

# Livestock Management Strategy Affects Net Ecosystem Carbon Balance of Subhumid Pasture

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## Abstract

Temperate grasslands are generally considered carbon (C) sinks, but climate and management likely affect whether they accumulate or lose C on an annual time step. The North Central Region of the United States contains highly productive improved pasture that is used exclusively for livestock grazing and mechanical harvest. The objective of this study was to use a net ecosystem carbon balance (NECB) approach to estimate C accumulation or loss in subhumid pastures under four typical livestock management practices: management-intensive rotational grazing (MIRG), continuous grazing (CONT), haymaking (HARV), and land set aside with no harvests (NONE). MIRG lost significantly less C in 2006 than all other treatments, and in 2007 MIRG was the only treatment that had a positive NECB. For 2006, our model resulted in an average change of  $-236 \pm 15$  (CONT),  $-100 \pm 24$  (MIRG),  $-391 \pm 11$  (HARV), and  $-276 \pm 28$  (NONE)  $\text{g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$ . For 2007, the change was  $-234 \pm 56$  (CONT),  $106 \pm 69$  (MIRG),  $-200 \pm 25$  (HARV), and  $-171 \pm 38$  (NONE)  $\text{g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$ . Increased C fixed as net primary production (NPP) and C imported as hay and grain resulted in the MIRG treatment having the most favorable C balance. Even with imported hay and grain, reduced NPP in the CONT treatment led to a less favorable C balance. In the HARV treatment, high biomass removal drove the negative C balance, while the relationship between reduced NPP and heterotrophic respiration alone drove the negative C balance in the NONE treatment. Climate change mitigation services provided from ecosystem C accumulation relative to cultivation may be warranted for pastures, but when all cross-boundary transfers of C are not considered, significant misconceptions can occur regarding how different management strategies affect the NECB of subhumid pasture.

**Key Words:** improved pasture, management-intensive rotational grazing, net ecosystem carbon balance, net primary production, soil respiration

## INTRODUCTION

Depending on definition, grasslands occupy 31% to 43% of the global land area (Gibson 2009), and store 28% to 37% of the terrestrial soil organic carbon (SOC) pool (Lal 2004). Improved management of grasslands is believed to increase their potential to accumulate carbon (C) (Follett et al. 2001; Follett and Reed 2010; Franzluebbers et al. 2012). Practices that increase forage production, such as fertilization, irrigation, sowing favorable grasses and forbs, intensive grazing management, and conversion from cultivation to well-managed pasture, provide the opportunity to sequester atmospheric C and enhance soil organic matter (SOM; Conant et al. 2001; Soussana et al. 2004; Franzluebbers et al. 2012). Conversion from cultivation to permanent pasture has been estimated to increase C storage by 50 to 100  $\text{g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  (Intergovernmental Panel on Climate Change 2007).

The North Central Region of the United States contains  $\sim 9$  M ha of pasture, and within this region Wisconsin contains just

less than 1 M ha of highly productive subhumid improved pasture used exclusively for livestock grazing and mechanical harvest of forage (Vough 1990). Taylor and Foltz (2006) estimated  $\sim 25\%$  of all dairy operations in Wisconsin maintained some form of grazing as a management strategy. In addition to dairy operations, 73% of Wisconsin's beef industry, which accounts for  $\sim 23\%$  of the state's total cattle population, also use grazing as a management tool (Taylor and Lehmkuhler 2008). While pasture management in this region historically focused on productivity and forage quality by sowing improved plant species and fertility management, increasingly they have been managed with short-duration, high-intensity grazing practices (Brock and Barham 2008). Management-intensive rotational grazing (MIRG) has the potential to enhance pasture SOC through increased plant production (Oates et al. 2011) and the rapid and homogeneous incorporation of plant-derived C in the form of manure (Follett and Reed 2010; Morgan et al. 2010).

In a study comparing intensive rotational grazing to extensive grazing and harvesting for hay in the southeastern United States, total soil C was 22% higher under the intensive treatment (Conant et al. 2003). In contrast, a study in Manitoba, Canada found no differences in soil C between rotationally grazed pastures and pastures grazed continuously regardless of stocking rate (Banerjee et al. 2000). Wienhold et al. (2001) found that soil C content ( $\text{g} \cdot \text{kg}^{-1}$ ) to 15-cm under heavy grazing in a North Dakota mixed-prairie pasture was comparable to an ungrazed enclosure, while moderate grazing reduced soil C. An update to this study by Liebig et al. (2006),

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while not including the ungrazed pasture, observed a similar pattern in SOC to 20-cm between the heavy and moderately grazed pastures.

While studies have considered prairie restoration, grazing effects on riparian systems, conversion of cultivated fields to pasture, and grazing comparison studies under other climatic conditions such as semi-arid rangelands, quantitative studies looking at C balance under intensive rotational grazing systems in subhumid improved pastures are lacking (Schnabel et al. 2001; Follett and Reed 2010). In this study we sought to estimate the net ecosystem C balance (NECB) in subhumid cool-season grass pastures under four typical livestock management practices of the North Central Region: MIRG, continuous grazing (CONT), haymaking (HARV), and land that had been set aside with no defoliation (NONE). We define NECB as the net rate of C accumulation in, or loss from, the ecosystem. But, as proposed by Chapin et al. (2006), in addition to estimating the difference between C fixation (net primary production [NPP]) and C loss (heterotrophic respiration), we included C imported into and exported from the ecosystem as a direct or indirect result of management. The pasture ecosystem was delineated by the plant canopy, lower boundary of the rooting zone, and the pasture boundary fences.

