were deployed to measure CO_2, H_2O, and energy fluxes from the winter wheat (35.5685, -98.0558) and tallgrass prairie (35.54865°, -98.03759°) ecosystems. Continuous measurements of CO_2 and H_2O fluxes are presented in this study from October 2014 to September 2016 for winter wheat and from January 2015 to December 2016 for tallgrass prairie, respectively.

The measurement system at each site consisted of a three-dimensional sonic anemometer (CSAT3, Campbell Scientific Inc., Logan, UT, USA) and an open path infrared gas analyzer (LI-7500, LI-COR Inc., Lincoln, NE, USA). The sensors were mounted at a 2.5 m height from the ground and the system was set up at the center of each site facing south, towards the prevalent wind direction. Storage fluxes were considered negligible because the tower height was maintained at higher than 2 m throughout the growing season. The fetch area was about 300 m in all directions. The *EddyPro* processing software (LI-COR Inc., Lincoln, NE, USA), was used to process the raw data, collected at 10 Hz frequency (10 samples sec^{-1}), to get 30-min fluxes. The software employed correction factors for coordinate alignment, temperature due to humidity influence, and compensation of density fluctuations in infrared gas analyzer using the Webb-Pearman-Leuning (WPL) theory to make necessary corrections in the high frequency data. Negative sign convention is used to denote CO_2 uptake by the ecosystem whereas positive sign denotes the CO_2 release by the ecosystem to atmosphere.

Auxiliary sensors measured other metrological and soil variables. Quantum sensors (LI-190, LI-COR Inc. Lincoln, NE, USA) were used to measure photosynthetic photon flux density (PPFD). Net radiometers (CNR1, Kipp and Zonen, Delft, The Netherlands) were used to measure net radiation (Rn) over plant canopy. Temperature and relative humidity were measured by using temperature and relative humidity probes (HMP45C, Vaisala, Helsinki, Finland). Similarly, self-calibrating heat flux sensors (HFT3.1, Radiation& Energy Balance Systems, Inc, Seattle, WA, USA) at 5-cm depth were used to measure soil heat fluxes (G). Soil moisture content was measured at about 5-cm depth using a Hydra probe (Delta-T, Lexington, MA, USA).

2.3. Vegetation measurements and phenology

Leaf area index (LAI) was measured at biweekly intervals during the active growing season using the LAI-2200 (LI-COR Biosciences, Lincoln, NE, USA). Six measurements were made within the eddy covariance footprint area at each site. Aboveground biomass (AGB) was measured by destructive sampling from 0.5 m^2 quadrats with three replicates at each site. The fresh samples were oven dried at 70 °C for 72 hours and total aboveground dry weight was measured by weighing the oven dried samples. The 8-day Enhanced Vegetation Index (EVI) (Huete et al., 2002) was computed from the land surface reflectance data from MOD9A1 data product downloaded from the University of Oklahoma data portal (http://eomf.ou.edu/modis/visualization/gmap/).

Winter wheat in this region is often a dual-purpose crop in which cattle grazing is allowed over the winter (generally November-February) and the crop is allowed to grow for grain harvest after removal of the cattle. However, the winter wheat site was not grazed during this study. The tallgrass prairie is generally used for grazing. Our tallgrass prairie site was grazed at different time periods and different grazing intensity (cows/ head) during the study period (Table S1). Based on EVI time series data, we divided the one-year cycle of winter wheat into two categories: growing season and non-growing season. The growing season for winter wheat comprises the period from planting to harvesting which is referred to as I-A (November-January) and I-B (February-June) whereas the non-growing season (summer fallow) is the period between harvesting (June) and the next planting (September), denoted as II. For the tallgrass prairie, the calendar year is divided into growing season (March to mid-October) and non-growing season (January, February and mid-October to December) based on phenology represented by the EVI time series data (Fig. 1a, b).

2.4. Data Screening and Gap filling for eddy flux tower data

Quality flags were applied for screening erroneous data. Data outside a ± 3.5 standard deviation range (consecutive six values) from a 14-day running mean window were identified as outliers and were removed (Wagle et al., 2015a). This allowed us to filter out the data outside of the accepted range of $-50 < CO2\ flux < 60$ μmol m^{-2} s^{-1}, $-20 < LE < 600$ W m^{-2}, and -100 < H <

400W m^{-2} (Joo et al., 2016; Ní Choncubhair et al., 2016; Zeri et al., 2011). We used the online R package "REddyProc" tool developed at the Max Planck Institute for Biogeochemistry, Jena, Germany (Moffat et al., 2007; Reichstein et al., 2005) for gap filling of flux data and partitioning of NEE into ecosystem respiration (ER) and gross primary production (GPP). The gaps in the datasets (18 and 22% at the winter wheat site and 24 and 32% at the tallgrass prairie site in 2015 and 2016, respectively), which were due to bad quality observations and unreliable values recorded by malfunctioning sensors, were gap filled. The average value of measurements immediately before and after the gap were used to fill half hourly gaps. Gaps of two hours or less were filled by linear interpolation. Mean diurnal variation, look up tables, and regression techniques were used to fill the larger gaps either in isolation or in combination based on the requirements described in previous studies (Amiro et al., 2006; Falge et al., 2001; Hui et al., 2004; Moffat et al., 2007; Wilson and Baldocchi, 2001). The NEE was partitioned into ER and GPP using the regression model constructed by plotting night time NEE versus air temperature. This model defines the temperature sensitivity of ER by considering the temporal auto-correlations of fluxes and the co-variation of fluxes with meteorological variables to separate ER and GPP from NEE (Lloyd and Taylor, 1994; Reichstein et al., 2005).More details on the methods employed on the "REddyProc" tool can be found at https://www.bgc-jena.mpg.de/bgi/index.php/Services/REddyProcWebR Package. The gap filled NEE and partitioned GPP and ER data were used to compute the daily, seasonal, and annual carbon budgets and the uncertainties associated in annual budgets were computed by summing up the standard errors.

2.5. Energy Balance Closure

The plausibility of fluxes from the EC system was assessed based on energy balance closure (EBC). According to the first law of thermodynamics, the sum of turbulent fluxes (latent, LE and sensible heat, H) should be equivalent to the available energy, (i.e., R_n-G). The imbalance between available energy and turbulent fluxes indicates inaccurate estimates of scalar fluxes. Many research studies reported EBC as a standard test of eddy covariance data (Foken, 2008; Twine et al., 2000; Wilson et al., 2002). Turbulent fluxes (LE +H) are commonly underestimated by about 10-30% relative to the estimates of available energy (Rn-G) (Aubinet et al., 1999; Barr et al., 1994; Foken, 2008; Wilson et al., 2002). We used half-hourly data of three months of growing season (March- May for winter wheat and June-August for tallgrass prairie) in our EBC calculation. The EBC was calculated when all four terms of the EBC were available. Canopy storage energy and the energy used in photosynthesis were not accounted in the calculation. The scatter plot between turbulent fluxes available energy were strongly correlated and yielded the slope (corresponds to EBC) of 83% and 85% in winter wheat and tallgrass prairie, respectively in 2015 and 77% for both sites in 2016 (Fig. 2). The annual Bowen ratio was 0.72 and 0.30 for winter wheat and tallgrass prairie, respectively in 2015 and 0.51 and 0.48 at winter wheat and tallgrass prairie in 2016.

2.6. Estimates of evapotranspiration (ET) and ecosystem water use efficiency (EWUE)

ET (mm 30 min^{-1}) was calculated from the H_2O fluxes (mmol m^{-2} s^{-1}) measured by the eddy covariance system using the following equation:

$$ET = (H_2O \text{ flux} \times 18.01528 \times 1800)/10^6 \quad (1)$$

The computed half hourly ET values after gap-filling were used to generate daily, monthly, and seasonal ET values. We estimated ecosystem water use efficiency (EWUE) at monthly and seasonal scales: a) monthly EWUE as the ratio of monthly GPP to monthly ET, and b) seasonal EWUE as the ratio of seasonal GPP to seasonal ET over the growing season (Tubiello et al., 1999; Wagle and Kakani, 2014). Only the daytime ET was considered in the calculation because the carbon sequestration by the vegetation occurs during daytime only. The ratio of nighttime ET to total ET were 0.24 and 0.18 in 2015 and 0.20 and 0.23 in 2016, respectively at winter wheat and tallgrass prairie sites.

3. Results

3.1. Seasonal dynamics of weather, soil moisture, and plant growth

Patterns of air temperature, rainfall, soil water content (SWC), and photosynthetically active radiation (PAR) for the two sites during the study period are shown in Fig. 1c, d. The highest daily mean air temperature reached approximately 32 °C in August 2015 (15-year average maximum air temperature of 28 °C). The study sites received above normal rainfall (1273 mm) in 2015, which have a 30-year average (1981-2010) annual precipitation of 925 mm. Notably, the sites received record high rainfall of 393 mm in May 2015 (30-year average May rainfall= 124 mm). Both sites showed similar trends in SWC fluctuations corresponding with rainfall events. Distinct seasonality in LAI and AGB were observed for winter wheat and tallgrass prairie in both years (Fig. 3). The maximum recorded LAI was 5.0 and 4.7 m^2m^{-2} for winter wheat and 5.4 and 4.3 m^2m^{-2} for tallgrass prairie in 2015 and 2016, respectively. The maximum recorded AGB was 881 and 865 g m^{-2} for winter wheat and 1048 and 670 gm^{-2} for tallgrass prairie in 2015 and 2016, respectively.

3.2. Diurnal dynamics of carbon dioxide and water vapor fluxes

The diurnal trends of NEE for winter wheat and tallgrass prairie for different months during the active growing season are compared in Fig. 4. Considerable variations in NEE rates were observed between sites as well as among the growing-season months. The rates of NEE were higher in 2015 than in 2016 at both sites. As expected, NEE rates became more negative with the plant development, reaching the most negative values (maximum sink capacity) during peak growth, and became less negative during the late growing season due to vegetation senescence. Themost negative NEE rates for winter wheat occurred in April (-24.22 ± 0.97 and -24.79 ± 0.53 μmol m^{-2} s^{-1} in 2015 and 2016, respectively), and for tallgrass prairie in July (-20.55 ± 0.74 and -14.40 ± 0.83 μmol m^{-2} s^{-1} in 2015 and 2016, respectively). Solar radiation was one of the primary drivers for determining diurnal rates of the fluxes within the growing season. With increasing maximum PAR from 40 $molm^{-2}day^{-1}$ in February to 48 $molm^{-2}day^{-1}$ in April (Fig. 1c), winter wheat achieved its maximum carbon uptake. Similarly, the maximum PAR in July (70 $molm^{-2}day^{-1}$) corresponded with the highest negative NEE rate in tallgrass prairie.

