# Agriculture accentuates interannual variability in water fluxes but not carbon fluxes, relative to native prairie, in the U.S. Corn Belt

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ABSTRACT

To decrease negative environmental impacts associated with row crop agriculture, the conversion of conventional agricultural lands to no-till with cover crops or to restored prairie in the Midwest U.S. has been proposed and has the potential to alter hydrologic behavior. Our understanding of the impacts of this conversion on water and carbon fluxes from (agro-)ecosystems, however, is limited. We deployed eddy covariance systems in a business-as-usual (BAU) tilled cropping system, an aspirational (ASP) no-till cropping system with cover crops, and a native tallgrass prairie (TP) ecosystem to measure evapotranspiration (*ET*) and carbon dioxide exchange with the atmosphere. Measurements began in 2015 and we have at least 4 complete years of observations at each site. The average annual *ET* is higher at the TP site than the BAU site, but not significantly different from the ASP site. The gross primary production and ecosystem respiration are highest at the ASP site. Average annual net ecosystem exchange is negative (carbon uptake) at both agricultural sites (-305 $\pm$ 25 and -311 $\pm$ 31 gC m$^{-2}$ yr$^{-1}$ at ASP and BAU, respectively) and neutral in the prairie (-11 $\pm$ 10 gC m$^{-2}$ yr$^{-1}$ at TP). We evaluate the sensitivity of fluxes to environmental conditions including soil water content, vapor pressure deficit, air temperature, and photosynthetic photon flux density and find that the BAU site is the most sensitive to changes in environmental conditions while the TP site is the most resilient to changes. The interannual variability in *ET* is accentuated by agricultural management and because of the more diverse cropping system, is highest at the ASP site. The interannual variability in carbon fluxes, however, is not increased by agricultural management. Our findings illustrate how conservation practices impact the water budget and highlights the value of these practices in a changing climate.

## 1. Introduction

Global food demand is expected to double by 2050, relative to 2005 levels, thereby increasing demand for ecosystem services from agro-ecosystems (Tilman et al., 2011). Globally, agricultural crops cycle large amounts of carbon through carbon uptake to produce food, fiber, and seed, which is eventually returned to the atmosphere via decomposition and respiration (Snyder et al., 2009). During the growing season, the Corn Belt in the Midwestern United States has the highest rate of Gross Primary Productivity (*GPP*) in the world (Guanter et al., 2014). This high *GPP*, along with increased global efforts to limit global warming, has generated interest in using agricultural lands to sequester carbon from the atmosphere, though evidence for the effectiveness of this strategy is mixed (Baker et al., 2007a; Baker and Griffis, 2005). In

addition to influencing the carbon budget, croplands in the Corn Belt modify the water and energy fluxes from the land surface, with far ranging impacts to the regional climate, groundwater contamination, streamflow, and flooding (Al-Qudah et al., 2016; Alter et al., 2018; Kelly et al., 2017).

In the U.S. Corn Belt, a large fraction of the land is used for agriculture, most of which is in a rotation of maize (*Zea mays* L.) and soybean (*Glycine max* L.), and this area is expanding (Lark et al., 2020). Historically, however, much of this land was covered with tallgrass prairie where a vast land conversion effort in the 19th century was one of the most rapid land transformations in the history of mankind (Robertson et al., 1997). The transition from native prairie to agricultural lands is estimated to have resulted in the depletion of 20 to 25% of soil carbon across the U.S. Corn Belt (Allmaras et al., 2000; Baker and Griffis, 2005).

This carbon loss has led many to propose efforts to sequester atmospheric carbon in agricultural soils (Al-Kaisi and Yin, 2005; Follett et al., 2012; Ogle et al., 2019; Sun et al., 2020), though the long-term efficacy of this idea to combat global warming is debatable (Baker et al., 2007b). While the conversion of land from cropland back to prairie and forest has been limited (Rhemtulla et al., 2007), recent efforts to restore prairies are accelerating (Schilling and Drobney, 2014). Efforts to incorporate prairie strips into agricultural fields show promise to improve multiple ecosystem services delivered by the agro-ecosystem (Schulte et al., 2017). Maize and soybean are the two most common crops in the United States. Thus, understanding carbon flux dynamics in maize/soybean rotations and prairie in the Corn Belt is an important challenge if carbon sequestration is to be considered.

Recent efforts to quantify the carbon budget of different crop types in the U.S. Corn Belt using eddy covariance (EC) systems have resulted in considerable progress in our understanding of how these agro-ecosystems function. Previous studies have shown that net ecosystem exchange (*NEE*) in soybean is either carbon neutral or a small carbon source, while maize is a carbon sink, but this does not account for carbon removed as yield (Dold et al., 2017, 2019; Gebremedhin et al., 2012). Crop growth in the Corn Belt is generally energy limited, rather than water limited (Hernandez-Ramirez et al., 2011). Maize has been shown to use more water than soybean as well as having higher carbon uptake (Prueger et al., 2004), meaning that maize generally has a higher water use efficiency (*WUE = GPP/ET*) than soybean (Anapalli et al., 2019). Prueger et al. (2004) also find that the daily fluxes of carbon and water vapor depend primarily on the incoming solar radiation, whereas the seasonal fluxes are controlled by the water holding capacity of the soil. Dold et al. (2017) demonstrate that *GPP* and *WUE* are both linearly related to precipitation, soil water content, and maximum air temperature. The differences between how carbon and water fluxes of managed and natural ecosystems respond to interannual climate variability, however, has not been widely explored. Measurements of evapotranspiration (*ET*) found that growing season water use is not significantly different between maize and tallgrass prairie at a study site in Michigan (Hamilton et al., 2015). Similar *ET* for maize, soybean and reconstructed prairie in Iowa has been found based on chamber measurements (Luo et al., 2018) and remote sensing (Baeumler et al., 2019).

The AmeriFlux network contains various sites to measure water and carbon fluxes in agricultural settings. Results from these sites have been used to examine fluxes in maize/soybean rotations, but there is a lack of understanding of how land management practices alter the agro-ecosystem response to climate conditions. Tillage practices can influence the carbon and water fluxes, though reduced tillage has little impact (Baker and Griffis, 2005). No-till agriculture results in more carbon uptake than conventional tillage practices (Chi et al., 2016; Wagle et al., 2019). No-till practices can limit soil $CO_2$ emissions and evaporation in agro-ecosystems (Feiziene et al., 2011; Hu et al., 2013; Ward et al., 2012). Cover crops have also been shown to alter the $CO_2$ fluxes and evapotranspiration. For soybean crops, which have a positive or near-zero *NEE*, a cover crop allows the cropping system to have a negative *NEE* (Gebremedhin et al., 2012). Nevertheless, our understanding of how fluxes from current and aspirational cropping practices and prairie ecosystems behave under a variety of environmental conditions is limited. Wagle et al. (2019) examined the optimal environmental conditions to maximize carbon fluxes in the southern Great Plains, but little is known about how cropping practices affect ecosystem resilience to changes in environmental conditions. In the U.S. Corn Belt, climate change is expected to result in wetter spring months and drier summer months, which poses a challenge for rainfed agriculture (Byun and Hamlet, 2018; Chen and Ford, 2022). (Agro)-ecosystems that utilize multiple portions of the year for growth may be more resilient to these precipitation shifts, but quantifying this resilience requires long-term observations. The Long-Term Agroecosystem Research (LTAR) network is a relatively new research initiative that is contributing up to 18 new sites to the AmeriFlux network (Menefee et al., 2022). These sites will be funded long-term to research systems level agricultural questions, making them valuable contributions to the AmeriFlux network (Sadler et al., 2015; Wallbridge and Shafer, 2011).

As part of the LTAR network, we installed EC towers at three locations at the Central Mississippi River Basin (CMRB) Long Term Agroecosystem Network (LTAR) site located in central Missouri. The three sites consist of a conventionally managed agricultural field, an aspirationally managed field using no-till and cover crops, and a native prairie that has never been tilled. We use these observations to compare the sites at a systems level to understand how two important cropping rotations impact water and carbon fluxes in the U.S. Corn Belt. Because of how no-till and cover crops individually impact agricultural fluxes we hypothesize that (1) water and carbon fluxes are highest in aspirational agriculture, relative to conventional agriculture and prairie. Previous work has found that the *ET* and *NEE* depend heavily on the crop type, so we hypothesize that (2) interannual variability in water and carbon fluxes is accentuated by more complex cropping systems, relative to a native prairie. Finally, (3) because the native prairie has more than 100 species of plants, we hypothesize that it maintains *GPP* over a wider range of environmental conditions than the agricultural fields.

## 2. Methods

### 2.1. Study sites

The LTAR network was developed in part to provide information about how different cropping systems behave (Kleinman et al., 2018). Within the LTAR network, the croplands common experiment aims to evaluate strategies to sustainably intensify agricultural production by comparing aspirational (ASP) and business-as-usual (BAU) cropping systems. In this study, we present observations from the CMRB LTAR site located in central Missouri, USA. Observations at the site began in 1971 when streamflow and rain gauges were installed, and several climatological variables were measured (Baffaut et al., 2015). An automated weather station was installed in 1993 and has been run continuously since (Sadler et al., 2015). Beginning in 2015, eddy covariance (EC) towers were installed in the ASP and BAU fields, as well as at a tallgrass prairie, Tucker Prairie (TP) to quantify the fluxes of $CO_2$ and water vapor (Fig. 1).

The TP site is a remnant tallgrass prairie that was bought by the University of Missouri in 1957 and has never been plowed and contains over 100 different vegetation species (Li et al., 2021). At the TP site, winter or early spring burns were conducted annually from 1958 to 2002. Since 2002, it has been divided into 5 units, where a burn rotation is applied, and each unit is burned twice in a 5-year period. The BAU agricultural site is managed by a local farmer who employs a maize/-soybean rotation which generally consists of two years of soybean followed by one year of maize. The field is conventionally tilled, and no cover crops are used. Nitrogen fertilizer is applied at a constant rate across the entire field. The ASP field, in contrast, uses a more complex cropping system and is managed by USDA-ARS. Beginning in 2015, a 3-year cropping system consisting of wheat-maize-soybean was applied. In 2019, the maize crop failed and soybean was planted. At the same time, the cropping system was extended to a 4-year system consisting of maize-soybean-wheat-hay (Table 1). The ASP site also utilizes cover crops, no-till, and precision fertilizer application (Yost et al., 2017). A timeline of management activities at the two agricultural sites is displayed in Table 1.

