



Effect of grassland afforestation on soil N mineralization and its response to soil texture and slope position



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ABSTRACT

Grassland afforestation, which is a major form of land-use change, has expanded considerably in the last few decades. However, the effect of conversion from grassland to shrubland on soil nitrogen (N) cycling and availability with regard to soil textures and slope positions is unclear. This lack of clarity has hindered our understanding of how grassland afforestation influences soil N availability and transformation on a larger spatial scale. In this study, we compared the net N mineralization rates and mineral N concentrations in soils from native grassland and under legume shrub (*Caragana korshinskii* Kom.) with contrasting soil textures (sandy-loam vs. loamy-sand) and slope positions (upper vs. lower) in the northern region of China's Loess Plateau. The objective of this study is to understand how soil texture and slope position regulate the response of soil N cycling to grassland afforestation. The results showed that the concentrations of soil nitrate and total mineral N and the rates of net nitrification and mineralization were higher in soils under shrub ($0.64 \pm 0.10 \text{ g N m}^{-2}$, $1.01 \pm 0.11 \text{ g N m}^{-2}$, $23.19 \pm 5.22 \text{ mg N m}^{-2} \text{ d}^{-1}$ and $22.07 \pm 5.44 \text{ mg N m}^{-2} \text{ d}^{-1}$) than in grassland ($0.22 \pm 0.03 \text{ g N m}^{-2}$, $0.56 \pm 0.04 \text{ g N m}^{-2}$, $1.48 \pm 1.51 \text{ mg N m}^{-2} \text{ d}^{-1}$ and $1.36 \pm 1.96 \text{ mg N m}^{-2} \text{ d}^{-1}$). However, soil ammonium concentration and net ammonification rate were not affected by grassland afforestation. The effect of afforestation on nitrate and total mineral N concentrations was greater in the lower slope than in the upper slope but was similar between the sandy-loam and loamy-sand soils. The effect of this land-use change on net nitrification and mineralization rates varied with soil texture and slope position, with greater effects in the upper slope than in the lower slope in more clayey soils, but with the opposite influencing pattern in more sandy soils. These results suggested that planting *C. korshinskii* in grassland ecosystems increased soil N mineralization and availability. More importantly, soil texture and slope position should be considered when assessing the effects of grassland afforestation in complex landscape conditions.

1. Introduction

As part of efforts to mitigate anthropogenic carbon (C) emission, prevent soil erosion and increase economic benefits, there has been considerable expansion of afforestation of grassland as a form of major land-use change in the past few decades (Fahey and Payne, 2017; Ozalp and Cavdar, 2016). With greater net primary productivity, the conversion of grassland to woodland can increase the sequestration of carbon dioxide in plant biomass (McKinley and Blair, 2008; Mercedes Vassallo et al., 2013) and influence soil organic C (OC) stock with gains in drier regions and losses in wetter regions (Berthrong et al., 2012).

This form of land-use conversion also changes soil nitrogen (N) storage, as C and N cycling are closely related (Calazans et al., 2018). Given that soil N availability is often a limiting factor in many terrestrial ecosystems, it is necessary to understand the ecological consequences of grassland afforestation on soil N dynamics (Chapin et al., 2011).

Over 90% of the N in soils is in the organic form (Kelley and Stevenson, 1995), most of which needs to be transformed into inorganic N via microorganisms (mineralization) before assimilation by plants (Booth et al., 2005). Generally, increases in the net primary productivity and C allocation from belowground to aboveground result in variations in the quantity and quality of litters that would alter soil N

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mineralization (McKinley and Blair, 2008; Uri et al., 2008). Soil N mineralization was also regulated by soil temperature and moisture. Soil N mineralization was enhanced by increasing the temperature (Dalias et al., 2002) and was highest at 80–100% of field capacity (Guntinas et al., 2012). Moreover, soil pH has an influence on N mineralization by altering the activity of nitro bacteria and increasing pH-stimulated N mineralization in slightly acidic soils (Curtin et al., 1998). Afforestation often results in the shifts of these factors, and thus alters soil N mineralization and availability (McKinley and Blair, 2008). Compared with the number of studies on afforestation of croplands or bare-lands (e.g. Li et al., 2014; Rahman et al., 2017), few studies have focused on changes in soil N mineralization after conversion grassland to woodland (but see McKinley and Blair, 2008; Peichl et al., 2012; Scowcroft et al., 2004). The establishment of woodland on grassland can result in the accumulation of soil organic matter (SOM), which, in turn, can increase the total N pool (Wei et al., 2010). However the rate of N mineralization can be reduced due to immobilization (McKinley and Blair, 2008) or be increased due to the priming effect (Kuzyakov et al., 2000), thus influencing the dynamics of mineral N in soil solution.

The mineralization and availability of N in soils are largely dependent on topography, soil texture, soil depth and their interactions with environments (Hook and Burke, 2000; Uri et al., 2008; Wang et al., 2014a,b; Ge et al., 2019). The slope position influences soil N cycling by redistributing SOM and sediment along landscapes and by altering soil moisture and temperature (Bennie et al., 2008; Scowcroft et al., 2004). The net N mineralization rate and the mineral N concentration are higher in the lower slope than in the upper slope as SOM and soil moisture conditions decrease with the increasing slope position (Hishi et al., 2014; Soon and Malhi, 2005). However, greater vegetation cover and higher soil temperature in the upper slope can result in a higher mineral N concentration (Liu et al., 2018). The association of SOM with clay particles protects SOM from mineralization (Côté et al., 2000; Hassink, 1994), while this protection varies with soil environments (Scott Bechtold and Naiman, 2006). Soil texture varies greatly with spatial scales of the landscape (Campos et al., 2007), while soil moisture and temperature not only vary in space and time but also respond significantly to land-use change (Jiang et al., 2015). For example, Takahashi and Murayama (2014) found that the coarseness of soil texture increases with the increasing slope position on Mount Norikura in central Japan. Kidron (2009) showed that planting shrubs on bare land reduced evaporation and soil temperature and extended soil wetness after rain in the west Negev Desert. Moreover, due to variations of organic matter inputs, temperature, moisture and microbial activity with soil depth, N mineralization and availability often decrease with soil depth in woodland (Wang et al., 2011) and grassland (Liu et al., 2018), and the response of N turnover and availability to land-use change was greater in top soils than in deep soils (Yao et al., 2019). How grassland afforestation interacts with soil texture, soil depth and the landscape position in influencing soil N cycling in complex landscape conditions is limited.

In this study, we presented the dynamics of soil N availability and mineralization during the growing season, as affected by the planted shrub (*Caragana korshinskii* Kom.) on grasslands in the northern region of China's Loess Plateau. We also related the effect of grassland afforestation to soil textures (sandy-loam and loamy-sand) and slope positions (upper and lower slope). The legume shrub *C. korshinskii* has been widely planted on the Loess Plateau since the 1980s to serve as wind breaker and sand fixer (He et al., 2008; Wei et al., 2013). The *C. korshinskii* can survive under harsh conditions (e.g., dry and cold weather) and has the potential to increase soil fertility by fixing N from the atmosphere. China's Loess Plateau has an area of 640,000 km², with soils varying from sandy soil in the northwest to clayey soil in the southeast. The region is characterized by undulating hills and gullies (Liu, 1985), and the vegetation cover has increased from 31.6% in 1999 to 59.6% in 2013 due to vegetation restoration (Chen et al., 2015). The variable soil

texture, complex landscape and extensive land-use change have made the Loess Plateau an ideal platform with which to investigate the interactions of these factors on soil N cycling. The objectives of this study are to understand how conversion of grassland to *C. korshinskii* shrubland influenced soil N availability and mineralization, and to investigate whether these effects depended on soil texture and slope position.

2. Materials and methods

2.1. Site description

This study was carried out in the Liudaogou watershed (with latitude of 38°46'–38°51'N, longitude of 110°21'–110°23'E and elevation of 1094–1274 m above sea level) in the northern region of China's Loess Plateau. The watershed is characterized by a semiarid continental monsoon climate. Based on the climate data from 1961 to 2014, the mean annual precipitation is 420.8 mm, 76.6% of which occurs between June and September. The mean annual air temperature is 9.0 °C, with the monthly minimum of –9.0 °C (in January) and maximum of 24.0 °C (in July).

