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Soil type and moisture regime control microbial C and N mineralization in grassland soils more than atmospheric CO₂-induced changes in litter quality

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ABSTRACT

Global change-induced alterations in litter quality and soil moisture regime will likely impact grassland C and N dynamics, but how these changes interact with edaphic properties across the landscape is unclear. We measured the effects of litter quality, soil type, soil moisture level, and soil drying-rewetting frequency (D-RW) on microbial C and N mineralization of litter and soil organic matter (SOM) in a full-factorial, controlled incubation experiment. Four levels of litter quality (no litter; or litter from *Bouteloua curtipendula* grown under 280, 380, 500 μ L L⁻¹ CO₂) were surface-applied to three contrasting soils common to Blackland Prairie landscapes: an upland Mollisol, a lowland Vertisol, and a fluvial Alfisol. Different soil moisture regimes were tested by incubating soils at four moisture levels (air-dry, 25%, 35%, or 50% water-holding capacity, WHC) and by drying-rewetting soils 0, 1, 2, 4 or 8 times over the 112d incubation period. Litter additions stimulated microbial activity, increasing total CO₂ production (i.e. C mineralized from litter + SOM decomposition) up to $17 \times$ more than no-litter controls (average $3 \times$) and decreasing net N mineralization up to $-3 \times$ less (average $-0.5 \times$) due to greater microbial N immobilization. Neither C nor N mineralization, however, was affected by litter quality. For all soils, litter decomposition increased with increasing WHC and D-RW frequency, but the average percent of total CO₂ derived from litter was a negative function of SOM content. Similarly, net N mineralization also was positively correlated with soil WHC and affected most strongly by soil type (Alfisol < Mollisol < Vertisol). Net N mineralization responses to D-RW events was also soil-specific, with Alfisol soils showing no response and Mollisol and Vertisol soils decreasing after 4 D-RW events. Our results suggest that predicted changes in rainfall patterns and its interactions with soil type across the landscape will control short-term C and N mineralization responses in grasslands to a greater extent than atmospheric CO₂induced changes in litter C:N ratio for this common species of prairie grass.

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1. Introduction

Global changes such as rising atmospheric CO₂ and altered precipitation regimes are expected to impact grassland productivity by directly or indirectly altering the availability of water, nutrients, and energy (i.e. carbon, C; Dukes et al., 2005; Soussana and Lüscher, 2007; Leakey et al., 2009). The interaction of these global changes with edaphic characteristics on soil microbial activity likely will affect whether grasslands sequester or release C. Little data, however, are available on more spatially-explicit responses to global changes, such as those driven by landscape variation in soils.

Small-scale spatial variation in soils caused by vegetation can significantly affect plant-soil responses to elevated atmospheric CO₂. For example, microbially-mediated C and N processes in soil microsites under shrubs are accelerated under elevated atmospheric CO₂ compared to soils in plant-free interspace microsites in a desert rangeland, particularly when water is less limiting (Billings et al., 2004; Jin and Evans, 2007; Schaeffer et al., 2007; Jin et al., 2011). In more mesic rangelands and grasslands, plant growth responses are mediated largely by elevated CO₂-induced increases in plant water-use efficiency and soil moisture availability (Morgan et al., 2004; Körner, 2006; Polley et al., 2008) and, in some cases, greater N mineralization (Dijkstra et al., 2007, 2010). Furthermore, in a Blackland prairie system, soil type strongly affects how plant productivity and grassland species composition respond to increased soil water availability and subsequent changes to soil N





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availability under elevated CO_2 (Polley et al., 2003, 2012a, 2012b; Fay et al., 2012).

Soil CO₂ effluxes are high in tallgrass prairies compared to other temperate terrestrial ecosystems because of their relatively high plant productivity, annual precipitation, and soil organic matter (SOM) content (Mielnick and Dugas, 2000). Changes in the amount and variability of precipitation, however, are likely to regulate soil CO₂ effluxes and their underlying drivers (Fay et al., 2011) and responses to elevated atmospheric CO₂ (Jastrow et al., 2000). For example, grassland soil CO₂ fluxes decreased with increased time interval between rainfall events (Fay et al., 2000; Harper et al., 2005), in part due to reduced soil moisture and plant productivity and expected C input to soils (Fay et al., 2008).

More than half of the CO₂ emitted by grassland soils is from microbial SOM decomposition (Wan and Luo, 2003; Millard et al., 2008). Litter inputs and litter decomposition, however, confer substantial variability to overall soil CO₂ fluxes and the transfer of plant C into SOM. Increases in litter inputs may lead to more C stored as SOM (Franck et al., 1997; Gorissen and Cotrufo, 2000; Jastrow et al., 2000), and changes in litter quality affect the partitioning of C into different SOM pools (Melillo et al., 1982; Cotrufo et al., 1998; Guo and Gifford, 2002). Increased C inputs, however, can stimulate SOM decomposition (i.e. priming effect) and limit grassland C storage potential (Kuzyakov et al., 2000; Xie et al., 2005; Niklaus and Falloon, 2006; Carney et al., 2007). Further, the relationship between litter C:N ratio, which typically increases in plants grown under elevated CO₂, and litter decomposition rate is highly variable (Franck et al., 1997; Gorissen and Cotrufo, 2000; Norby et al., 2001). Changes in the quantity and/or quality of plant litter inputs into soil can result in either net mineralization or immobilization of N and feed back to affect litter decomposition (Torbert et al., 2000). Interactions between altered litter inputs and precipitation regime, therefore, could affect litter decomposition and grassland soil C storage.