The three study sites have similar climate and geologic conditions. Topographic relief is minimal with slopes at each of the three sites ranging from 0-3%. Runoff potential, however, remains high because of the presence of claypan restrictive layers between 10-50 cm depth that limit infiltration (Baffaut et al., 2015; Sadler et al., 2015). Soils at the ASP and BAU sites include Adco silt loam (fine, smectitic, mesic Vertic Albaqualfs) in summit positions with 0% to 1% slopes, and Mexico silt loam (fine, smectitic, mesic Vertic Epiaqualfs) in backslope (1% to 3%)
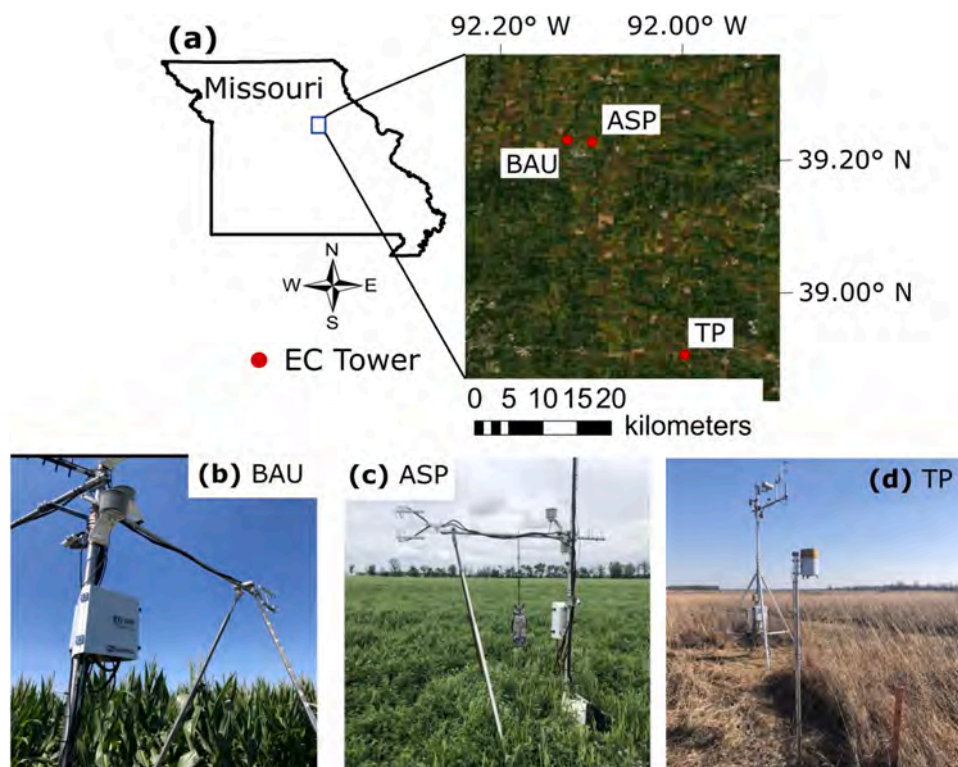
**Fig. 1.** The (a) three study sites located within central Missouri, USA including photographs of the EC deployments at (b) the business-as-usual (BAU) field (July 24, 2020), (c) the aspirational management (ASP) field (May 25, 2021), and (d) tallgrass prairie (TP; December 9, 2021).

**Table 1**
Dates when crop transitions occur, typically seeding or harvest (month/day), and management activities at the ASP and BAU fields.

| Site | Year | Crop | Crop Start | Crop End | Fertilizer | Tillage | Herbicide |
|------|------|------|-----------|----------|-----------|---------|-----------|
| ASP | 2016 | Maize | 4/15/2016 | 9/30/2016 | 4/15, 6/3, 6/22 | - | 5/23, 6/8 |
| | 2017 | Soybean | 5/17/2017 | 10/19/2017 | 10/18, | - | 5/10, 6/19, 7/10, 10/18 |
| | 2018 | Wheat | 10/19/2017 | 7/2/2018 | 3/21, 4/30 | - | 8/8, |
| | 2019* | Soybean | 6/10/2019 | 10/23/2019 | 5/16, 10/19 | - | 6/11, 7/16 |
| | 2020 | Wheat | 10/23/2019 | 7/8/2020 | 3/6, 5/6, 9/22 | - | |
| | 2021 | Hay | 3/6/2020 | 10/5/2021 | 4/5 | - | |
| | | | | | | | |
| BAU | 2016 | Soybean | 5/22/2016 | 10/25/2016 | 11/21, | | 5/5, |
| | 2017 | Maize | 4/19/2017 | 10/13/2017 | 2/14, | 4/19,11/2,11/6 | 5/3, |
| | 2018 | Soybean | 5/12/2018 | 10/30/2018 | | 4/27, | 5/8, 6/7 |
| | 2019 | Soybean | 6/4/2019 | 10/18/2019 | 10/25, | | 6/3, 10/25 |
| | 2020 | Maize | 5/10/2020 | 10/31/2020 | 4/8, 11/20 | 4/15, 5/10 | 5/31, 11/21 |
| | 2021 | Soybean | 5/19/2021 | 11/10/2021 | | | |

* In 2019 the cropping system at ASP was changed from a 3-year wheat-maize-soybean rotation to a 4-year maize-soybean-wheat-hay rotation. The maize crop in 2019 failed at the ASP site due to poor germination, and soybean was planted instead.

and footslope (1% to 2%) positions (Veum et al., 2015). The poorly drained soils cause challenges for crop grown and lead to soil degradation and water quality problems (Baffaut et al., 2020). Soils at the TP site have lower bulk density and higher infiltration rates at the surface than either of the agricultural sites (Mudgal et al., 2010). Detailed descriptions of the vegetation and soils at the TP site can be found in (Kucera, 1956, 1958).

### 2.2. Eddy covariance data collection and processing

An EC tower was installed at each site; data collection began at the ASP field on October 1, 2015, the BAU field on June 1, 2016, and the TP site on January 1, 2018. At both ASP and BAU sites, the EC system consists of an integrated infrared gas analyzer and 3D sonic anemometer (IRGASON, Campbell Scientific Inc., Logan, UT) while the TP site uses a separate 3D sonic anemometer (WindMaster Pro, Gill Instruments Ltd.,

Lymington, UK) and gas analyzer (LI-7500A, LI-Cor Inc., Lincoln, NE). At the TP site, the installation height is constant 3.4 m with a maximum canopy height of 1.4 m that is reduced following management burns. At both BAU and ASP sites, the height of the EC instruments is varied between 1.8 and 3.6 m in response to crop growth in order to preserve a consistent measurement footprint that does not intersect the edge of the field. Stabilizer arms are built into each tower to prevent movement of EC instruments in the wind. To measure the energy balance, a four-component net radiometer measures incoming and outgoing components of both short and longwave radiation (CNR-4, Kipp and Zonen USA Inc. Bohemia, NY). Additionally, incoming and outgoing photosynthetically active radiation is measured with a quantum sensor (LI-190R, LI-Cor Inc.). Ground heat flux ($G$) is measured at 5 cm depth using four heat flux plates at each site (HFP01, Huskeflux USA, Center Moriches, NY). The $G$ at the soil surface was computed using the calorimetric method and taken as the sum of $G$ measured at a depth of 5 cm using a soil heat

flux plate (Hukseflux USA, East Moriche NY) and heat storage above the plate. Heat storage above the plate was determined from the time derivative of soil temperature measurements at 2 and 4 cm depth, and heat capacity estimated based on observed volumetric water content (CS615, Campbell Scientific). This setup was replicated 4 times (with the exception of the water content reflectometer as there is only 1 at each site), and we report mean values of G at the surface. Ancillary measurements of air temperature and relative humidity are taken using a heat shielded sensor (HC2A, Rotronic Measurement Solutions USA, Hauppauge, NY) above the canopy height and varied between 1.5 and 3.3 m height with the EC instruments. Precipitation is measured with a tipping bucket rain gauge located at the ASP site (TI-525, Texas Electronics, Dallas TX). Finally, canopy temperatures are measured with an infrared thermometer at each site (SI-111-SS, Apogee Instruments Inc., Logan, UT).

The EC systems measure high frequency data at 10 Hz resolution, from which covariances are computed at 30-minute time steps. We perform 2-D coordinate rotation into the natural wind coordinate because the height of the EC instruments is adjusted as the crops grow (Finnigan, 2004). This approach is appropriate for systems such as this one with flat topography and homogeneous land cover (Rannik et al., 2020). Finally, we apply the 'WPL' terms to account for air density effects due to heat and water vapor transfer (Webb et al., 1980). Spikes in measurements are filtered at the 30-minute time step when the signal strength of the gas analyzer measurement is less than 80% at the ASP and BAU sites and less than 100% at the TP site. The stricter threshold at the TP site is due to the different gas analyzer and was selected based on a visual inspection of the data; lower thresholds resulted in unrealistic spikes in carbon fluxes. An additional filter removes net ecosystem exchange (*NEE*) values where $NEE < -90$ or $> 60$ µmol m$^{-2}$ s$^{-1}$, and latent heat (*LE*) or sensible heat (*H*) values when *LE* or $H < -200$ W m$^{-2}$ or $> 800$ W m$^{-2}$. Open path EC systems are less likely to suffer from high frequency losses than closed path systems, and our spectral analysis confirmed that these losses are not important at any of the three sites. Further filtering, gap-filling and partitioning of NEE is accomplished using in-house scripts developed using the REddyProc program (Wutzler et al., 2018). Periods of insufficient turbulence representing conditions for which EC assumptions fail are identified using an objective friction velocity (u∗) filtering method (Papale et al., 2006). Determination of the u∗ threshold is accomplished using the moving point method (Papale et al., 2006; Wutzler et al., 2018). Nighttime data (incoming solar radiation $< 10$ W m$^{-2}$) are split into different times of the year ('seasons') based on the surface roughness. These 'seasons' are delineated at the ASP and BAU sites with images from PhenoCams, harvest and planting dates, and crop height measurements, while at the TP site 'seasons' are not delineated because the vegetation height remains relatively constant. From here, the u∗ threshold is determined following Wutzler et al. (2018). The percent of missing data due to gaps from all sources is 16%, 21%, and 21% at ASP, BAU, and TP, respectively, which is a relatively low amount comparted to previous EC studies. Gaps in the turbulent fluxes are filled using the marginal distribution sampling (MDS) method with default parameters (Wutzler et al., 2018). Finally, the net ecosystem exchange (*NEE*) is separated into *GPP* and ecosystem respiration (*R_eco*) using the nighttime partitioning method (Reichstein et al., 2005).

We assess the EC measurements with the energy balance closure. The energy balance closure is calculated as $(LE + H + G)/R_n$ at each site and has values of 0.79, 0.73, and 0.84 at the ASP, BAU, and TP sites, respectively. This closure at the sites is within the range of reported values across the globe, where the energy balance closure for crops and wetlands is generally lower than other land cover classes and in the 0.7 – 0.78 range (Stoy et al., 2013; Wilson et al., 2002). Additionally, we assess the partitioning of carbon fluxes using the daytime partitioning method (Lasslop et al., 2010) and compare the results to the nighttime partitioning method. Finally, we assess the uncertainty in the EC measurements arising from three sources: random instrument error,

uncertainty in the selection of u∗ thresholds, and the lack of energy budget closure. We assess the uncertainty due to random error in annual sums of NEE using the daily differencing approach (Hollinger and Richardson, 2005). We compute the uncertainty associated with the selection of u∗ thresholds following Wutzler et al. (2018). For carbon flux measurements, the random measurement error and the uncertainty from the selection of u∗ thresholds are summed in quadrature for each 30-minute observation and propagated into annual sums following the approach laid out for autocorrelated measurements (Zięba and Ramza, 2011). For measurements of *ET*, we also consider the uncertainty from lack of energy budget closure by applying the Bowen ratio method to force energy budget closure (Twine et al., 2000).

### 2.3. Analysis techniques

We define several periods of analysis to draw robust conclusions. The initial data is collected at 30-minute resolution. We aggregate this data to the daily scale to examine inter-annual patterns in fluxes. Additionally, we examine inter-annual patterns by dividing the year into seasons. We define climatological seasons following divisions based on months; winter consists of JFM, spring is AMJ, summer is JAS, and fall is OND. We also define growing and non-growing seasons based on crop planting and harvest dates. Precise planting and harvest dates vary between the ASP and BAU files, so we define the growing season as May 15 – October 31. We use this definition to calculate the growing season *ET* (*ET_GS*) and non-growing season *ET* (*ET_NGS*). We present the standard error of the mean annual *ET* and carbon fluxes. We apply an ANOVA test with a Tukey HSD post-hoc test to test for statistically significant differences in the mean annual values of *ET* and carbon flux. We calculate the evaporative fraction (*EF*) at an annual and growing vs. non-growing season scale as the sum of the latent heat flux (*LE*) divided by the sum of the turbulent energy fluxes (latent heat and sensible heat, *H*). We calculate the albedo for each day of the study period using noontime observations of incoming and reflected outgoing shortwave solar radiation.

