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Influence of management and precipitation on carbon fluxes in great plains grasslands☆

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ABSTRACT

Suitable management and sufficient precipitation on grasslands can provide carbon sinks. The net carbon accumulation of a site from the atmosphere, modeled as the Net Ecosystem Productivity (NEP), is a useful means to gauge carbon balance. Previous research has developed methods to integrate flux tower data with satellite biophysical datasets to estimate NEP across large regions. A related method uses the Ecosystem Performance Anomaly (EPA) as a satellite-derived indicator of disturbance intensity (e.g., livestock stocking rate, fire, and insect damage). To better understand the interactions among management, climate, and carbon dynamics, we evaluated the relationship between EPA and NEP data at the 250 m scale for grasslands in the Central Great Plains, USA (ranging from semi-arid to mesic). We also used weekly estimates of NEP to evaluate the phenology of carbon dynamics, classified by EPA (i.e., by level of disturbance impact). Results show that the cumulative carbon balance over these grasslands from 2000 to 2008 was a weak net sink of 13.7 g C m^{-2} yr^{-1}. Overall, NEP increased with precipitation ($R^2 = 0.39$, $P < 0.05$) from west to east. Disturbance influenced NEP phenology; however, climate and biophysical conditions were usually more important. The NEP response to disturbance varies by ecoregion, and more generally by grassland type, where the shortgrass prairie NEP is most sensitive to disturbance, the mixed-grass prairie displays a moderate response, and tallgrass prairie is the least impacted by disturbance (as measured by EPA). Sustainable management practices in the tallgrass and mixed-grass prairie may potentially induce a period of average net carbon sink until a new equilibrium soil organic carbon is achieved. In the shortgrass prairie, management should be considered sustainable if carbon stocks are simply maintained. The consideration of site carbon balance adds to the already difficult task of managing grasslands appropriately to site conditions. Results clarify the seasonal and interannual dynamics of NEP, specifically the influence of disturbance and moisture availability.

1. Introduction

Grasslands can be a carbon sink with site-appropriate management (i.e., stocking rates) (Owensby et al., 2006; Svejcar et al., 2008; Zhu et al., 2011) and adequate precipitation. Carbon balance is often assessed with the Net Ecosystem Productivity (NEP). NEP is defined as CO_2 uptake by photosynthesis minus CO_2 lost by ecosystem respiration, and represents the net carbon accumulation over a given time interval (Randerson et al., 2002; Chapin et al., 2006). Positive NEP values represent a net carbon uptake by the ecosystem from the atmosphere, and negative values occur when ecosystems release carbon to the atmosphere (Zhang et al., 2010). NEP does not consider lateral flows of carbon (e.g., leaching loss and removal [culling] of herbivores) (Chapin et al., 2006). Carbon removed by grazing is mostly returned through excreta, with only a small fraction exported through the culling of livestock (Soussana et al., 2004). Harvesting biomass, grazing, and fires are additional pathways in which carbon could be removed from a site (Zhu et al., 2011).

Grasslands in the central and northern Great Plains tend to have a long-term cumulative carbon balance near zero (i.e., equilibrium) (Haferkamp and MacNeil, 2004; Owensby et al., 2006; Wylie et al., 2007; Zhang et al., 2010; Chimner and Welker, 2011) or slightly positive (i.e., sink) (Frank and Karn, 2003; Zhang et al., 2011). Root turnover in grassland soils constitutes the largest input to soil

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organic carbon (SOC), the below-ground C pool, due to the high concentration of lignin and polyphenols, which are resistant to degradation (Soussana et al., 2004). Decomposition of plant and animal residue and deposition by rhizomes also comprise major pathways of SOC deposition, and are directly related to NEP. Precipitation deficits and interannual variability often limit NEP, causing grasslands to become a net carbon source during drought years (Meyers, 2001; Zhang et al., 2010). Net ecosystem exchange in the northern Great Plains was reported to vary widely (spatio-temporally) from −537 to 610 g C m^{-2} yr^{-1} (Gilmanov et al., 2005).

Improved understanding of grassland carbon fluxes is needed to determine their potential role in mitigating (or contributing to) increasing atmospheric CO_2 concentrations and global change (Gilmanov et al., 2005) and the influence of management on ecosystem carbon cycling and equilibrium SOC (Owensby et al., 2006; Svejcar et al., 2008). Management (e.g., grazing intensity) can have a strong impact on grassland carbon balance (Soussana et al., 2004; Owensby et al., 2006; Svejcar et al., 2008; Zhu et al., 2011). Grazing affects ecosystem carbon flux by influencing species composition and the rate of carbon incorporation into soil, while grazing exclusion allows litter to accumulate and often reduces the rate of carbon assimilation (Milchunas and Lauenroth, 1993; Reeder and Schuman, 2002; Haferkamp and MacNeil, 2004). Though grazing often reduces a substantial portion of aboveground live biomass (Haferkamp and MacNeil, 2004; Derner et al., 2006), most grassland carbon is belowground (Schuman et al., 1999), thus NEP does not have a consistently strong correlation with aboveground live biomass. The effect of grazing intensity on carbon balance is non-linear and varies by ecosystem (Owensby et al., 2006).

