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Effect of precipitation frequency on litter decomposition of three annual species (*Setaria viridis, Artemisia sacrorum*, and *Chenopodium acuminatum*) in a semi-arid sandy grassland of northeastern China

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ABSTRACT

Annual total precipitation typically fails to predict litter decomposition rates, especially in semi-arid regions, where precipitation frequency (*PF*) significantly affects decomposition. We hypothesized that low *PF* would decrease decomposition rates. We performed a litterbag decomposition experiment with litter of three annual species at three depths (aboveground litter, litter at 0–10 and 10–20 cm below the surface). We used the same total amount (280 mm), but three precipitation frequencies (PF1, 10 mm every 5 days; PF2, 20 mm every 10 days; PF3, 40 mm every 20 days) during the growing season. We measured the remaining mass, carbon (C) content, nitrogen (N) content, C:N ratio, and C and N losses of each species at the three positions. Litter decomposition and the C and N dynamics were influenced by species, depth, and *PF*. Low *PF* significantly decreased litter decomposition (mass loss rates of 49.7, 49.1, and 41.2% for PF1, PF2, and PF3, respectively). This effect interacted with placement-depth but not with species. This interaction also existed for C and N dynamics except for the N loss rate. The influence of *PF* on C loss mainly resulted from its effect on mass loss, whereas N loss was not affected by *PF*. Our results highlight the importance of *PF* for litter decomposition in a semi-arid region. The relationship between *PF* and litter decomposition provides a theoretical basis for regional carbon-cycle models and carbon budget predictions. Our results also suggest that non-graminaceous species showed higher potential than grasses for improving soil carbon in semi-arid sandy grasslands due to faster decomposition rates, especially below ground.

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Introduction

Semi-arid ecosystems are an important component of the global carbon (C) cycle (Campos, Germino, and Graaff 2017), as they cover about 15% of the Earth land surface and are undergoing continuous expansion (Huang et al. 2016). Their importance is increasing because they are fragile and sensitive to human activities and climate change. Sandy ecosystems are also a dominant landscape in China, where they cover an estimated $36.9 \times 10^4\,km^2$ in 2000. These ecosystems play a significant role in carbon sequestration, but are vulnerable to both biotic and abiotic factors, including climate change, that affect many processes such as plant growth (Luo et al. 2017), C sequestration (Campos, Germino, and Graaff 2017), and litter decomposition (Joly, Kurupas, and Throop 2017).

Litter decomposition is a key process in a terrestrial ecosystem matter exchanges and energy fluxes (Vázquez-de-Aldana, García-Ciudad, and García-Criado 2008). During these exchanges, fixed C stored in plant matter after uptake from the atmosphere is released back into the environment. The release of C and other elements from litter during decomposition can affect the composition of atmosphere and soil nutrient status. Nutrient elements such as nitrogen (N), phosphorus (P), and potassium (K) that are released from plant litter into the soil can also directly promote plant growth, especially in impoverished soil (Li et al. 2016; Wang et al. 2017). In addition, the accumulation of organic matter in the soil during litter decomposition can greatly increase the soil cation exchange capacity, which can improve the nutrient retention capacity (Berg and McClaugherty 2008). This process would indirectly enhance plant growth.

Litter decomposition comprises both biotic and abiotic processes, including physiological and chemical processes. In the biotic processes, soil microbes and their enzymes are the main drivers. Both enzyme activity and microbial activity are controlled by moisture to different degrees. In abiotic processes, which include physical fragmentation, leaching, and transportation, water also plays important roles through its effects on wetting and drying cycles, rainwater infiltration, and evaporation (Berg and McClaugherty 2008).

Soil moisture has been shown to strongly control litter decomposition in many types of ecosystems. However, studies of the relationship between precipitation, which is a dominant factor that affects soil moisture, and the litter decomposition rate have shown contradictory results. In a long-term (10-year) field investigation, Vanderbilt et al. (2008) found no relationship between the decomposition constant (k, the rate of decomposition) and the total amount of precipitation for many species in arid environments in central New Mexico, USA. In an incubation experiment, Austin, Araujo, and Leva (2009) also found that the aboveground litter decomposition rate was not affected by the size of the water pulse. In contrast, other researchers have found that more frequent precipitation pulses led to increased decomposition rates (Schuster 2016; Joly, Kurupas, and Throop 2017).

Previous studies have shown that the mean annual precipitation (*MAP*) varies at both global and regional scales, but this variation may have been smaller than the variation in precipitation frequency (*PF*) and duration in recent decades (Intergovernmental Panel on Climate Change (IPCC) 2007). For example, a previous analysis of climate data from 1971 to 2013 in the Horqin Sandy Land of northeastern China (Yue et al.

2016) revealed that both *MAP* and the total rainfall during the growing season (from May to September) varied slightly. However, the precipitation duration and quantity during each precipitation event both increased significantly over time, whereas the number of days with precipitation showed a significant linear decreasing trend.