To assess resilience of the ecosystems to environmental conditions, we apply a threshold-based approach on binned data. We bin the *ET* and carbon flux data based on the environmental condition in question (e.g., soil moisture) and calculate the bin with the maximum magnitude of the flux and define this as the optimum value of that environmental condition. Then, we find the continuous range of that environmental condition where the bin average flux magnitudes are within 50% of the maximum value and define this as the range of values over which fluxes are resilient to changes in the condition. Sites with a larger range of resiliency are defined as more resilient to changes in that environmental condition. To test for differences between the three study sites, we perform one-way ANOVA tests to determine if bin means have statistically significant differences between the three treatments. To determine if flux values are changing with environmental conditions, we perform repeated measures ANOVA within each site to test that the flux in question (ET, NEE, GPP, or Reco) does vary with changes in environmental conditions. We also calculate the ecosystem water use efficiency (EWUE) at the annual scale as the GPP/ET.

### 3. Results

### 3.1. Temporal trends in land surface fluxes

An ANOVA test showed that there were significant differences in the mean annual *ET* ($p = 0.005$). The post-hoc Tukey HSD test showed that the *ET* at the TP site was significantly larger than the BAU site, while the ASP site was not significantly different from either TP or BAU at a significant level of $p < 0.05$. The lack of significance in differences between ASP and TP is partially attributable to the small sample size of 4 years. The TP site has perennial coverage, the ASP site also has perennial coverage, but plants are harvested or killed multiple times per year, and the BAU site has bare soil for roughly half the year. The observed

differences in annual *ET*, however, are not because the $ET_{NGS}$ is higher at sites with perennial coverage, indeed the $ET_{NGS}$ is lowest at the TP site and the ANOVA analysis reveals that it is significantly less than $ET_{NGS}$ at the ASP site (Table 2). The plants at the TP site have mature root systems, however, and this allows them to take advantage of the growing season and have higher $ET_{GS}$ than either the ASP or BAU sites (demonstrated by ANOVA). Additionally, the soil permeability is 1 to 2 orders of magnitude higher at the TP site than the ASP site, and various soil health indicators, such as bulk density and soil organic carbon, show that soils at the TP site can absorb more water than the ASP or BAU sites. (Mudgal et al., 2010; Veum et al., 2015). The uncertainty in annual estimates of *ET* resulting from random measurement error and the lack of energy budget closure is presented in Table 3. The primary component of uncertainty results from the lack of energy budget closure. Because of this, we repeated the ANOVA and post-hoc Tukey HSD tests described above using *ET* estimates with forced energy budget closure. None of the significant results change when values of *ET* are calculated with the energy budget forced closed.

The full year *EF* is similar at the two agricultural sites (i.e., no significant difference) indicating that the higher *ET* at the ASP site is a result of more energy available for the turbulent fluxes. This energy availability is primarily caused by lower ground heat flux; the ASP site has a slightly higher average albedo (0.218) than the BAU site (0.211), leading to only negligible differences in $R_n$. The low *EF* at the TP site, relative to the agricultural sites, however, is due to low *ET* during the non-growing season and higher *H* throughout the year. The low $ET_{NGS}$ is likely caused by a lower average albedo (0.19) and the insulating effect of the vegetation mat at the site. The annual *ET* values are higher at the ASP site than the BAU site for all years except 2018 and 2019. At the ASP site, wheat and soybean were planted in 2018 and 2019, respectively, while soybean was planted both years at the BAU site. It is noteworthy that at the BAU site, the years with soybean have higher *ET* than years with maize (soybean years had 641 mm/yr while maize years had 583 mm/yr), as this is contrary to previous findings (Prueger et al., 2004). Due to crop rotations, however, our study period is not long enough to draw robust interpretations regarding which crop uses more water.

The average annual carbon fluxes at the agricultural sites show more active carbon cycles with both ASP and BAU sites having higher magnitude of *NEE* than at the TP site, and the ASP site has higher *GPP*,

**Table 3**
Sources of uncertainty in annual estimates of *ET* including random error and uncertainty from the lack of energy budget closure.

| Site | Year | ET | ET Forced Closure | Closure Uncertainty | Random Uncertainty | Total Uncertainty |
|------|------|------|------|------|------|------|
| ASP | 2015 | 91 | 106 | 15 | 1 | 15 |
| | 2016 | 648 | 747 | 99 | 4 | 100 |
| | 2017 | 684 | 766 | 82 | 5 | 82 |
| | 2018 | 630 | 717 | 87 | 5 | 87 |
| | 2019 | 671 | 776 | 105 | 4 | 105 |
| | 2020 | 766 | 893 | 127 | 5 | 127 |
| | 2021 | 788 | 918 | 131 | 5 | 131 |
| BAU | 2015 | - | - | - | - | - |
| | 2016 | 482 | 582 | 100 | 3 | 100 |
| | 2017 | 556 | 667 | 111 | 4 | 111 |
| | 2018 | 644 | 772 | 129 | 3 | 129 |
| | 2019 | 713 | 859 | 146 | 5 | 146 |
| | 2020 | 602 | 721 | 119 | 5 | 119 |
| | 2021 | 582 | 685 | 103 | 4 | 104 |
| TP | 2015 | - | - | - | - | - |
| | 2016 | - | - | - | - | - |
| | 2017 | - | - | - | - | - |
| | 2018 | 659 | 759 | 100 | 4 | 100 |
| | 2019 | 724 | 841 | 117 | 4 | 117 |
| | 2020 | 702 | 817 | 115 | 5 | 115 |
| | 2021 | 684 | 739 | 55 | 5 | 56 |

and both ASP and BAU have higher $R_{eco}$ than the TP site (Table 4). Average annual *NEE* is near zero at the TP site (-11 g m$^{-2}$ yr$^{-1}$) and a t-test reveals that it is not significantly different from 0. The ASP site has higher *GPP* and $R_{eco}$ than the BAU site while the *NEE* values are similar (-305 and -311 gC m$^{-2}$ yr$^{-1}$ at the ASP and BAU sites, respectively). The number of days with *NEE*<0 is lowest at the TP site. This low number of days is likely due to the annual average *NEE* being near-zero so that *GPP* is balanced with $R_{eco}$.

The annual cumulative sums of *ET, NEE, GPP*, and $R_{eco}$ are presented in Fig. 2 and we describe the general patterns observed, we do not test for statistical significance when discussing differences in single year values. The *ET* at the TP site is higher than the BAU site in every year, but the ASP site has the highest annual *ET* in 2020 and 2021 when wheat and hay were planted, respectively. During the growing season, the *ET* is

**Table 2**
Annual, growing season, and non-growing season sums of observed evapotranspiration (*ET*) and evaporative fraction (*EF*). Uncertainty in annual estimates is presented as the random measurement errors propagated into the annual sum in *ET* for each year. The uncertainty in the annual average is presented as the standard error of the mean values.

| | Variable [unit] | 2015* | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | Average |
|------|------|------|------|------|------|------|------|------|------|
| ASP | $ET$ [mm] | 91±1 | 648±4 | 684±5 | 630±5 | 672±4 | 766±5 | 787±5 | 698±10 |
| | $ET_{GS}$ [mm] | 32±1 | 422±4 | 445±4 | 378±5 | 442±3 | 486±5 | 478±4 | 442±6 |
| | $ET_{NGS}$ [mm] | 58±1 | 221±2 | 237±2 | 246±2 | 226±2 | 275±2 | 306±3 | 252±5 |
| | ET/P | - | 0.75 | 0.83 | 0.64 | 0.60 | 0.80 | 0.68 | 0.72 |
| | EF [-] | 0.84 | 0.68 | 0.69 | 0.64 | 0.74 | 0.81 | 0.86 | 0.74 |
| | $EF_{GS}$ [-] | 0.61 | 0.70 | 0.69 | 0.59 | 0.78 | 0.78 | 0.79 | 0.72 |
| | $EF_{NGS}$ [-] | 0.99 | 0.70 | 0.76 | 0.78 | 0.73 | 0.90 | 0.98 | 0.81 |
| BAU[1] | $ET$ [mm] | - | 482±3 | 556±4 | 644±3 | 713±5 | 602±5 | 582±4 | 619±11 |
| | $ET_{GS}$ [mm] | - | 439±3 | 361±3 | 450±3 | 490±4 | 373±3 | 376±3 | 410±10 |
| | $ET_{NGS}$ [mm] | - | 42±1 | 194±2 | 188±2 | 219±2 | 225±3 | 203±2 | 206±3 |
| | ET/P | - | - | 0.68 | 0.65 | 0.64 | 0.63 | 0.51 | 0.62 |
| | EF [-] | - | 0.86 | 0.66 | 0.74 | 0.81 | 0.72 | 0.68 | 0.72 |
| | $EF_{GS}$ [-] | - | 0.85 | 0.67 | 0.79 | 0.82 | 0.70 | 0.68 | 0.73 |
| | $EF_{NGS}$ [-] | - | 0.90 | 0.65 | 0.65 | 0.78 | 0.75 | 0.68 | 0.70 |
| TP[2] | $ET$ [mm] | - | - | - | 685±4 | 773±4 | 751±5 | 711±5 | 730±9 |
| | $ET_{GS}$ [mm] | - | - | - | 511±4 | 554±4 | 537±5 | 513±5 | 529±4 |
| | $ET_{NGS}$ [mm] | - | - | - | 168±2 | 211±2 | 211±2 | 195±3 | 196±4 |
| | ET/P | - | - | - | 0.70 | 0.69 | 0.78 | 0.62 | 0.70 |
| | EF [-] | - | - | - | 0.65 | 0.76 | 0.73 | 0.68 | 0.71 |
| | $EF_{GS}$ [-] | - | - | - | 0.77 | 0.83 | 0.83 | 0.77 | 0.80 |
| | $EF_{NGS}$ [-] | - | - | - | 0.44 | 0.61 | 0.56 | 0.52 | 0.53 |

[1] Data available 6/1/2016 – 9/30/2021
[2] Data available 1/1/2018 – 9/30/2021
* Data collection began 9/30/2015

**Table 4**

Annual sums of observed $CO_2$ fluxes from the three study sites, as well as the average values. Carbon fluxes are reported with units of g C m$^{-2}$ yr$^{-1}$. Uncertainty in *NEE* is presented as the random measurement and u* errors, propagated into the sum for each year. The uncertainty value associated with annual averages is the standard error of the mean values.

| | Variable | 2015* | 2016 | 2017 | 2018 | 2019 | 2020 | 2021 | Average |
|---|---|---|---|---|---|---|---|---|---|
| ASP[1] | *NEE* | -25±7 | -362±17 | -331±17 | -282±31 | -34±41 | -533±18 | -338±34 | -305±25 |
| | *GPP* | 206±4 | 1522±25 | 1472±16 | 1426±36 | 1342±52 | 1876±18 | 1944±19 | 1566±38 |
| | $R_{eco}$ | 181±8 | 1160±33 | 1141±16 | 1144±54 | 1308±90 | 1343±16 | 1606±12 | 1261±28 |
| | Days *NEE*<0 | 65 | 165 | 206 | 197 | 104 | 255 | 235 | 196±8 |
| BAU[2] | *NEE* | - | -274±12 | -400±13 | -290±14 | -195±13 | -513±15 | -68±10 | -311±31 |
| | *GPP* | - | 1277±10 | 1236±11 | 1036±17 | 1253±11 | 1240±14 | 917±12 | 1246±27 |
| | $R_{eco}$ | - | 1003±10 | 836±7 | 746±16 | 1058±4 | 727±8 | 850±10 | 935±34 |
| | Days *NEE*<0 | - | 106 | 128 | 145 | 131 | 139 | 114 | 137±2 |
| TP[3] | *NEE* | - | - | - | 8±17 | -7±14 | -77±40 | 33±19 | -11±10 |
| | *GPP* | - | - | - | 969±43 | 1158±13 | 1383±86 | 1174±55 | 1171±37 |
| | $R_{eco}$ | - | - | - | 975±43 | 1102±17 | 1305±121 | 1208±63 | 1147±31 |
| | Days *NEE*<0 | - | - | - | 107 | 120 | 125 | 107 | 115±2 |