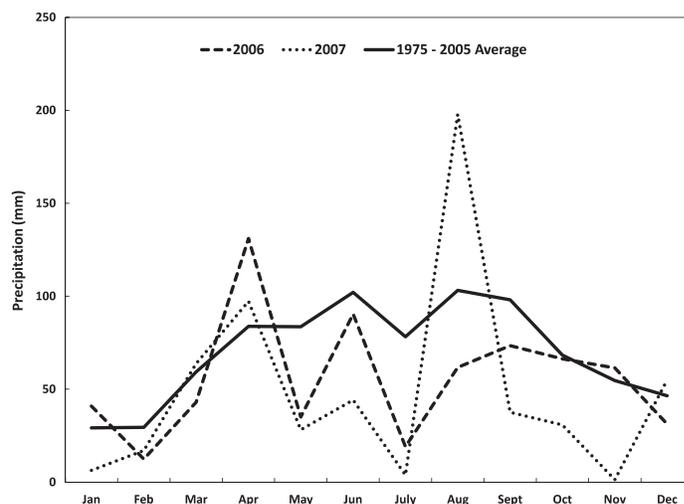
## METHODS

### Study Site

This study was conducted at the Franbrook Farm, a research property in south central Wisconsin, United States (lat 42°44'16.65"N, long 89°45'13.27"W). Elevation range at the farm is 265 to 320 m above sea level. The climate is continental with warm summers and cold winters. Average monthly temperatures range from a low of -7°C in January to a high of 22°C in July. For our study period, 2006 and 2007, temperature ranged from 0°C to 22°C and -5°C to 21°C, respectively. Mean annual precipitation is 900 mm, of which 100 mm is from snow. Approximately two-thirds of the annual precipitation falls during the growing season, April through October (Fig. 1). Yearly precipitation totals for 2006 and 2007 were 691 and 583 mm, respectively. The research area is ~26 ha of valley bottom pasture dominated by cool-season grasses (~75% of the pasture had a 0–2% slope, with the remaining area at 0–6%). Soils in this area are ~90 cm deep and are classified as Otter silt loam (Cumulic Endoaquolls), Arenzville silt loam (Typic Udifluvents), and Huntsville silt loam (Cumulic Hapludolls; Soil Survey Staff 2010). In the surface 15-cm of soil, bulk density was 1.15 mg·m<sup>-3</sup>, pH 6.8 (1:1 soil–water), organic carbon 42 g·kg<sup>-1</sup>, and total nitrogen 3.1 g·kg<sup>-1</sup>.

### Experimental Design

In April 2005, we established a randomized complete block design field experiment with four treatments within three blocks on existing pasture. The land, which had previously been cultivated with corn as the primary crop, was converted to pasture in 1999. Following conversion, and up until the treatments were established, the primary use was continuous grazing of livestock. The four treatment levels mimicked MIRG, CONT, periodic harvesting of pasture forages for hay



**Figure 1.** Average monthly precipitation for 2006 and 2007, and long-term average precipitation from 1975 through 2005.

(HARV), and active management removal (NONE). Beginning in May 2005, MIRG paddocks (0.6 ha) were grazed monthly (six grazing cycles to ~15 cm residual stubble height) by separate 25 cow–calf pair herds (1 pair = 1.3 animal units [AU]) at high animal stocking rates (i.e., instantaneous stocking rate of 54 AU·ha<sup>-1</sup>) for a brief duration of ~2 d (i.e., a stocking rate of 108 AU d·mo<sup>-1</sup>), and then allowed to rest for ~28 d. Continuous pastures (8.1 ha) had lower instantaneous stocking rates (i.e., 4 AU·ha<sup>-1</sup>), but pastures were rested only during the 2 d that the herd was confined to the MIRG paddocks, resulting in a comparable stocking rate to MIRG (i.e., a stocking rate of 112 AU d·mo<sup>-1</sup>). Plant biomass was mechanically harvested to ~6 cm residual stubble height and removed from the HARV plots (0.3 ha) to mimic the making of hay typical to confinement operations. The first harvest took place at boot stage in May 2006 and 2007. A second cutting took place in late July when plant biomass was 30 to 35 cm high. Finally, we set aside 0.3 ha plots for NONE to mimic a Conservation Reserve Program site. To represent standard management practice, granular ammonium phosphate (11-44-0) fertilizer was applied at the University of Wisconsin Extension recommended rate (57 kg N·ha<sup>-1</sup>) to all treatment areas except NONE in early June in 2005, 2006, and 2007.

### Net Primary Production

Starting in May 2006 and continuing through October 2007, aboveground net primary production (ANPP) was estimated monthly for both MIRG and CONT. In the HARV plots, biomass production was estimated for three growth periods: April through May, June to mid-July in 2006 and June through July in 2007, and from the end of the second growth period through October. Biomass in the NONE treatment was estimated for three growing periods: April through June, July through September, and September through October. Biomass was estimated using Leaf Area Index (LAI), which is defined as the amount of leaf area in a canopy per unit ground area. In 2005, at multiple time points throughout the season, LAI was calculated as a function of intercepted photosynthetically active

radiation (IPAR). IPAR was determined by measuring incoming PAR above and below the leaf canopy in randomly placed 0.1-m<sup>2</sup> quadrats with an AccuPAR LP-80 Ceptometer (Decagon Devices, Inc, Pullman, WA). Biomass was harvested from these quadrats, dried at 60°C for 48 h, and weighed. An allometric equation was developed from the relationship between LAI measured and biomass ( $r^2=0.72$ ). We calculated production in the MIRG and HARV paddocks as the difference between pregrazing/harvest event measurements and postgrazing/harvest measurements of the previous grazing/harvest event. For CONT pasture, five grazing exclusion cages were randomly located in each block and moved each month. Biomass was estimated inside and outside the cage, and monthly production was calculated as the difference in biomass estimated from inside the cage minus biomass estimated outside the cage.

Live and dead plant material fractions were calculated using a line-point transect at the beginning of the season in all treatments, and the middle and the end of the season in NONE treatment. Five 10-m transects were randomly located in each treatment, and point determinations of live or dead were made at each decimeter resulting in 100-point determinations in each treatment at each sampling date. Biomass estimates were adjusted by multiplying total biomass by percentage of live biomass to reflect incremental growth of live biomass for the given growth period.

Belowground net primary production (BNPP) was estimated in 2006 and 2007 from root in-growth cores (Fahey et al. 1999; Steingrobe et al. 2001). Five 5-cm diameter × 15-cm deep mesh cores containing a neutral soil medium (75% field soil and 25% sand) were installed within each treatment at the beginning of the growing season (April). The cores were harvested at the end of the season (October), and the roots were washed free of soil over a 1-mm sieve, dried at 60°C for 48 h, and weighed.

### Plant Community

Plant cover was estimated in late July (2005 through 2007) and late September (2006 and 2007). Five 10-m transects were randomly located in a quincunx pattern within each treatment plot. At 20 1-dm intervals along each transect, a sharpened point was lowered from above the vegetation and the first plant species intercepting the point was recorded (Heady et al. 1959). Absolute cover was calculated by dividing species intercepts by total intercepts. The cover estimates by species were categorized into the following functional groups: C3 cool-season grasses, perennial legumes, and nonleguminous forbs.