Diurnal ET trends at the tallgrass prairie and winter wheat sites across the growing season are compared in Fig. 5. Like NEE rates, ET rates were higher in 2015 than in 2016 for both ecosystems. The ET rates reached a maximum in April for winter wheat and in June for tallgrass prairie. Peak hourly ET was 0.86 ± 0.06 (2015) and 0.44 ± 0.06 (2016) for winter wheat, and was 0.62± 0.02 (2015) and 0.65 ± 0. 03 (2016) mm hr^{-1} for tallgrass prairie. The average SWC at both ecosystems was above 15 % by volume (Fig. 1d) during most of the growing season, suggesting that the ecosystems did not experience severe drought during the study period. However, the rates of fluxes were impacted by grazing in the tallgrass prairie (. In 2015, from January to July the site was grazed with stocking density of about 0.4 heads/hac with the average cattle weight of 500kg and the intensity was increased to 0.96 heads/hac from August to

December. In 2016, the site was grazed intermittently grazed (January, May—June, August-October) with the same grazing intensity of 0.72 heads/hac with the cows weighing about 576 kg.

3.3. Seasonal dynamics of carbon dioxide and water vapor fluxes

Figure 6 shows the seasonal dynamics of daily NEE, ER, GPP, and ET from the winter wheat and tallgrass prairie sites. Winter wheat was a sink of carbon (negative NEE) for ~100 days between DOY 32 (February 1) and 132 (May 12), while tallgrass prairie was a sink of carbon for 144 days between DOY 105 (April 15) and 248 (September 5). The carbon uptake rate (negative NEE) by winter wheat began to increase from early February and reached a maximum in April, followed by a rapid decrease as the crop senesced (mid- May). For tallgrass prairie, CO_2 uptake began to increase in mid-March and reached a maximum in July before it rapidly decreased as the vegetation senesced (early October), when the release of CO_2 (ER) was more than CO_2 assimilation (GPP). Substantial rates of carbon uptake were observed during the active growing season by both ecosystems (Fig. 6) because of rates of carbon assimilation (GPP) exceeded carbon release (ER). However, the magnitude of carbon uptake (NEE) was greater in winter wheat (-9.24 g C m^{-2} d^{-1} in 2015 and – 8.69 g C m^{-2} d^{-1} in 2016) than in tallgrass prairie (-6.23 g C m^{-2} d^{-1} in 2015 and -7.52 g C m^{-2} d^{-1} in 2016). The higher negative NEE in winter wheat resulted from lower ER and higher GPP in winter wheat as compared to tallgrass. However, the relatively higher carbon uptake occurred for a short period (only during April) in winter wheat, while the carbon uptake was consistently higher for three months (June-August) in tallgrass prairie.

High variability in ET was observed between two ecosystems and the ET rates showed a clear seasonal pattern corresponding to the seasonality of the respective crops (Fig. 6d). At the winter wheat site, the magnitude of daily ET was the highest (6.0 and 5.4 mm day^{-1}) on the 10th and 23rd May in 2015 and 2016, respectively, while in tallgrass prairie, the highest daily ET (7.2 and 8.2 mm day^{-1}) was observed on the 10th and 31st June in 2015 and 2016, respectively. Higher ET was observed during the period of higher LAI values, which is earlier in the year for winter wheat than tallgrass prairie. However, two significant peaks (~29th June and ~29th July) of ET can be seen for winter wheat in 2015 during the summer, after winter wheat was harvested (June 10), due to the growth of weeds.

3.4. Seasonal and annual (calendar year) sums of carbon dioxide and water vapor fluxes

The growing season, non-growing season, and annual (based on Fig. 1a) values of GPP, ER, and NEE for both ecosystems are shown in Table 1. Cattle grazing is generally allowed in first half of the growing season (GS I-A) of winter wheat. This period had low plant activity indicated by lower cumulative GPP. Cumulative growing seasonal values (GS) of GPP and ER fluxes from the tallgrass prairie were larger than those from the winter wheat in both years. The NEE during the 2015 growing season for tallgrass prairie (-276 ± 43 g C m^{-2}) was similar to winter wheat (-251 ± 43 g C m^{-2}). However, it was more than double in winter wheat (-403 ± 73 g C m^{-2}) than in tallgrass prairie (-159 ± 61 g C m^{-2}) during the 2016 growing season. The growing season GPP total was 921 ± 169 and 996 ± 137 g C m^{-2} in winter wheat as compared to 1663 ± 233 and 1346 ± 103 g C m^{-2} in tallgrass prairie in 2015 and 2016, respectively. Similarly, the total growing-season ER for winter wheat was 672 ± 154 and 603 ± 102 g C m^{-2} compared to 1386 ± 221 and 1186 ± 145 g C m^{-2} for the tallgrass prairie in 2015 and 2016, respectively. The differences in the fluxes between two years was due to the rainfall variations during the growing season (Fig.) These results show that both ecosystems were carbon sinks (negative NEE) on a seasonal scale. However, the winter wheat site was a carbon source (positive NEE) on an annual scale when the carbon fluxes of the fallow period were considered. The non-growing season (NGS II) of winter wheat, which is comprised of mostly summer fallow, had larger positive NEE values attributed to higher ER and lower GPP. On the other hand, the growing season was longer and ER rates were lower in tallgrass prairie during the non-growing season (NGS II). Thus, the winter wheat ecosystem released about 56 ± 13 and 33 ± 9 g C m^{-2} (positive NEE), while tallgrass prairie ecosystem gained about -128 ± 69 and -119 ± 53 g C m^{-2} (negative NEE) on an annual scale in 2015 and 2016, respectively.

Annual ET was greater in tallgrass prairie (919 ± 89 mm) than in winter wheat site (651± 69 mm) in 2015, but was similar in 2016 at both sites (winter wheat = 644 ± 111 mm and, tallgrass prairie = 669 ± 117 mm). Monthly ET was also generally higher for tallgrass prairie except in March and April (Fig. 7 b,e). Winter wheat had the highest GPP (334 g C m^{-2}) during the peak growth (AGB= 400 g m^{-2}) in April when the total ET reached the maximum (101 mm). The highest ET in the winter wheat was observed in April (71 and 53 mm $month^{-1}$ in 2015 and 2016, respectively) when the winter wheat was in the initial phase of the growing season (AGB=300 g m^{-2}) with a cumulative GPP of 127 and 71 g C m^{-2}. For tallgrass prairie, ET reached its maximum (2015= 180 mm and 2016= 127 mm) in the month of July, with the corresponding AGB of 900 and 482 g m^{-2} and a cumulative GPP of 379 and 314 g C m^{-2} in 2015 and 2016, respectively.

The GPP to ER ratio for the study period at the two sites are presented in Fig. S2. Generally, the ratio was greater than one (net carbon uptake) from February to June in winter wheat and from April to October in tallgrass prairie. In winter wheat, the ratio decreased after June when the crop was harvested, while in tallgrass prairie the ratio decreased after October with the onset of senescence.

3.5. Seasonal dynamics of ecosystem-level water use efficiency (EWUE)

EWUE was lower in the early and late growing season and higher during the peak growth of the vegetation at both sites (Fig. 7 c, f). In winter wheat, the EWUE reached a maximum of 3.9 and 4.1 g C mm^{-1} ET in March of 2015 and 2016, respectively, while the highest EWUE in tallgrass prairie was 2.4 and 3.2 g C mm^{-1} ET in August 2015 and June 2016, respectively. The peak growing season EWUE was substantially higher for winter wheat (2.63 and, 2.32 g C mm^{-1} ET in 2015 and 2016, respectively) than for tallgrass prairie (2.01 and, 2.29 g CO_2 mm^{-1} ET in 2015 and 2016, respectively) (Fig.7, Table 1).

3.6. Rainfall, management activities, and carbon flux rates

We demonstrated four specific cases from the two study sites to illustrate the impact of climate and management activities on carbon fluxes (Fig. 8). Case I: A week before a July 26th rain event, carbon fluxes (NEE, GPP and ER) in tallgrass prairie site started declining. It took few days after rainfall for carbon fluxes to increase. The rainfall event contributed about 32% increase in sink capacity (more negative NEE) of grassland compared to the average NEE before rain event. Case II: In the 1st week of June 2016, at the tallgrass prairie site, NEE decreased by about 0.8 $\mu molm^{-2}$ s^{-1} during the first week of grazing. Grazing of cattle 0.40 head per hectare for a week caused lower carbon uptake (less negative NEE) by about 28 %. After about one week of grazing, the carbon uptake rate increased again to about 3 $\mu molm^{-2}$ s^{-1} similar to the rate before grazing. Case III: In winter wheat site, herbicide RT 3 glyphosate, Weedmaster (dicamba and 2,4 D) was applied on July 29, 2015, to kill the weeds. This caused a reduction in GPP at a higher rate than ER resulting in positive NEE (3.78 µmolm^{-2} s^{-1}) which was negative a week (-1.2 µmolm^{-2} s^{-1}) before the herbicide was applied. Case IV: The winter wheat site was tilled using a tandem disc harrow on June 30, 2015, to inhibit the growth of weeds for maintaining fallow. This management activity also caused changes in the carbon fluxes, particularly GPP and NEE. GPP was reduced at a higher rate, while ER remained unchanged, making NEE of the site positive. The weekly average NEE rate changed from -1.07 µmolm^{-2} s^{-1}(before harrowing) to 4.09 µmolm^{-2} s^{-1} (after harrowing).

4. Discussion

4.1. Comparison of CO_2 and H_2O fluxes of winter wheat and tallgrass prairie

Management activities, weather conditions, and soil types at the study sites influenced the magnitudes of the CO_2 and H_2O fluxes. At the study sites, the year 2015 was wetter and hotter than normal, whereas 2016 was close to the 30-year average temperature and precipitation. Similarly, the grazing events at the tallgrass prairie sites impacted the rates of fluxes recorded in this study. The maximum diurnal peak rate of -24 (2015) and – 25 (2016) µmol m^{-2} s^{-1} measured in the winter wheat ecosystem was close to the maximum NEE of -25 to -30 µmol m^{-2} s^{-1} measured for winter wheat ecosystem of Ponca City, Oklahoma (Fischer et al., 2007; Gilmanov et al., 2003)**.** Additionally, the daily peak NEE value (2015= -9.24 and 2016= -8.8 g C m^{-2} d^{-1}) of winter wheat measured in our study was similar to the daily peak NEE value of -9.3 g C m^{-2} d^{-1} at Billings, Oklahoma (Fischer et al., 2007) and -8.18 g C m^{-2} d^{-1} at Ponca City, Oklahoma (Gilmanov et al., 2003). The daily peak NEE values of about -11 to -13 g C m^{-2} d^{-1} from the

Europe (Belgium and Germany) and China (Yucheng) were more negative than that those -8 to -9.3 g C m^{-2} d^{-1} from winter wheat ecosystems in Oklahoma (Table 2).