[1] Data available 9/30/2015 – 12/31/2021
[2] Data available 6/1/2016 – 12/31/2021
[3] Data available 1/1/2018 – 12/31/2021
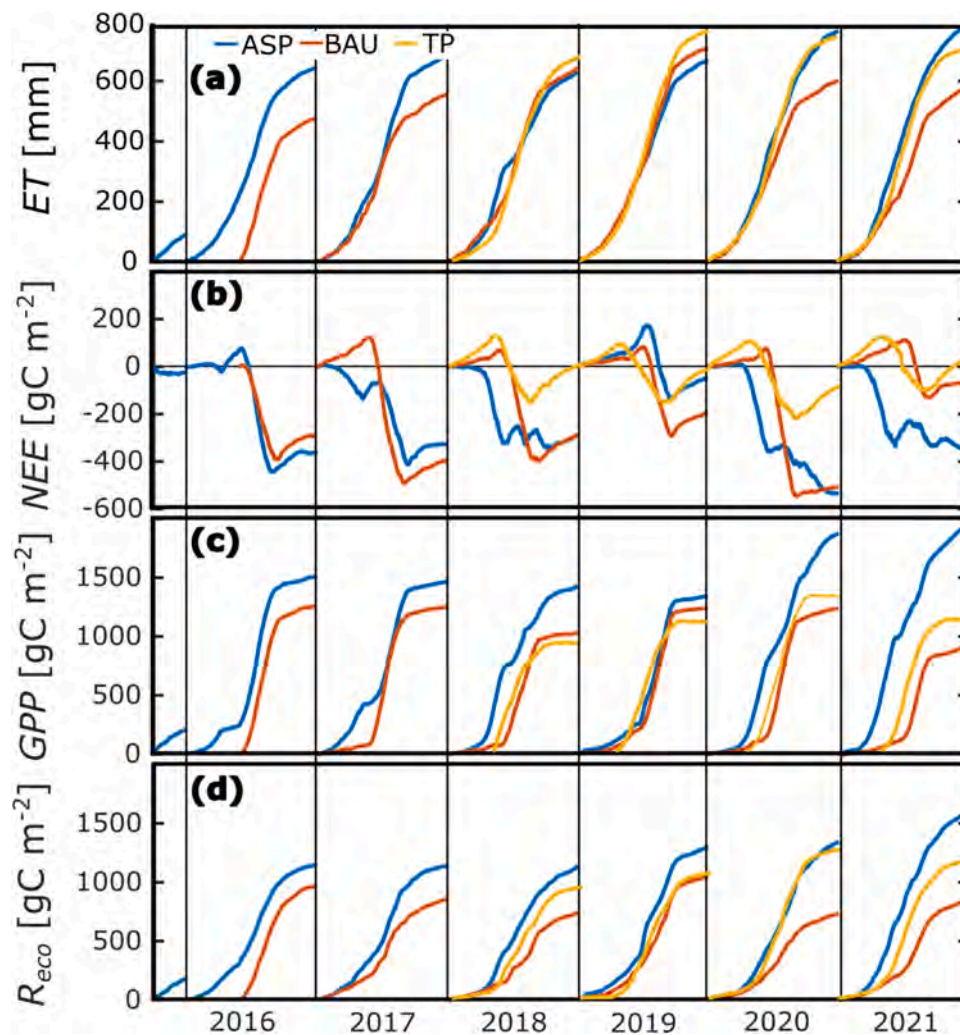


**Fig. 2.** Annual cumulative sums of (a) evapotranspiration, (b) net ecosystem $CO_2$ exchange, (c) gross primary productivity, and (d) ecosystem respiration from each of the three study sites. Annual values are defined by the calendar year (January 1 – December 31) and all available data are presented. See Table 1 for crop types planted during each year.

6% higher at the ASP site (442 mm/GS) than the BAU site (415 mm/GS), and during the non-growing season the *ET* is 18% higher at the ASP site (252 mm/NGS) than the BAU site (206 mm/NGS). As the BAU field is fallow during the non-growing season, it is not surprising that the ASP

site has higher *ET*. But it is noteworthy that the TP site has lower *ET* during the non-growing season than both the agricultural sites, even though the TP site has perennial vegetation cover. During the NGS most of the vegetation at the TP site goes dormant, leaving behind a 0.5-1 m

thick layer of dried grass and plant stems, some of which remain standing, some of which lie flat on the soil surface. This dormant vegetation creates a higher albedo surface, relative to bare soil, and although wind can travel through the standing plant stems, they form a semi-permeable barrier to mass transfer from soil to atmosphere. Similar patterns are observed in the carbon fluxes, where carbon uptake via *GPP* at the ASP site is greater than the *GPP* at the BAU site during both the growing and the non-growing season, but the difference is larger in the non-growing season. The high *GPP* at the ASP site for 2020 and 2021 is noteworthy. In both years crop growth starts early in the year and is sustained throughout the year. The crops are winter wheat (2020) and hay (2021). The $R_{eco}$ at all sites is the highest during the warm, humid summer period (July-September). The high levels of *GPP* at the ASP site are indicative of higher rates of metabolism and thus autotrophic respiration, as well as production of substrate for heterotrophic respiration, which collectively sustain the highest levels of $R_{eco}$.

The average annual cycle of water and carbon fluxes, using all available data at each site, supports the above finding that the sites with agricultural management have higher carbon fluxes, but lower water flux, than the prairie site (Fig. 3). Daily values of *LE* are highly variable as they are affected by several factors including temperature, soil water availability, and incoming radiation. The carbon cycle shows differences between the behavior of the three sites. The differences are most obvious during the agricultural growing season when carbon uptake at the two agricultural sites peaks (Fig. 3b). The TP site has perennial vegetation, but *GPP* is negligible during the winter, when temperatures are often below freezing, but the warm season carbon uptake is similar to the agricultural sites (Fig. 3c). The ASP and BAU sites have high carbon uptake during the growing season, consistently higher than 10 g C m$^{-2}$ d$^{-1}$. The ASP field has considerable carbon uptake during the period from approximately day of year (DOY) 50 through DOY 150 when a cover crop is growing, that is not present at the BAU site. The drop in *GPP* at the ASP field between DOY 150 and DOY 175 is due to herbicide applications to kill the cover crop. The higher *GPP* at the ASP field is balanced by higher $R_{eco}$, particularly during the period from DOY 75 through DOY 200, resulting in a total NEE that is similar to the BAU field.

In addition to the annual cycle of water and carbon fluxes, land use practices impact the annual average diel cycles of energy fluxes from the land surface. The net radiation ($R_n$) is similar across the three sites with minor differences during the peak daytime hours (Fig. 4a). These differences are caused by small differences in the albedo between the sites; the mean albedo at the ASP site (0.218) is higher than that of the BAU (0.211) or TP sites (0.19). The timing of peak values and diel patterns of latent energy (*LE*) and sensible heat (*H*) are similar across the three sites, with the primary difference being the magnitude of the energy fluxes. As noted above, the TP site has the highest *LE* but more of the available radiation is directed to *H* than at the two agricultural sites, resulting in a lower evaporative fraction (Fig. 4 b,c). The ASP site has higher average values of both *LE* and *H* than the BAU site, which is partly explained by higher values of *G* at the BAU site (Fig. 4). The *G* has larger differences in the temporal dynamics than the other values. At the TP site, the peak flux occurs at 13:00 (1 PM), 1 hour later than at the two agricultural sites, and *G* is elevated at the TP site throughout the afternoon, evening, and night. This delay is likely caused by the dense grassy vegetation at the site which provides shading and thermal buffering. Differences in the energy budget closure across the sites result in some uncertainty in direct comparisons. We examined the diel cycles in the energy budget for the growing and non-growing seasons separately (data not shown), and for each season, the patterns are very similar to those for the full year. During the non-growing season, the TP site has the lowest peak *LE* of the three sites and significantly higher peak values of *H*.

### 3.2. Factors affecting land surface fluxes

As the three study sites are located near each other, the average annual environmental conditions are similar (Table 5). Precipitation measurements are only available at the ASP site and average precipitation is assumed to be representative for all sites and is 983 mm/yr during the study period. Differences in the average conditions between the sites are minor and are partially attributable to different measurement periods. During the time periods with available data, the TP site has the highest average $T_{air}$, lowest *VPD* and $T_{soil}$, and highest *SWC*, which is consistent with its higher *H* and lower *LE* than the other sites. The primary difference between the two agricultural sites is that the BAU site has a higher *PPFD* than the ASP site.

We use the temporal variability in environmental conditions that are observed at each site to assess the sensitivity of land surface fluxes to different conditions. The analyses presented use all years available, so environmental relationships in the managed systems average over different crop types. Fig. 5 presents binned average values of water and
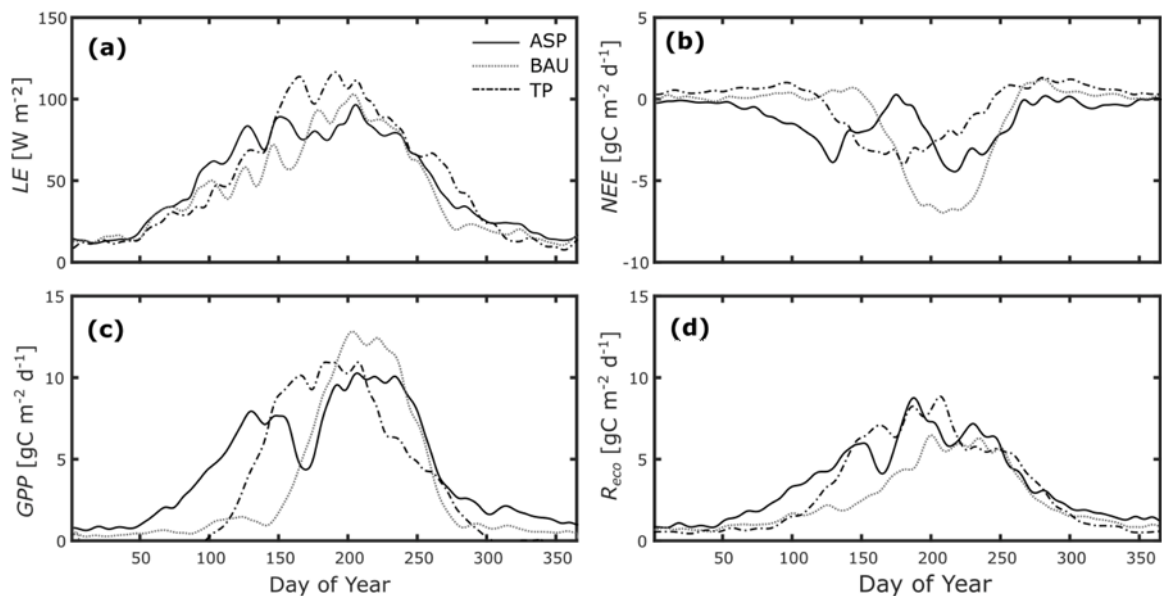