MAP has traditionally been considered to be a dominant control on ecosystem processes. Many studies have found a significant relationship between *MAP* and ecological processes such as soil respiration (McCulley, Boutton, and Archer 2007; Zhou, Talley, and Luo 2009), plant growth (Lauenroth, Burke, and Paruelo 2000; O'Connor, Haines, and Snyman 2001; Zhou, Talley, and Luo 2009), ecosystem matter exchanges (Zhou et al. 2002), and litter decomposition (Zhou, Talley, and Luo 2009). Moreover, *MAP* has been widely and successfully used in models to predict net primary productivity (Nemani et al. 2003; Li et al. 2015), carbon dioxide emission (Li and Ji 2010; Cox et al. 2013; Wang, Zeng, and Wang 2015), and water balances (Rochdane et al. 2012). Significant relationships exist between *MAP* and ecological processes at multiple scales, but especially at large spatial scales (Zhou et al. 2002; Zhou, Talley, and Luo 2009) and long time scales (Vázquez-de-Aldana, García-Ciudad, and García-Criado 2008). However, at meso and micro temporal scales, there have been contradictory results for the relationships between *MAP* and ecological processes. For example, some researchers have found significant positive relationships between *MAP* and plant litter decomposition (Yahdjian, Sala, and Austin 2006; Brandt, King, and Milchunas 2007), whereas others found no significant relationship (Vanderbilt et al. 2008; Gallo et al. 2009). These contrasting results may be due to the fact that cumulative precipitation, which is typically defined at monthly or annual time scales, is a coarse-grained measure that fails to reflect short-term variations of moisture levels in the surface litter and shallow soil horizons, and may thus fail to account for changes in the activity of the associated decomposers (Joly, Kurupas, and Throop 2017). Another key factor relates to the soil texture, which determines the water-holding capacity. For example, Luo et al. (2017) found that in a soil with a sandy texture and a high saturated hydraulic conductivity, the soil water content decreased rapidly after a rainfall event; as a result, increased precipitation (30% greater than in the control) had no effect on plant growth. Where the surface soil or the shallowness of the upper soil layers promotes rapid drainage, the increased soil moisture that results from precipitation events disappears rapidly (Yao, Zhao, et al. 2013; Yao, Zhang, et al. 2013); as a result, soil microbial activity, which is strongly controlled by soil moisture, would be more strongly correlated with *PF* than with *MAP*. Because *MAP* cannot account for these finer-resolution changes in soil moisture, it may fail to predict litter decomposition rates in semi-arid regions with sandy soils. In these regions, rainfall frequency and amount varies widely.

A few studies have directly demonstrated a significant relationship between *PF* and the litter decomposition rate (Anaya et al. 2012; Schuster 2016; Joly, Kurupas, and Throop 2017). Some studies have demonstrated that precipitation pulses result in pulses of soil respiration (Liu et al. 2002; Sponseller 2007; Chen et al. 2008, 2009; Yan et al. 2014). These pulses can be explained by the rapid increase in activity of the microbial community in response to a sudden increase in the availability of water, which is often a limiting factor for their metabolic processes in dry soils (Griffiths and Birch 1961; Orchard and Cook 1983; Jarvis et al. 2007). These increased microbial populations can

accelerate litter decomposition when water availability is no longer a limiting factor. Furthermore, the duration of this pulse effect depends on the pulse size, since large pulses increase soil moisture for a longer time (Chen et al. 2008). However, in a soil with a low water-holding capacity or with a high saturated hydraulic conductivity, the microbial response would be controlled by the soil texture (primarily through its effects on drainage of water into deeper horizons) or by human activities (Yao, Zhao, et al. 2013; Yao, Zhang, et al. 2013) such as grazing (Chen et al. 2008) that would decrease the duration of favorable water conditions.

Nitrogen in plant litter is commonly a key limiting nutrient for populations of microbial decomposers, as N is usually immobilized or mineralized by microorganisms in terrestrial ecosystems during litter decomposition (Li et al. 2016). Nitrogen dynamics during litter decomposition are controlled by many factors, such as species (Wang et al. 2017), litter placement-depth in the soil (*PD*), tissue type (Parton et al. 2007), microclimate, initial litter quality (Brandt, King, and Milchunas 2007; Austin, Araujo, and Leva 2009), and the length of the decomposition (Yahdjian, Sala, and Austin 2006). However, in semi-arid sandy grassland, the factors that control N dynamics and their relative importance during litter decomposition are poorly understood, especially regarding the influence of *PF* on N release during litter decomposition.

Based on this review of the literature, we hypothesized that (1) *PF* would play an important role in litter decomposition and N dynamics in a semi-arid region with sandy soils, and that high *PF* would increase litter decomposition, with the magnitude of the increase varying among species and litter placement-depths; (2) litter of non-graminaceous species would decompose faster; and (3) the aboveground litter would decompose slowly. We tested these hypotheses by conducting a litterbag decomposition experiment using litter from three dominant annual species in our study area (graminaceous *Setaria viridis* (L.) Beauv., and non-graminaceous *Artemisia sacrorum* Ledeb. and *Chenopodium acuminatum* Wild.) at three depths in the soil. Our goal was to strengthen knowledge of the relationship between litter decomposition and related abiotic factors for dominant species in the Horqin Sandy Land of northeastern China, which is a typical agro-pastoral ecotone, but is also a region that is highly sensitive to human activities and climate change. Our results will provide more detailed information to support ecological models of terrestrial organic matter cycles in semi-arid ecosystems, but will also provide guidance for management of the sandy grassland (i.e. to support species selection in artificial vegetation restoration) in semi-arid regions of northeastern China.

Methods

Study site

We conducted our study at the Naiman Desertification Research Station of the Chinese Academy of Sciences (42°58′N, 120°43′E; elevation 360 m a.s.l.), in the eastern part of Inner Mongolia autonomous region. This region belongs to the cold-temperate zone, and has a semi-arid continental monsoon climate. From 1971 to 2015, *MAP* averaged 335 mm, with high variability (CV = 28.0%), and 84% of the total precipitation (i.e. an average of 280 mm) falling from May to August (Yue at al. 2016). The mean annual

temperature is 6.7 °C, with a minimum monthly mean temperature of −12.6 °C in January and a maximum of 24.3 °C in July (Yue at al. 2016). The annual frost-free period is approximately 141 days and the average annual wind speed ranges from 3.4 to 4.5 m s^{-1}, but the mean wind speed during the early growing season (March to May), when the most severe wind erosion occurs, ranges from 5.0 to 6.0 m s^{-1} (Zhao et al. 2010). The landscape is characterized by sand dunes alternating with gently undulating lowlands between the dunes (Li et al. 2011). The soil is classified as a Cambic Arenosol in the FAO soil classification system (Food and Agriculture Organization (FAO), 2006). The surface soil (0–20 cm) organic carbon as well as inorganic carbon ($CaCO_3$) contents, pH and bulk density values were 1.04 g·kg^{-1} (determined using the Walkley–Black dichromate oxidation procedure), 0.07 g·kg^{-1} (determined using the gasometric method), 7.8 and 1.6 g·cm^{-3}, respectively (Luo et al. 2020). The native vegetation in this region is dominated by the sub-shrub *Artemisia halodendron* Turcz. ex Bess. and the leguminous shrub *Caragana microphylla* Lam. The herbaceous vegetation is dominated by annual species, with high variation of the species composition. The most common species are *S. viridis, A. sacrorum*, and *C. acuminatum*, which together account for 62.1% of the plants, as well as *Corispermum macrocarpum* Bunge and *Salsola collina* Pall.