The peak diurnal NEE rates of -20 μmol m^{-2} s^{-1} (2015) and -15μmol m^{-2} s^{-1} (2016) in tallgrass prairie in our study were slightly lower than the values of -28 μmol m^{-2} s^{-1} and -22 μmol m^{-2} s^{-1} in 2005 and 2006 reported for a tallgrass prairie ecosystem at El Reno, Oklahoma (Fischer et al., 2012). The maximum NEE daily values of tallgrass prairie varied from -5.2 to -8.1 g C m^{-2} d^{-1} at various sites in Southern Plains (Suyker and Verma, 2001; Wagle et al., 2015b) (Table. 2), which agreed with the maximal NEE daily value of –6.3 g C m^{-2} d^{-1} (2015) and -7.5 g C m^{-2} d^{-1} (2016) measured in our study.

In the winter wheat ecosystem, the maximum daily ET of 6 mm d^{-1} (2015) and 5.3 mm d^{-1} (2016) measured in our study was similar with the maximal daily ET values of 7 mm day^{-1} measured in winter wheat at Ponca City, Oklahoma, but the maximum daily ET of 7.2 mm d^{-1} (2015) and 8.2 mm d^{-1} (2016) in tallgrass prairie ecosystem was slightly higher than 5 mm d^{-1} reported for tallgrass prairie at Ponca city, Oklahoma (Burba and Verma, 2005). Similarly, the annual ET totals of 651 and 644 mm measured at our winter wheat site in 2015 and 2016, respectively, was slightly lower than that of 750, 714, and 742 mm of ET at the winter wheat site of Oklahoma in 1997, 1998, and 1999, respectively (Burba and Verma, 2005). On the other hand, the annual ET from tallgrass prairie site in our study was relatively higher (919 mm) in 2015 and was similar (679 mm) in 2016 compared to the range (485 to 716 mm) of ET values reported for six different tallgrass prairie sites by Wagle et al. (2017). The higher values of ET were most likely due to the high amount of rainfall received in 2015, which agreed with the higher ET values (807 mm yr^{-1}) reported for tallgrass prairie when Oklahoma received higher rainfall in 1997 (Burba and Verma, 2005). Although weather conditions and management activities (e.g., grazing) are site-specific, the impact of these conditions and activities on the rate of atmospheric exchanges, the CO_2 and H_2O fluxes, reported in our study are comparable to the values reported in the literature.

4.1.1 Impacts of management activities on carbon fluxes

Application of herbicide and tillage for keeping the land fallow at the winter wheat site during summer months impacted the carbon fluxes. These activities contributed to the change in annual carbon budgets. For example, the weekly average of NEE was changed from -0.39 g C m^{-2} to 3.79 g C m^{-2} after herbicide was applied to kill the weeds. A similar switch in NEE was observed when the site was tilled for maintaining it as fallow (Fig. 8). Summer fallow contributed only about 25% and 11% GPP to the annual budget whereas the carbon loss due to ER was about 48% and 47% in 2015 and 2016, respectively, resulting in positive annual NEE (carbon source) at the winter wheat site. This loss of carbon from the fallow in winter wheat-fallow system was consistent with the study conducted in Montana, USA. About 135 g C m^{-2} was lost between April to September from the fallow field of Montana in 2013/2014 (Vick et al., 2016). Livestock grazing in prairie pasture is a common practice in the Southern Plains (Gillen et al., 1998; Hickman et al., 2004; Luo et al., 2012; Zhou et al., 2017a). Grazing plays an important role in modifying the vegetation phenology, canopy structure, and productivity of grasslands which, in turn, alters the magnitude and temporal patterns of CO_2 and H_2O fluxes of the ecosystem (Luo et al., 2012; Owensby et al., 2006; Wayne Polley et al., 2008). For example, in the 1st week of June 2016, the tallgrass prairie NEE decreased by about 3 μmol$m^{-2}s^{-1}$ during the first week of grazing (Fig. 8). After about one week of grazing, the ecosystem again increased the carbon uptake rate. Although the effects of grazing are not quantified completely in this study, it can be argued that the tallgrass prairie ecosystem in our study would be a larger sink (more negative NEE) with less or no grazing. However, low productivity has been reported for prairie that was ungrazed for long periods due to senesced vegetation that shades out new green leaves (Belsky, 1986; Dalgleish and Hartnett, 2009). Bailing of the grasses stimulated new growth of the vegetation when the site (El Reno, OK) received good rainfall and sunshine(Zhou et al., 2017a).

4.2. Change in seasonal patterns and magnitudes of water vapor fluxes

Some researchers have reported that the utilization of summer cover crops rather than making the field fallow increased the ecosystem productivity and resulted to less evaporative water loss (Farahani et al., 1998a; Farahani et al., 1998b; McGee et al., 1997).While the objective of summer fallow is to accumulate water for the subsequent crop, the wheat-fallow system has been found to be inefficient in storing soil water due to greater loss by soil evaporation, transpiration from weeds, deep percolation, and increased runoff (Black et al., 1981; Farahani et al., 1998b). When land use is converted from grassland to winter wheat with a summer fallow, the resulting ecosystem became less water efficient (low EWUE) at annual scale due to the resulting amount of moisture loss to evaporation when no or minimal amounts of carbon are fixed. In our study, the EWUE in winter wheat declined to about 1.5 – 1.6 g C mm^{-1} ET at the annual scale from about 3 – 3.5g C mm^{-1} ET at the seasonal scale due to loss of water during the fallow period with more release of carbon than the uptake. Although more water was lost as ET from tallgrass prairie than from the winter wheat, results showed that tallgrass prairie was more water efficient (EWUE= 1.8 g C mm^{-1} ET in 2015 and 2.2 g C mm^{-1} ET in 2016) than winter wheat (EWUE= 1.7 g C mm^{-1} ET in 2015 and 1.8 g C mm^{-1} ET in 2016 g CO_2) at the annual scale. Throughout the Southern Plains, the dominant agricultural crop is winter wheat, which is planted in early fall and harvested in June. This pattern contrasts sharply with the seasonal cycle of tallgrass prairie, which is most active from May to August. The change from prairie to winter wheat shifts the magnitude and seasonal timing of energy, momentum, H_2O, and CO_2 fluxes between the atmosphere and ecosystem. Many studies in the past have examined the role of ET variability in relation to the atmospheric processes determining the change in the regional climate (Clark et al., 2001; Katul et al., 2012; Shukla and Mintz, 1982; Wang and Eltahir, 2000). The soil-plant system is embedded within the atmospheric boundary layer where change in ET due to change in land surface influences the precipitation patterns and frequency at the regional scale (Katul et al., 2012). Currently, there are known impacts of the winter wheat in the Southern Plains on surface-layer and boundary layer processes (Haugland and Crawford, 2005; McPherson and Stensrud, 2005; McPherson et al., 2004). Further, the Southern Plains region is located where strong feedbacks between the land surface and the atmosphere across various spatial and temporal scales occur during the growing season (Basara and Christian, 2017; Basara and Crawford, 2002; Ferguson and Wood, 2011; Ford et al., 2015a; Ford et al., 2015b; Guo and Dirmeyer, 2013; Guo et al., 2006; Koster et al., 2004; Ruiz-Barradas and Nigam, 2013; Santanello Jr et al., 2013; Santanello Jr et al., 2009; Santanello Jr et al., 2015). Thus, the shift in the ET (latent heat flux) resulting from land use change (tallgrass to winter wheat) could impact the overall water balance of terrestrial ecosystems, atmospheric circulations, and the regional climate of the Southern Plains, especially given expansion of the winter wheat within the region. Such impacts could also influence the timing and severity of convective storms in the region. Recent research has found that the overall variability of precipitation in the region is increasing (Christian et al., 2015; Flanagan et al., 2017; Weaver et al., 2016), which could have additional downstream impacts related to excessive precipitation (McCorkle et al., 2016) and the rapid development of drought (Bajgain et al., 2017; Otkin et al., 2013; Zhou et al., 2017b).

4.3. Land use change, winter wheat-summer fallow, and carbon sink potential

Despite large differences in carbon uptake (NEE) in 2015 and 2016 at both ecosystems, winter wheat and tallgrass prairie ecosystems were carbon sinks in both years during their respective growing seasons (Table. 1). The higher carbon uptake during the growing season in 2016 in winter wheat than 2015 was due to the good crop growth resulted from higher amount of fall rainfall (Fig. 1d). Similarly, the higher amount of rainfall during May in 2015 contributed to the higher carbon uptake during the 2015 growing season by tallgrass prairie than in 2016. The carbon fluxes showed large differences (winter wheat released 56 and 33 g C m^{-2} in 2015 and 2016, respectively, and tallgrass prairie accumulated -128 and -119 g C m^{-2} in 2015 and 2016, respectively) when accounted for at the annual scale. This difference in carbon fluxes between the two sites suggests that although the tallgrass prairie had a longer growing season (March to mid-October) than winter wheat (October-May), the carbon sink potential was similar during the growing season in 2015 and the carbon sink of tallgrass prairie was smaller in 2016 than that of winter wheat. This is due to less loss of carbon via ER displayed by the higher ratio of GPP over ER in winter wheat ecosystem (Fig. S2 and Table 1). The average GPP to ER ratios for winter wheat during the growing season were 1.6 (2015) and 1.7 (2016), while the same ratios during the growing season were 1.2 (2015) and 1.1 (2016) for tallgrass prairie. Until the harvest of winter wheat, the ecosystem was a carbon sink due to higher GPP than ER. The transition from a carbon sink to a carbon source resulted from lower GPP and higher ER (low GPP:ER ratio) after harvesting during the summer month with increased temperature and decomposition of winter wheat residue. Consequently, the winter wheat ecosystem was a potential carbon source offsetting the growing season carbon sink magnitude when accounted for the annual time scale. On the other hand, the annual GPP in the tallgrass prairie ecosystem was sufficient to cover the carbon expense caused by ER with a GPP:ER ratio of about 1.07 (both years) and resulting in a net cumulative carbon balance (NEE) of -128 and –119 g C m^{-2} in 2015 and 2016, respectively. This differential capacity in carbon uptake potential between these two ecosystems suggested that the Southern Plains could contribute a substantial amount of carbon to atmosphere which otherwise would have been a potential carbon sink indicating that the land use change from grassland to winter wheat has a significant effect on the carbon cycle of the Southern Plains. The prevailing practice of keeping land fallow after harvesting the winter wheat for capturing moisture from summer rainfall for the following winter wheat crop caused the ecosystem to release more carbon to the atmosphere. Although the main goal of fallow is to ensure soil moisture for the subsequent winter wheat, it has been found that summer fallow rotation system is not effective with respect to productivity, economic risk, organic matter storage, and even soil water storage (Kolberg et al., 1996; McGee et al., 1997; Peterson et al., 1996). Only 25% precipitation efficiency was achieved from the summer fallow in terms of soil water storage (McGee et al., 1997; Peterson et al., 1996). The use of cover crops after winter wheat during summer could be a better practice to compensate for carbon loss via ER by fixing more carbon into the ecosystem via photosynthesis from cover crops. However, any changes in the summer fallow system must consider the effect on the soil moisture availability required to stabilize production for the next crop cycle.