**Fig. 3.** Average annual pattern of daily fluxes of latent energy (a), net ecosystem exchange (b), gross primary productivity (c), and ecosystem respiration (d). Values are the average flux on each day of the year (Jan. 1 – Dec. 31) for all years with available data.
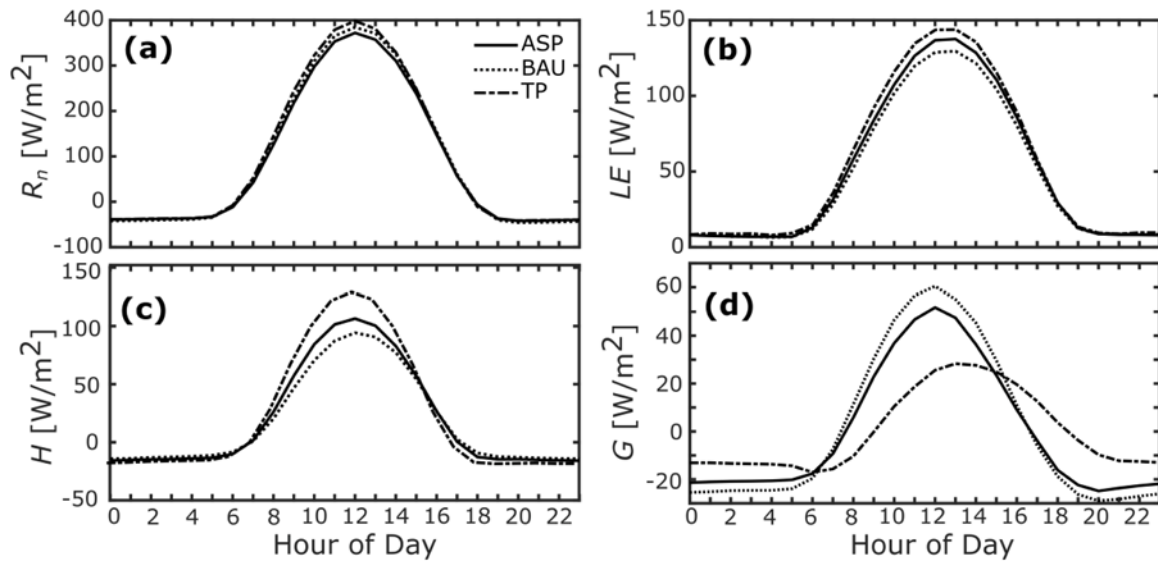
**Fig. 4.** Annually averaged diel cycle in surface energy fluxes for the aspirational (ASP), business as usual (BAU), and Tucker Prairie (TP) sites. Net radiation (a), latent energy (b), sensible energy (c), and ground heat flux (d) are presented.

**Table 5**

Annual average and standard error of the mean of precipitation, daytime photosynthetic photon flux density (*PPFD*), air temperature ($T_{air}$), daytime vapor pressure deficit (*VPD*), soil water content at 12.5 cm depth (*SWC*), and soil temperature at 12.5 cm depth ($T_{soil}$) for the three study sites. Precipitation is only recorded at the ASP site.

| Meteorological Variable [unit] | ASP | BAU | TP |
|---|---|---|---|
| Precipitation [mm/yr] | $983 \pm 53$ | | |
| *PPFD* [µmol m$^{-2}$ s$^{-1}$] | $643 \pm 18$ | $707 \pm 21$ | $620 \pm 6$ |
| $T_{air}$ [°C] | $13.0 \pm 0.14$ | $13.1 \pm 0.08$ | $13.8 \pm 0.4$ |
| *VPD* [kPa] | $0.93 \pm 0.1$ | $0.93 \pm 0.02$ | $0.35 \pm 0.003$ |
| *SWC* at 12.5 cm [m$^3$/m$^3$] | $0.33 \pm 0.01$ | $0.35 \pm 0.01$ | $0.37 \pm 0.01$ |
| $T_{soil}$ at 12.5 cm [C] | $13.3 \pm 0.2$ | $13.9 \pm 0.2$ | $12.9 \pm 0.1$ |

carbon fluxes during the growing season across a variety of soil moisture conditions, measured at 12.5 cm depth. A repeated measures ANOVA determines that there are significant differences between bin mean values within each site, for each of *ET, NEE, GPP,* and $R_{eco}$. Generally, the fluxes are lowest at both low ($<15\%$) and high ($>35\%$) soil moisture and peak between 20-25% *SWC*. Water and carbon fluxes remain relatively constant for *SWC* values between 12-40% at the TP site, suggesting that it is the least sensitive to changes in soil moisture. This is observed as decreases relative to the peak value at each site when soil water is either too low or too high for the crops to grow. We test the hypothesis that the BAU site is least resilient to changes in soil moisture by quantifying the range of soil moisture values where fluxes remain within 25% of the maximum flux. The BAU site has *ET* and the three carbon fluxes above this threshold for a 5.1%, and 8.1% range of soil moisture, respectively. At the TP site, *ET* and all three carbon fluxes are above that threshold for an 18% range of soil moisture and at the ASP site *ET* is above the
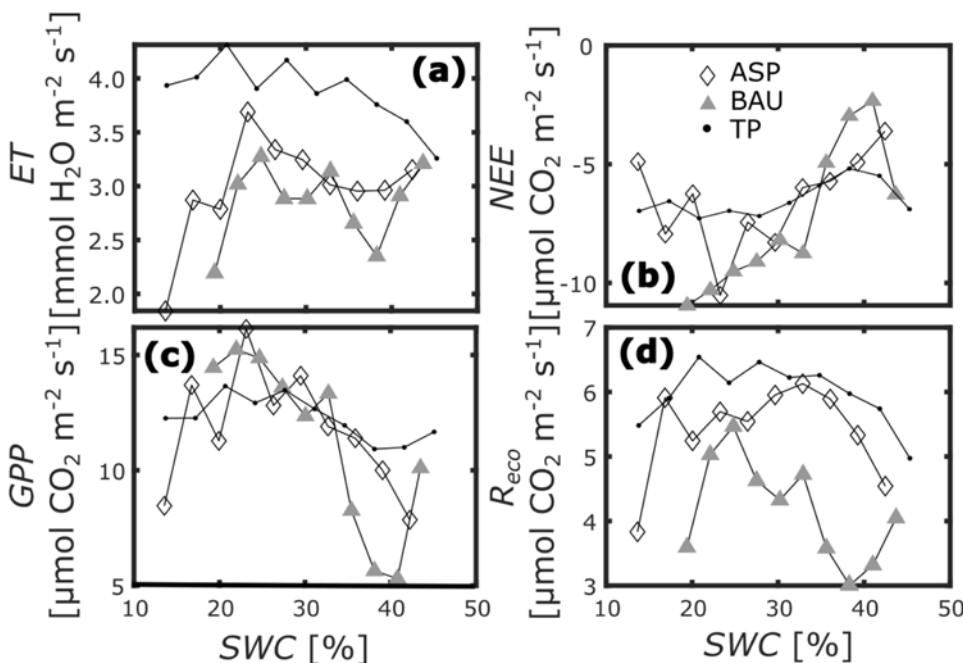


**Fig. 5.** Soil water content (*SWC*, at 12.5 cm) impacts on measured water and CO$_2$ fluxes; (a) evapotranspiration (*ET*), (b) net ecosystem exchange (*NEE*), (c) gross primary productivity (*GPP*), and (d) ecosystem respiration ($R_{eco}$). Data from the growing season only is binned by soil water content (*SWC*). Standard error bars are not presented, but across the three sites the standard error for any bin does not exceed 0.0059 mmol H$_2$O m$^{-2}$ s$^{-1}$ for *ET*, 0.0194 µmol CO$_2$ m$^{-2}$ s$^{-1}$ for *NEE*, 0.0234 µmol CO$_2$ m$^{-2}$ s$^{-1}$ for *GPP*, and 0.0076 µmol CO$_2$ m$^{-2}$ s$^{-1}$ for $R_{eco}$.

threshold for a 16.5% range in soil moisture and *NEE, GPP*, and $R_{eco}$ are above the threshold for ranges of soil moisture of 9.9%, 13.2%, and 19.8%, respectively. Similar results are obtained when data from the full year is analyzed; the TP site has the largest range of resiliency and the BAU site has the smallest (data not shown).

Photosynthesis is controlled by light; therefore, we present light responses in Fig. 6. The response of *ET* to different values of *PPFD* is similar, and significantly different according to a repeated measures ANOVA, across the three sites (Fig. 6a), though differences exist. For values of *PPFD* > 1,800 µmol photon $m^{-2}$ $s^{-1}$, the *ET* decreases at all three sites. The ASP site has higher *ET* than the BAU site when *PPFD* is between 200 – 1,600 µmol photon $m^{-2}$ $s^{-1}$. The differences in *NEE* to *PPFD* are smaller between the three sites, however, and differences in the carbon cycle are primarily expressed in the components *GPP* and $R_{eco}$ (Fig. 6). Like the ET, the repeated measures ANOVA finds significant differences in carbon flux for changes in *PPFD* at all three sites, and the ASP site has larger carbon uptake than the BAU site for values of *PPFD* between 200 - 1,600 µmol photon $m^{-2}$ $s^{-1}$ (Fig. 6c). When *PPFD* > 1,600 µmol photon $m^{-2}$ $s^{-1}$, however, the two agricultural sites have similar values of *GPP*. The $R_{eco}$ is lowest at the BAU site across all values of *PPFD*, which makes the total carbon uptake of the BAU site higher than the ASP site when *PPFD* > 1,600 µmol photon $m^{-2}$ $s^{-1}$. It should be noted that the bin for *PPFD* between 1,800 – 2,000 µmol $m^{-2}$ $s^{-1}$ has considerably fewer data points than other bins and results there may be less reliable.

Two important environmental controls on water and carbon fluxes are *TA* and *VPD*. In Fig. 7, we present water and carbon fluxes with changes in *TA* or *VPD*, using data from the full year. At all three sites, the repeated measures ANOVA reveals that there are significant differences between bin means for all fluxes and both *VPD* and *TA*. At all three sites, increases in air temperature result in increasing *GPP* and *ET*, and increasingly negative *NEE*, but when *TA* is greater than 33° C, fluxes decline at the ASP and TP sites (Fig. 7). The TP site has the highest *ET* of the three sites only when *TA* is greater than 20° C, highlighting that its higher annual average *ET* is a function of higher *ET* during the warm season. When *TA* is below 0° C, all sites have negligible carbon fluxes and *ET* is less than 1 mmol $H_2O$ $m^{-2}$ $s^{-1}$. Between temperatures of 0 and 20° C, carbon uptake, and to a lesser degree *ET*, is higher at the ASP site than the BAU site. This higher carbon uptake is in part because these temperatures are common during the spring and fall when the BAU field

is fallow. The *VPD* has a more complex relationship with water and carbon fluxes. When *VPD* is lowest, there is not sufficient atmospheric demand for water and both water and carbon fluxes are minimal. Flux magnitudes increase at all three sites as *VPD* increases to ~1.75 kPa. The peak fluxes at the ASP site occur at a slightly higher VPD (1.61 kPa for *NEE* and *GPP*; 2.07 kPa for *ET*) than at the BAU site (1.54 kPa for *NEE, GPP*, and *ET*) while the peak fluxes are at the highest *VPD* for the TP site (1.98 kPa for *NEE* and *GPP*; 3.3 for *ET*). When *VPD* is greater than 2 kPa, the behavior of the three sites diverges. Both *GPP* and *ET* decrease at the BAU site. The *GPP* falls by greater than 50% from a high of 12.8 µmol $m^{-2}$ $s^{-1}$ when *VPD* is 1.54 kPa to approximately 6 µmol $m^{-2}$ $s^{-1}$ when VPD is greater than 3.3 kPa. The *GPP* at the ASP site, however, only drops by 30% from a high of 14.5 µmol $m^{-2}$ $s^{-1}$ to 10.0 µmol $m^{-2}$ $s^{-1}$ while the *GPP* at the TP site decreases by 22% from its peak. The response of *ET* to increases in *VPD* is similar to the patterns in *GPP* for the agricultural sites, but *ET* does not decrease at high *VPD* for the TP site. The patterns observed suggest that prairie is the most resilient to high *VPD* and *TA* as plants continue to exchange water and carbon dioxide with the atmosphere and that a perennial agricultural site (ASP site) mimics this resilience, but to a lesser degree.