Sampling

We selected three dominant annual species in the Horqin Sandy Land for the litter decomposition study, *S. viridis*, *A. sacrorum*, and *C. acuminatum*. Detailed characteristics of these three species are provided Table S1. We collected samples at the end of the 2014 growing season (late September), when all these annual plants were dead. We randomly selected plots in which the three species grew separately, and were naturally distributed at a high density, to facilitate sampling. First, we cut the aboveground biomass at the soil surface. We then excavated the belowground mass in two layers: from 0 to 10 cm and from 10 to 20 cm below the surface. We brought the samples to the laboratory, and gently washed them with tap water to remove any attached dust or soil. We manually collected all visible fragments of roots and other organic matter from the soil and used these samples in the incubation experiments. After washing, we oven-dried all samples at 65 °C for 48 h. We used the samples to create a single composite sample for each species and depth before filling the litterbags. We created control litter samples (undecomposed) by grinding the samples (which we collected from each depth for each of the three species) to pass through a 0.5-mm mesh and then stored them in a fridge at 5 °C in the dark. We cut each of the remaining samples into 2-cm lengths and determined the C and N contents with dry combustion method using a Vario Macro Cube elemental analyzer (Elementar, Hanau, Germany), Table 1 summarizes the litter quality. We placed 5 g of the litter from each species and from each depth in the soil in nylon mesh bags (each 10 × 10 cm, with a 0.2 mm mesh opening).

Experimental design

Our litter decomposition experiment was conducted under a rain shelter with simulated precipitation. The rain shelter (10 × 5 m) comprised of a steel frame that supported a

Table 1. Litter initial carbon content (C), nitrogen content (N), and C:N ratio in three dominant annual species at the study site (mean value ± standard deviation, $n = 5$).

Species	Litter depth	C (% *w/w*)	N (% *w/w*)	C/N
Setaria viridis	A	41.12 ± 0.73a	0.64 ± 0.06b	64.25 ± 4.70a
	$B_{0–10}$	43.07 ± 1.40b	0.54 ± 0.05a	79.75 ± 7.68b
	$B_{10–20}$	42.91 ± 0.99b	0.55 ± 0.05a	77.97 ± 5.23b
	F	5.085	6.019	9.958
	P	0.025	0.015	0.003
Artemisia sacrorum	A	44.43 ± 1.11a	1.04 ± 0.09b	42.72 ± 2.64a
	$B_{0–10}$	44.53 ± 0.96a	0.78 ± 0.05a	57.08 ± 4.69b
	$B_{10–20}$	44.13 ± 0.76a	0.80 ± 0.08a	55.16 ± 4.27b
	F	0.245	18.552	19.286
	P	0.786	<0.001	<0.001
Chenopodium acuminatum	A	40.96 ± 1.17a	0.59 ± 0.02b	69.42 ± 1.85a
	$B_{0–10}$	44.06 ± 1.56b	0.48 ± 0.05a	91.79 ± 6.98b
	$B_{10–20}$	44.22 ± 0.9b	0.47 ± 0.06a	94.09 ± 8.95b
	F	10.956	9.766	21.069
	P	0.002	0.003	<0.001

"A" represents aboveground litter; "$B_{0–10}$" and "$B_{10–20}$" represent litter buried at depths of 0–10 and 10–20 cm, respectively. Values for a species labeled with different letters differ significantly ($p < 0.05$) between the placement-depths for each species.

transparent roof. The shelter sides remained open to maximize air movement. Light intensity under the shelter was about 95% of ambient light intensity at the top of the vegetation, and had a similar spectrum to ambient light. Detailed information about this shelter is provided by Yue et al. (2016). Under the shelter, we installed 54 PVC buckets with open tops (each with a 30-cm inside diameter and a 40-cm height), oriented vertically, with the top edge at the ground surface. All buckets were filled with homogenized samples of sandy soil collected from a semi-fixed dune near the study site. The soil was passed through a 1-mm mesh to eliminate plant debris, including roots and coarse materials. The soil texture is medium to coarse sand, with some fine sand (particles between 1 and 0.05, as well as 0.05 and 0.002 mm accounted for 97.1 and 2.8% of the total mass, respectively); the saturated water content was 26.2% *w/w*. We randomly allocated the buckets among three precipitation-frequency treatments (i.e. $n = 18$ for each *PF* water treatment), with the buckets separated by 30 cm to avoid any disturbance from watering. We then installed the nylon mesh litterbags in the buckets (i.e. $n = 6$ buckets per species for each water treatment). In each bucket, we vertically installed litterbags containing litter from each species collected at depths from 0 to 10 cm and from 10 to 20 cm in the buckets at 0–10 and 10–20 cm, respectively, below the soil surface, and a bag filled with the aboveground litter at the soil surface, where it was fixed in place with thin bamboo sticks. We therefore installed 9 bags in each bucket (3 species × 3 placement-depths). Before the surface bag was installed, all soil that had been removed was replaced in the bucket and gently pressed down to ensure that each bucket had approximately the same soil bulk density.