### Soil Moisture, Temperature, and Bulk Density

Soil samples were collected monthly from 28 April through 24 October 2006, and from 14 April through 25 October 2007. Five soil cores (2.5-cm diameter × 15-cm depth) were collected from within each plot. The samples were hand composited, placed into plastic bags, and immediately placed in coolers for transport to University of Wisconsin, Madison. Fifteen grams of soil were weighed, oven-dried for 24 h at 105°C, and then reweighed for gravimetric soil moisture determination. Gravimetric moisture content was converted to volumetric water content using soil bulk density from each treatment within block. Due to moisture sensor failure in 2006 and data logger failure in 2007, daily volumetric moisture contents were

interpolated between collection dates beginning in early April through late November or early December based on whether soil temperature was above 0°C. Daily soil moisture values were validated and corrected where necessary using local precipitation data and the change in soil moisture ( $\Delta SW$ ) equation adapted from Martin and Gilley (1993):

$$\Delta SW = P_e - ET_c, \quad [1]$$

where  $P_e$  is equal to effective precipitation and  $ET_c$  is the pasture evapotranspiration value for the calculated period. In April 2007, we estimated soil bulk density by carefully excavating 260.24-cm<sup>3</sup> (4.72-cm diameter × 15-cm depth) soil cores that were returned to the laboratory, dried at 105°C for 48 h, and weighed to determine mass of soil per unit volume soil (Elliot et al. 1999). Soil bulk density was  $1.16 \pm 0.08$ ,  $1.19 \pm 0.06$ ,  $1.14 \pm 0.07$ , and  $1.11 \pm 0.06$  (mean ± SE;  $n=3$ ) for CONT, MIRG, HARV, and NONE respectively. Soil temperature (15-cm depth) was measured monthly at the same time soil respiration measurements were made. Temperature was recorded between 1000 and 1600 hours using a temperature probe attached to a portable infrared gas analyzer (IRGA; Li-Cor Biosciences, Lincoln, NE) within 0.5-m of the soil respiration measurement collar. For the period of 1 January 2006 through 31 December 2007, daily maximum, minimum, and average soil temperatures (0 to 10-cm) for Arlington, Wisconsin, were downloaded from the Wisconsin–Minnesota (WI-MN) Cooperative Extension Agricultural Weather webpage.<sup>1</sup> Daily maximum temperatures from WI-MN were correlated with daily temperatures interpolated from monthly on-site measurements (slope and 95% confidence intervals: 1.02 [0.7, 1.85];  $r^2=0.89$ ) using the ordinary least-squares (OLS) bisector method (Isobe et al. 1990; see method explanation below). To represent soil temperature over a 24-h period, daily average plot temperatures were then calculated using the WI-MN daily average soil temperature, corrected to plot temperatures by difference between WI-MN daily maximum and the interpolated daily maximum.

### Soil Respiration

Soil respiration ( $R_s$ ) was measured using a Li-Cor 6400 portable CO<sub>2</sub> IRGA (Li-Cor Biosciences) equipped with a LI-6400-09  $R_s$  chamber (Norman et al. 1992, 1997). As with soil temperature, the  $R_s$  measurements were made monthly from the beginning of April through late November in both years. The number of measurements made required sampling over several hours. Forty-eight discreet measurements, four in each treatment plot, were measured as closely to mid-morning as possible but were always made between 1000 and 1600 hours. Sampling of blocks, and the order of measuring treatments within a block, were randomized to avoid confounding time-of-day variability with treatment effects (Davidson et al. 2002). Four circular thin-walled (3.2 mm) polyvinyl chloride collars (10-cm diameter × 5-cm height) were randomly distributed within each treatment plot and repositioned for each sample date. Plant shoots within soil collars were clipped prior to soil CO<sub>2</sub> efflux measurement, so  $R_s$  excludes shoot respiration. The measurement protocol produces four estimates of CO<sub>2</sub> flux

<sup>1</sup><http://www.soils.wisc.edu/wimnext/weather.html>

(each estimate an average of three CO<sub>2</sub> flux determinations) for each treatment plot per sample date.

To assemble annual C budgets, it was necessary to estimate daily  $R_s$  to gap fill the data when no direct measurements have been made (Norman et al. 1992; Cahill et al. 2009; Vargas et al. 2011). Two nonlinear models for predicting  $R_s$  were developed by Norman et al. (1992). They compared modeling  $R_s$  as: 1) a function of soil temperature ( $T$ ) and volumetric water content ( $\theta$ ), and 2) soil temperature, volumetric water content, and LAI. In our study, both models were found to be an acceptable predictor of  $R_s$ , although the inclusion of LAI slightly improved the predictive ability of the model (model 1,  $r^2=0.52$ ; model 2,  $r^2=0.53$ ). Mowing and grazing can significantly alter  $R_s$  by reducing the contribution of canopy photosynthesis to root respiration (Bremer and Ham 2002). In systems managed with grazing and haying, including LAI in models of  $R_s$  can improve the predictive accuracy of gap filling models (Bremer and Ham 2002). To estimate daily rates of flux, nonlinear least squares regression was used to model  $R_s$ . Daily rates were then summed to estimate annual  $R_s$ , the mass of C per unit area respired. Visual inspection of correlation plots of  $R_s$  against temperature showed that variability explained behaved quite differently by season. For dormant season flux (number of days when soil temperature was 0°C or below), 24 late-season observations of  $R_s$  were modeled as a function of  $T$  ( $r^2=0.77$ ) and the intercept was used to estimate flux (0.4 g C · m<sup>-2</sup> · d<sup>-1</sup>). Variability explained in observed data for spring ( $r^2=0.7$ ) and fall ( $r^2=0.87$ ) was strong, while for summer ( $r^2=0.27$ ) was weak. Splitting summer data by year improved variability explained (2006,  $r^2=0.48$ ; 2007,  $r^2=0.47$ ) and allowed us to build more precise predictive models. Separate models for spring (April and May) and fall (September, October, and November) were fit to 48 observations for 2006, and 2007. For summer (June, July, and August), models were fit to both 2006 and 2007 data separately (36 observations for each year). The exponential model for nondormant season  $R_s$  was as follows:

$$R_s = a + b \exp(cT + d\theta + eLAI). \quad [2]$$

To validate the predictive ability of each of the  $R_s$  models developed, the relationship of the slopes between observed  $R_s$  (oRs) and predicted  $R_s$  (pRs) were tested using the OLS bisector method (Isobe et al. 1990). An accurate model will result in a slope not significantly different from 1 when regressing oRs against pRs. Since there is uncertainty in both oRs and pRs, it is not clear whether oRs should be regressed on pRs, or vice versa. The OLS bisector method alternatively regresses oRs on pRs and pRs on oRs, which ascertains the slope of the ordinary least squares fit by bisecting the fit from the separate regressions. Confidence intervals (95%) were calculated by bootstrapping (Crawley 2002) the OLS bisector slope 500 times to determine whether it bounded the 1:1 line (Fig. S1 available online at <http://dx.doi.org/10.2111/REM-D-12-00151.s1>).