The watershed has an area of 6.89 km² and is mainly characterized by sloping land, which accounts for 76.5% of the total area. Major soil types in this watershed are Castanozes, Skeletal, Loessial and Aeolian, which cover 92% of the total area of the watershed and have a wide range of physical clay (< 0.01 mm) content (from 13.9%–36.1%) (Wei et al., 2010). The dominant vegetation is xerophytic shrub (e.g., *Artemisia ordosica* Krach., *C. korshinskii* and *Salix psammophila* C.) and grass (e.g., *Stipa bungeana* Trin., *Stipa breviflora* Griseb., *Artemisia gmelinii* and *Medicago sativa* L.) (She, 2009; Wei et al., 2013). The harsh natural conditions and poor land management have resulted in severe environmental problems, such as soil erosion and land degradation. In the 1980s, a leguminous shrub species, *C. korshinskii*, was widely introduced to grass slopes in the northern Loess Plateau by the local government to control soil erosion and to restore degraded lands, including nearly half of the grass slopes in the Liudaogou watershed (Wei et al., 2013).

2.2. Experimental design, field and laboratory analyses

This study was designed to examine the effects of afforestation, soil texture, slope position, soil depth and time of the season on soil N mineralization and availability. The experiment included two soils (Skeletal vs. Aeolian) of contrasting texture, two slope positions (upper vs. lower slopes) and two soil depths (0–10 cm vs. 10–20 cm). At each combination of soil textures and slope positions, shrubland and grassland were selected to compare the effect of grassland afforestation. The experiment was conducted throughout a growing season, which allowed us to test the effect of the time of season.

Field investigations were carried out between April and October in 2008 on a Skeletal slope (260 m in length, 200 m in width, 45 m in altitude gradient) and on an Aeolian slope (200 m in length, 140 m in width, 35 m in altitude gradient). Both slopes have similar gradient (10°) and aspect (northwest). The Skeletal and Aeolian soils account respectively for 35.9% and 13.5% of the study watershed. The soils belong to Chernozems and Arenosols according to the FAO/ISRIC/ISSS taxonomy, with texture of sandy-loam and loamy-sand. The *C. korshinskii* was established on the native grassland approximately 30 years ago. The dominant native grass is benge needlegrass (*S. bungeana*) (He et al., 2008; Wei et al., 2013).

For each slope, the sampling transects were set at the summit (upper slope) and toe (lower slope) positions. Three *C. korshinskii* were randomly selected as replicates for each slope position to determine the growth conditions, including canopy area, height, base diameter and branch number. The aboveground biomass of *C. korshinskii* was calculated according to the allometric model (Yang et al., 2016, Method S1),

which was developed for the same species in the same study site as those of this study. The growth conditions of *C. korshinskii* are presented in Table S1. In mid-August, the aboveground biomass of fresh grass beneath *C. korshinskii* and that at the center of the open interspaces (grassland) was determined by clipping an area of 1 × 1 m, and the litters were collected along with the leaves of *C. korshinskii*. The plant samples were oven-dried at 65 °C for 72 h, weighted and analyzed for C and N concentrations (Table S2).

For each combination of soil texture, slope position and vegetation type, 3 plots (3 × 3 m) were established for the *in situ* measurement of soil N mineralization by using the undisturbed buried core method (Robertson et al., 1999). In each plot, a sharpened PVC core (5 cm in diameter and 22 cm in length) was driven into the soil to the depth of 20 cm. The PVC core was pegged at the location of the mid-canopy beneath *C. korshinskii* and at the center of the grass interspace. The core was covered with permeable plastic film so that it prevented precipitation from soaking the ground, but allowed gas exchange. Soils inside the cores were incubated for five consecutive periods (25 Apr to 25 May, 25 May to 21 Jun, 21 Jun to 23 Jul, 23 Jul to 23 Aug and 23 Aug to 16 Oct). Soils near the core were collected in the 0–10 and 10–20 cm soil layers at the start of each incubation period, and soils inside the core were collected in the same layers at the end of each incubation period. The soil samples were immediately put into plastic bags and stored at 4 °C before extraction with 2 M KCl in the laboratory (Kachurina et al., 2000). The concentrations of nitrate (NO₃⁻) and ammonium (NH₄⁺) in the extractions were analyzed by using a continuous flow analyzer (AutoAnalyzer-AA3, Seal Analytical, Norderstedt, Germany). Total mineral N (Min-N) concentration was calculated as the sum of NO₃⁻-N and NH₄⁺-N. Soil moisture was measured by oven-dry method at 105 °C for 24 h. The precipitation, air temperature and soil moisture during the experimental period are presented in Fig. 1. The concentrations of NO₃⁻-N, NH₄⁺-N and Min-N in samples before incubation were subtracted from those after incubation, and then were divided by the days of incubation for the calculation of the net nitrification (R_n), ammonium (R_a) and mineralization (R_m) rates, respectively.

2.3. Statistical analyses

A multiway analysis of variance (ANOVA) was conducted to test the direct effect of grassland afforestation (grassland and *C. korshinskii*), soil texture (sandy-loam and loamy-sand), slope position (lower and upper slope), soil depth (0–10 and 10–20 cm soil depth), sampling date (25 Apr, 25 May, 21 Jun, 23 Jul, 23 Aug and 16 Oct) or incubation period (25 Apr to 25 May, 25 May to 21 Jun, 21 Jun to 23 Jul, 23 Jul to 23 Aug and 23 Aug to 16 Oct) and their interactions on soil mineral N concentrations and mineralization rates. Student's *t*-test was used to examine whether the difference of soil metrics between grassland and *C. korshinskii* on a given sampling date or during a given incubation period was significant. A simple linear correlation was conducted to establish the relationships of R_m, R_n and R_a to soil moisture, bulk density (BD) and C/N of litters. Statistical analyses were performed using JMP 10.0 (SAS Institute, Cary, USA).

3. Results

3.1. Soil mineral nitrogen

Soil NO₃⁻-N and Min-N concentrations were not influenced by soil texture, slope position, soil depth or their interactions (*p* > 0.05). However, the NO₃⁻-N and Min-N concentrations generally decreased throughout the growing season, probably due to the plant uptake of NO₃⁻-N and leaching of NO₃⁻-N during the rains (Table 1, Figs. 2 and 3). For example, when averaged across all sources of variation, the NO₃⁻-N and Min-N concentrations decreased from 0.93 and 1.29 g m⁻² at the start of the growing season to 0.09 and 0.37 g m⁻² at the end of the

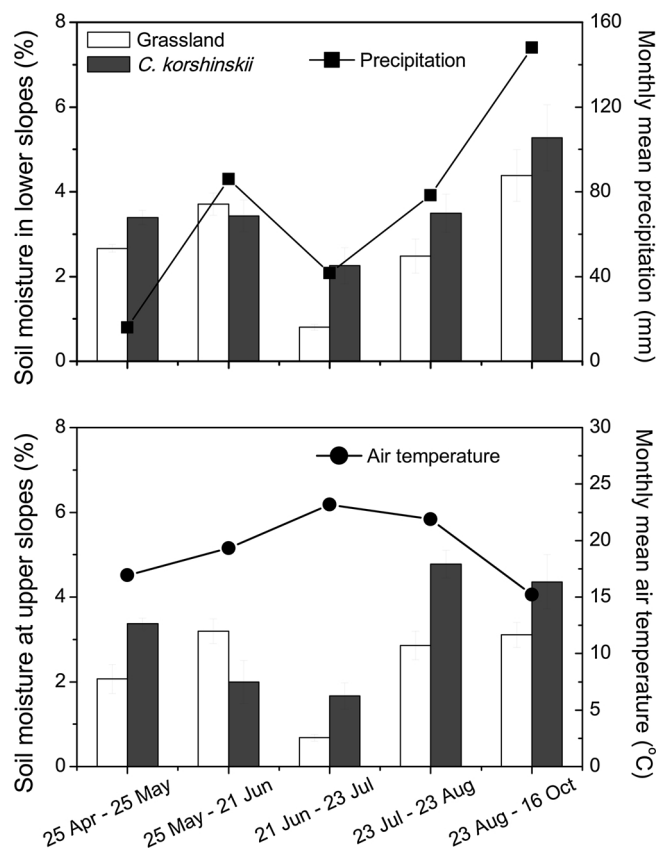


Fig. 1. Mean precipitation and air temperature during the 2008 growing season in the study area, and monthly dynamics of soil moisture averaged across the 0–10 and 10–20 cm soil layers in grassland and beneath *C. Korshinskii* in the lower and upper slope positions. Error bar indicates two standard errors of the mean.

Table 1

Results of ANOVA test (*p* values) for the concentrations of nitrate (NO₃⁻), ammonium N (NH₄⁺) and total mineral N (Min-N) in soil solution and rates of net nitrification (R_n), ammonification (R_a) and mineralization (R_m). The independent variables include vegetation (V), soil texture (T), slope position (P), soil depth (D) and time of season (S). The time of season is sampling dates for mineral N and incubation periods for net N mineralization, respectively. For three- and four-way interactions, only those with significant interaction (*p* < 0.05) are reported in the table. Bold values indicate a statistical significance (*p* < 0.05).