It is important to mention the uncertainties associated with the rates of land use change and the spatial heterogeneity of land management, soil properties, and weather variables across the region. However, the ecosystems chosen are the representative of the practices of the Southern Plains. While the size of the potential carbon sink/source at the regional level can`t be estimated with greater confidence from this study, it can be inferred that the change of grassland to winter wheat with a summer fallow reduced the carbon sink potential and made the ecosystem less water efficient (more water loss for less carbon fixed). Also, the winter wheat fields in our study had been in wheat for many years and had depleted soil carbon relative to tallgrass prairie. For the first few years after conversion there would be an even greater loss of carbon, and then at some new semi-equilibrium, the estimated carbon loss become more relevant. Therefore, it appears that fallow land after harvesting of winter wheat is a factor that needs to be considered for managing the ecosystem sustainably.

Conclusions

Carbon dioxide and water vapor fluxes were measured using the eddy covariance system from two major ecosystems of the Southern Plains (winter wheat and tall grass prairie) in 2015 and 2016. Both ecosystems were carbon sinks during their active growing seasons. Despite winter wheat having a greater carbon sink potential at the hourly and daily timescales during the growing season, winter wheat ecosystems were a carbon source when the carbon budgets for the summer fallow period were included. Similarly, the significant water loss due to evaporation from the fallow land (winter wheat-fallow rotation), when little carbon was fixed, caused the winter wheat ecosystem to be less water efficient than the tallgrass prairie ecosystem despite higher growing season EWUE. Results suggest that the differences in magnitudes and patterns of carbon dioxide and water vapor fluxes between winter wheat and tallgrass prairie can exert influence on the carbon and water budgets of the whole region under land use change scenarios.

Acknowledgements

This study was supported in part by research grants from the USDA National Institute of Food and Agriculture (NIFA) (award # 2013-69002-23146 and 2016-68002-24967), the US National Science Foundation (NSF) EPSCoR (IIA-1301789), and NOAA Climate Office's Sectoral Applications Research Program (SRP) grant NA130AR431012, and the USDA ARS Long Term Ecosystem Research (LTAR) network.

References

Amiro, B. et al., 2006. Carbon, energy and water fluxes at mature and disturbed forest sites, Saskatchewan, Canada. Agricultural and forest meteorology, 136(3): 237-251.

Anthoni, P.M., Freibauer, A., Kolle, O. and Schulze, E.-D., 2004. Winter wheat carbon exchange in Thuringia, Germany. Agricultural and Forest Meteorology, 121(1): 55-67.

Aubinet, M. et al., 1999. Estimates of the annual net carbon and water exchange of forests: the EUROFLUX methodology. Advances in ecological research, 30: 113-175.

Bajgain, R. et al., 2017. Assessing agricultural drought in summer over Oklahoma Mesonet sites using the water-related vegetation index from MODIS. International journal of biometeorology, 61(2): 377-390.

Barr, A.G., King, K., Gillespie, T., Den Hartog, G. and Neumann, H., 1994. A comparison of Bowen ratio and eddy correlation sensible and latent heat flux measurements above deciduous forest. Boundary-Layer Meteorology, 71(1-2): 21-41.

Basara, J.B. and Christian, J.I., 2017. Seasonal and interannual variability of land–atmosphere coupling across the Southern Great Plains of North America using the North American regional reanalysis. International Journal of Climatology.

Basara, J.B. and Crawford, K.C., 2002. Linear relationships between root-zone soil moisture and atmospheric processes in the planetary boundary layer. Journal of Geophysical Research: Atmospheres, 107(D15).

Belsky, A., 1986. Does herbivory benefit plants? A review of the evidence. The American Naturalist, 127(6): 870-892.

Black, A., Brown, P., Halvorson, A. and Siddoway, F., 1981. Dryland cropping strategies for efficient water-use to control saline seeps in the northern Great Plains, USA. Agricultural Water Management, 4(1): 295-311.

Burba, G.G. and Verma, S.B., 2005. Seasonal and interannual variability in evapotranspiration of native tallgrass prairie and cultivated wheat ecosystems. Agricultural and Forest Meteorology, 135(1): 190-201.

Christian, J., Christian, K. and Basara, J.B., 2015. Drought and pluvial dipole events within the great plains of the United States. Journal of Applied Meteorology and Climatology, 54(9): 1886-1898.

Claassen, R., Carriazo, F., Cooper, J.C., Hellerstein, D. and Ueda, K., 2011. Grassland to cropland conversion in the Northern Plains. US Department of Agriculture, Economic Research Report(120).

Clark, D.B., Xue, Y., Harding, R.J. and Valdes, P.J., 2001. Modeling the impact of land surface degradation on the climate of tropical North Africa. Journal of Climate, 14(8): 1809-1822.

Cooley, H., Riley, W., Torn, M. and He, Y., 2005. Impact of agricultural practice on regional climate in a coupled land surface mesoscale model. Journal of Geophysical Research: Atmospheres, 110(D3).

Dalgleish, H.J. and Hartnett, D.C., 2009. The effects of fire frequency and grazing on tallgrass prairie productivity and plant composition are mediated through bud bank demography. Plant Ecology, 201(2): 411-420.

Dhuyvetter, K., Thompson, C., Norwood, C. and Halvorson, A., 1996. Economics of dryland cropping systems in the Great Plains: A review. Journal of Production Agriculture, 9(2): 216-222.

Falge, E. et al., 2001. Gap filling strategies for defensible annual sums of net ecosystem exchange. Agricultural and forest meteorology, 107(1): 43-69.

Farahani, H., Peterson, G. and Westfall, D., 1998a. Dryland cropping intensification: A fundamental solution to efficient use of precipitation. Advances in agronomy (USA).

Farahani, H., Peterson, G., Westfall, D., Sherrod, L. and Ahuja, L., 1998b. Soil water storage in dryland cropping systems: The significance of cropping intensification. Soil Science Society of America Journal, 62(4): 984-991.

Ferguson, C.R. and Wood, E.F., 2011. Observed land–atmosphere coupling from satellite remote sensing and reanalysis. Journal of Hydrometeorology, 12(6): 1221-1254.

Fischer, M.L., Billesbach, D.P., Berry, J.A., Riley, W.J. and Torn, M.S., 2007. Spatiotemporal variations in growing season exchanges of CO_2, H_2O, and sensible heat in agricultural fields of the Southern Great Plains. Earth Interactions, 11(17): 1-21.

Fischer, M.L. et al., 2012. Carbon, water, and heat flux responses to experimental burning and drought in a tallgrass prairie. Agricultural and forest meteorology, 166: 169-174.

Flanagan, P.X., Basara, J.B. and Xiao, X., 2017. Long-term analysis of the asynchronicity between temperature and precipitation maxima in the United States Great Plains. International Journal of Climatology.

Foken, T., 2008. The energy balance closure problem: an overview. Ecological Applications, 18(6): 1351-1367.

Ford, T.W., Quiring, S.M., Frauenfeld, O.W. and Rapp, A.D., 2015a. Synoptic conditions related to soil moisture-atmosphere interactions and unorganized convection in Oklahoma. Journal of Geophysical Research: Atmospheres, 120(22).

Ford, T.W., Rapp, A.D. and Quiring, S.M., 2015b. Does afternoon precipitation occur preferentially over dry or wet soils in Oklahoma? Journal of Hydrometeorology, 16(2): 874-888.

Gillen, R.L., McCollum III, F.T., Tate, K.W. and Hodges, M.E., 1998. Tallgrass prairie response to grazing system and stocking rate. Journal of Range Management: 139-146.

Gilmanov, T.G. et al., 2003. Gross primary production and light response parameters of four Southern Plains ecosystems estimated using long-term CO_2-flux tower measurements. Global Biogeochemical Cycles, 17(2).

Guo, Z. and Dirmeyer, P.A., 2013. Interannual variability of land–atmosphere coupling strength. Journal of Hydrometeorology, 14(5): 1636-1646.

Guo, Z. et al., 2006. GLACE: the global land–atmosphere coupling experiment. Part II: analysis. Journal of Hydrometeorology, 7(4): 611-625.

Haugland, M.J. and Crawford, K.C., 2005. The diurnal cycle of land–atmosphere interactions across Oklahoma's winter wheat belt. Monthly weather review, 133(1): 120-130.

Hickman, K.R., Hartnett, D.C., Cochran, R.C. and Owensby, C.E., 2004. Grazing management effects on plant species diversity in tallgrass prairie. Journal of Range Management, 57(1): 58-65.

Huete, A. et al., 2002. Overview of the radiometric and biophysical performance of the MODIS vegetation indices. Remote sensing of environment, 83(1): 195-213.

Hui, D. et al., 2004. Gap-filling missing data in eddy covariance measurements using multiple imputation (MI) for annual estimations. Agricultural and Forest Meteorology, 121(1): 93-111.

Joo, E. et al., 2016. The influence of drought and heat stress on long term carbon fluxes of bioenergy crops grown in the Midwestern US. Plant, cell & environment.

Katul, G.G., Oren, R., Manzoni, S., Higgins, C. and Parlange, M.B., 2012. Evapotranspiration: A process driving mass transport and energy exchange in the soil-plant-atmosphere-climate system. Reviews of Geophysics, 50(3).

Kolberg, R., Kitchen, N., Westfall, D. and Peterson, G., 1996. Cropping intensity and nitrogen management impact of dryland no-till rotations in the semi-arid western Great Plains. Journal of Production Agriculture, 9(4): 517-521.

Koster, R.D. et al., 2004. Regions of strong coupling between soil moisture and precipitation. Science, 305(5687): 1138-1140.