We used three simulated precipitation treatments, which differed in *PF*, that is water amount per event. The total precipitation in each treatment was 280 mm. This was based on the historical precipitation pattern during the growing season (May to September) from 1971 to 2013 (Yue et al. 2016). The potential evaporation during the study period was 1260.6 mm. Supplemental Figure S1 provides details of the precipitation pattern. We designed the following *PF* treatments with rainwater collected by 5 × 5 m plastic sheets. PF1 represented high *PF*, in which 10 mm of the collected

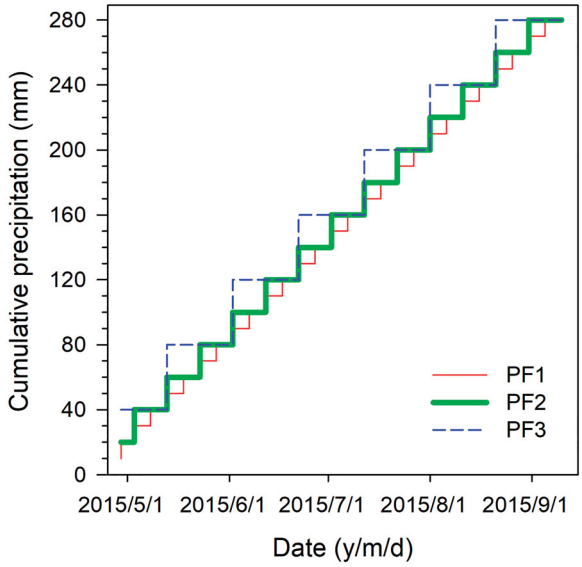

Figure 1. Simulated precipitation patterns throughout the litter incubation experiment. Each treatment received the same total amount of simulated rainfall (280 mm), but in the PF1, PF2, and PF3, treatments, the precipitation was provided as 10 mm every 5 days (i.e. 28 rainfall events), 20 mm every 10 days (i.e. 14 events), and 40 mm every 20 days (i.e. 7 events), respectively.

rainwater was added to the bucket every 5 days (i.e. 28 rainfall events); PF2 and PF3 represented medium and low *PF*, with 20 mm of water added every 10 days (i.e. 14 events) or 40 mm of water added every 20 days (i.e. 7 events), respectively (Figure 1). For comparison, effective precipitation (defined as minimum precipitation that enters the rooting zone) is about 5 mm in our study area (Yue et al. 2016). The rain water was added to each bucket uniformly within 5 minutes using a plastic watering pot. The precipitation simulation was conducted from 29 April to 10 September 2015.

Measurements

At the end of the growing season (25 September 2015), all litterbags were collected and washed gently with tap water to remove soil. They were then dried for 48 h at 65 °C. The remaining mass of the decomposed litter in each bag was measured to allow calculation of the mass loss rate, and the litter was then ground to pass through a 0.5-mm mesh. We then measured the C and N concentrations in all samples and in the control using the Vario Macro Cube device.

Statistical analysis

We conducted statistical analysis using version SPSS 20.0 (www.ibm.com/software/analytics/spss/). We analyzed differences in the remaining mass, in the C and N contents, in the C:N ratio, and in the C and N losses among the three species (S), three litter placement-depths (*PD*), and three *PF* treatments (PF) by three-way ANOVA, with S,

Table 2. Results of the three-way ANOVA test, with species (S), placement-depth (PD), and precipitation frequency treatment (PF) as the factors.

		F value					
Source	d.f.	Mass loss	C content	N content	C:N ratio	C loss	N loss
S	2	51.67**	28.05**	93.21**	46.28**	59.76**	16.35**
PD	2	198.39**	36.12**	129.83**	109.86**	198.81**	31.22**
PF	2	22.90**	4.03*	9.74**	17.33**	26.12**	0.44
S × PD	4	6.47**	32.26**	7.70**	21.32**	10.70**	3.44*
S × PF	4	1.09	1.17	0.28	3.85*	1.72	0.69
PD × PF	4	3.66*	6.08**	4.00*	4.41*	4.56*	0.27
S × PD × PF	8	0.91	1.07	3.27*	4.38*	0.79	1.05

* and ** represent significance at $p < 0.05$ and $p < 0.01$, respectively.

PD, and PF as factors. We performed multiple comparisons using the least-significant-difference (LSD) test whenever the ANOVA indicated a significant difference ($p < 0.05$).

Results

Mass loss

The mass loss in this study was significantly affected by the litter species, litter placement-depth (*PD*), and precipitation frequency (*PF*) (Table 2). The mass loss of *S. viridis* was significantly lower than that of *A. sacrorum* and *C. acuminatum*, and the mass loss of the surface litter was significantly lower than that of the buried litter for all three species (Table 3). *PF* significantly affected litter decomposition rates, and the mass loss decreased significantly at low *PF* (i.e. in PF3). Furthermore, we found significant species × *PD* and *PD* × *PF* interactions (Table 2). For example, the mass loss in the deeper soil layer (10–20 cm below the soil surface) was generally not affected by the *PF* treatment, but did not differ significantly among the *PF* treatments for all species at the soil surface and at a depth of 10 cm below the soil surface (Table 3).

C and N contents and C:N ratio

The C and N contents of the litter after one growing season of decomposition differed significantly among the species, *PD*, and *PF* treatments (Tables 2 and 4). In *A. sacrorum*, the C and N contents were both significantly higher than in the other species, and the C:N ratio was significantly lower (Table 4). In the PF3 treatment, the N content was generally lower than in the PF1 and PF2 treatments, so the C:N ratio was generally significantly higher than in PF1 and PF2 (Table 4). The C and N contents and C:N ratio differed significantly among species, and we found a significant *PD* × *PF* interaction for the C and N contents and for their ratio (Table 2). This indicated that the magnitude of the differences in C and N contents among species and *PF* treatments differed among the litterbag *PD* treatment variants.

C and N losses

After one growing season of decomposition, the C loss differed significantly among the species, and was significantly affected by *PD* and *PF*; there was also a significant species

Table 3. Mass loss due to litter decomposition during one growing season, under the three precipitation-frequency (PF) treatments of three dominant annual species *Setaria viridis*, *Artemisia sacrorum*, and *Chenopodium acuminatum*.