### Net Ecosystem Carbon Balance

NECB is defined as the net rate of C accumulation or loss from an explicitly defined ecosystem (Chapin et al. 2006). For this

study, the ecosystem was delineated by the plant canopy, the lower boundary of the rooting zone, and the pasture boundary fences. Verification of soil C change may require long-term experiments, while changes over short time periods can be measured over one or more growing seasons using the net balance of inputs and outputs. NECB measured annually for our pasture ecosystem is described by the equation:

$$NECB = NPP + I_h + I_g - R_b - E_h - E_{lb} - F_{ch4} - F_{co2} - DOC, \quad [3]$$

where NPP is equal to the sum of annual ANPP+BNPP and equals total litter inputs to the soil,  $I_h$  and  $I_g$  are C imported as hay and grain, respectively,  $R_b$  is heterotrophic respiration,  $E_h$  and  $E_{lb}$  are C exported as hay and livestock biomass, respectively,  $F_{ch4}$  and  $F_{co2}$  are C lost through enteric fermentation and livestock respiration, respectively, and DOC is soluble C loss. Grassland intra-annual root turnover is highly variable with reported values ranging from 0.25 to 1.51 (Gill and Jackson 2000). Root turnover was not measurable with root in-growth cores harvested at season end; therefore, NECB was calculated using root production estimates from harvested root ingrowth cores increased 53% to adjust for turnover (Gill and Jackson 2000). Total root profile was not captured with root ingrowth cores to 15-cm depth, so in addition to the fine root turnover correction we provide estimates of BNPP where 66% of root production was realized in the upper 15-cm (Jackson et al. 1996; Crush et al. 2005; Kucharik et al. 2006). Root C exudations represent a significant and important contribution to grassland BNPP. Estimates range from 5% to 21% of net fixed C in biomass (Kumar et al. 2006). To estimate root exudate C, we used the value presented in a 2009 review by Jones et al. where root exudate C is 11% of net fixed C in above- and belowground grassland biomass. The equation used to calculate BNPP was:

$$BNPP = Fr \times Dc \times \tau + C_e, \quad [4]$$

where  $Fr$  is live fine root biomass (estimated from root ingrowth cores),  $D_c$  is a correction for sampling depth,  $\tau$  is turnover rate, and  $C_e$  is root exudate C. Additional inputs to the ecosystem included import of hay ( $I_h$ ) and grain ( $I_g$ ) as supplemental feed. An accounting of hay and grain supplementation was kept on a daily basis and summed for the year. Fractional C contents of aboveground and belowground plant biomass for each treatment, and imported hay, were determined by flash combustion on finely ground subsamples of dried plant material (~10 mg) using a Flash EA 1112 CN Automatic Elemental Analyzer (Thermo Finnigan, Milan, Italy). The C fraction of grain was assumed to be 45% (Latshaw and Miller 1924; Ma and Dwyer 2001).

Carbon fixed in the ecosystem and lost as a result of heterotrophic respiration was estimated from soil respiration where  $R_b$  is fixed C loss and is soil CO<sub>2</sub> efflux (heterotrophic respiration [ $R_b$ ]+autotrophic respiration [ $R_a$ ]) -  $R_a$  (adapted from Hanson et al. 2000). Separating  $R_b$  and  $R_a$  for quantification is difficult, and investigation of methodologies is beyond the scope of this study. Results from grasslands have varied, with reported seasonal  $R_a$  ranging from 10% to 80% and annual  $R_a$  ranging from 40% to 60% (Hanson et al. 2000). We adopt a value near the middle of this range, 53%

from pasture as reported by Robertson et al. (1995). This value for  $R_a$  is similar to other findings for pasture and glasshouse experiments from cool-season grasses (Byrne and Kiely 2006; Chen et al. 2006), and from modeling approaches taken by Bond-Lamberty et al. (2004) and Raich and Schlesinger (1992). Our estimates of NECB also included C exported across the pasture boundary. In the HARV treatments, hay exports ( $E_b$ ) were estimated as ANPP produced over two cutting cycles. Hay was harvested for the period of April through mid-May, and mid-May through mid-August in both years. Carbon fraction was determined by near infrared reflectance spectroscopy. Carbon leaving the pasture as livestock biomass ( $E_{lb}$ ) was determined as calf body weight  $\times$  23% (Emsley 1989). Average weight of brood cows was the same when they were taken off as when they were turned out to pasture (Ralph Siegenthaler, Franbrook Farm manager, personal communication, February 4, 2008).

Following the Food and Agriculture Organization of the United Nations, and Johnson and Johnson (1995), C lost as  $CH_4$  was estimated as 6% of digestible energy (DE):

$$CH_4 - C(g\dot{s}d^{-1}) = DE \times 0.06, \quad [5]$$

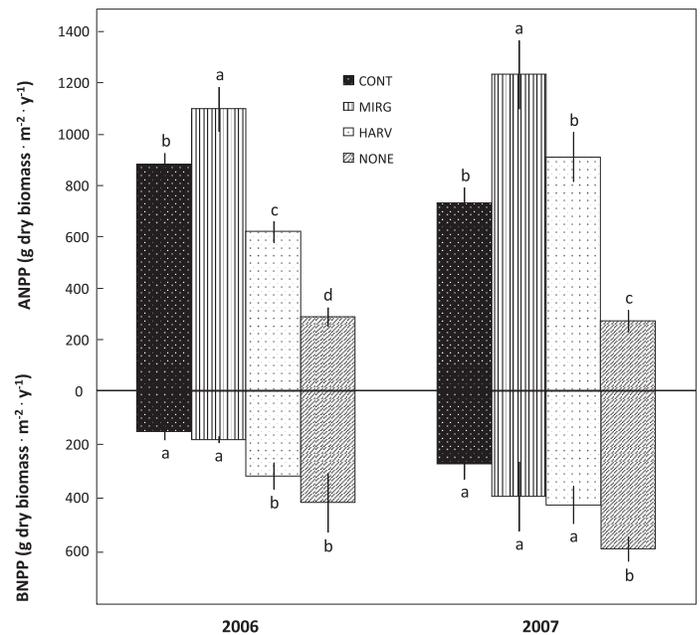
where DE is estimated as 6% of net energy intake, which in turn is estimated as 62.9% of dry matter intake (DMI). To estimate C lost through livestock respiration, we use the equation developed by Kirchgessner et al. (1991):

$$CO_2 - C(kg\dot{s}d^{-1}) = -1.4 + 0.42DMI + 0.045W, \quad [6]$$

where  $W$  is metabolic live weight ( $kg \text{ weight}^{0.75}$ ). Finally, we assume soluble C loss was negligible (Brye et al. 2002; Klumpp et al. 2007). Brye et al. (2002) found leached soluble C from a grassland site in Wisconsin accounted for only 0.1% of total C lost from the system. Also, a study by Klumpp et al. (2007) showed dissolved organic C loss from temperate pasture soil monoliths averaged  $5.5 \text{ g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$ , an amount that would have had no significant effect on the results of our C budget.

### Statistical Analysis

Means calculated from subsamples within each experimental unit (paddock) were used in analysis of variance (ANOVA) linear mixed-effects modeling using the maximum likelihood algorithm. All modeling was done using S-Plus 7.0 (Insightful Corp, Seattle, WA, 2010). Saturated models were constructed to analyze response variables as a function of management treatments, once we accounted for the random effect of block. To test the significance of the fixed effect, we dropped the treatment term and compared a model with only the intercept term to the model including the treatment term using likelihood ratios (Crawley 2002). If models were significantly different ( $P < 0.05$ ), the model with the lower Akaike's Information Criterion was retained. If not, we selected the simpler model. If treatment was significant, treatment levels were sequentially collapsed and subsequent models were compared with likelihood ratio tests using the same approach of model selection. If treatment was not significant, no further comparison of treatment levels was made. Separate model selection procedures were run for each year (2006 and 2007) to test for treatment effects on ANPP, BNPP, soil respiration, and NECB.