Sources of variance	NO ₃ ⁻	NH ₄ ⁺	Min-N	R _n	R _a	R _m
V	< 0.001	0.954	0.001	< 0.001	0.345	< 0.001
T	0.968	0.141	0.699	0.017	0.085	0.010
P	0.255	0.888	0.349	0.042	0.084	0.024
D	0.828	0.647	0.768	< 0.001	0.014	< 0.001
S	0.003	0.036	0.002	0.097	0.666	0.146
V × T	0.419	0.380	0.633	0.464	0.232	0.672
V × P	0.015	0.012	0.008	0.156	0.451	0.138
V × D	0.917	0.980	0.934	0.001	0.820	0.002
V × S	0.033	0.288	0.039	0.136	0.732	0.143
T × P	0.418	0.062	0.256	0.007	0.665	0.015
T × D	0.056	0.945	0.107	0.687	0.194	0.512
T × S	0.851	0.153	0.615	0.729	0.531	0.852
P × D	0.104	0.296	0.103	0.687	0.949	0.718
P × S	0.250	0.361	0.232	0.003	0.179	0.002
D × S	0.320	0.712	0.448	0.294	0.878	0.312
V × T × D	0.346	0.250	0.281	0.435	0.030	0.799
V × P × S	0.062	0.774	0.127	0.011	0.840	0.015
T × P × D	0.311	0.101	0.210	0.004	0.183	0.003
V × T × P × D	0.262	0.849	0.317	0.005	0.367	0.015

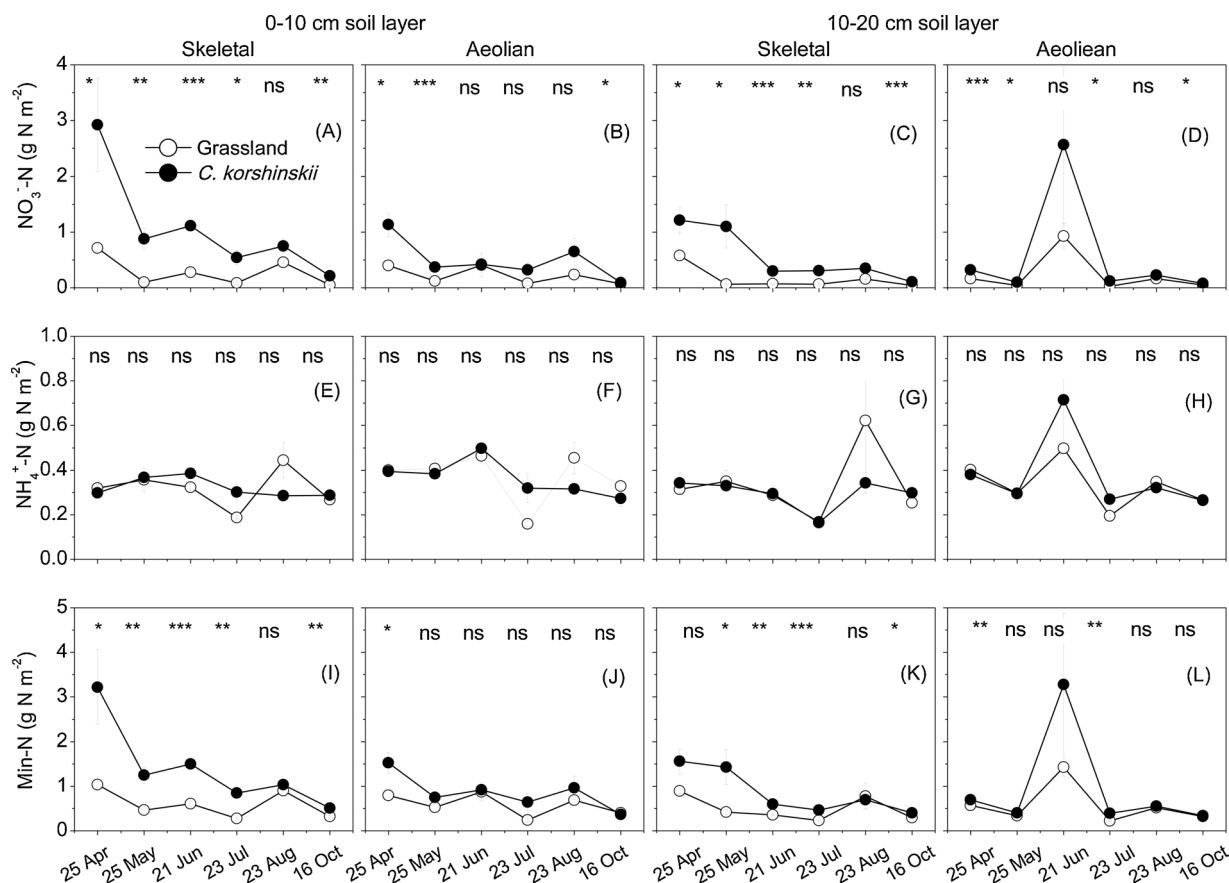


Fig. 2. Monthly dynamics of nitrate (NO_3^-), ammonium (NH_4^+) and total mineral nitrogen (Min-N) concentrations in soil solution in grassland and beneath *C. Korshinskii* for Skeletal and Aeolian soils during the growing season. Error bar indicates two standard errors of the mean. *, **, *** indicate the differences between grassland and *C. korshinskii* were significant at $p < 0.05$, 0.01 and 0.001 , respectively. ns: not significant.

season, respectively. Furthermore, seasonal patterns of NO_3^- -N and Min-N were similar among soil texture, slope position and soil depth (Table 1). Soil NH_4^+ -N concentration was significantly higher on 21 Jun and 23 Aug but lower on 23 Jul ($p = 0.036$) either across or within soil texture, slope position and soil depth (Figs. 2 and 3).

The conversion of grassland to *C. korshinskii* shrubland significantly increased soil NO_3^- -N by 0.45 g m^{-2} (+202%, $p < 0.001$) and Min-N by 0.45 g m^{-2} (+81%, $p = 0.001$) (Table 1, Figs. 2 and 3). The effect of afforestation was consistent across soil texture and soil depth ($p > 0.05$ for the interactions of vegetation and soil texture or soil depth, Table 1). For example, when averaged across slope position and sampling date, the increase of Min-N concentration in the 0–10 and 10–20 cm soil layers after conversion of grassland to shrubland were 132% and 73%, respectively, in Skeletal soils and 47% and 67%, respectively, in Aeolian soils (Fig. 2). However, the effect of this land-use conversion on NO_3^- -N and Min-N concentrations varied with slope position and sampling date ($p < 0.05$ for the interactions of vegetation and slope position or sampling date, Table 1). The increases in NO_3^- -N and Min-N concentrations due to land-use conversion were higher in the lower slope (+407% and +158%, $p < 0.05$) than in the upper slope (+74% and +22%, $p > 0.05$). These increases were also higher during the period from 25 Apr to 21 Jun (average of +334% and +109%, $p < 0.05$) than during the period from 23 Jul to 16 Oct (+127% and +42%, $p > 0.05$). Moreover, soil NH_4^+ -N concentration was affected by the interaction between land-use conversion and slope position ($p = 0.012$), with 20% higher in shrubland than in grassland in the lower slope but 19% lower in the upper slope. Thus, the effect of land-use change on soil mineral N concentrations depended on slope position and time of the season, irrespective of soil texture and soil depth.

3.2. Soil nitrogen mineralization

The R_n and R_m were significantly higher in the 0–10 cm (22.07 and $22.77 \text{ mg N m}^{-2} \text{ d}^{-1}$) than those in the 10–20 cm (2.59 and $0.66 \text{ mg N m}^{-2} \text{ d}^{-1}$) soil layer ($p < 0.05$, Table 1, Figs. 4 and 5). The R_n and R_m were significantly affected by soil texture, slope position and their interactions ($p < 0.05$, Table 1). For Skeletal soils, R_n and R_m were significantly higher in the upper slope (28.75 and $29.73 \text{ mg N m}^{-2} \text{ d}^{-1}$) than those in the lower slope (6.89 and $6.50 \text{ mg N m}^{-2} \text{ d}^{-1}$). For Aeolian soils, R_n and R_m were similar between the upper (5.28 and $4.89 \text{ mg N m}^{-2} \text{ d}^{-1}$) and lower (8.41 and $5.74 \text{ mg N m}^{-2} \text{ d}^{-1}$) slope (Fig. 6). Furthermore, R_n and R_m were significantly influenced by the interaction of slope position and incubation period ($p < 0.05$, Table 1). The R_n and R_m increased over the season in the lower slope, but unchanged over the season in the upper slope, particularly for the 0–10 cm soil layer (Fig. 5).