		Mass loss (%)				
Species	Litter depth	PF1	PF2	PF3	*F*	*P*
S. viridis	A	26.32 ± 3.44 bA	27.61 ± 3.10 bA	17.16 ± 1.7 7aA	23.747	<0.001
	B_{0-10}	48.10 ± 4.26 cB	40.40 ± 4.54 bB	29.28 ± 3.91 aB	29.785	<0.001
	B_{10-20}	56.98 ± 9.21 aC	52.06 ± 10.16 aC	49.03 ± 8.10 aC	1.143	0.345
	F	39.005	20.143	55.412		
	P	<0.001	<0.001	<0.001		
A. sacrorum	A	42.74 ± 4.26 bA	43.09 ± 3.16 bA	32.76 ± 4.85 aA	11.996	0.001
	B_{0-10}	58.36 ± 5.50 abB	62.20 ± 4.63 bB	50.10 ± 10.36 aB	4.333	0.033
	B_{10-20}	60.44 ± 4.81 aB	56.60 ± 8.60 aB	58.01 ± 8.28 aB	0.410	0.671
	F	23.535	16.497	15.065		
	P	<0.001	<0.001	<0.001		
C. acuminatum	A	35.91 ± 2.53 cA	32.81 ± 1.55 bA	23.50 ± 2.60 aA	48.418	<0.001
	B_{0-10}	55.89 ± 11.99 aB	61.79 ± 10.98 aB	46.86 ± 16.67 aB	1.875	0.188
	B_{10-20}	62.24 ± 7.68 aB	65.82 ± 7.54 aB	64.46 ± 6.19 aC	0.382	0.689
	F	16.240	32.476	23.527		
	P	<0.001	<0.001	<0.001		

"PF1", "PF2", and "PF3" represent 10 mm precipitation every 5 days (i.e. 28 rainfall events), 20 mm every 10 days (i.e. 14 events), and 40 mm every 20 days (i.e. 7 events), respectively; summing to a total of 280 mm. "A" represents aboveground litter, whereas "B_{0-10}" and "B_{10-20}" represent belowground litter at depths of 0–10 and 10–20 cm, respectively. Values labeled with different lower-case letters differed significantly ($p < 0.05$) between *PF* treatments for a given species; values labeled with different capital letters differed significantly between litter depths for a given species. Values are mean value ± standard deviation (ANOVA, $n = 6$).

× *PD* interaction (Table 2, Figure 2). The N loss also differed significantly among the species and the litter *PD* treatment variants, and showed a significant species × *PD* interaction, but unlike the C and mass losses, N loss was not significantly affected by *PF* (Table 2, Figure 3). We found strong and significant positive linear relationships between the mass loss and C loss in all three *PD* treatments (Figure 4(A–C)), but the rate of C loss for *S. viridis* was lower than for the other species (Figure 2). The influence of species but not *PF* on C loss also showed a significant interaction with litter *PD* (Tables 2 and 4). The N loss also showed a significant positive linear relationship with mass loss for all three *PD* treatment variants (Figure 4(D–F)), but the strength of the relationship was much weaker than that for C loss, and the loss rates (slopes) were also much smaller. The N loss in *A. sacrorum* was significantly higher than that in the other species, and was also significantly higher at the surface than in depth (Figure 3).

Discussion

Our study confirms previous research in which more frequent precipitation pulses led to increased decomposition rates (Schuster 2016; Joly, Kurupas, and Throop 2017), as we found that litter decomposition was greatly reduced in the low *PF* treatment for all three species (Table 3). This result can be explained by the soil water-holding capacity and the duration of high levels of soil moisture. Our study was conducted in a semi-arid grassland with a coarse sandy soil, in which soil water evaporation and infiltration were both rapid after precipitation events (Yao, Zhao, et al. 2013; Yao, Zhang, et al. 2013). A greater frequency of precipitation would create moisture pulses more frequently, thereby promoting microbial activity during the brief period when water

Table 4. Litter carbon (C) contents, nitrogen (N) contents, and C:N ratios after decomposition during one growing season, under the three precipitation-frequency (PF) treatments of three dominant annual species *Setaria viridis*, *Artemisia sacrorum*, and *Chenopodium acuminatum*.

		C content (% *w/w*)			N content (% *w/w*)			C/N		
Species	Litter depth	PF1	PF2	PF3	PF1	PF2	PF3	PF1	PF2	PF3
S. viridis	A	39.39 ± 2.13 aA	40.71 ± 1.41 abA	41.99 ± 1.21 bA	0.71 ± 0.10 aA	0.72 ± 0.08 aA	0.59 ± 0.17 aA	56.74 ± 10.06 aB	57.10 ± 7.67 aB	76.39 ± 20.82 bA
	B_{0-10}	45.60 ± 0.75 aB	44.99 ± 1.98 aB	45.48 ± 1.03 aB	1.02 ± 0.13 aB	0.93 ± 0.16 aB	0.79 ± 0.36 aA	45.28 ± 4.93 aA	49.15 ± 7.28 abAB	65.34 ± 21.74 bA
	B_{10-20}	45.82 ± 0.98 aB	44.85 ± 1.21 aB	44.81 ± 1.31 aB	1.03 ± 0.05 bB	0.99 ± 0.13 abB	0.88 ± 0.08 aA	44.68 ± 2.56 aA	46.04 ± 5.84 aA	51.42 ± 5.82 aA
A. sacrorum	A	44.83 ± 0.59 aA	45.27 ± 0.88 aA	45.00 ± 0.56 aA	1.00 ± 0.27 aA	0.84 ± 0.17 aA	1.07 ± 0.16 aA	47.35 ± 10.79 aB	56.24 ± 14.04 aB	42.97 ± 6.82 aB
	B_{0-10}	45.38 ± 0.78 aA	44.05 ± 1.16 aA	44.74 ± 0.63 aA	1.42 ± 0.17 bB	1.62 ± 0.06 cB	1.23 ± 0.18 aAB	32.39 ± 4.06 bA	27.18 ± 1.32 aA	37.00 ± 5.58 bAB
	B_{10-20}	44.97 ± 0.53 aA	44.34 ± 0.26 aA	44.49 ± 0.45 aA	1.64 ± 0.20 aB	1.35 ± 0.3 2aB	1.33 ± 0.17 aB	27.70 ± 3.13 aA	34.74 ± 10.09 aA	33.82 ± 4.25 aA
C. acuminatum	A	42.32 ± 0.65 aA	42.94 ± 0.48 aA	43.09 ± 0.56 aA	0.65 ± 0.05 cA	0.51 ± 0.07 bA	0.40 ± 0.10 aA	65.26 ± 5.46 aB	86.13 ± 11.45 aB	114.32 ± 31.85 bB
	B_{0-10}	44.48 ± 0.33 aB	41.97 ± 2.91 aA	43.72 ± 0.45 aA	1.13 ± 0.21 bB	1.21 ± 0.11 bB	0.90 ± 0.15 aB	40.35 ± 7.42 aA	34.97 ± 4.25 aA	49.59 ± 7.98 bA
	B_{10-20}	43.91 ± 0.65 aB	42.79 ± 1.24 aA	43.43 ± 0.84 aA	1.18 ± 0.20 aB	1.19 ± 0.12 aB	1.31 ± 0.21 aC	38.07 ± 7.16 aA	36.1 ± 3.23 aA	33.86 ± 5.4 2aA