Lark, T.J., Salmon, J.M. and Gibbs, H.K., 2015. Cropland expansion outpaces agricultural and biofuel policies in the United States. Environmental Research Letters, 10(4): 044003.
Lloyd, J. and Taylor, J., 1994. On the temperature dependence of soil respiration. Functional ecology: 315-323.
Luo, G. et al., 2012. Moderate grazing can promote aboveground primary production of grassland under water stress. Ecological Complexity, 11: 126-136.
Lyon, D.J., Nielsen, D.C., Felter, D.G. and Burgener, P.A., 2007. Choice of summer fallow replacement crops impacts subsequent winter wheat. Agronomy journal, 99(2): 578-584.
McCorkle, T.A., Williams, S.S., Pfeiffer, T.A. and Basara, J.B., 2016. Atmospheric contributors to heavy rainfall events in the Arkansas-Red River Basin. Advances in Meteorology, 2016.
McGee, E., Peterson, G. and Westfall, D., 1997. Water storage efficiency in no-till dryland cropping systems. Journal of Soil and Water Conservation, 52(2): 131-136.
McPherson, R.A. et al., 2007. Statewide monitoring of the mesoscale environment: A technical update on the Oklahoma Mesonet. Journal of Atmospheric and Oceanic Technology, 24(3): 301-321.
McPherson, R.A. and Stensrud, D.J., 2005. Influences of a winter wheat belt on the evolution of the boundary layer. Monthly weather review, 133(8): 2178-2199.
McPherson, R.A., Stensrud, D.J. and Crawford, K.C., 2004. The impact of Oklahoma's winter wheat belt on the mesoscale environment. Monthly Weather Review, 132(2): 405-421.
Meyers, T.P., 2001. A comparison of summertime water and CO 2 fluxes over rangeland for well watered and drought conditions. Agricultural and Forest Meteorology, 106(3): 205-214.
Moffat, A.M. et al., 2007. Comprehensive comparison of gap-filling techniques for eddy covariance net carbon fluxes. Agricultural and Forest Meteorology, 147(3): 209-232.
Ní Choncubhair, Ó., Osborne, B., Finnan, J. and Lanigan, G., 2016. Comparative assessment of ecosystem C exchange in Miscanthus and reed canary grass during early establishment. GCB Bioenergy.
Otkin, J.A. et al., 2013. Examining rapid onset drought development using the thermal infrared–based evaporative stress index. Journal of Hydrometeorology, 14(4): 1057-1074.
Owensby, C.E., Ham, J.M. and Auen, L.M., 2006. Fluxes of CO 2 from grazed and ungrazed tallgrass prairie. Rangeland Ecology & Management, 59(2): 111-127.
Peterson, G., Schlegel, A., Tanaka, D. and Jones, O., 1996. Precipitation use efficiency as affected by cropping and tillage systems. Journal of Production Agriculture, 9(2): 180-186.
Reichstein, M. et al., 2005. On the separation of net ecosystem exchange into assimilation and ecosystem respiration: review and improved algorithm. Global Change Biology, 11(9): 1424-1439.
Riebsame, W.E., 1990. The United States Great Plains. The earth as transformed by human action: Global and regional changes in the biosphere over the past 300 years: 561-575.
Ruiz-Barradas, A. and Nigam, S., 2013. Atmosphere–land surface interactions over the Southern Great Plains: Characterization from pentad analysis of DOE ARM field observations and NARR. Journal of Climate, 26(3): 875-886.
Santanello Jr, J.A., Peters-Lidard, C.D., Kennedy, A. and Kumar, S.V., 2013. Diagnosing the nature of land–atmosphere coupling: a case study of dry/wet extremes in the US Southern Great Plains. Journal of Hydrometeorology, 14(1): 3-24.
Santanello Jr, J.A., Peters-Lidard, C.D., Kumar, S.V., Alonge, C. and Tao, W.-K., 2009. A modeling and observational framework for diagnosing local land–atmosphere coupling on diurnal time scales. Journal of Hydrometeorology, 10(3): 577-599.
Santanello Jr, J.A., Roundy, J. and Dirmeyer, P.A., 2015. Quantifying the land–atmosphere coupling behavior in modern reanalysis products over the US Southern Great Plains. Journal of Climate, 28(14): 5813-5829.

Schmidt, M., Reichenau, T.G., Fiener, P. and Schneider, K., 2012. The carbon budget of a winter wheat field: An eddy covariance analysis of seasonal and inter-annual variability. Agricultural and Forest Meteorology, 165: 114-126.

Shukla, J. and Mintz, Y., 1982. Influence of land-surface evapotranspiration on the earth's climate. Science, 215(4539): 1498-1501.

Song, J., Willmott, C.J. and Hanson, B., 1997. Simulating the surface energy budget over the Konza Prairie with a mesoscale model. Agricultural and forest meteorology, 87(2): 105-118.

Suyker, A.E. and Verma, S.B., 2001. Year-round observations of the net ecosystem exchange of carbon dioxide in a native tallgrass prairie. Global Change Biology, 7(3): 279-289.

Suyker, A.E., Verma, S.B. and Burba, G.G., 2003. Interannual variability in net CO2 exchange of a native tallgrass prairie. Global Change Biology, 9(2): 255-265.

Tubiello, F.N., Lin, G., Druitt, J. and Marino, B.D., 1999. Ecosystem-level evapotranspiration and water-use efficiency in the desert biome of Biosphere 2. Ecological Engineering, 13(1): 263-271.

Turner, B.L. and Meyer, W.B., 1994. Global land-use and land-cover change: an overview. Changes in land use and land cover: a global perspective, 4(3).

Twine, T.E. et al., 2000. Correcting eddy-covariance flux underestimates over a grassland. Agricultural and Forest Meteorology, 103(3): 279-300.

Vick, E.S., Stoy, P.C., Tang, A.C. and Gerken, T., 2016. The surface-atmosphere exchange of carbon dioxide, water, and sensible heat across a dryland wheat-fallow rotation. Agriculture, Ecosystems & Environment, 232: 129-140.

Wagle, P. and Kakani, V.G., 2014. Growing season variability in evapotranspiration, ecosystem water use efficiency, and energy partitioning in switchgrass. Ecohydrology, 7(1): 64-72.

Wagle, P., Kakani, V.G. and Huhnke, R.L., 2015a. Net ecosystem carbon dioxide exchange of dedicated bioenergy feedstocks: Switchgrass and high biomass sorghum. Agricultural and Forest Meteorology, 207: 107-116.

Wagle, P. et al., 2015b. Biophysical controls on carbon and water vapor fluxes across a grassland climatic gradient in the United States. Agricultural and Forest Meteorology, 214: 293-305.

Wang, G. and Eltahir, E.A., 2000. Role of vegetation dynamics in enhancing the low-frequency variability of the Sahel rainfall. Water Resources Research, 36(4): 1013-1021.

Wayne Polley, H., Frank, A.B., Sanabria, J. and Phillips, R.L., 2008. Interannual variability in carbon dioxide fluxes and flux–climate relationships on grazed and ungrazed northern mixed-grass prairie. Global Change Biology, 14(7): 1620-1632.

Weaver, C.P. and Avissar, R., 2001. Atmospheric disturbances caused by human modification of the landscape. Bulletin of the American Meteorological Society, 82(2): 269-281.

Weaver, S.J., Baxter, S. and Harnos, K., 2016. Regional Changes in the Interannual Variability of US Warm Season Precipitation. Journal of Climate, 29(14): 5157-5173.

Wilson, K. and Baldocchi, D., 2001. Comparing independent estimates of carbon dioxide exchange over 5 years at a deciduous forest in the southeastern United States. Journal of Geophysical Research. D. Atmospheres, 106: 34.

Wilson, K. et al., 2002. Energy balance closure at FLUXNET sites. Agricultural and Forest Meteorology, 113(1): 223-243.

Wright, C.K. and Wimberly, M.C., 2013. Recent land use change in the Western Corn Belt threatens grasslands and wetlands. Proceedings of the National Academy of Sciences, 110(10): 4134-4139.

Zeri, M. et al., 2011. Carbon exchange by establishing biofuel crops in Central Illinois. Agriculture, ecosystems & environment, 144(1): 319-329.

Zhao, F.-H. et al., 2007. Canopy water use efficiency of winter wheat in the North China Plain. agricultural water management, 93(3): 99-108.

Zhou, Y. et al., 2017a. Examining the short-term impacts of diverse management practices on plant phenology and carbon fluxes of Old World bluestems pasture. Agricultural and Forest Meteorology, 237: 60-70.

Zhou, Y. et al., 2017b. Quantifying agricultural drought in tallgrass prairie region in the US Southern Great Plains through analysis of a water-related vegetation index from MODIS images. Agricultural and Forest Meteorology, 246: 111-122.

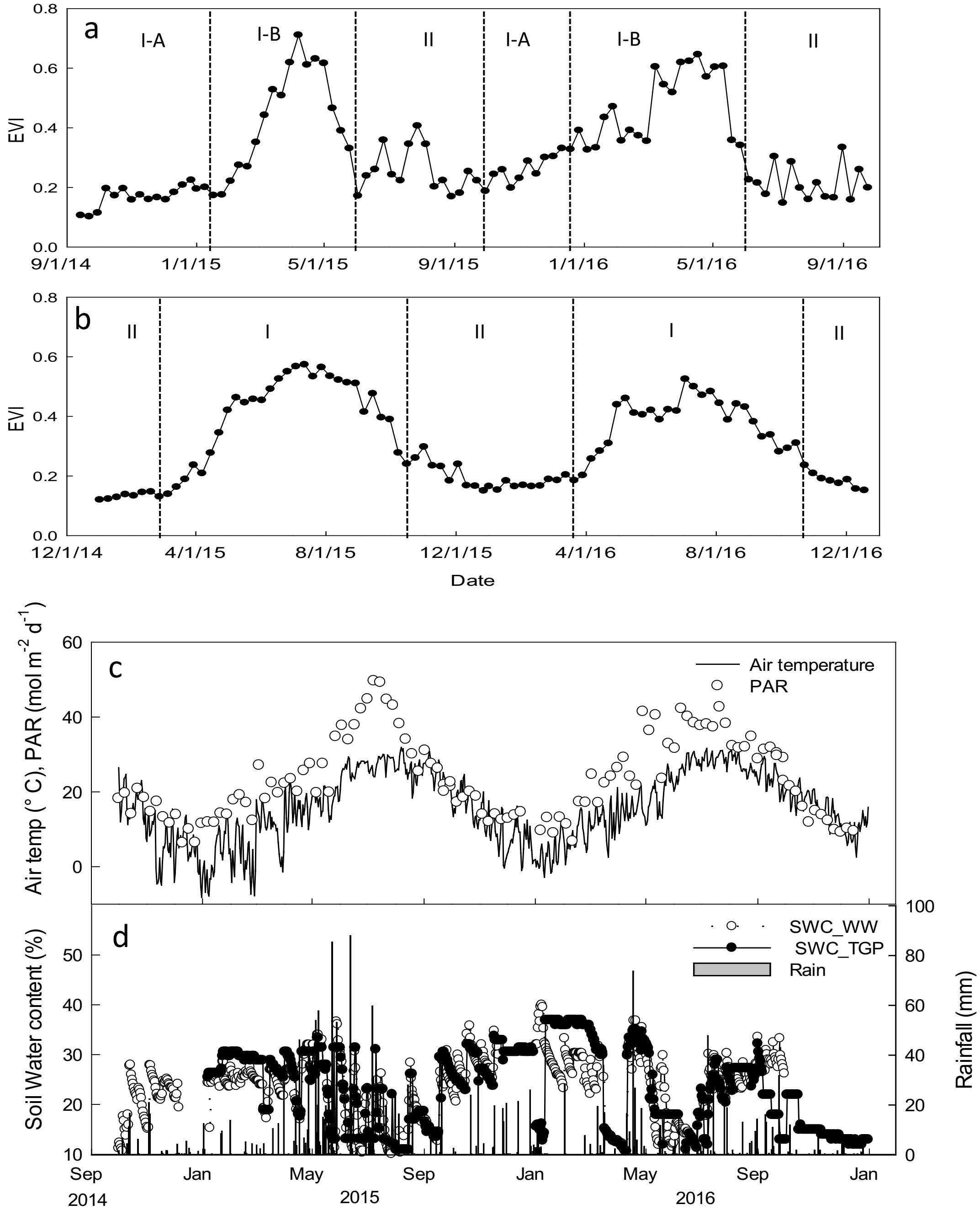

Fig.1. Seasonal dynamics of enhanced vegetation index (EVI) of (a) winter wheat, (b) tallgrass prairie (c) mean air temperature and photosynthetically active radiation (PAR); (d) Soil water content (SWC) and rainfall at winter wheat and tallgrass prairie sites respectively. I and II represents growing is non-growing season. The A and B represent green up before and after freezing respectively in winter wheat. Each data point for PAR and EVI represents 8-day mean.