By grouping data together, R_n and R_m were significantly increased by $21.71 \text{ mg N m}^{-2} \text{ d}^{-1}$ (+1467%) and $20.71 \text{ mg N m}^{-2} \text{ d}^{-1}$ (+1522%), respectively, after conversion of grassland to shrubland ($p < 0.05$, Table 1, Figs. 4 and 5). These increases were significant for the 0–10 cm soil layer (37.74 and $36.51 \text{ mg N m}^{-2} \text{ d}^{-1}$, $p < 0.05$) but were not significant (5.66 and $4.90 \text{ mg N m}^{-2} \text{ d}^{-1}$, $p > 0.05$) for the 10–20 cm soil layer. Furthermore, the responses of N mineralization to grassland afforestation varied with slope position, soil texture and soil depth ($p < 0.05$ for their interactions). For the 0–10 cm soil layer, the effects of this land-use change on R_n and R_m were higher in the upper slope (+68.11 and +59.94 $\text{mg N m}^{-2} \text{ d}^{-1}$) than in the lower slope (+21.19 and +19.74 $\text{mg N m}^{-2} \text{ d}^{-1}$) for Skeletal soils but were higher in the lower slope (+42.81 and +44.05 $\text{mg N m}^{-2} \text{ d}^{-1}$) than in the upper slope (+18.89 and +22.33 $\text{mg N m}^{-2} \text{ d}^{-1}$) for Aeolian soils (Fig. 6).

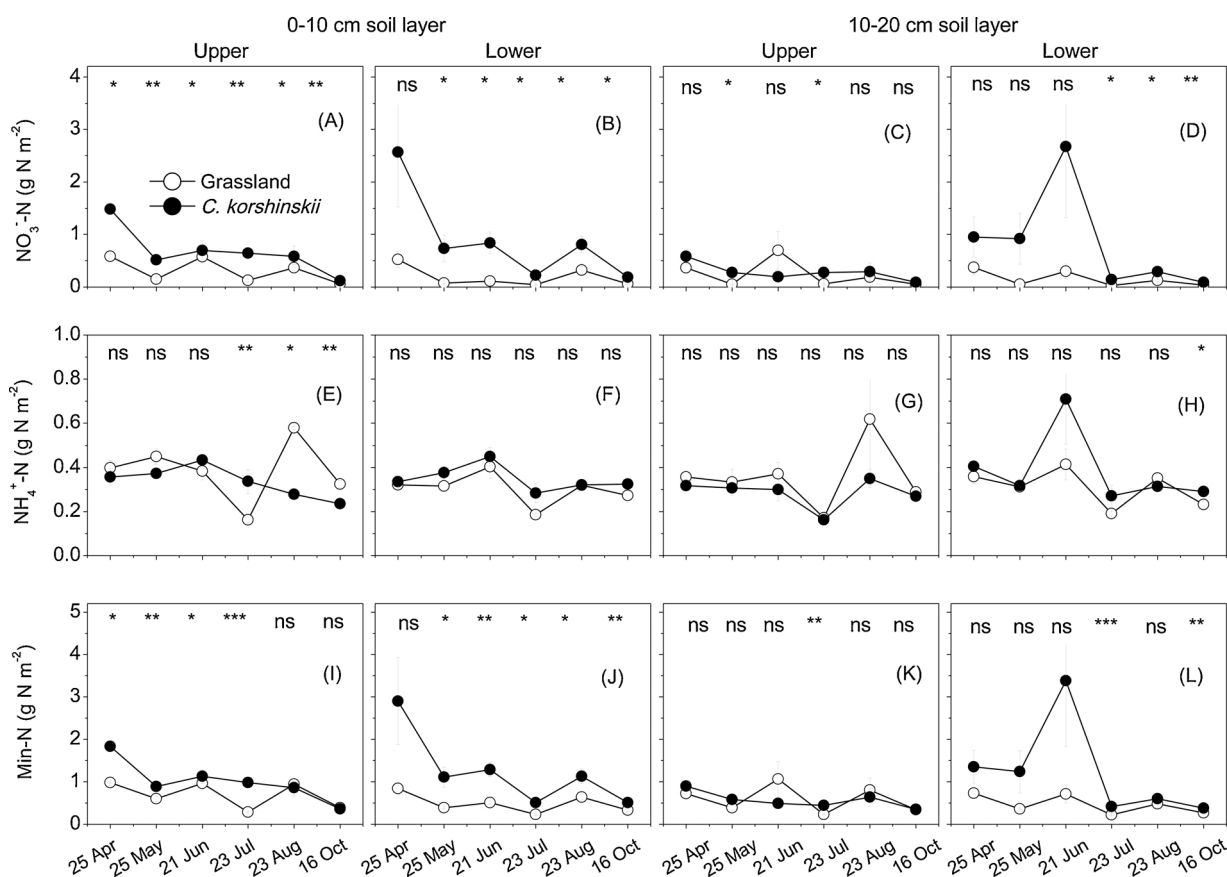


Fig. 3. Monthly dynamics of nitrate (NO_3^-), ammonium (NH_4^+) and total mineral nitrogen (Min-N) concentrations in soil solution in grassland and beneath *C. korshinskii* in the upper and lower slopes during the growing season. Error bar indicates two standard errors of the mean. *, **, *** indicate the differences between grassland and *C. korshinskii* land were significant at $p < 0.05$, 0.01 and 0.001 , respectively. ns: not significant.

The effect of grassland afforestation on R_n and R_m also depended on the interaction of slope position and incubation period (Table 1). After conversion of grassland to shrubland, R_n and R_m in the lower slope significantly increased for the incubation period from 23 Jul to 16 Oct, but significantly increased for the incubation period from 21 Jun to 23 Jul in the upper slope (Fig. 5).

The R_a was significantly higher in the 0–10 cm ($0.70 \text{ mg N m}^{-2} \text{ d}^{-1}$) than in the 10–20 cm ($-1.93 \text{ mg N m}^{-2} \text{ d}^{-1}$) soil layer ($p < 0.05$), regardless of slope position and time of the season ($p > 0.05$ for their interactions, Table 1, Figs. 4 and 5). The R_a was significantly affected by the interaction among vegetation type, soil texture and soil depth ($p = 0.03$, Table 1). For example, the conversion of grassland to shrubland decreased R_a by 102% in the 0–10 cm soil layer and remained unchanged in the 10–20 cm soil layer in Skeletal soils, but did not affect R_a in both soil layers in Aeolian soils (Fig. 6). Therefore, the effect of grassland afforestation on soil N mineralization rates at the 0–10 cm soil depth interacted with soil texture, slope position, and time of season.

3.3. Soil moisture

Soil moisture varied significantly over time of the season, with lower soil moisture during the period from 21 Jun to 23 Jul than in all other periods, and this seasonal dynamics was consistent with that of precipitation for the growing season (Fig. 1). Soil moisture was not affected by soil texture or soil depth ($p > 0.05$), but varied significantly with slope position and vegetation type ($p < 0.05$). Specifically, soil moisture was significantly higher in the lower slope than the upper slope and was higher in soils beneath shrub than in grassland (Fig. 1). Furthermore, grassland afforestation increased soil moisture,

and this effect varied with soil texture and time of the season ($p < 0.05$ for their interactions), with a higher increase in Skeletal than in Aeolian soils, and with the highest increase occurring during the period of 21 Jun to 23 Jul throughout the growing season (Fig. 1). Furthermore, R_n and R_m were related with soil moisture and increased with increasing soil moisture (SM) ($R_n = 5.16 \times \text{SM} - 3.14$, $p < 0.01$; $R_m = 4.95 \times \text{SM} - 3.12$, $p < 0.01$). However, R_a was not affected by soil moisture ($R_a = -0.21 \times \text{SM} + 0.02$, $p > 0.05$).