"PF1", "PF2", and "PF3" represent 10 mm precipitation every 5 days (i.e. 28 rainfall events), 20 mm every 10 days (i.e. 14 events), and 40 mm every 20 days (i.e. 7 events), respectively; summing to a total of 280 mm. "A" represents aboveground litter, whereas "B_{0-10}" and "B_{10-20}" represent belowground litter at depths of 0–10 and 10–20 cm, respectively. Values labeled with different lower-case letters differed significantly ($p < 0.05$) between *PF* treatments for a given species; values labeled with different capital letters differed significantly between litter depths for a given species. Values are mean value ± standard deviation (ANOVA, $n = 6$).

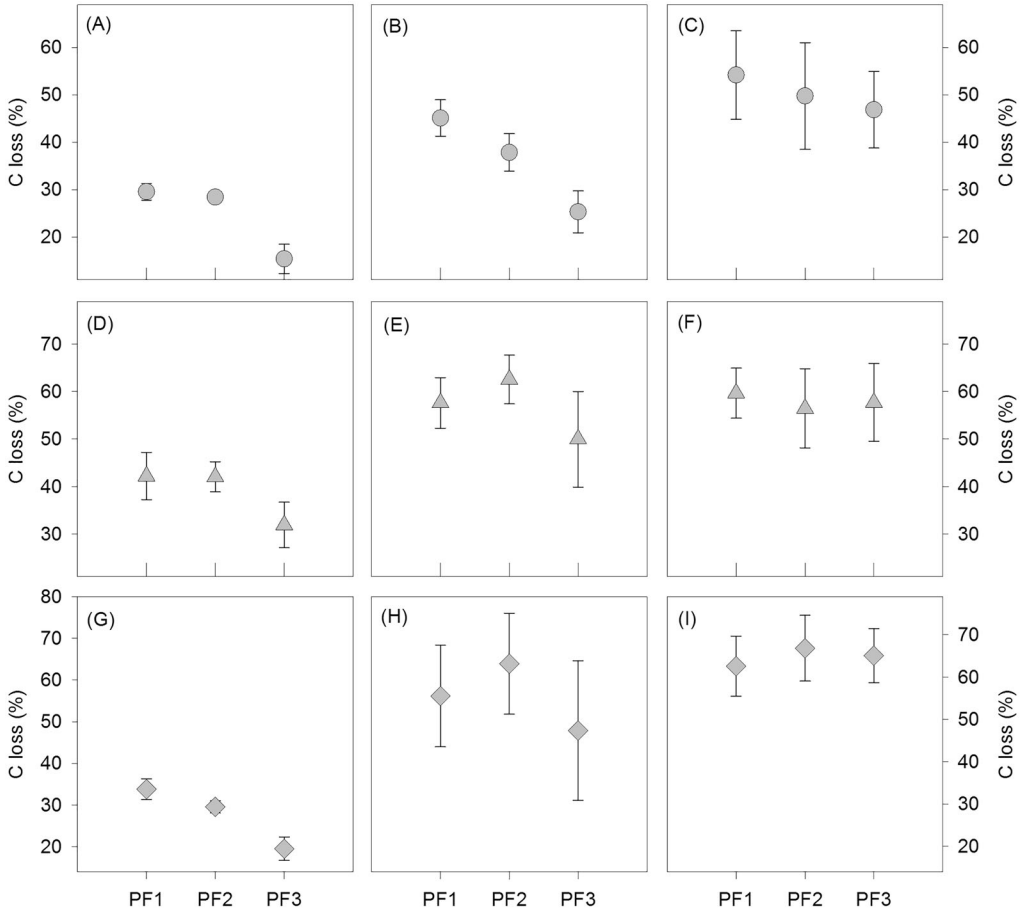

Figure 2. Litter carbon (C) loss after decomposition during one growing season under the three precipitation-frequency (PF) treatments of litter from three dominant annual species, *Setaria viridis* (A, B, and C), *Artemisia sacrorum* (D, E, and F), and *Chenopodium acuminatum* (G, H, and I). PF1, PF2, and PF3 represent precipitation that was provided as 10 mm every 5 days (i.e. 28 rainfall events), 20 mm every 10 days (i.e. 14 events), and 40 mm every 20 days (i.e. 7 events), respectively. Columns of graphs represent the same placement-depth for each species, aboveground (A, D, and G),) 0–10 cm (B, E, and H, and 10–20 cm below the surface (C, F, and I). Values are means ± standard deviation (ANOVA, $n = 6$).

remained available; thus, smaller and more frequent effective rainfall would promote litter decomposition more effectively than larger but less-frequent rainfall, even though the total amount of rainfall during the growing season did not change. The lower rate of litter decomposition at lower *PF* may also result from saturation of the litter water-holding capacity and from microbial activity (Joly, Kurupas, and Throop 2017).