The objectives of this study were to evaluate how global changeinduced alterations in litter quality and soil moisture regime interact with edaphic characteristics to affect microbial C and N mineralization. Microbial C mineralization was quantified as the decomposition of organic C in litter and SOM to CO_2 . Microbial N mineralization, the conversion of organic N to inorganic N (i.e. ammonium + nitrate), was quantified as the net change in soil inorganic N over the incubation period. A positive net change indicates microbial mineralization of N, and a negative net change indicates microbial immobilization of N. We removed the confounding influences of root respiration, climatic variation in the

field, and the cumulative impact of CO₂ enrichment on soils by incubating plant litter grown under three CO₂ levels with soils that had no exposure to elevated atmospheric CO₂ (Paul et al., 2001; Collins et al., 2000). Because short-term C mineralization rapidly quantifies litter quality differences (Haney et al., 2001; Haney and Franzleubbers, 2009), controlled incubations were conducted using four litter treatments (no litter; or litter from Bouteloua cur*tipendula* grown under 280, 380, 500 μ L L⁻¹ CO₂) on three contrasting soils common to Blackland Prairie landscapes (Alfisol, Mollisol, Vertisol). Total CO₂ production and net N mineralization were measured in soils incubated in a full-factorial experiment with four levels of soil moisture and five levels of drying-rewetting frequency to represent different precipitation regimes. In addition, water-extractable organic C and organic N concentrations in preand post-incubated soils were measured to evaluate whether changes in these potentially labile pools correlated with microbial C and N activities. We hypothesized that: (1) litter additions would stimulate microbial activities (decomposition, net N mineralization) in all soils; (2) differences in litter C:N ratio due to growth-CO2 level would be reflected by differences in microbial activity; (3) C and N mineralization would increase with greater soil moisture; and (4) soil-specific properties would interact with soil moisture regime as the dominant factor controlling litter decomposition and net N mineralization.

2. Materials and methods

2.1. Soil and litter collection

Soils (0–15 cm) from three soil orders commonly found in the Blackland Prairie region of Texas were collected in Spring 2008. These soils represent a broad range of texture, N and C contents, and hydrologic properties for grasslands in the southern portion of the U.S. Central Plains. An Alfisol (Bastsil series), Mollisol (Austin series), and Vertisol (Houston Black series) were collected from U.S. Army Corp of Engineers Stillhouse Hollow Lake area (Belton, TX), the USDA-ARS's Grassland, Soil, and Water Research Laboratory (GSWRL; Temple, TX), and the GSWRL watershed network (Riesel, TX), respectively. Bastsil soils are loamy fine sands (fine-loamy, siliceous, active, thermic Udic Paleustalfs) formed of alluvial sediments and commonly found on stream terraces. Austin soils are silty clays (fine-silty, carbonatic, thermic Udorthentic Haplustolls), typical of erosional uplands. Houston Black soils are heavy shrink-swell clays (very-fine, smectitic, thermic Udic Haplusterts) that dominate lowland areas. Selected soil properties are shown in Table 1. Fresh

Table 1

Initial properties for soils (0–15 cm) and *Bouteloua curtipendula* litter used in laboratory incubations (mean \pm se; n = 3). Different letters for each property indicate significant differences between soil series or between growth-CO₂ levels ($P \le 0.05$). Soil texture data from Fay et al. (2009).

Soil order	Unit	Alfisol	Mollisol	Vertisol
Soil series name	_	Bastsil	Austin	Houston Black
Sand	%	$71.9\pm0.9a$	$11.7\pm0.4b$	$10.9\pm0.6b$
Silt	%	$20.8\pm0.5a$	$45.3\pm0.7b$	$43.0\pm0.6b$
Clay	%	$7.3\pm0.4a$	$39.3 \pm \mathbf{1.0b}$	$49.8 \pm 1.4 c$
Total inorganic C	$g C kg^{-1}$	$0.3\pm0.1a$	$68.2 \pm 1.0 b$	$64.6 \pm \mathbf{1.0c}$
Total soil organic C (SOC)	$g C kg^{-1}$	$4.1\pm0.4a$	$17.8\pm0.4b$	$21.8 \pm \mathbf{0.6c}$
Total N	$g N kg^{-1}$	$0.6\pm0.1a$	$1.7\pm0.1b$	$5.6\pm0.1c$
Water-extractable SOC	mg C kg ⁻¹	$137\pm62a$	$226\pm24b$	$376\pm 38c$
Water-extractable organic N (ON)	mg N kg ⁻¹	$16.3\pm0.2a$	$3.9\pm0.3b$	$9.8\pm0.3c$
Water-extractable inorganic N (IN)	mg N kg^{-1}	$7.4\pm0.4a$	$9.0\pm0.5b$	$\textbf{7.9} \pm \textbf{0.3a}$
<i>B. curtipendula</i> growth-CO ₂ level		280 μL L ⁻¹	380 μL L ⁻¹	$500 \ \mu L \ L^{-1}$
Initial litter C	mg C g ⁻¹	377 ± 3a	397 ± 1b	$384 \pm 2c$
Initial litter N	$mg N g^{-1}$	$5.6\pm0.1a$	$7.3\pm0.1b$	$18.9\pm2.0c$
Initial litter C:N	_	$67.3\pm0.3a$	$53.1\pm2.0b$	$22.7\pm1.9c$
C added in 0.25 g dry litter	mg C	94	99	96
N added in 0.25 g dry litter	mg N	1.4	1.8	4.7

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