4. Discussions

4.1. Effects of afforestation

Our study suggested that planting leguminous shrub species (*C. korshinskii*) increased soil mineral N concentration and N mineralization. These increases were attributed to the accumulation of organic matter in the surface soils after conversion of grassland to shrubland. The decomposition of organic matter accelerates N mineralization and mineral N release, which in turn increases soil mineral N concentration (Kuzayakov et al., 2000). In this study, the OC was 78% higher in soils under *C. korshinskii* than soils in interspace grassland (Table S3). Secondly, the legume shrub fixed N into the soil, which led to higher mineral N concentration and mineralization rate in shrubland than in grassland soils due to the priming effect of new input N (Luce et al., 2016). This explanation was supported by the meta-analysis that N addition increased soil net N mineralization in the terrestrial ecosystem (Lu et al., 2011) and by the field observations that N addition increased soil inorganic N pools and net N mineralization in the grassland ecosystem (Wang et al., 2014a, 2014b; Wei et al., 2017). In this study, soil total N concentration was 81% higher in soils beneath *C. korshinskii*

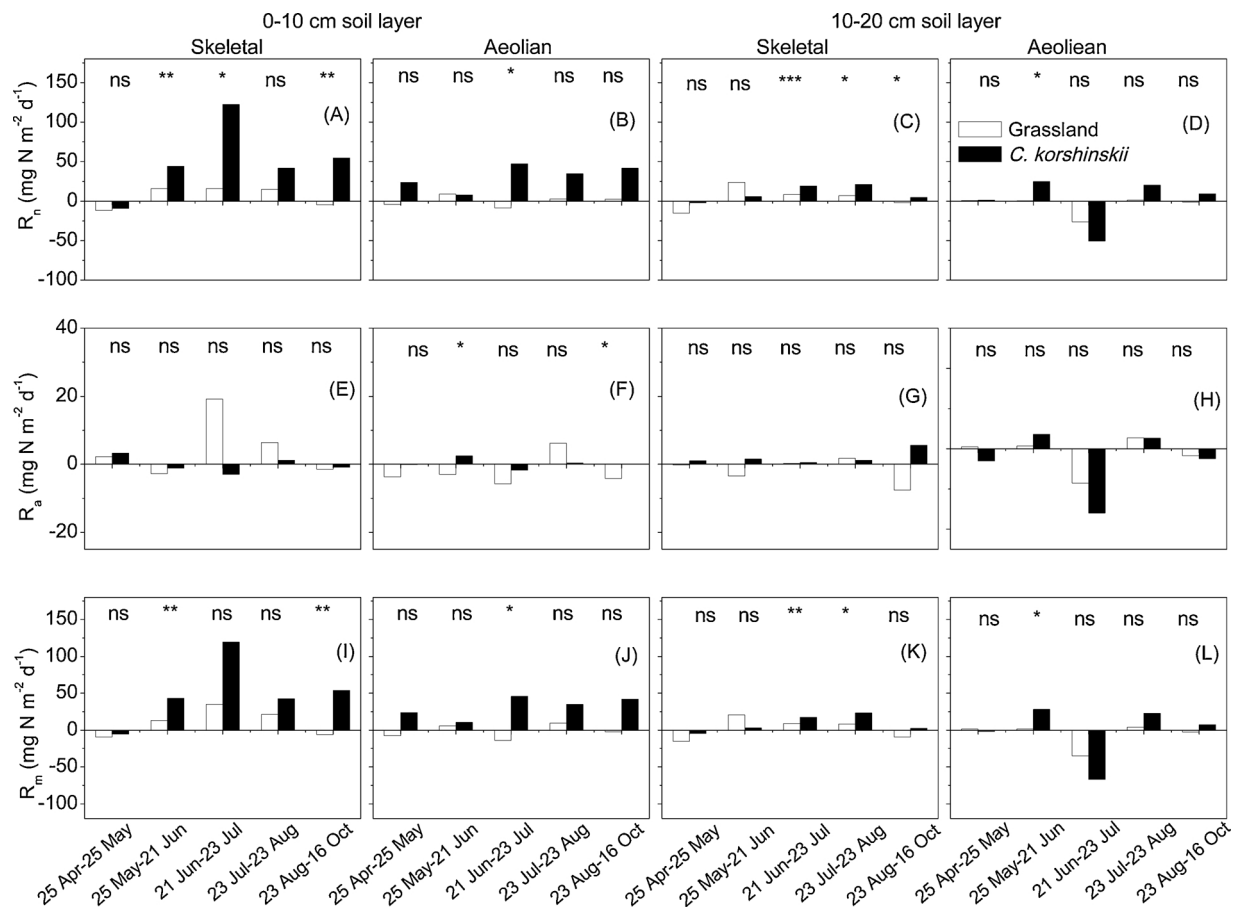


Fig. 4. Monthly dynamics of soil net nitrification (R_n), ammonification (R_a) and mineralization (R_m) rates in grassland and beneath *C. korshinskii* for Skeletal and Aeolian soils during the growing season. Error bar indicates two standard errors of the mean. *, **, *** indicate the differences between grassland and *C. korshinskii* land were significant at $p < 0.05$, 0.01 and 0.001 , respectively. ns: not significant.

than in grassland (Table S3), further supporting this explanation. Additionally, the higher N concentration in leaves and litters of legume shrub than those of grass could also contribute to the increased N mineralization and, thus, the concentrations of mineral N. The leaves of *C. korshinskii* have higher N concentration (42.94 g kg^{-1}) but lower C/N ratio (12.44) than those of grass (13.51 g kg^{-1} and 38.13 , respectively). The litter of *C. korshinskii* also had a lower C/N ratio (20.92) than that of grassland litter (37.06) (Table S2), and R_m negatively related with C/N ratio of litters ($R_m = 23.31 - 0.57 \times \text{C/N}$, $p < 0.05$) in this study. Based on the regulation of C/N on decomposition of organic residues, the net N mineralization occurs when the C/N ratio is smaller than 20, while net immobilization occurs when the ratio is larger than 30 (Stevenson and Cole, 1999). Our results were also consistent with other findings that soil N mineralization and availability are significantly higher in soils under legumes than in soils under non-legumes (Gei and Powers, 2013; Raiesi, 2006; Yao et al., 2019). Finally, conversion of grassland to shrubland reduced BD but increased moisture of the surface soil (Table S3 and Fig. 1). These changes in soil BD and moisture may stimulate soil microbial activity, which, in turn, increase soil N mineralization (Uri et al., 2008). In this study, we found that R_m was negatively related with BD ($R_m = 316.65 - 192.00 \times \text{BD}$, $p < 0.01$) but was positively related with soil moisture ($p < 0.01$), which supports this explanation. Given that the effect of legume plants on soil N mineralization was greater than the effect of non-legume plants (Yao et al., 2019), the effect of legumes vs. non-legumes on N mineralization in grassland merits further study, which would link the soil response to plant functional groups.

4.2. Effects of soil texture and slope position

In this study, soil R_n and R_m were higher in Skeletal soils than in Aeolian soils (Table 1, Fig. 4), probably because of the higher clay content in Skeletal soils (sandy-loam) than in Aeolian soils (loamy-sand). This result agreed with findings from other studies that clay soils have higher net N mineralization rate than sandy soils in the semi-arid savanna (Scott Bechtold and Naiman, 2006) and in coastal Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forest (Prescott et al., 2000). Generally, soil C and N mineralization decrease with increasing clay content due to the chemical protection by fine soil particles (Côté et al., 2000; Hassink, 1994). Strong et al. (1999) found that such protection from microbial attack disappears in dry and rewetted conditions, thus leading to a positive correlation between clay content and soil net N mineralization rate. Soil moisture in the surface soil is vulnerable to external factors and is not always high (Fig. 1), which explains the higher R_n and R_m in Skeletal soils than in Aeolian soils. Furthermore, higher SOM, lower BD (Table S3) and thus higher soil aeration resulted in greater N mineralization in Skeletal soils than in Aeolian soils, providing an alternative explanation for the variations of R_n and R_m between two soils (Kuzyakov et al., 2000; Uri et al., 2008). In this study, the similar NH_4^+ -N concentration between the two soils was ascribed to the fixation of the crystal structures of clay minerals (Nieder et al., 2011), and similar NO_3^- -N concentrations could be attributed to the leaching of NO_3^- -N into deep soil layers.