Our results showed that aboveground litter decomposed significantly slower than the belowground litter (Table 3). This result was inconsistent with previous results. Li et al. (2016) demonstrated that litter with relatively high quality, such as litter with a low C:N ratio or a low lignin content, enhanced aboveground litter decomposition rate more than the belowground rate. However, in the present study, the C:N ratios of the

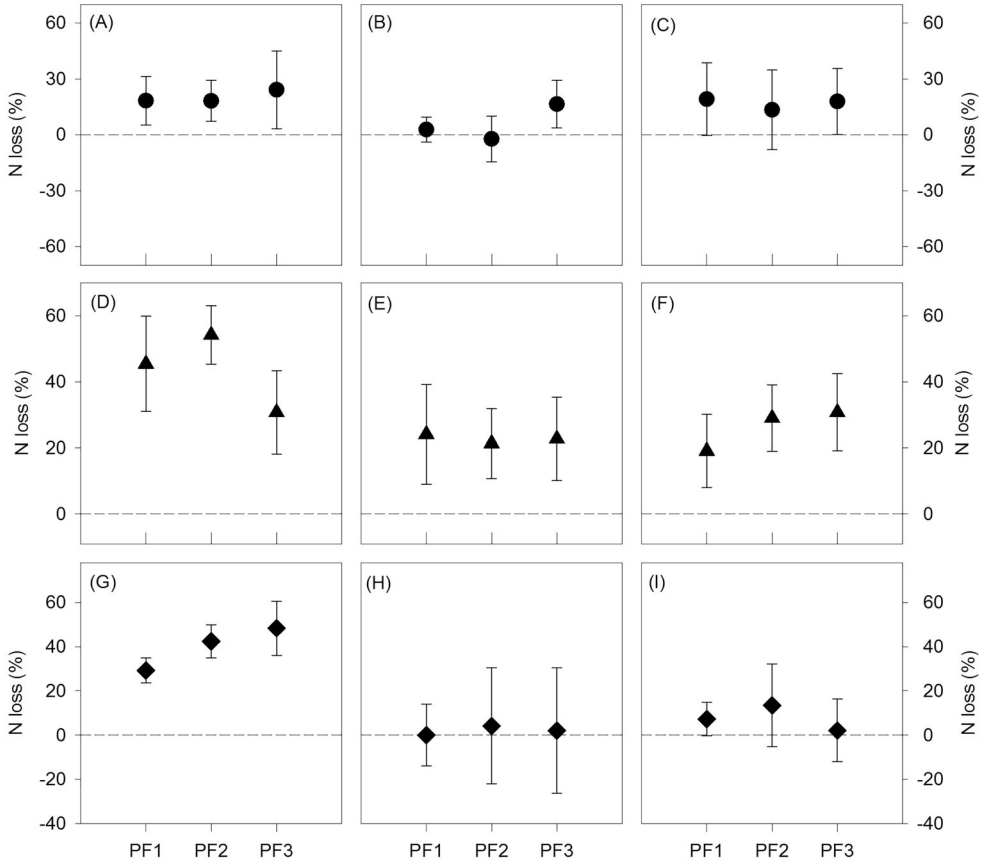

Figure 3. Litter nitrogen (N) loss after decomposition during one growing season under the three precipitation-frequency (PF) treatments of litter from three dominant annual species, *Setaria viridis* (A, B, and C), *Artemisia sacrorum* (D, E, and F), and *Chenopodium acuminatum* (G, H, and I). Negative losses represent an increase in the N content. PF1, PF2, and PF3 represent precipitation that was provided as 10 mm every 5 days (i.e. 28 rainfall events), 20 mm every 10 days (i.e. 14 events), and 40 mm every 20 days (i.e. 7 events), respectively. Columns of graphs represent the same placement-depth for each species, aboveground (A, D, and G),) 0–10 cm (B, E, and H, and 10–20 cm below the surface (C, F, and I). Values are means ± standard deviation (ANOVA, $n = 6$).

aboveground litter of all three species were lower than those of the belowground litter, and in all three *PF* treatment variants, the aboveground litter decomposed significantly slower than the belowground litter. This difference may be explained by the litter *PD*, which Austin, Araujo, and Leva (2009) considered to be a fundamental control on litter decomposition. They found that leaf litter and root litter decomposed equally fast when placed at the soil surface, regardless of significant differences in a number of litter quality parameters (Austin, Araujo, and Leva 2009). In the present study, the buried litter decomposed faster than the aboveground litter. This indicated that the tight contact between buried litter and the soil may have accelerated litter decomposition. This result echoes those of other studies, which demonstrated that mixing soil with litter consistently increased litter decomposition (Hewins et al. 2013; Lee et al. 2014; Joly, Kurupas, and Throop 2017; Levi et al. 2020). The buried litter decomposes more rapidly because