**Figure 2.** Biomass contribution ( $g \text{ dry biomass} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$ ) to net ecosystem carbon balance from aboveground net primary production (ANPP) and belowground net primary production (BNPP) for 2006 and 2007. Error bars show  $\pm$  standard error (SE),  $n=3$ . Means of ANPP and BNPP shown with different letters were determined to be significantly different using ANOVA linear mixed-effects model selection,  $P=0.05$ .

## RESULTS

### Ecosystem Carbon Inputs

Treatment effect on ANPP was significant ( $P < 0.001$ ) with MIRG contributing significantly greater aboveground biomass in both 2006 and 2007 (Fig. 2). Aboveground NPP in CONT was significantly greater than HARV in 2006 ( $P=0.001$ ), but there were no differences between these treatments in 2007 ( $P=0.10$ ). In both years, the NONE treatment produced the least biomass. Treatment had very little effect on the plant functional group component of ANPP with the exception of an increase in grass cover in the HARV treatment ( $P=0.04$ ). At the species level, the CONT treatment had greater Kentucky bluegrass cover ( $P=0.02$ ) but less orchard grass cover ( $P < 0.001$ ). Also, the proportion of bare ground was greater in the CONT treatment ( $P=0.002$ ; Table S1 available online at <http://dx.doi.org/10.2111/REM-D-12-00151.s2>). As a result of defoliation, LAI was quite variable in the CONT, MIRG, and HARV treatments, while there was little variability in the NONE treatment (Fig. S2 available online at <http://dx.doi.org/10.2111/REM-D-12-00151.s3>).

Treatment also had a significant effect on BNPP in both years ( $P=0.002$ ). Root production in CONT and MIRG was significantly less than HARV ( $P < 0.001$ ) and NONE in 2006 ( $P < 0.001$ ), and CONT, MIRG, HARV were all significantly less than NONE in 2007 ( $P=0.01$ ). Across both years, C contribution to NPP from root biomass in the NONE treatment was  $\sim 3.5$  times that of the grazed treatments and  $\sim 2$  times the HARV treatment (Fig. 2). Within treatment variability among root cores resulted in a coefficient of variation (CV) of 79%, 48%, 45%, and 46% for 2006, and

**Table 1.** Net ecosystem carbon balance (NECB) ( $\text{NECB} = \text{net primary production [NPP]} + \text{imported hay [I}_h\text{]} + \text{imported grain [I}_g\text{]} - \text{heterotrophic respiration [R}_h\text{]} - \text{exported hay [E}_h\text{]} - \text{exported livestock biomass [E}_{lb}\text{]} - \text{enteric fermentation [F}_{ch4}\text{]} - \text{livestock respiration [F}_{co2}\text{]} - \text{dissolved organic carbon [DOC}^1\text{]}$ ) for all treatments in 2006 and 2007. All units are in  $\text{g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$ .

Year	Treatment	NPP	$I_h^2$	$I_g^2$	$R_h$	$E_h$	$E_{lb}^2$	$F_{ch4}$	$F_{co2}$	NECB <sup>3</sup>
2006	CONT <sup>4</sup>	481 (12)	101	17	603 (8)a	0	4	21 (0.8)	207 (7)	-236 (15)b
	MIRG	614 (39)	101	17	599 (5)a	0	4	20 (0.8)	209 (7)	-100 (24)a
	HARV	446 (19)	0	0	583 (1)b	254 (16)	0	0	0	-391 (11)c
	NONE	285 (30)	0	0	561 (2)b	0	0	0	0	-276 (28)b
2007	CONT	467 (50)	167	31	647 (20)a	0	5	24 (0.6)	223 (7)	-234 (56)b
	MIRG	774 (80)	167	31	613 (4)b	0	5	24 (0.6)	225 (6)	106 (69)a
	HARV	636 (34)	0	0	579 (3)c	257 (20)	0	0	0	-200 (25)b
	NONE	392 (42)	0	0	564 (3)c	0	0	0	0	-171 (38)b

<sup>1</sup>DOC was considered negligible, and zero was used in the NECB calculation.

<sup>2</sup>Grazing treatments were implemented with similar stocking rates on a yearly basis. Therefore, total hay and grain inputs, and exported livestock biomass, were divided equally between grazing treatments.

<sup>3</sup>Mean values with standard errors indicated in parentheses. Significant NECB and  $R_h$  differences shown by different lowercase letters were determined by ANOVA linear mixed effects models,  $P < 0.05$ .

<sup>4</sup>CONT indicates continuous grazing; MIRG, management-intensive rotational grazing; HARV, haymaking; and NONE, land set aside with no harvests.

46%, 71%, 49%, and 50% for 2007, in CONT, MIRG, HARV, and NONE, respectively.

Only the grazed treatments received supplemental feed inputs, which represent a major import of C into the system. Across both years, CONT and MIRG received C imports in the form of hay and grain that were 25% (CONT) and 18% (MIRG) of the total C inputs (Table 1).

### Ecosystem Carbon Exports

Heterotrophic respiration represents the major source of C loss from our pasture C budget and treatment resulted in significant differences in soil respiration for both years ( $P < 0.001$ ). Carbon lost as soil respiration was greatest in CONT and MIRG for 2006, but this pattern changed in 2007 with significantly greater C loss from CONT than all other treatments (Table 1). In 2006, an early decline in spring soil moisture and lower than average precipitation through the summer growing season appeared to have depressed the summer  $\text{CO}_2$  flux plateau creating a bimodal flux distribution (Fig. 3A). While in 2007 (Fig. 3B), daily soil  $\text{CO}_2$  flux reached a plateau in all treatments from approximately day of year (DOY) 150 through 240, which corresponded to low but consistently available monthly soil moisture and the highest monthly soil temperatures (Figs. 4A and 4B).

The second greatest loss of C was in the HARV treatment as biomass removed as hay (Table 1). Carbon lost as exported hay was 254 and 257  $\text{g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  in 2006 and 2007, respectively. In both years, these losses significantly altered the NECB for the HARV treatment; without biomass export, the value for 2006 would have been less negative and similar to the loss realized in MIRG (-137 and -100  $\text{g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  for HARV and MIRG, respectively). In 2007, the biomass export was of even greater relevance as the sign for NECB would have changed from -200 to 57  $\text{g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$ .

Carbon lost through livestock respiration was also a major export pathway. In 2006 livestock respiration produced losses of 207 and 209  $\text{g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  for CONT and MIRG, respectively. This loss was slightly greater in 2007, with CONT losing 223 and MIRG losing 225  $\text{g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  (Table 1).