Soil net N mineralization rate is reported to be higher in the lower slope than the upper slope due to higher soil moisture and organic matter (Hishi et al., 2014; Soon and Malhi, 2005). For Aeolian soils, soil moisture and BD were higher in the lower slope than in the upper slope

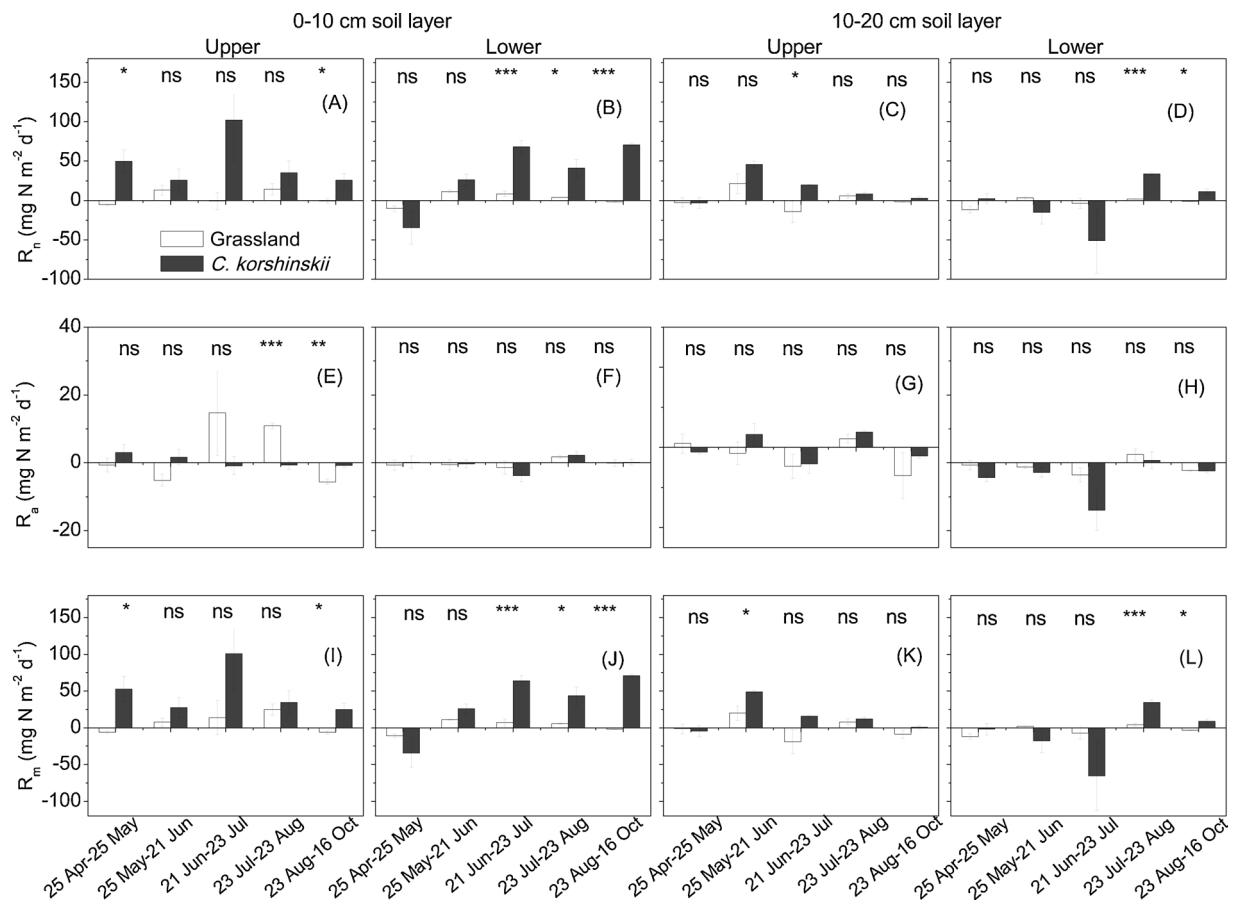


Fig. 5. Monthly dynamics of soil net nitrification (R_n), ammonification (R_a) and mineralization (R_m) rates in grassland and beneath *C. korshinskii* in the upper and lower slopes during the growing season. Error bar indicates two standard errors of the mean. *, **, *** indicate the differences between grassland and *C. korshinskii* land were significant at $p < 0.05$, 0.01 and 0.001, respectively. ns: not significant.

(Fig. 1, Table S3). The positive effect of soil moisture on net N mineralization ($R_m = 4.95 \times SM - 3.12$, $p < 0.01$) could be offset by the negative effect of BD ($R_m = 316.65 - 192.00 \times BD$, $p < 0.01$), and thus, the two slope positions had similar net N mineralization rates. For Skeletal slope, soil moisture, OC, N, C/N ratio and BD were similar between the lower and upper slopes (Table S3). The higher soil N mineralization in the upper slope in Skeletal soils might be related with higher soil temperature. Generally, soil temperature is higher in the upper slope than in the lower slope in the Loess Plateau (Liu et al., 2018), and soil net N mineralization rate increases with increasing temperature (Dalias et al., 2002). The higher NO_3^- -N and Min-N concentrations in lower slope than in the upper slope observed in this study were attributed to the smaller plant uptake of them in the lower slope. This is because the concentration and dynamics of soil mineral N during the growing season were dominated by plant growth and mineral N uptake (Kong et al., 2019). In this study, the plant biomass was higher in the upper slope than in the lower slope (Table S1). An alternative explanation was the transportation of mineral N with runoff and sediment along the slope (Scowcroft et al., 2004; Takahashi and Murayama, 2014; Xue et al., 2013), which usually resulted in the accumulation of mineral N in the lower slope. For example, Takahashi and Murayama (2014) reported a decreasing trend in soil mineral N concentrations from the toe to the summit on Mount Norikura in central Japan.

4.3. Interactive effects of afforestation with soil texture and slope position

The effect of grassland afforestation on soil NO_3^- -N, NH_4^+ -N and Min-N concentrations was higher in the lower slope than in the upper slope, regardless of soil texture. Such dependence of land-use change's

effect on slope position was ascribed to the variation of plant uptake on soil mineral N between the upper and lower slopes. In this study, soil mineral N concentrations were not affected by slope position in grassland, but were significantly higher in the lower slope than the upper slope in soils beneath *C. korshinskii* (Fig. 3). This is consistent with our observation that *C. korshinskii* produces more biomass in the upper slope than in the lower slope, irrespective of soil texture (Table S1), because the formation of biomass increases with increasing mineral N uptake (Kong et al., 2019). Therefore, the differences in soil mineral N concentrations between grassland and shrubland were higher in the lower slope than in the upper slope (Fig. 3). These results further indicated that the effect of plant growth on soil mineral N concentrations was greater than that of soil texture in the northern region of the Loess Plateau.

The effect of grassland afforestation on R_n and R_m was greater in the upper slope than the lower slope for Skeletal soils, but was greater in the lower than the upper slope for Aeolian soils at the 0–10 cm soil depth, which suggested an interactive effect of vegetation type, slope position and soil texture on N mineralization. For Skeletal soils in the study area, the accumulated litter under shrub canopy was markedly higher than that in grassland in the upper slope but was similar in the lower slope (Table S2). The effect of litter on soil N mineralization (priming effect) after afforestation was, therefore, higher in the upper slope than in the lower slope. However, for Aeolian soils, the accumulated litter under the shrub canopy was markedly smaller than that in grassland in the lower slope but was similar in the upper slope (Table S2). The effect of litter was, therefore, smaller in the upper slope than in the lower slope. Additionally, soil BD and C/N ratio decreased greatly after afforestation in the upper slope of Skeletal soils and the lower