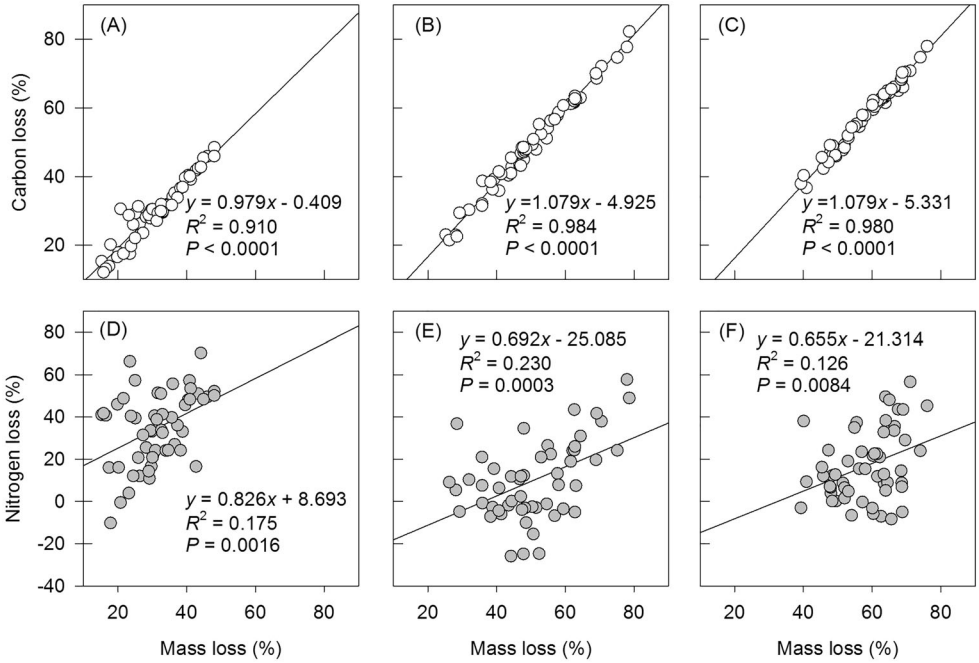

Figure 4. Relationships between mass loss and the C and N loss of the three species combined. Graphs in the same column represent the same placement-depth, such as aboveground (A, D), 0–10 cm (B, E), and 10–20 cm (C, F) below the surface.

following precipitation events moisture remains available longer to support microbial activity (Joly, Kurupas, and Throop 2017). This explanation was also supported by the fact that the deeper litter decomposed faster than the shallower litter in all three species (Table 3). In addition, the significant *PD* × *PF* interaction for all loss rates and contents (Table 2) can also be explained by an extended period of moisture availability; at low *PF* (i.e. PF3), rapid evaporation in the semi-arid environment would decrease water near the surface faster than in deeper layers, thereby decreasing litter decomposition rates near the surface. As a result, mean mass losses for all three species at 0–10 and 10–20 cm under low *PF* treatment variants averaged 42.1 and 57.2%, respectively.

A previous study (Parton et al. 2007) demonstrated that, during decomposition, the aboveground litter and roots exhibited different N dynamics. This agrees with our results (Figure 3), which showed lower rates of N loss in the deeper soil for two of the three species. Our study is consistent with this finding, because the aboveground litter showed greater N loss than the belowground litter during decomposition at all *PF* treatment variants and in all species, especially for *A. sacrorum* and *C. acuminatum* (Figure 3). A study of global-scale similarities in nitrogen release patterns during long-term decomposition found that, during decomposition, roots exhibited little net N immobilization (Parton et al. 2007). In the present study, the belowground litter showed N release less than about 20% and net N accumulation (a negative loss) for some *S. viridis* and *C. acuminatum* litter treatment variants (Fig. 3B,C,H,I), and this release rate interacted significantly with species (Table 2), with higher release rates for *A. sacrorum*

and lower rates for the other species (Figure 3). This contradiction between aboveground and belowground litter N dynamics may be due to the length of the decomposition period. Yahdjian, Sala, and Austin (2006) found that litter N contents initially decreased to about 70% of the starting value after 7 months (versus 5 months in the present study), then increased slightly. Li et al. (2016) conducted a study near our study site and found that during the decomposition of fine roots of *Artemisia halodendron*, the remaining N content increased, with slight variation, but finally showed a net loss of litter N in the control treatment (under natural precipitation).

Precipitation has complex effects on litter N during decomposition. For example, water addition significantly increased N loss from both leaves and fine roots of *A. halodendron*, *Phragmites australis* (Cav.) Trin. ex Steud., and *S. viridis* in the Horqin Sandy Land (Li et al. 2016). However, rainfall interception by rainout shelters (ranging from 30 to 80%) on the Patagonian steppe had no significant effect on litter N loss (Yahdjian, Sala, and Austin 2006). This disagreement between studies may be due to the effect of different *PF* values in the present study, since *PF* alters litter decomposition dynamics (Joly, Kurupas, and Throop 2017), and this was demonstrated by the fact that N loss was significantly linearly related to the mass loss during decomposition (Parton et al. 2007). A significant linear relationship between mass loss and N loss also existed in our study (Figure 4).

Conclusions

Our study demonstrated that *PF* played an important role in litter decomposition, as lower *PF* significantly inhibited litter decomposition of the three dominant annual species in a semi-arid sandy grassland in northeastern China after decomposition during one growing season. The effect of lower *PF* on litter decomposition differed among litter *PD* treatment variants. This supports our first research hypothesis. The belowground litter decomposed faster than the aboveground litter, and the litter of graminaceous *S. viridis* decomposed slower than non-graminaceous *A. sacrorum* and *C. acuminatum*. These findings support our second and third hypotheses, respectively. The litter C content after decomposition was affected by the *PD* and the *PF* treatments, and their interaction; and litter C loss was closely related to the mass loss. After decomposition the litter N content increased greatly, especially for belowground litter. However, there was no accumulation of litter N after decomposition. Litter N loss, which was linearly related to the litter mass loss, differed among species and *PD* treatment variants in the soil, but *PF* treatment did not significantly affect litter N loss. Precipitation frequency should therefore be accounted for in future research on the relationship between precipitation and litter decomposition, and quantifying this relationship would provide theoretical support for the establishment of regional carbon cycle models and carbon budget predictions, especially in semi-arid regions. The differences in decomposition rate between species and *PD* treatment variants suggested that, in semi-arid regions in northeastern China, non-graminaceous species may have greater potential in grassland management due to their faster decomposition rates (i.e. their greater ability to increase soil C content), especially for their belowground litter.

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