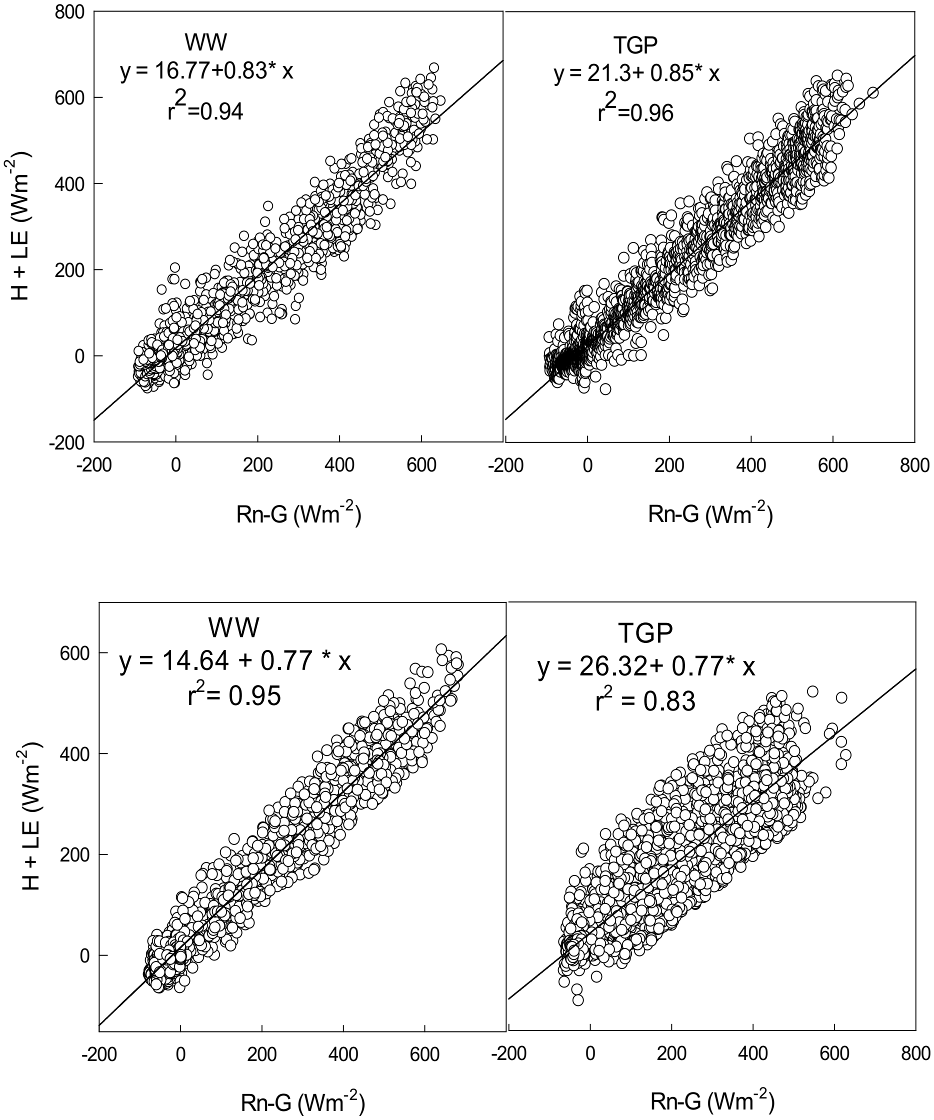

Fig.2. Relation between the available energy [net radiation (Rn) – soil heat flux(G)] and the sum of turbulent fluxes [(latent heat (LE) + Sensible heat (H)] at winter wheat (WW) and tall grass prairie (TGP) sites in 2015 (upper panel) and 2016 (lower panel).

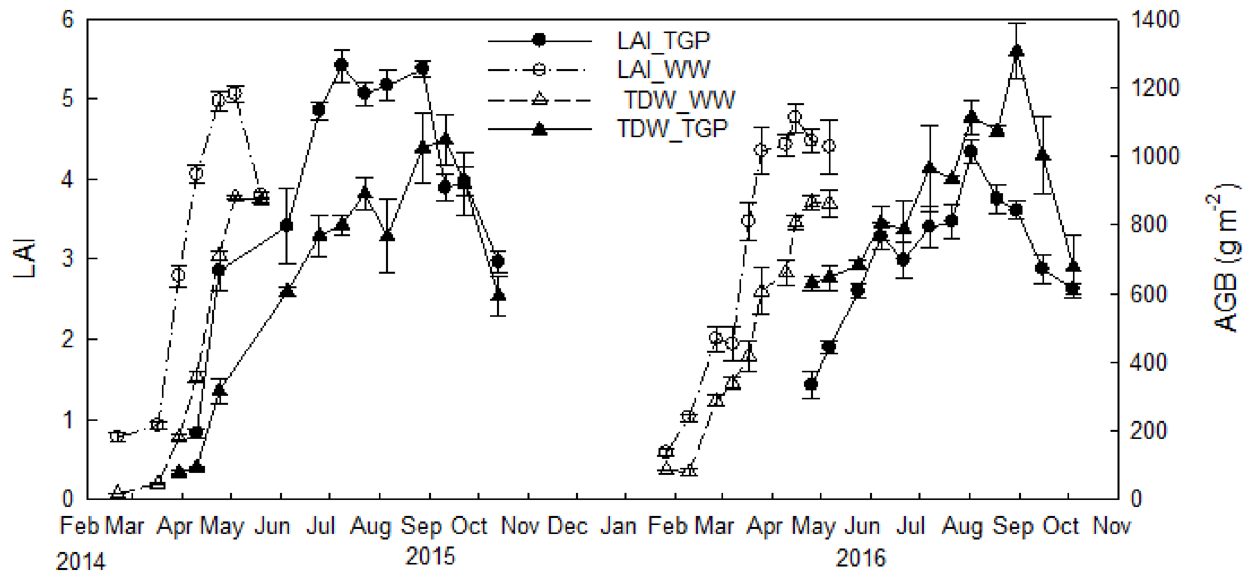

Fig.3. Evolution of leaf area index (LAI) and aboveground biomass (AGB) of winter wheat (WW) and tallgrass prairie (TGP) in 2015 and 2016.

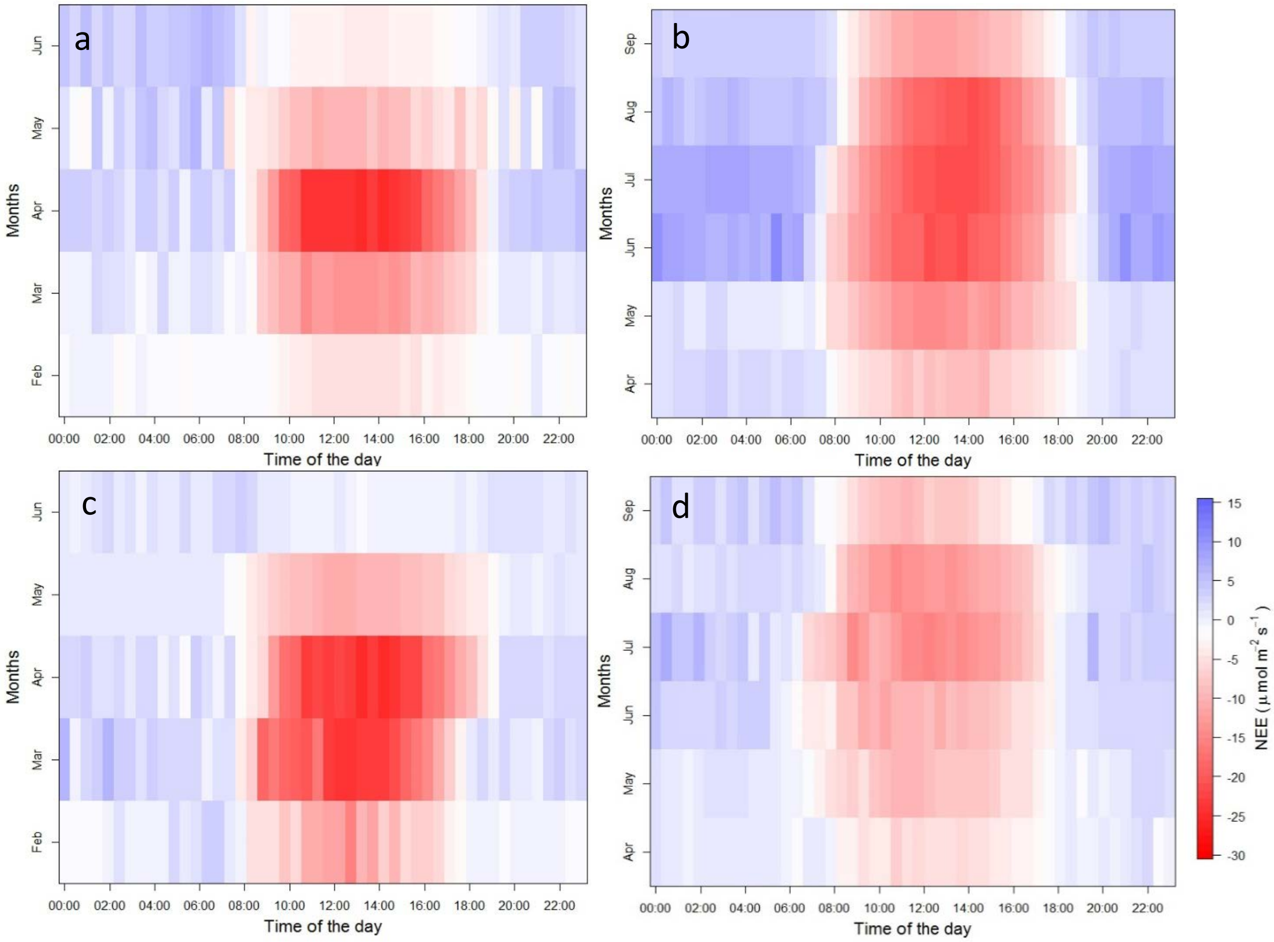

Fig.4. Half-hourly binned diurnal courses of net ecosystem CO_2 exchange (NEE) in winter wheat (WW) (left) and tall grass prairie (TGP) (right) across the active growing season. Each data point is a 30-min time-stamp average value for the entire month in 2015 (top) and 2016 (bottom).