To check that the differences between ASP and BAU are not simply due to observations occurring when cover crops are present and/or better environmental conditions at the ASP and TP sites, we plot the relations between *TA* and *VPD* and the fluxes during the growing season when optimal environmental conditions occur (Fig. 8). Fig. 5 demonstrates that fluxes are highest when soil moisture values are between 15 and 35%; therefore, we limit the analysis to periods with 15%< *SWC*<35% and high light availability, defined as incoming solar radiation > 300 W/$m^2$. Fig. 8 reveals that the patterns observed in Fig. 7 for the full year are reproduced during the growing season when there is sufficient light and soil water availability. The ASP system is more resilient to *TA* that is lower than the optimum values as evidenced by the higher *GPP* and *ET* than the BAU site when the *TA* is in the 5-20° C range. The TP site is the least sensitive to high values of *VPD*, but its *GPP* is reduced at values of *VPD* < 1.5 kPa (Fig. 8d). The BAU site exhibits sensitivity to *VPD* higher than ~1.5 kPa, after which the *GPP* and *ET* both decline. These values decline at the ASP site when *VPD* is greater than ~1.5 kPa as well, but the impact is smaller, which suggests that the ASP cropping system is more resilient to adverse environmental
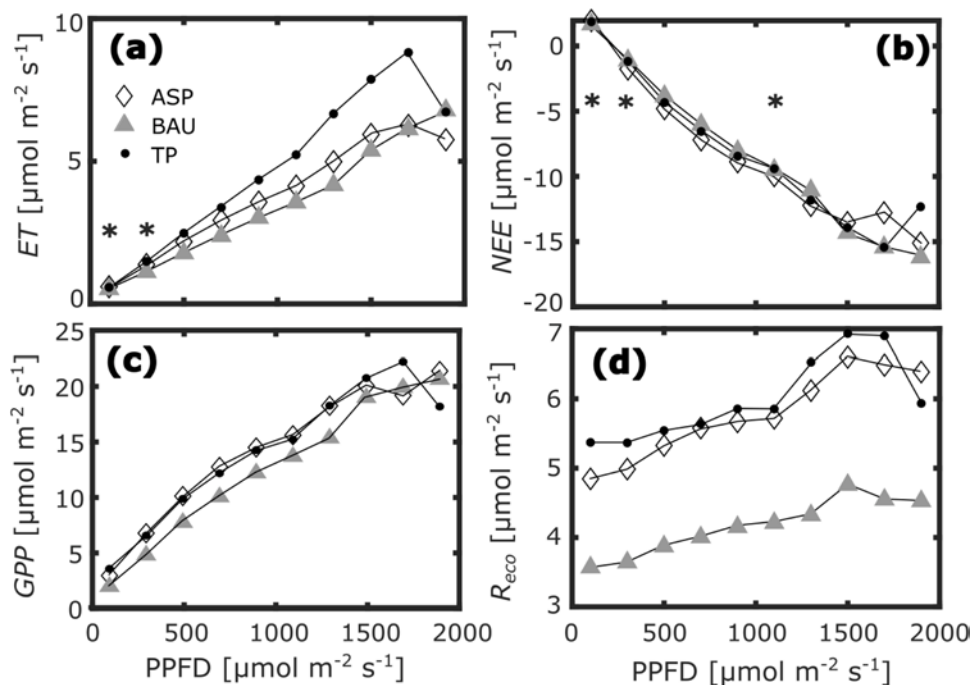


**Fig. 6.** Photosynthetic photon flux density (PPFD) controls on measured water and $CO_2$ fluxes; (a) evapotranspiration (*ET*), (b) net ecosystem exchange (*NEE*), (c) gross primary productivity (*GPP*), and (d) ecosystem respiration ($R_{eco}$). Only daytime ($R_g > 10$ W/$m^2$) data that has not been gap-filled is used. Data from the growing season only is binned by *PPFD*. Standard error bars are not presented, but across the three sites the standard error for any bin does not exceed 0.108 mmol $H_2O$ $m^{-2}$ $s^{-1}$ for *ET*, 0.249 µmol $CO_2$ $m^{-2}$ $s^{-1}$ for *NEE*, 0.306 µmol $CO_2$ $m^{-2}$ $s^{-1}$ for *GPP*, and 0.072 µmol $CO_2$ $m^{-2}$ $s^{-1}$ for $R_{eco}$. Asterisks denote bins where no significant difference between the three sites was observed.
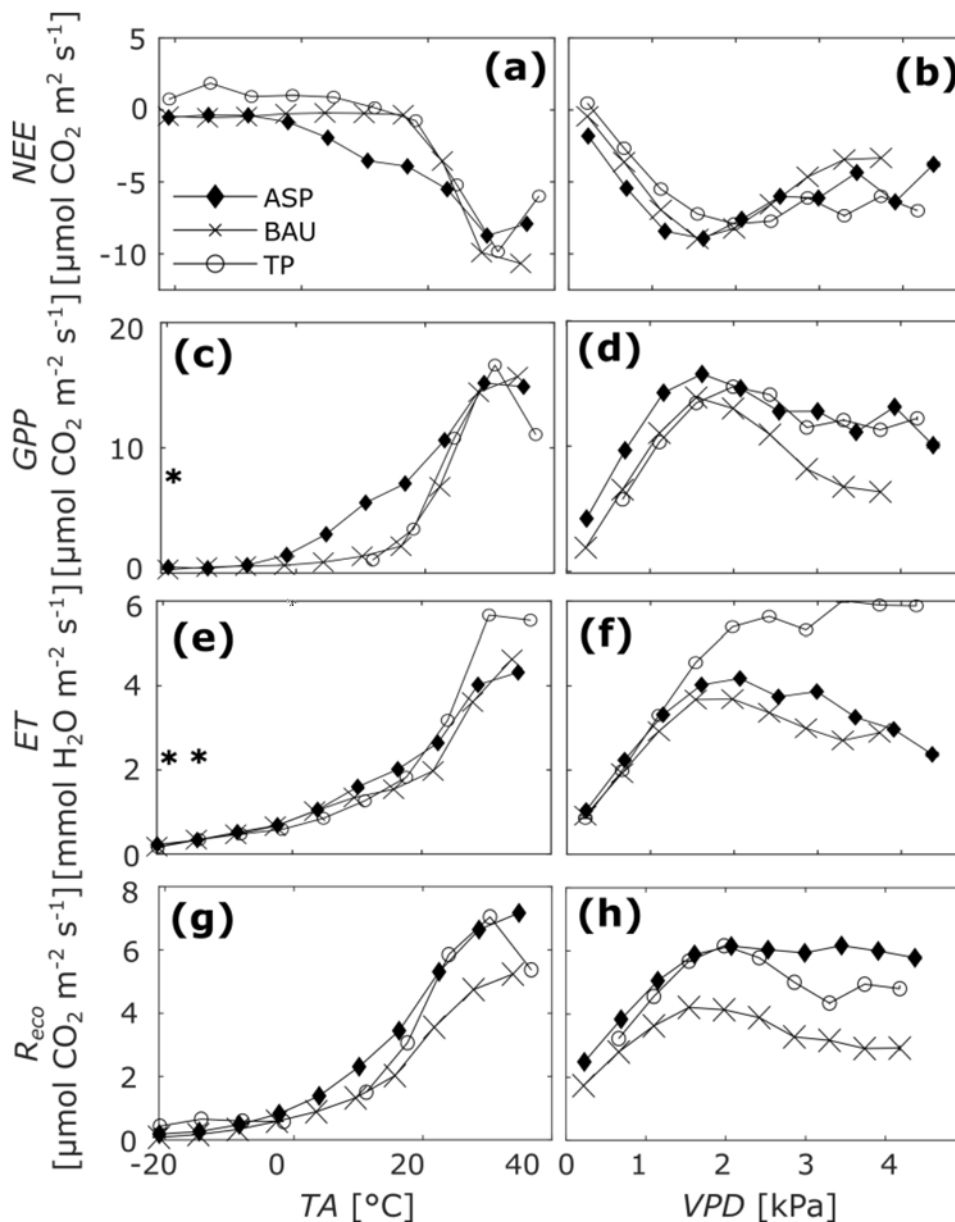
**Fig. 7.** Response of net ecosystem $CO_2$ exchange (*NEE*), gross primary production (*GPP*), soil respiration (*$R_{eco}$*) and evapotranspiration (*ET*) to air temperature (*TA*) and vapor pressure deficit (*VPD*). Daytime *NEE, GPP,* and *ET* were aggregated in classes of increasing air temperature or *VPD* for the period with data available at each site. Data from the full year is binned by *TA* and *VPD*. Standard error bars are not presented, but across the three sites the standard error for any bin does not exceed 0.161 mmol $H_2O$ $m^{-2}$ $s^{-1}$ for *ET*, 0.333 µmol $CO_2$ $m^{-2}$ $s^{-1}$ for *NEE*, 0.39 µmol $CO_2$ $m^{-2}$ $s^{-1}$ for *GPP*, and 0.109 µmol $CO_2$ $m^{-2}$ $s^{-1}$ for *$R_{eco}$*. Asterisks denote bins where no significant difference between the three sites was observed.

conditions even during the growing season when sufficient light and soil water are available.

### 3.3. Interannual variability in fluxes is affected by management

To assess the degree to which land management practices affect interannual variability in water and carbon fluxes we present metrics demonstrating the variability in annual fluxes in Fig. 9. We present data from 2018 – 2021, when data are available at all three sites simultaneously, so that we do not bias the results. The precipitation (*P*) has a range of 432 mm/yr during the study period, and the C.V. is 0.19. Irrespective of the metric (range, standard deviation, or C.V.), variability in *P* is larger than variability in *ET* at each of the three sites. The lower variability in *ET* may be caused by energy limitations to the *ET,* or it may be because the runoff-prone soils have higher proportions of *P* go to runoff during wetter than average years. Of the three sites, the ASP site has the highest interannual variability in *ET*. This is unsurprising because the ASP site has a wider variety of crops than the BAU site. The TP site has the lowest interannual variability in *ET* of the three sites (C.