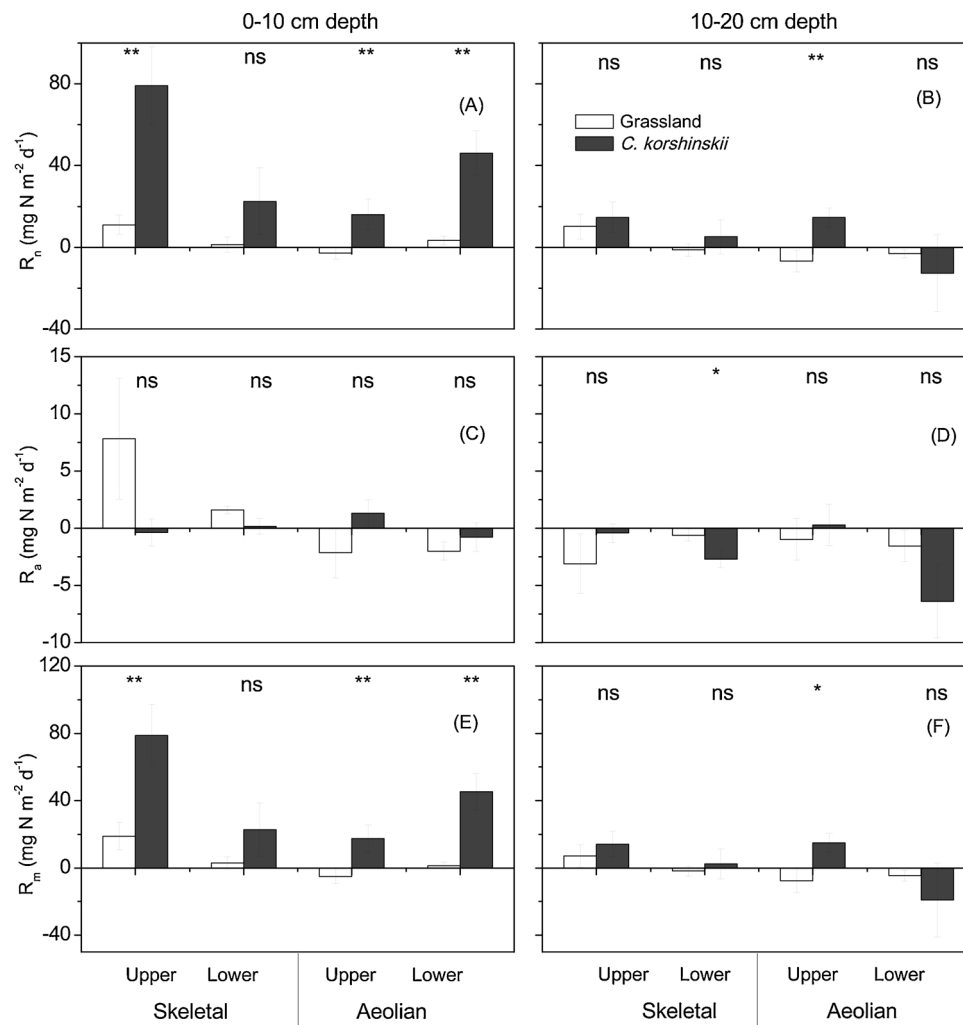


Fig. 6. Average rates of soil net nitrification (R_n), ammonification (R_a) and mineralization (R_m) in grassland and beneath *C. korshinskii* in the lower and upper slopes in Skeletal and Aeolian soils across the growing season. Error bar indicates two standard errors of the mean. *, **, *** indicate the differences between grassland and *C. korshinskii* land were significant at $p < 0.05$, 0.01 and 0.001, respectively. ns: not significant.

slope of Aeolian soils. However, these variables were less affected in the lower slope of Skeletal soils and upper slope of Aeolian soils (Table S3). These variations in the litter, soil BD and C/N ratio between shrubland and grassland provided an explanation for the variation in the effect of afforestation on net N mineralization with slope position and soil texture.

4.4. Implications

In this study, our one growing season investigation showed that the plantation of legume shrub *C. korshinskii* increased soil N availability and net mineralization rates. Although the seasonal patterns of precipitation and air temperature in the study year were similar to the average values from 2004 to 2013 (Liu and Shao, 2014), the results from our study might not have captured all of the variations of the land-use effect among years. Further research should be conducted to examine the temporal patterns of such effect. Moreover, afforestation has negative ecohydrological consequences in arid/semiarid regions. For example, Yao et al. (2012) found that afforestation was the major driver of soil desiccation along the 0–4 m soil layer across the semiarid area of the Loess Plateau, China. Duan et al. (2016) showed that insufficient water replenishment existed for seven vegetation types on a steep slope on the Loess Plateau, which could result in soil desiccation. Jia et al. (2019) reported severe depletion of soil moisture due to afforestation on the Loess Plateau and recommended a reasonable range of net

primary biomass for *C. korshinskii* ($83.7\text{--}201.7 \text{ g C m}^{-2} \text{ yr}^{-1}$) to mitigate soil desiccation. The soil water carrying capacity for afforestation species should, thus, be considered to maintain soil water balance and sustainable vegetation development.

Declarations of interest

None.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:<https://doi.org/10.1016/j.agee.2019.02.017>.