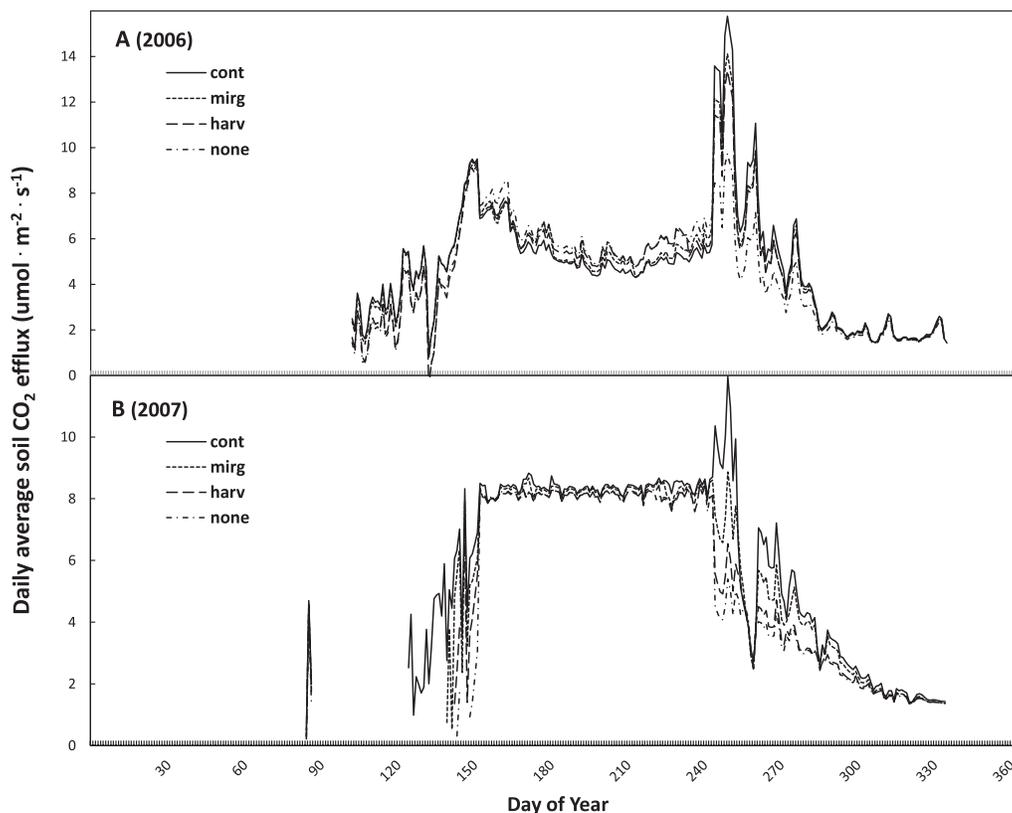
Across both years, C allocated to respiration accounted for 25% of C lost in both CONT and MIRG. Other pathways for C loss, albeit of less magnitude, were C lost as  $\text{CH}_4$  (< 3% for both CONT and MIRG), and C exported as livestock biomass (< 1% for CONT and MIRG; Table 1).

### Net Ecosystem Carbon Balance

There was a significant effect of treatment on NECB ( $P < 0.001$ ). MIRG lost significantly less C in 2006 than all other treatments, and in 2007 MIRG was the only treatment that had a favorable C balance (Table 1). Our model of NECB resulted in C loss in both years for all other treatments. Even though the magnitude of soil respiration was similar (range, 561 to 647  $\text{g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$ ), there were significant differences between treatments in both 2006 ( $P < 0.001$ ) and 2007 ( $P < 0.001$ ; Table 1). Even with additional inputs of C in the form of hay and grain, lower NPP production and higher respiration rates resulted in a negative NECB for the CONT treatment. In the MIRG treatment, while soil respiration was no different than CONT in 2006, it was significantly lower in 2007. This coupled with significantly greater NPP and the addition of hay and grain reduced the magnitude of C lost in 2006 and led to a positive balance in 2007. While respiration rates were lower in the HARV treatment relative to the grazed treatments, NPP was also less than both MIRG and CONT in 2006, but greater than CONT in 2007. Removal of biomass as hay was the main driver of the negative C balance in the HARV treatment. In the NONE treatment, the relationship between reduced C fixed as NPP and C lost as heterotrophic respiration alone drove C balance. Average NPP: $R_s$  ratio across both years was 0.76, 1.15, 0.94, and 0.6 for CONT, MIRG, HARV, and NONE, respectively, while the average ratio of total inputs ( $\text{NPP} + I_h + I_g$ ) to total outputs ( $R_s + E_h + E_{lb} + F_{ch4} + F_{co2}$ ) was 0.68 (CONT), 0.92 (MIRG), 0.59 (HARV), and 0.55 (NONE).

### Sources of Variability in the Net Ecosystem Carbon Budget

The two major sources of variability in our budget were NPP (55%) and soil respiration (22%). A number of parameter estimates were used to calculate NPP, especially BNPP. In this



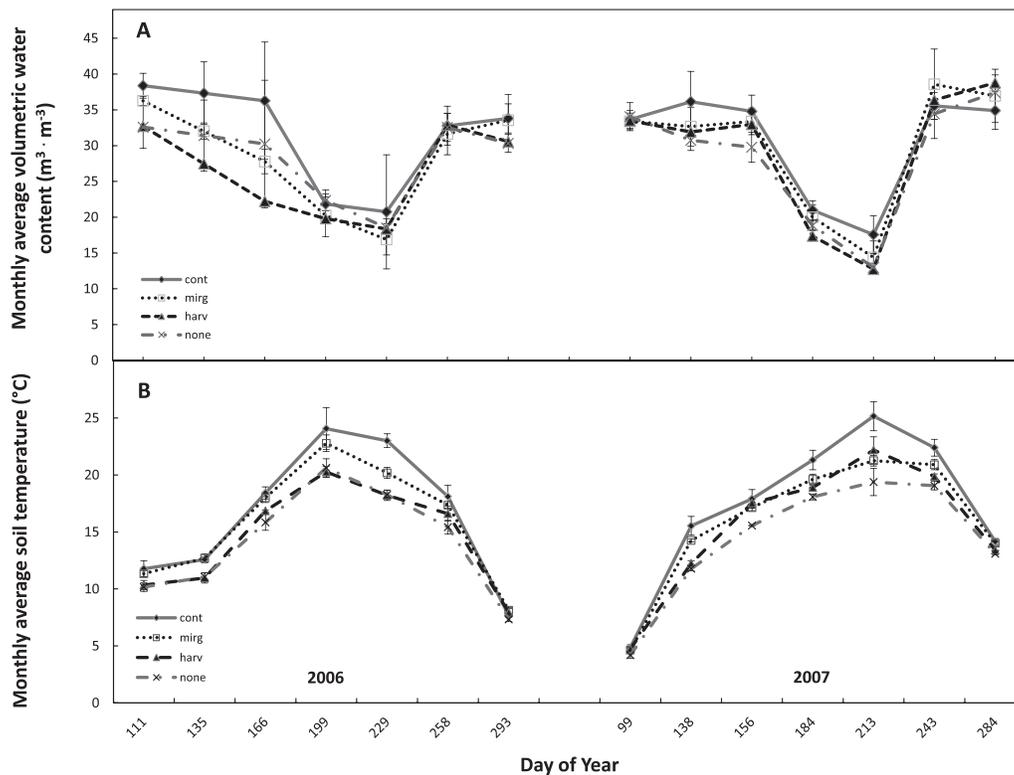
**Figure 3.** Modeled daily average CO<sub>2</sub> efflux for 2006 (A), and 2007 (B). All treatment fluxes are based on the average of four replicate measurement collars in each treatment by block. Individual measurements from 48 collars for spring and fall 2006 and 2007 were combined to model CO<sub>2</sub> efflux. For summer, 36 individual collar measurements from each year were modeled independently to predict yearly CO<sub>2</sub> efflux.