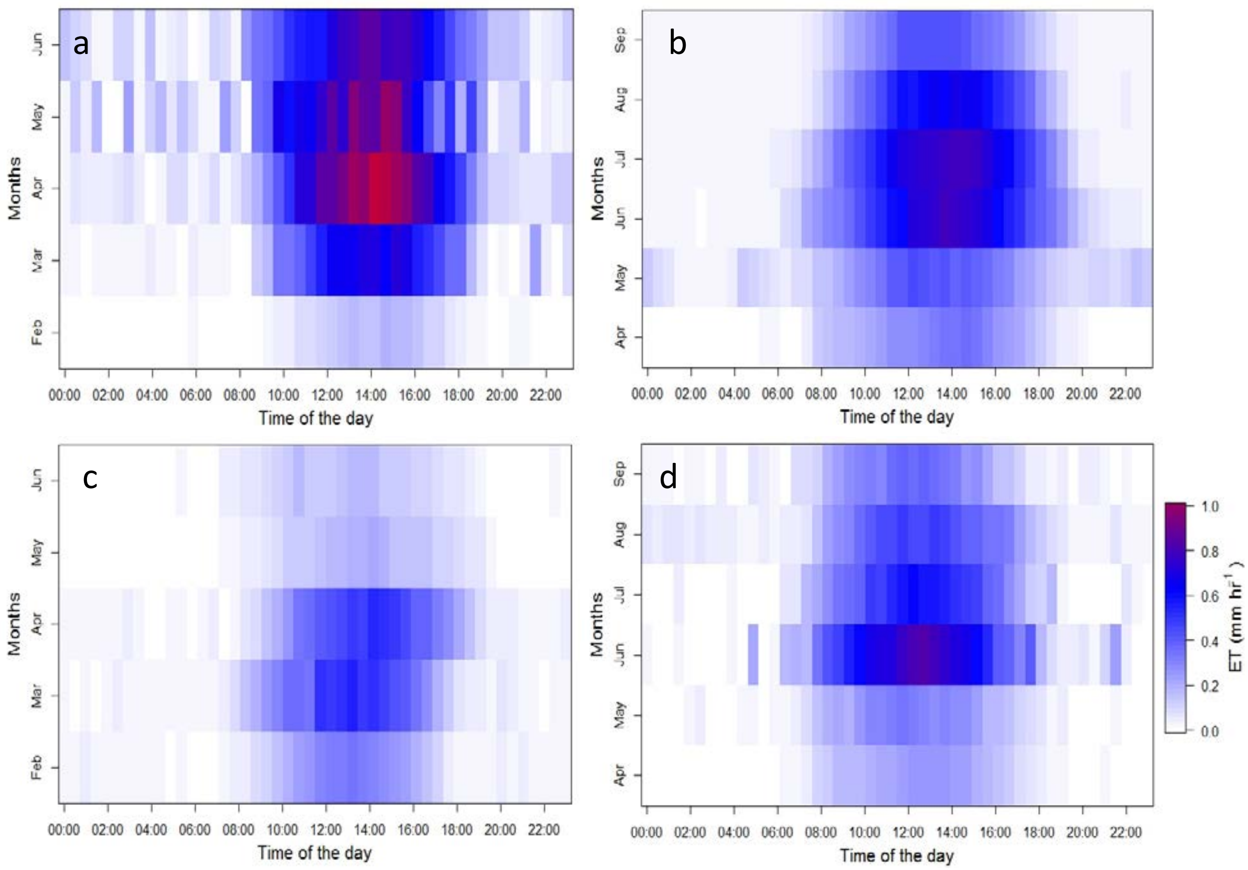

Fig.5. Half-hourly binned diurnal courses of evapotranspiration (ET) in winter wheat (WW) (left) and tall grass prairie (TGP) (right) across the active growing season. Each data point is a 30-min time-stamp average value for the entire month in 2015 (top) and 2016 (bottom).

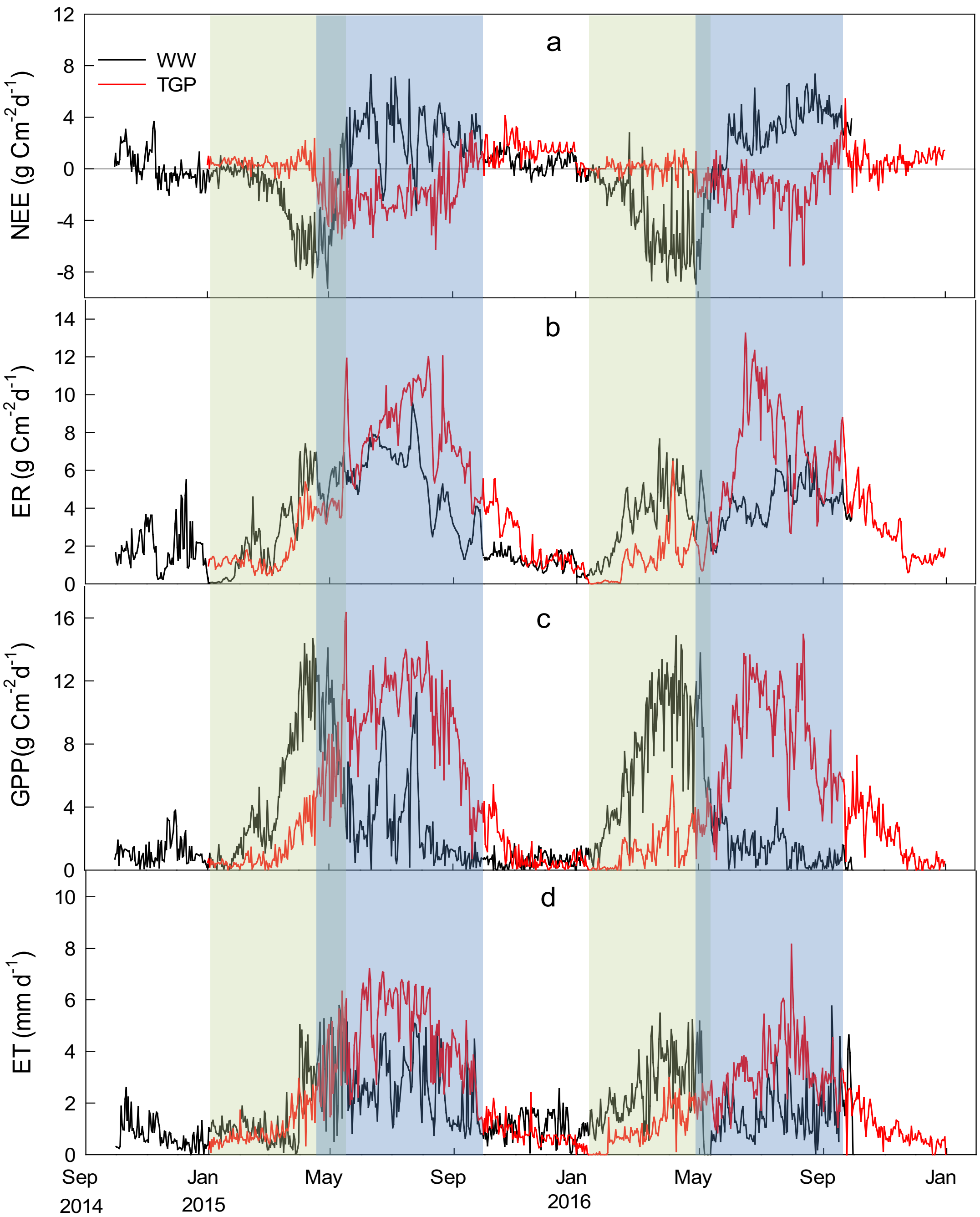

Fig.6. Growing season patterns of: (a) net ecosystem CO_2 exchange (NEE), (b) ecosystem respiration (ER), (c) gross primary productivity and (GPP) (d) Evapotranspiration (ET) in winter wheat (WW) and tall grass prairie (TGP) ecosystems. Data lines represent daily values of CO_2 and ET fluxes and the growing seasons are represented by shaded regions (greenish: WW, bluish: TGP).