Equilibrium SOC levels are only altered with management and environmental changes. Schuman et al. (2002), for example, estimated that implementing proper grazing management could increase carbon sinks on U.S. grasslands by 10–30 g C m^{-2} $year^{-1}$, for a period of time until a new equilibrium SOC is achieved. Gebhart et al. (1994) demonstrated that these changes can be rapid; with significant increases to SOC within 5 years of altered management practices (e.g., cropland to grassland conversion or change in livestock management practices), though decades may be required to reach a new equilibrium (Soussana et al., 2004). Minimal increases to SOC would be expected beyond this point (Schuman et al., 2002).

Land managers have the difficult decision of setting a stocking rate appropriate to site conditions (Schuman et al., 1999), which is even more difficult with the consideration of carbon fluxes. One tool available to assist rangeland managers in setting a sustainable stocking rate is the Ecosystem Performance Anomaly (EPA) (Wylie et al., 2008; Rigge et al., 2013). The EPA is calculated using ecosystem models and satellite data to describe the departure of annual biomass production from the level expected by annual weather conditions and site potential (driven by soils, aspect, slope, climate, etc.) at each pixel, a method that has been successfully applied to several ecosystems and validated by a variety of field data (Wylie et al., 2008; Rigge et al., 2013). By simultaneously monitoring the NEP and EPA of a site (i.e., measuring the carbon flux and proxy of disturbance intensity/stocking rate) we can more fully describe the complex connection between the two. Moreover, this technique may guide recommendations for improving the suitability of grassland management practices that enhance carbon sequestration in the ecoregions of the Great Plains.

1.1. Objectives

Our objectives were to assess the interactions among NEP (Zhang et al., 2010), EPA (Wylie et al., 2008), and precipitation for grasslands in the Greater Platte River Basin. This was accomplished using three techniques. First, we determined the quantity of precipitation needed to induce a carbon sink (i.e., NEP $\geq$ 0 g C m^{-2} yr^{-1}) at each pixel. Second, the temporal correlation between EPA and NEP was calculated at each pixel. Finally, we used weekly NEP data to develop average NEP phenologies for grassland types and ecoregions (Omernik, 1987), which were subsequently stratified by EPA class (i.e., level of grazing and/or disturbance impact), to better understand the connections between management, climate, and carbon dynamics across a range of environmental conditions.

2. Materials and methods

2.1. Study area

We evaluated grasslands, using the areas classified by the 2006 National Land Cover Dataset (NLCD) (Fry et al., 2011) as herbaceous and hay/pasture, in the Greater Platte River Basin. This Basin includes the Platte River watershed in addition to the ecologically similar Republican and Niobrara River watersheds in the central and northern Great Plains, USA (Fig. 1C). Omernik Level III ecoregions were used for comparison and reference in our analysis (Omernik, 1987). Our study area included the grassland portions of the High Plains, Nebraska Sand Hills, Northwestern Great Plains, Northwestern Glaciated Plains, Southwestern Tablelands, Central Great Plains, Western Corn Belt Plains, and Flint Hills ecoregions within the Greater Platte River Basin. Ecoregions ranged from 21% grassland cover in the Western Corn Belt Plains to 83% in the Northwest Great Plains, with a study area average of 58% grassland cover. The portions of the Southern Rockies and Wyoming Basin ecoregions that were present within the Greater Platte hydrologic basin were excluded from this study due to their ecological dissimilarity (i.e., dominance of woody species cover and/or mountainous topography).

Evaluated grassland ecosystems range from mesic tallgrass prairie in the east to semi-arid shortgrass prairie in the west. We defined the shortgrass prairie as the High Plains and Southwestern Tablelands ecoregions; mixed-grass as the Nebraska Sand Hills, Northwestern Great Plains, and Northwestern Glaciated Plains ecoregions; and tallgrass as the Central Great Plains, Western Corn Belt Plains, and Flint Hills ecoregions (Sims et al., 1978; Omernik, 1987; Table 1; Fig. 1C). Shortgrass, mixed-grass, and tallgrass types occupied 41%, 39%, and 20% of total grassland areas, respectively. We recognize that this grouping of ecoregions into grassland types does not exactly match the spatial distribution of the defined grassland types. For example, tallgrass prairie occurs in parts of the Nebraska Sand Hills ecoregion, and mixed-grass prairie occurs in parts of the Central Great Plains ecoregion. However, our defined grassland types correspond closely with those described by Johnson and Larson (2007).

The tallgrass prairie is dominated by big bluestem (*Andropogon gerardi* Vitman), switchgrass (*Panicum virgatum* L.), and Indiangrass (*Sorghastrum nutans* L.), with some areas invaded by Kentucky bluegrass (*Poa pratensis* L.) and smooth brome (*Bromus inermis* Leyss). The mixed-grass prairie includes sand bluestem (*Andropogon hallii* Hack.), switchgrass, porcupine grass (*Hesperostipa spartea* Trin.), western wheatgrass (*Pascopyrum smithii* Rybd.), and junegrass (*Koeleria pyramidata* Lam). Finally, the shortgrass prairie is dominated by blue grama (*Bouteloua gracilis* Kunth) and buffalo grass (*Bouteloua dactyloides* Nutt.) (Sims et al., 1978; Zhang et al., 2010). Total biomass production, precipitation, and the percentage of total biomass production contributed by C_4 plants all increase from northwest to southeast across the study area (Sims et al., 1978; Tieszen et al., 1997). Average precipitation, based on 30-year average Parameter-Elevation Regressions on Independent Slopes Model (PRISM) weather data (PRISM Climate Group, Oregon State University, http://www.prismclimate.org) ranged from approximately 200 mm in the west to 600 mm in the east.