study, turnover of fine roots was assumed to be 53%, proportion of roots below 15 cm was assumed to be 44%, and contribution from exudates was assumed to be 11%. Also, the autotrophic portion of soil respiration was assumed to be 53%. Kucharik et al. (2006) used fine root turnover estimates of 0.21 to 1.58 from a study of prairie ecosystems of the upper Midwest, while fine root turnover modeled for grasslands with mean annual temperature and precipitation similar to the upper Midwest ranged from 0.25 to 1.5 (Gill and Jackson 2000). Proportion of fine root production in the upper 15 cm ranges from 50% (Dahlman and Kucera 1965) to 66% (Jackson et al. 1996; Crush et al. 2005; Kucharik et al. 2006), while exudates contribute from 5% to 21% of net fixed C in biomass (Kumar et al. 2006). For soil respiration reported for grasslands, the annual  $R_a$  contribution ranges from 40% to 60% (Hanson et al. 2000). Substituting these values into our budget gives values ranging from  $-440 \text{ g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  to  $54 \text{ g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  (CONT),  $-297 \text{ g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  to  $523 \text{ g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  (MIRG),  $-455 \text{ g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  to  $229 \text{ g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  (HARV), and  $-439 \text{ g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  to  $364 \text{ g C} \cdot \text{m}^{-2} \cdot \text{y}^{-1}$  (NONE). The more negative balances reflect the lower estimates of fine root turnover and root C exudates, the lower estimate of fine root production below 15 cm, and the lower estimate of plant contribution to soil respiration. The lower NECB values were also for 2006 when NPP was lowest. Conversely, the positive NECB values were for 2007, which had greater NPP and reflect the most optimistic estimates of fine root turnover, root exudate C, fine root production, and plant contribution to soil respiration.

## DISCUSSION

In this study, we showed that by differentially affecting NPP and soil temperature and moisture, management strategy affects NECB by altering the ratio of NPP to soil respiration. While soil respiration rates were greater in the grazing treatments, using a rotational grazing strategy can reduce C loss in subhumid pasture by significantly increasing NPP. But even with greater C inputs from biomass production, MIRG has a moderate likelihood of being a C source without the import of C in the form of extra feed. In the other defoliation treatments, CONT and HARV, higher  $R_b$  coupled with reduced NPP (CONT), and high biomass removal even when  $R_b$  is lower (HARV), resulted in a high to moderate likelihood of significant C loss. When the plots were not managed for biomass production or removal (NONE), reduced rates of respiration could not overcome a significant reduction in NPP and led to a moderate likelihood the system is losing C in the short term.

The relatively high uncertainty in estimates of belowground C inputs, mainly root turnover and to a lesser extent proportion of roots below 15 cm, along with uncertainties in heterotrophic contribution to C efflux, make it important to recognize there is a range of potential NECB outcomes. We used conservative values for unmeasured parameter estimates and report NECB outcomes that are representative of this approach. Without refinement of belowground methodologies, using a biometric approach to estimate C balance as we have



**Figure 4.** Monthly average volumetric water content (VWC) to 15-cm depth for all treatments (A), and monthly average soil temperature (°C) at 10-cm depth for all treatments (B). Mean values are the average of four replicate measurements per treatment by block for soil temperature, and five replicate measurements per treatment by block for VWC. Error bars show  $\pm$  standard error (SE),  $n=3$ .

done here results in greater uncertainty than is desired for policy applications (such as assigning C credit based on absolute amounts of C gained or lost), but is useful for comparing relative management effects on C balance.

### Contribution of NPP to NECB

NPP from plant tissue above- and belowground is a fundamental component of potential C accumulation or loss in terrestrial ecosystems (Schlesinger 1997). Across both years, our estimates of C allocated to the NECB as biomass (NPP) ranged from 285 to 774 g C · m<sup>-2</sup> · y<sup>-1</sup>, with the greatest contribution coming from the MIRG treatment and the NONE treatment contributing the least. Greater production in MIRG was most likely the result of managing the timing, intensity, and frequency of defoliation by livestock. Recent studies have shown herbivore-plant interactions in subhumid pasture are coupled and timing and frequency of defoliation enhances ANPP by 1) maintenance of the plant community in a homogeneous vegetative state, 2) retention of a favorable plant community by excluding unpalatable species, and 3) increasing the rate of N mineralization (Woodis and Jackson 2008; Oates et al. 2011). ANPP in the CONT treatment was 66% of MIRG and was likely due to an increase in low stature and less productive plant species and a significant increase in bare ground relative to the other treatments as shown in Table S1. While the HARV treatment was also relatively productive, 66% of MIRG ANPP, and maintained a favorable plant community, allowing plants to gain greater maturity with longer intervals between defoliation, and reduced rates of N

cycling negatively affected overall production. In the NONE treatment, significantly less ANPP (24% of MIRG ANPP) was measured. It has been well recognized that subhumid pastures can be managed with favorable plant species, grazing, and fertilization to increase production (Paine et al. 1999; Woodis and Jackson 2008; Oates et al. 2011), concomitantly; unmanaged pasture results in lower ANPP with a build-up of residual plant biomass, slower rates of N cycling, and an increase in unfavorable plant species (Oates et al. 2011).

A significant proportion of NPP in temperate grasslands, relative to other ecosystems, is produced belowground (Jackson et al. 1997). ANPP was greater than BNPP in all treatments except where no harvest occurred (NONE). In the defoliated treatments, ANPP was two to five times BNPP, while in the NONE treatment root productivity equaled shoot productivity in 2006 and was slightly less than three times as productive in 2007. This discrepancy in belowground production was likely the result of defoliation effects on plant allocation of resources (see Holland et al. 1992, for references). Defoliation accelerates rates of carbon and nitrogen cycling, which reduces fine root turnover (Pucheta et al. 2004) and root growth (Bardgett et al. 1998; McNaughton et al. 1998).