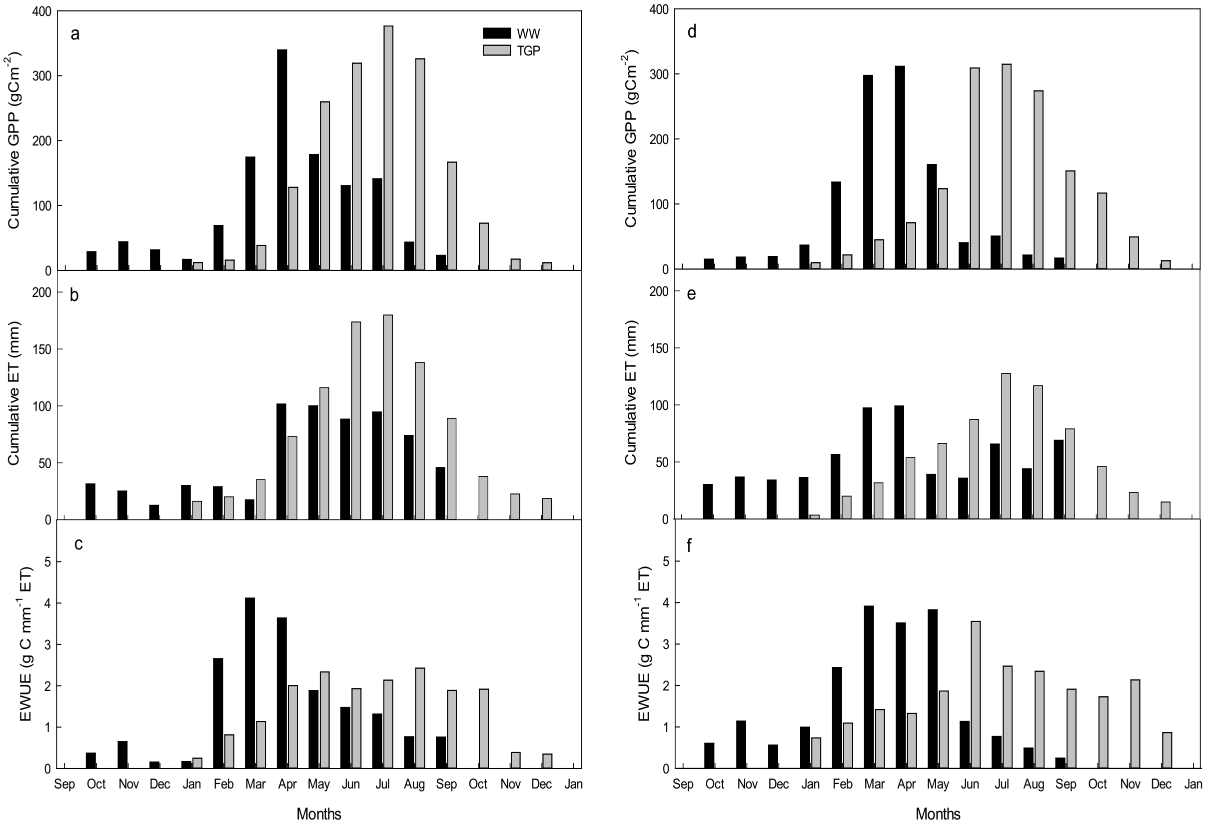

Fig. 7. Monthly cumulative gross primary productivity (GPP), evapotranspiration (ET) and average ecosystem water use efficiency (EWUE) at winter wheat (WW) and tallgrass prairie (TGP) ecosystems in 2015 (a-c) and 2016 (d-f).

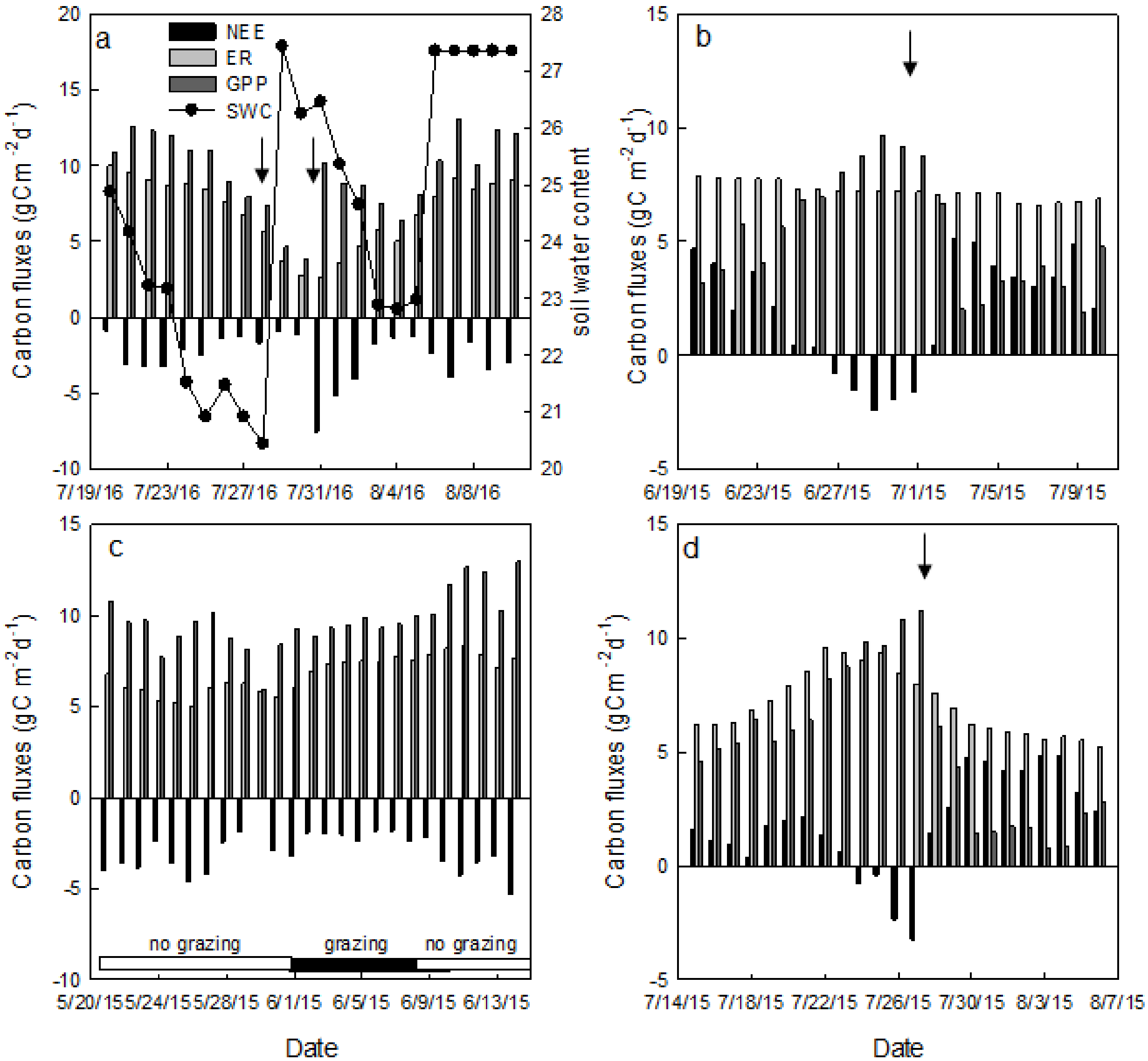

Fig. 8. Changes in carbon fluxes at winter wheat and tallgrass prairie ecosystem due to climate and management activities: (a) rainfall events, (b) tillage, (c) grazing and (d) herbicide application. The black solid arrows represent the occurrence of the events.

Graphical Abstract

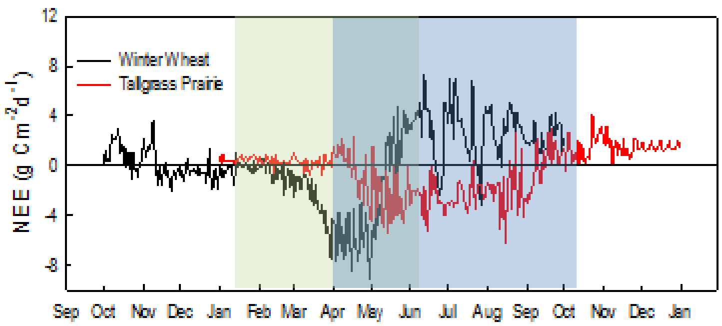

Table 1. Sums of net ecosystem CO_2 exchange (NEE), gross primary production (GPP), ecosystem respiration (ER), and evapotranspiration (ET) (± Standard Error), and average ecosystem water use efficiency (EWUE= GPP/ET) from Winter Wheat (WW) and Tallgrass prairie (TGP) fields during their respective growing seasons. Growing season (GS) refers to Oct - May (WW) and March - mid-Oct (TGP). Non-growing season refers to Jun-Sep (WW) and Jan-Feb & mid-Oct- Dec (TGP), and a whole year is an integrated flux for 12 months (Oct-Sep and Jan-Dec, respectively for WW and TGP).

		Winter Wheat					Tallgrass Prairie		
		Growing season (GS)			Non-growing season (NGS)	Annual	Growing Season (GS)	Non-growing season (NGS)	Annual
	Year	I-A (Fall)	I-B (Spring)	Total (I-A + I-B)					
GPP (g C m^{-2})	2015	150 ± 23	771 ± 146	921 ± 169	312 ± 81	1233 ±168	1663 ± 233	41 ± 19	1704 ± 252
	2016	93 ± 14	903 ± 123	996 ± 137	129 ± 48	1125 ± 185	1346 ± 103	152 ± 80	1498 ± 183
ER (g C m^{-2})	2015	176 ± 32	496 ± 122	672 ± 154	638 ± 143	1311 ± 153	1386 ± 221	203 ± 29	1589 ± 250
	2016	139 ± 19	464 ± 93	603 ± 102	555 ± 97	1158 ± 209	1186 ± 145	192 ± 23	1378 ± 168
NEE (g C m^{-2})	2015	26 ± 6	-277 ± 37	-251 ± 43	325 ±96	56 ± 13	-276 ± 43	148 ± 26	-128± 69
	2016	46 ± 12	-439 ± 61	-403 ± 73	357 ± 101	33 ± 9	-159 ± 61	40 ± 8	-119 ± 53
ET (mm m^{-2})	2015	94 ± 15	256 ± 81	350 ± 96	301 ± 102	651 ± 69	826 ± 72	93 ± 17	919 ± 89
	2016	135 ±74	294 ± 40	429 ± 114	214 ± 56	644 ± 111	588 ± 102	81 ± 15	669 ± 117
EWUE	2015	1.60	3.01	2.63	1.04	1.89	2.01	0.44	1.85
(g C mm^{-1}ET)	2016	0.69	3.07	2.32	0.60	1.75	2.29	1.88	2.24

Table. 2. The maximum rates of net ecosystem exchange (NEE, g C m^{-2} d^{-1}) and evapotranspiration (ET, mm d^{-1}) of winter wheat and tallgrass prairie at different study sites.

Sites	Year	Vegetation	NEE_{max}	ET_{max}	References
Ponca City, OK	1997-1998	Winter Wheat	-8.2	7.0	(Gilmanov et al., 2003)
Billings, OK	2001-2003	Winter Wheat	-9.3	5.2	(Fischer et al., 2007)
Selhausen, Belgium	2007-2009	Winter Wheat	-12.0	NA	(Schmidt et al., 2012)
Thuringia, Germany	2001	Winter Wheat	-13.3	5.7	(Anthoni et al., 2004)
Yucheng, China	2003-2005	Winter Wheat	-11.99	5.07	(Zhao et al., 2007)
El Reno, OK	2015	Winter Wheat	-9.2	6.0	This study
El Reno (Burned), OK	2005-2006	Tallgrass Prairie	-6.9	5.5	(Wagle et al., 2015b)
El Reno (Unburned), OK	2005-2006	Tallgrass Prairie	-5.2	5.7	(Wagle et al., 2015b)
Fermi, IL	2005-2007	Tallgrass Prairie	-9.5	5.6	(Wagle et al., 2015b)
Konza Prairie, KS	2007-2012	Tallgrass Prairie	-9.1	7.6	(Wagle et al., 2015b)
Shilder, OK	1997	Tallgrass Prairie	-8.1	-	(Suyker and Verma, 2001)
El Reno, OK	2015	Tallgrass Prairie	-6.3	7.0	This study