V. = 0.05). In contrast, the TP site has higher C.V. for *NEE* than either of the agricultural sites and a slightly higher C.V. for *GPP* than the BAU site (Fig. 9b). The ASP site has the highest range for all of the carbon fluxes, which is a result of the higher annual values of carbon fluxes at the site (Fig. 2). Additionally, the high C.V. of *NEE* at the TP site is misleading because the mean value is close to 0. The interannual variability of $R_{eco}$, however, is lowest at the TP site and highest at the BAU site. It is not surprising that the BAU site has high variability in $R_{eco}$ because following years when maize was planted there is a large amount of additional crop residue left on the field compared to years when soybeans are planted, leading to interannual variability. Additionally, the ecosystem water use efficiency (EWUE) is most variable at the BAU site. The C.V. of EWUE is 0.19, 0.10, and 0.07 at the BAU, TP, and ASP sites, respectively.
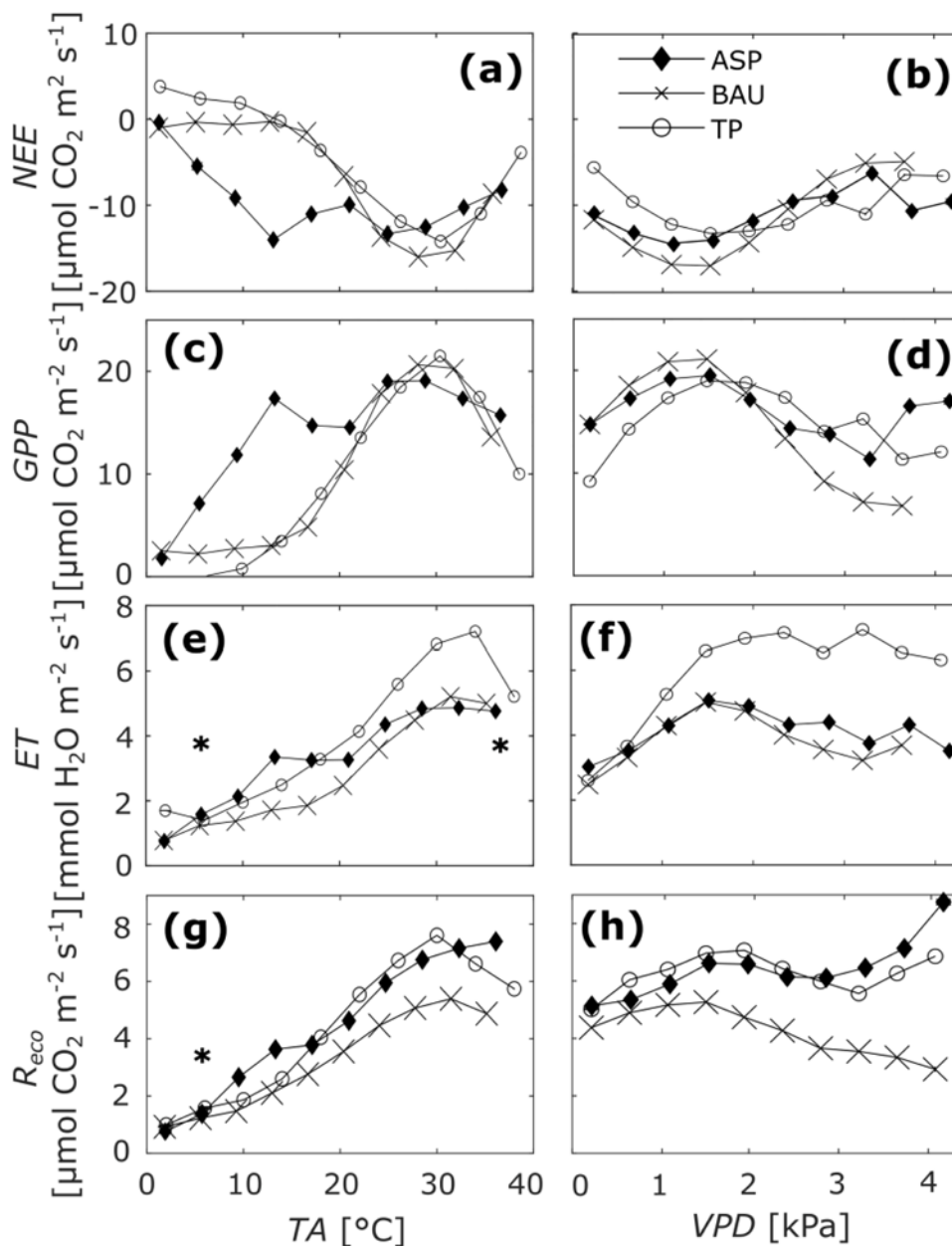
**Fig. 8.** Response of net ecosystem $CO_2$ exchange (*NEE*), gross primary production (*GPP*), soil respiration (*$R_{eco}$*), and evapotranspiration (*ET*) to air temperature (*TA*) and vapor pressure deficit (*VPD*). Values of *NEE, GPP,* and *ET* during the growing season with optimum growing conditions ($R_g$ > 300 W/m$^2$ and 0.15<*SWC*<0.35) were aggregated in classes of increasing *TA* or *VPD* for the period with data available at each site. Standard error bars are not presented, but across the three sites the standard error for any bin does not exceed 0.091 μmol $H_2O$ m$^{-2}$ s$^{-1}$ for *ET*, 0.163 μmol $CO_2$ m$^{-2}$ s$^{-1}$ for *NEE*, 0.397 μmol $CO_2$ m$^{-2}$ s$^{-1}$ for *GPP*, and 0.23 μmol $CO_2$ m$^{-2}$ s$^{-1}$ for $R_{eco}$. Asterisks denote bins where no significant difference between the three sites was observed.

## 4. Discussion

### 4.1. Water and carbon fluxes from important Corn Belt land use types

Despite the poorly drained soils in the claypan region, the carbon budget observed at the three study sites has similar values to other Midwestern sites with maize, soybean, and/or prairie (Chen et al., 2018; Dold et al., 2017; Gilmanov et al., 2013, 2014; Wagle et al., 2015). In this study we measure fluxes from cropping systems, which makes direct comparison to previous studies that have focused on carbon fluxes of specific crop types uninformative. Other studies have found that *GPP* for maize ranges from 731 to 1,525 g C m$^{-2}$ yr$^{-1}$ (Gilmanov et al., 2013) and 1,305, 630, and 916 g C m$^{-2}$ yr$^{-1}$ for a maize, soybean, and prairie site, respectively (Dold et al., 2017). So, generally, the maize/soybean rotations at the sites observed here have higher magnitudes of annual *GPP* and *NEE* than other sites in the U.S. Corn Belt as maize is planted relatively infrequently in the cropping systems in this study (Dold et al., 2017; Gilmanov et al., 2013). All the values discussed do not account for

carbon removal as crop yield, erosion, or prairie burning. In contrast, the annual *ET* at the sites in this study is similar to other sites in the U.S. Corn Belt, where values range from approximately 550 – 850 mm yr$^{-1}$ (Suyker and Verma, 2009; Zeri et al., 2013). As the spatial variations in carbon fluxes in the U.S. Corn Belt are largely controlled by temperature and precipitation (Hernandez-Ramirez et al., 2011), the relatively warm temperatures in Missouri allow for higher *NEE* despite the presence of a claypan layer and low values of 15 measured soil health indicators (Veum et al., 2015). Due to the long-term nature of the LTAR project, this study allows a comparison of two different cropping systems, rather than a simpler comparison of different crop types. The incorporation of long-term research sites from LTAR into the Ameriflux network will facilitate further studies regarding the impact of agricultural land use on water resources in the U.S. Corn Belt.

Of the three study sites, the TP site had the highest annual average *ET* (730 mm/yr), which was significantly higher than the BAU site (641 mm/yr), but not distinguishable from the ASP site (684 mm/yr). Thus hypothesis 1 is partially substantiated, the ASP site had the highest
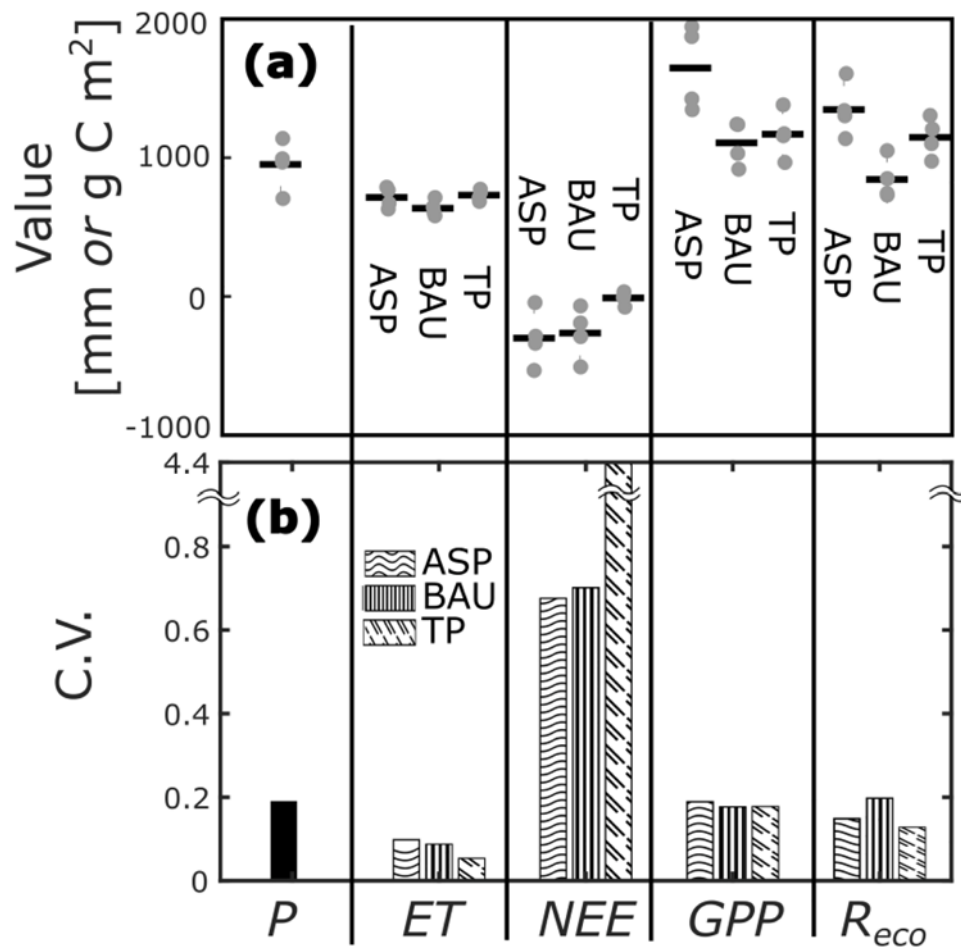
**Fig. 9.** Interannual variability in precipitation (*P*; mm/yr), evapotranspiration (*ET*; mm/yr), net ecosystem exchange (*NEE*; gC m$^{-2}$ yr$^{-1}$), gross primary productivity (*GPP*; gC m$^{-2}$ yr$^{-1}$), and ecosystem respiration (*R$_{eco}$*; gC m$^{-2}$ yr$^{-1}$). Metrics provided are (a) Annual values (grey dots) and mean values (black line) and (b) the coefficient of variation (C.V.). Data from 2018-2021 is presented.

magnitude of carbon fluxes, but not the highest *ET*. The three sites in our study represent major land use classes in the Midwest U.S.; the BAU site represents current land use, while the ASP represents a potential future cropping system. Prairie restoration is a commonly proposed conservation technique, and the TP site allows us to quantify how these sites might behave after they are restored. The sites have varied amounts of perennialization where the TP site is perennial, the ASP site has perennial cover but not perennial plants, and the BAU site is not perennial. The ASP site always has vegetation cover, but it is killed or harvested and reseeded twice per year, so unlike the TP site, the ASP site does not have mature plants at all times of the year. Previous research shows that perennial vegetation does not always increase *ET* (Abraha et al., 2015). The *ET* during the non-growing season is similar at the TP and BAU sites, even though the BAU site is fallow; the higher annual *ET* at the TP site occurs during the growing season. This difference suggests that the higher *ET* at the TP site is not simply due to vegetation cover during the non-growing season. Soil infiltration rates vary between the three sites with the highest infiltration at the TP site and the lowest at the BAU site, resulting in the most water available for *ET* at the TP site and the least at the BAU site (Baffaut et al., 2020). Between the two agricultural sites, the difference in *ET* is significant during the non-growing season but not during the growing season, suggesting that the cover crops play a role in increasing the annual *ET* at the ASP site. Research on the impact of cover crops on *ET* and soil water storage in the Midwest is limited (Sharma et al., 2017). The increase in *ET$_{NGS}$* at the ASP site suggests that within agricultural lands, adding cover crops to a traditional cropping system should increase *ET*, which is consistent with

remote sensing results (Hankerson et al., 2012). This further suggests that if the ASP practices become widely adopted, there may be increased *ET* at the regional scale, which can further alter regional climate as water vapor is a potent greenhouse gas (Alter et al., 2018). More data is required to quantify the impact of aspirational cropping systems that include cover crops on annual *ET*.