### Magnitude of NECB Driven by Management

Carbon imports as supplemental feed (hay and grain) were of primary importance in modulating C balance in the grazed treatments. Without contribution to CONT and MIRG treatments of 118 and 198 g C · m<sup>-2</sup> · y<sup>-1</sup> for 2006 and 2007 (Table 1), respectively, it is likely the CONT treatment would

be a stronger source, and a moderate likelihood the MIRG treatment would also be a source of C to the atmosphere.

Carbon lost through the process of heterotrophic respiration accounted for the greatest export between the ecosystem and the atmosphere. Treatment affected the magnitude, but not the seasonal pattern of these fluxes. While precipitation was lower than average in late spring and early summer for both years (DOY 105 through 195), a period when cool-season grasses put on a significant portion of their biomass, late summer precipitation was delivered in a consistent pattern in 2007 (DOY 195 through 240). In 2007, daily soil CO<sub>2</sub> efflux reached a plateau in all treatments from approximately DOY 150 through 240, which corresponded to highest monthly soil temperatures and low but consistently replenished soil moisture. While overall precipitation was greater in 2006, the early decline in spring soil moisture and lower than average precipitation through the summer growing season appeared to depress biomass growth and summer CO<sub>2</sub> efflux, creating a bimodal flux distribution. This pattern was also found in Pennsylvania grasslands (Skinner 2008), 20 European grasslands (Bahn et al. 2008), and a semi-arid grassland in China (Xu and Wan 2008). While our models found temperature had the greatest influence, soil moisture also influenced CO<sub>2</sub> efflux. In both years, spikes in CO<sub>2</sub> flux around day 250 corresponded to increased fall soil moisture and suggest heterotrophic and autotrophic respiration in this system may be modulated by low soil moisture. Garten et al. (2009) in a constructed grass dominated old-field experiment found similar results in which soil respiration was unaffected by increased temperature when adequate soil moisture was available, but declined under increased soil temperature in a water reduced condition. Similarly, factors such as canopy status can impact soil respiration (Craine et al. 1999). Biomass removal leads to greater light penetration, higher soil temperature, and lower transpiration rates resulting in potentially greater volumetric water content. The CONT and MIRG treatments emitted the greatest CO<sub>2</sub> flux while exhibiting higher soil temperatures, greater soil moisture for CONT in both years and for MIRG in 2007, and lower monthly LAI. All these are factors that have led to greater observed C loss in treatments with biomass removal.

Carbon export from metabolic respiration and enteric fermentation of livestock in the CONT and MIRG treatments, and plant biomass harvest for export in the HARV treatment were similar in magnitude ranging from 229 g C · m<sup>-2</sup> · y<sup>-1</sup> to 250 g C · m<sup>-2</sup> · y<sup>-1</sup>, and 254 g C · m<sup>-2</sup> · y<sup>-1</sup> to 257 g C · m<sup>-2</sup> · y<sup>-1</sup> for respiration and enteric fermentation, and biomass export, respectively. Without changing stocking rate, the losses from respiration and fermentation are unavoidable. For example, in the most productive treatment (MIRG), a reduction in stocking rate from 4 AU · ha<sup>-1</sup> to 2.5 AU · ha<sup>-1</sup> would have been required for the MIRG plots to be C neutral without C imports; an average yearly C loss of 90 g C · m<sup>-2</sup> compared to a loss of 245 g C · m<sup>-2</sup>. One possible option for beef and dairy farmers to offset organic C export from harvest would be to supplement their livestock with forage grown within the bounds of their pasture “system.” This would require a system of pastures where some are used to rotate livestock through for grazing, and some that would be harvested to supplement those grazed pastures. For the given year, the C exported from HARV plots

would be imported to other parts of the pasture system, which could be rotated through the paddock system on a yearly basis. This would result in no net C lost to harvested biomass from the pasture system other than livestock biomass. While not completely negating C losses, a farmer with adequate land base would be able to reduce overall system loss.

Most of the variability in our NECB was from the NPP and soil respiration components. This confirms the idea that grasslands have the potential for accumulation of C if we can manage pastures to increase NPP, especially the conversion of cropland to pasture. But managing pastures with inherently high SOM to increase NPP may lead to greater rates of soil respiration and not lend themselves to C accumulation to the degree to which grasslands of lower productivity might (Peichl et al. 2011). In a meta-analysis, Conant et al. (2001) found productive pasture without a long history of grazing lost an average of 1.8% of their C per year, and changes in soil C were less likely in soil with high initial C. In the plots where no defoliation took place, respiration rates were lower but the magnitude of NPP loss without management could not overcome C lost as  $R_b$ . Even our most productive pastures, those under intensive grazing management, were likely C neutral over the 2-yr study, but this was only with the addition of imported C in the form of hay and grain feed. Substituting the most optimistic values for the various components of NPP and  $R_b$  into our calculations resulted in all treatments having a positive C balance in the most productive year 2007, but in 2006 NECB was positive for only the MIRG treatment. All treatment C balance values were strongly negative for both years when using the most stringent estimates of these parameters.

## IMPLICATIONS

The opportunity for C sequestration by pasture through improved management must be tempered. Conversion from high-intensity managed ecosystems such as maize to pasture reduces C loss (see Conant et al. 2001), and increased C storage estimates for pastures with improved management are in the range of 10 to 130 g C · m<sup>-2</sup> · y<sup>-1</sup> (Follett et al. 2001). But, when pasture NECB budgets consider all cross-boundary transfers of C, management strategies other than those focused on increasing production can also affect where and how C is gained or lost. Our results indicate that grazing management strategy can have significantly different impacts on potential C accumulation or loss in cool-season grass pasture mainly through their influence on NPP and secondary effects on soil respiration, while export of biomass from the haying treatment, and reduced NPP in the nondefoliated treatment, were the main influences on C balance. Within the pasture boundary, under any parameter scenario MIRG had the most positive C balance in both years, while in both years the CONT treatment was strongly negative. The import of supplemental feed had a great impact on the magnitude of C balance in these treatments and shows cross boundary inputs must be considered when determining the NECB status of grazing treatments. Studies that have accounted for C imports and exports have found managed grassland to be C neutral (Suyker and Verma 2001; Owensby et al. 2006) or C sources (Bellamy et al. 2005;

Kucharik et al. 2006; Schipper et al. 2007; Skinner 2008), while some studies that did not have a full accounting of imports and exports report managed grasslands as C sinks (Frank and Dugas 2001; Sims and Bradford 2001; Soussana et al. 2007). Our data show that pasture management does influence the magnitude of change in NECB, and on shorter temporal and smaller spatial scales NPP and ecosystem soil respiration are the primary regulators, but to accurately determine NECB over any time or spatial scale, it is necessary to account for all fluxes and cross boundary transfers of C.

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