References

- Bennie, J., Huntley, B., Wiltshire, A., Hill, M.O., Baxter, R., 2008. Slope, aspect and climate: spatially explicit and implicit models of topographic microclimate in chalk grassland. *Ecol. Model.* 216, 47–59.
- Berthrong, S.T., Pineiro, G., Jobbagy, E.G., Jackson, R.B., 2012. Soil C and N changes with afforestation of grasslands across gradients of precipitation and plantation age. *Ecol. Appl.* 22, 76–86.
- Booth, M.S., Stark, J.M., Rastetter, E., 2005. Controls on nitrogen cycling in terrestrial ecosystems: a synthetic analysis of literature data. *Ecol. Monogr.* 75, 139–157.
- Calazans, S.O.L., Morais, V.A., Scolforo, J.R.S., Zinn, Y.L., Mello, J.M., Mancini, L.T., Silva, C.A., 2018. Soil organic carbon as a key predictor of N in forest soils of Brazil. *J. Soils Sediment* 18, 1242–1251.
- Campos, M.C.C., Marques Junior, J., Pereira, G.T., Montanari, R., Siqueira, D.S., 2007. Soil texture spatial variability of different parent material in pereira barreto, Sao Paulo state. *Rev. Cienc. Agron.* 38, 149–157.
- Chapin, F.S., Matson, P.A., Vitousek, P.M., 2011. Nutrient cycling. *Principles of Terrestrial Ecosystem Ecology*. Springer, New York.
- Chen, Y., Wang, K., Lin, Y., Shi, W., Song, Y., He, X., 2015. Balancing green and grain trade. *Nat. Geosci.* 8, 739–741.
- Côté, L., Brown, S., Paré, D., Fyles, J., Bauhus, J., 2000. Dynamics of carbon and nitrogen mineralization in relation to stand type, stand age and soil texture in the boreal mixedwood. *Soil Biol. Biochem.* 32, 1079–1090.
- Curtin, D., Campbell, C.A., Jalil, A., 1998. Effects of acidity on mineralization: pH-dependence of organic matter mineralization in weakly acidic soils. *Soil Biol. Biochem.* 30, 57–64.
- Dalias, P., Anderson, J.M., Bottner, P., Coûteaux, M.-M., 2002. Temperature responses of net nitrogen mineralization and nitrification in conifer forest soils incubated under standard laboratory conditions. *Soil Biol. Biochem.* 34, 691–701.
- Duan, L.X., Huang, M.B., Zhang, L.D., 2016. Differences in hydrological responses for different vegetation types on a steep slope on the Loess Plateau, China. *J. Hydrol.* 537, 356–366.
- Fahey, B., Payne, J., 2017. The Glendhu experimental catchment study, upland east Otago, New Zealand: 34 years of hydrological observations on the afforestation of tussock grasslands. *Hydrol. Process.* 31, 2921–2934.
- Ge, N., Wei, X., Wang, X., Liu, X., Shao, M., Jia, X., Li, X., Zhang, Q., 2019. Soil texture determines the distribution of aggregate-associated carbon, nitrogen and phosphorus under two contrasting land use types in the Loess Plateau. *Catena* 172, 148–157.
- Gei, M.G., Powers, J.S., 2013. Do legumes and non-legumes tree species affect soil properties in unmanaged forests and plantations in Costa Rican dry forests? *Soil Biol. Biochem.* 57, 264–272.
- Guntinas, M.E., Leiros, M.C., Trasar-Cepeda, C., Gil-Sotres, F., 2012. Effects of moisture and temperature on net soil nitrogen mineralization: a laboratory study. *Eur. J. Soil Biol.* 48, 73–80.
- Hassink, J., 1994. Effects of soil texture and grassland management on soil organic C and N and rates of C and N mineralization. *Soil Biol. Biochem.* 26, 1221–1231.
- He, S., Qiu, L., Jiang, D., Lamusa, A., Liu, Z., Luo, Y., 2008. Sand-fixing effects of *Caragana microphylla* shrub in Horqin sandy land, North China. *Front. For. China* 3, 31–35.
- Hishi, T., Urakawa, R., Tashiro, N., Maeda, Y., Shibata, H., 2014. Seasonality of factors controlling N mineralization rates among slope positions and aspects in cool-temperate deciduous natural forests and larch plantations. *Biol. Fert. Soils* 50, 343–356.
- Hook, P.B., Burke, I.C., 2000. Biogeochemistry in a shortgrass landscape: control by topography, soil texture, and microclimate. *Ecology* 81, 2686–2703.
- Jia, X., Shao, M., Yu, D., Zhang, Y., Binley, A., 2019. Spatial variations in soil-water carrying capacity of three typical revegetation species on the Loess Plateau, China. *Agric. Ecosyst. Environ.* 273, 25–35.
- Jiang, Y., Fu, P., Weng, Q., 2015. Assessing the impacts of urbanization-associated land use/cover change on land surface temperature and surface moisture: a case study in the midwestern United States. *Remote Sens.* 7, 4880–4898.
- Kachurina, O.M., Zhang, H., Raun, W.R., Krenzer, E.G., 2000. Simultaneous determination of soil aluminum, ammonium- and nitrate-nitrogen using 1 M potassium chloride extraction. *Commun. Soil Sci. Plant Anal.* 31, 893–903.
- Kelley, K.R., Stevenson, F.J., 1995. Forms and nature of organic N in soil. *Fertil. Res.* 42, 1–11.
- Kidron, G.J., 2009. The effect of shrub canopy upon surface temperatures and evaporation in the Negev Desert. *Earth Surf. Process. Landf.* 34, 123–132.
- Kong, W., Yao, Y., Zhao, Z., Qin, X., Zhu, H., Wei, X., Shao, M., Wang, Z., Bao, K., Su, M., 2019. Effects of vegetation and slope aspect on soil nitrogen mineralization during the growing season in sloping lands of the Loess Plateau. *Catena* 172, 753–763.
- Kuzyakov, Y., Friedel, J.K., Stahr, K., 2000. Review of mechanisms and quantification of priming effects. *Soil Biol. Biochem.* 32, 1485–1498.
- Li, M., Zhou, X., Zhang, Q., Cheng, X., 2014. Consequences of afforestation for soil nitrogen dynamics in central China. *Agric. Ecosyst. Environ.* 183, 40–46.
- Liu, D.S., 1985. *Loess and the Environment*. Science Press, Beijing.
- Liu, B., Shao, M., 2014. Estimation of soil water storage using temporal stability in four land uses over 10 years on the Loess Plateau, China. *J. Hydrol.* 517, 974–984.
- Liu, J., Qiu, L., Wang, X., Wei, X., Gao, H., Zhang, Y., Cheng, J., 2018. Effects of wildfire and topography on soil nutrients in a semiarid restored grassland. *Plant Soil* 428, 123–136.
- Lu, M., Yang, Y., Luo, Y., Fang, C., Zhou, X., Chen, J., Yang, X., Li, B., 2011. Responses of ecosystem nitrogen cycle to nitrogen addition: a meta-analysis. *New Phytol.* 189, 1040–1050.
- Luce, M.S., Whalen, J.K., Ziadi, N., Zebarth, B.J., 2016. Net nitrogen mineralization enhanced with the addition of nitrogen-rich particulate organic matter. *Geoderma* 262, 112–118.
- McKinley, D.C., Blair, J.M., 2008. Woody plant encroachment by *Juniperus virginiana* in a mesic native grassland promotes rapid carbon and nitrogen accrual. *Ecosystems* 11, 454–468.
- Mercedes Vassallo, M., Dieguez, H.D., Garbulsky, M.F., Jobbagy, E.G., Paruelo, J.M., 2013. Grassland afforestation impact on primary productivity: a remote sensing approach. *Appl. Veg. Sci.* 16, 390–403.
- Nieder, R., Benbi, D.K., Scherer, H.W., 2011. Fixation and defixation of ammonium in soils: a review. *Biol. Fert. Soils* 47, 1–14.
- Ozalp, M., Cavdar, G., 2016. Responses of some soil properties to 14th-year afforestation efforts in the semi-arid grasslands of Ankara, Turkey. *Fresen. Environ. Bull.* 25, 4589–4597.
- Peichl, M., Leava, N.A., Kiely, G., 2012. Above- and belowground ecosystem biomass, carbon and nitrogen allocation in recently afforested grassland and adjacent intensively managed grassland. *Plant Soil* 350, 281–296.
- Prescott, C.E., Chappell, H.N., Vesterdal, L., 2000. Nitrogen turnover in forest floors of coastal Douglas-fir at sites differing in soil nitrogen capital. *Ecology* 81, 1878–1886.
- Rahman, M.M., Barcena, T.G., Vesterdal, L., 2017. Tree species and time since afforestation drive soil C and N mineralization on former cropland. *Geoderma* 305, 153–161.
- Raiesi, F., 2006. Carbon and N mineralization as affected by soil cultivation and crop residue in a calcareous wetland ecosystem in Central Iran. *Agric. Ecosyst. Environ.* 112, 13–20.
- Robertson, G.P., Wedin, D., Groffman, P., Blair, J., Holland, E., Nedelhoff, K., Harris, D., Coleman, D., Bledsoe, C., Sollins, P., 1999. Soil carbon and nitrogen availability. Nitrogen mineralization, nitrification, and soil respiration potentials. In: Robertson, G.P., Coleman, D.C., Bledsoe, C.S., Sollins, P. (Eds.), *Standard Soil Methods for Long-Term Ecological Research*. Oxford University Press, New York, pp. 258–271.
- Scott Bechtold, J., Naiman, R.J., 2006. Soil texture and nitrogen mineralization potential across a riparian toposequence in a semi-arid savanna. *Soil Biol. Biochem.* 38, 1325–1333.
- Scowcroft, P.G., Haraguchi, J.E., Hue, N.V., 2004. Reforestation and topography affect montane soil properties, nitrogen pools, and nitrogen transformations in Hawaii. *Soil Sci. Soc. Am. J.* 68, 959–968.
- She, D.L., 2009. Soil-water Environmental Effects of Revegetation in a Small Catchment in the Wind-Water Erosion Crisscross Zone of the Loess Plateau. Northwest A&F University, Yangling (in Chinese with English Abstract).
- Soon, Y.K., Malhi, S.S., 2005. Soil nitrogen dynamics as affected by landscape position and nitrogen fertilizer. *Can. J. Soil Sci.* 85, 579–587.
- Stevenson, F.J., Cole, M.A., 1999. *Cycles of Soils: Carbon, Nitrogen, Phosphorus, Sulfur, Micronutrients*. John Wiley & Sons, Inc., New York.
- Strong, D.T., Sale, P.W.G., Helyar, K.R., 1999. The influence of the soil matrix on nitrogen mineralisation and nitrification. IV. Texture. *Aust. J. Soil Res.* 37, 329–344.
- Takahashi, K., Murayama, Y., 2014. Effects of topographic and edaphic conditions on alpine plant species distribution along a slope gradient on Mount Norikura, central Japan. *Ecol. Res.* 29, 823–833.
- Uri, V., Löhmus, K., Kund, M., Tullus, H., 2008. The effect of land use type on net nitrogen mineralization on abandoned agricultural land: silver birch stand versus grassland. *Forest Ecol. Manag.* 255, 226–233.
- Wang, W.J., Qiu, L., Zu, Y.G., Su, D.X., An, A., Wang, H.Y., Zheng, G.Y., Sun, W., Chen, X.Q., 2011. Changes in soil organic carbon, nitrogen, pH and bulk density with the development of larch (*Larix gmelinii*) plantations in China. *Glob. Change Biol.* 17, 2657–2676.
- Wang, C., Zhu, F., Zhao, X., Dong, K., 2014a. The effects of N and P additions on microbial N transformations and biomass on saline-alkaline grassland of Loess Plateau of Northern China. *Geoderma* 213, 419–425.
- Wang, W., Wang, H., Zu, Y., 2014b. Temporal changes in SOM, N, P, K, and their stoichiometric ratios during reforestation in China and interactions with soil depths: importance of deep-layer soil and management implications. *Forest Ecol. Manag.* 325, 8–17.
- Wei, X.R., Shao, M.A., Fu, X.L., Horton, R., 2010. Changes in soil organic carbon and total nitrogen after 28 years grassland afforestation: effects of tree species, slope position, and soil order. *Plant Soil* 331, 165–179.
- Wei, X., Huang, M., Shao, M., Li, L., Zhang, X., Horton, R., 2013. Shrubs increase soil resources heterogeneity along semiarid grass slopes in the Loess Plateau. *J. Arid Environ.* 88, 175–183.
- Wei, X., Reich, P.B., Hobbie, S.E., Kazanski, C.E., 2017. Disentangling species and functional group richness effects on soil N cycling in a grassland ecosystem. *Glob. Change Biol.* 23, 4717–4727.
- Xue, Z., Cheng, M., An, S., 2013. Soil nitrogen distributions for different land uses and landscape positions in a small watershed on Loess Plateau, China. *Ecol. Eng.* 60, 204–213.
- Yang, X., Wei, X., Shao, M., 2016. Stem biomass estimation models for dominant shrubs on the northern Loess Plateau of China. *Chin. J. Appl. Ecol.* 27, 3164–3172 (In Chinese with English Abstract).
- Yao, Y., Fu, B., Lü, Y., Chang, R., Wang, S., Wang, Y., Su, C., 2012. The multi-scale spatial variance of soil moisture in the semi-arid Loess Plateau of China. *J. Soils Sediment.* 12, 694–703.
- Yao, Y., Zhao, Z., Wei, X., Shao, M., 2019. Effects of shrub species on soil nitrogen mineralization in the desert-loess transition zone. *Catena* 173, 330–338.