Calls to store carbon in agricultural soils to mitigate climate change have resulted in increased research interest in the ability of agricultural soils to store carbon (Chenu et al., 2019; Dold et al., 2021). Both agricultural fields have carbon uptake of approximately 300 g C m$^{-2}$ yr$^{-1}$ and there is not a significant difference between the two. In this study, we do not account for carbon that is removed as grain yield, so average carbon uptake does not necessarily mean that the soils are storing more carbon at each site. As a result, further research is required to understand carbon storage in agricultural fields. The TP site, however, is on average, net carbon neutral and not significantly different from 0 (*NEE* = -11 g C m$^{-2}$ yr$^{-1}$). It does not accumulate or release carbon from the soils on a long-term basis because the *GPP* is consumed by the *R$_{eco}$*. This demonstrates that even if carbon storage practices are implemented in agricultural fields, there is a limit to the amount of carbon that ecosystems can store in the soil. Additionally, the TP site has never been plowed. Tillage alters soil properties, particularly the bulk density, in ways that may take decades to recover if conservation practices are enacted (Bugeja and Castellano, 2018; O'Brien and Jastrow, 2013). This makes the site valuable as a true end member of what conservation practices are capable of. Early on, a reconstructed prairie might present a misleading picture of the effect of conservation practices. Finally,

increased carbon uptake by ecosystems does not necessarily lead to increased carbon storage in soils. The response of heterotrophic respiration to rising global temperatures can negate long term gains of soil carbon (Naidu and Bagchi, 2021).

Unlike the *ET*, the ASP site has the highest carbon fluxes of the three sites. The *GPP* at the ASP site averaged 1,566 g C m$^{-2}$ yr$^{-1}$, approximately 300 g C m$^{-2}$ yr$^{-1}$ higher than at the BAU site, and approximately 396 g C m$^{-2}$ yr$^{-1}$ higher than the TP site. It should be noted that we do not directly measure *GPP*, we estimate it using common empirical approaches. Like previous studies, we find that a no-till cropping system had more carbon uptake than $R_{eco}$ (Chi et al., 2016; Gebremedhin et al., 2012). In this study, the BAU field had lower *GPP* and $R_{eco}$ than the ASP site, which is in contrast with Baker and Griffis (2005). This lower plant uptake of carbon is likely because the cropping system at the BAU field in this study uses two years of soybean and one year of maize, and soybean has lower values of *GPP* (and therefore less crop residue available for $R_{eco}$) than maize.

Uncertainty in our measurements is difficult to precisely quantify because secondary techniques to measure field-scale water and carbon fluxes over a long term do not exist. We estimate uncertainty in the eddy covariance method by checking the energy budget closure, results of which are presented in Table 3. Compared to global networks of flux measurements, the energy balance closure at the study sites is slightly below average (Reed et al., 2018; Wilson et al., 2002). This is somewhat surprising considering the sites are flat, homogeneous fields ideal for EC measurements. We do not measure all of the energy storage, such as canopy storage, which reduces the energy budget closure. We attribute a portion of the uncertainty to measurements of ground heat flux. Ground heat flux plates were installed at 5 cm depth, which in a humid region is shallow enough that wetting fronts frequently hit the plate and disrupt measurements. Additionally, because gaps in measurements of turbulent fluxes typically occur at night when there is not solar radiation, we use gap-filled values of *LE* and *H* to calculate the energy budget closure. The gap filling process adds further uncertainty to the measured turbulent fluxes. At the BAU site, both *GPP* and $R_{eco}$ derived from the daytime partitioning method are within 2% of the values derived with the nighttime partitioning method. At the ASP site, the daytime partitioning method results in a 24% decrease in $R_{eco}$ and a 9% decrease in *GPP*, relative to the nighttime partitioning method. The daytime partitioning method does not force $GPP + R_{eco} = NEE$ and at the ASP site $GPP - R_{eco}$ from the daytime partitioning method is -461 g C m$^{-2}$ yr$^{-1}$, while the observed *NEE* is -305 g C m$^{-2}$ yr$^{-1}$. As a result, we believe that the nighttime partitioning method is more accurate. At the TP site the difference between the daytime and nighttime partitioning methods is negligible. Finally, we assess the uncertainty in *NEE* caused by random errors with the daily differencing approach. The mean annual uncertainty in *NEE* is 25, 13, and 23 g C m$^{-2}$ yr$^{-1}$ for the ASP, BAU, and TP sites, respectively. The uncertainty in measurements is not accounted for when we perform statistical tests to test for differences in annual mean values of fluxes between the sites.

### 4.2. Resilience of land use classes to changes in environmental conditions

Our findings show that the BAU site is most affected by changes to environmental conditions including soil water availability and atmospheric demand for water while the TP site is the least affected. These findings suggest a continuum of sensitivity of fluxes to environmental conditions with the diversity of plant types within the (agro-)ecosystem, which affirms our hypothesis 3. The TP site has a highly diverse plant community comprised of over 100 species present (Kucera, 1956), and is perennial with continuous ground cover. The ASP site also has continuous ground cover and a more diverse ecosystem than the BAU site. As a result, vegetation at the ASP site continues to transpire and uptake carbon at a higher rate than the BAU site when the *VPD* is greater than ~2 kPa, when *TA* is 5-20 °C, when *PPFD* is 200 – 1,600 μmol m$^{-2}$ s$^{-1}$, and when *SWC* > 30%. Additionally, the annual values of EWUE are

most stable (i.e., least variable) at the ASP site and least stable at the BAU site. By mimicking some of the behavior of the native prairie, the ASP site is more resilient to changes in the environment. A conventional agricultural system relies on the summer months to grow maize and soybean crops. As the climate warms and precipitation patterns shift, summer droughts become more likely and maize and soybean crops are more susceptible to failure due to water and heat stress (Chen and Ford, 2022). By incorporating wheat and hay crops, the ASP system uses a variety of seasons to grow crops and utilizes the wetter spring conditions that hinder a BAU system.

Some of this improved resilience is likely due to improved soil conditions in perennial systems. Higher levels of soil organic matter have been reported under no-till management with cover crops (Nunes et al., 2020; Sainju et al., 2002). Additionally, the 'effective storage capacity' of soil water can be conceptualized as the storage capacity of water within the root zone (Baker et al., 2012). Maize and soybean crops both have relatively short growing seasons, and their rooting depths do not reach maximum levels until late in their growing season (Kirkham et al., 1998). Perennial vegetation cover, especially at the prairie site, therefore, increases the effective storage capacity of soils by increasing the average rooting depth throughout the year, which allows vegetation to better utilize soil water for growth. A larger effective capacity also makes an (agro-)ecosystem more resilient to short drought periods. The finding that the ASP system is more resilient to changes in environmental conditions is primarily based on binned average values of fluxes presented in Figs. 5–8. The low standard error values in all the plots highlight the statistical precision of the binned mean values. Additionally, statistically significant differences between bin mean values (p < 0.05) of different treatments support the conclusions we have drawn. We note, however, that the uncertainty estimates that we have presented throughout this manuscript are calculated using the random error of measurements. This does not represent a full accounting of the uncertainty, which is not fully understood for eddy covariance measurements. Sources of error arising from instrument bias and energy balance closure are not included, other than presenting the calculated energy balance closure.

### 4.3. Land management impacts interannual variability in carbon and water fluxes

The interannual variability in *ET* is highest at the ASP site and lowest at the TP site, providing evidence that partially supports hypothesis 2. This result is intuitive because, while the biodiversity is clearly highest at the TP site, the species composition is consistent year after year. Whereas at the agricultural sites, crop rotation is performed at an annual scale and results in higher interannual variability in *ET*. Because the BAU site crop rotation is only maize and soybean, while the ASP site includes wheat and hay as well, the ASP site has more interannual variability in *ET*. There has been recent debate about whether climate or land cover changes are the dominant cause of hydrologic trends in the Midwest (Frans et al., 2013; Mishra et al., 2010; Xu et al., 2013). While we do not attempt to answer that question, our results suggest that land cover and land management might have a larger impact, relative to climate, than has been suggested (Sun et al., 2017). The interannual variability in *ET* at the TP site is roughly half that of either the ASP or BAU sites (C.V. is 0.053 at TP, 0.091 at BAU, and 0.105 at ASP). Therefore, the interannual variability induced by cropping practices is similar to the variability created by annual variations in weather conditions. This suggests that land use management has a larger impact on *ET* than climate variability. Drawing the conclusion that land use management has a larger impact on *ET* than climate variability assumes that the response of a prairie ecosystem to climate variability is representative of all ecosystems, which is not necessarily true.

The interannual variability of carbon fluxes do not follow a consistent pattern across the three study sites. The range and standard deviation of *NEE, GPP*, and $R_{eco}$ are all highest at the ASP site. This is not

surprising because the ASP site has the highest average values of those fluxes. When the variability is normalized by the average fluxes, the story is less clear. The C.V. of *NEE* is higher at the BAU site than the ASP site and highest at the TP site. This is because the mean *NEE* is close to 0 at the TP site, making the high C.V. somewhat misleading. However, the TP site has the highest amount of soil carbon of the three sites. As a result, it is likely that carbon fluxes, particularly respiration, at TP site respond primarily to environmental conditions that vary from year to year, while the agricultural sites respond to management practices more than climate variability. The C.V. of the *GPP* is higher at the TP site than the BAU site, though all sites have very similar C.V. of *GPP*. The high variability in *GPP* at the TP site is noteworthy because the TP site has the same vegetation every year, and the lowest interannual variability in *ET*. The TP site has the lowest ecosystem water use efficiency (*EWUE*), defined as *GPP/ET*, and because interannual variability in *ET* is low at the TP site, variability in *EWUE* results in the similar variability in *GPP* with the agricultural sites. That the *EWUE* is higher and less variable at the ASP site than the BAU site could be attributed to improved soil health and cover crops at the ASP site.

## 5. Conclusions

In this study, we deployed eddy covariance towers in three (agro-) ecosystems important to the Midwest U.S., a business-as-usual maize/ soybean cropping rotation (BAU), an aspirational maize/soybean/ wheat/hay cropping system with cover crops (ASP), and a native tall-grass prairie (TP). This provides a comparison of energy, water, and carbon fluxes from three land cover classes that are currently or projected to be important in the U.S. Corn Belt. The observations demonstrate that on average, the TP site has higher annual *ET* than the BAU site while the average *ET* at the ASP site is between the two, but not significantly different. The ASP site had the highest values of *GPP* and $R_{eco}$. Both agricultural sites were carbon sinks, but this does not account for carbon removed as grain yield.

We find that the TP site fluxes are most resilient to changes in environmental conditions, defined as a lower sensitivity to changes. Fluxes at the ASP site are less sensitive to changes in environmental conditions than at the BAU site. We attribute the higher resilience of carbon uptake and water use at the TP and ASP sites to a combination of perennial vegetation cover and improved soil health relative to the BAU site. The interannual variability in water fluxes is consistent with this framework. The TP site has the lowest interannual variability in *ET*, which can be attributed to both the lack of sensitivity to environmental conditions and the consistent vegetation at the site. By changing the vegetation cover, agriculture increases the interannual variability in *ET*. The ASP site, with the higher variety of crop types has the highest interannual variability in *ET*. The impact of agricultural management on carbon fluxes is less clear, however. The TP site has the highest coefficient of variation in *NEE* and similar variability in *GPP* and $R_{eco}$ is seen at all three sites. This variability is likely because the TP site is carbon neutral, making the annual carbon balance depend on the environmental conditions. Results from this study highlight the value of perennial vegetation cover in the Corn Belt. This study also offers insights into the carbon and water fluxes of different (agro-)ecosystems in the Midwest and how they react to changing environmental conditions.

## Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

## Data availability

Data is available from the AmeriFlux website.

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