Soil nitrogen (N) availability is one of the limiting factors for plant growth on sandy lands. Little is known about impacts of afforestation on soil N availability and its components in southeastern Keerqin sandy lands, China. In this study, we measured N transformation under sandy Mongolian pine (Pinus sylvestris var. mongolica Litv.) plantations of different ages (grassland, young, middle-aged, close-to-mature) and management practices (non-grazing and free-grazing) during the growing seasons using the ion exchange resin bag method. Results showed that, for all plots and growing season, soil NH$_4^+$-N, NO$_3^-$-N, mineral N, and relative nitrification index, varied from 0/18 to 1/54, 0/96 to 22/05, 1/23 to 23/58 g d$^{-1}$ g$^{-1}$ dry resin, and 0/76 to 0/97, respectively, and NO$_3^-$-N dominated the available N amount due to intense nitrification in these ecosystems. In general, the four indices significantly increased in the oldest plantation, with corresponding values in non-grazing sites lower than those in free-grazing sites ($p < 0.05$). Our studies indicated that it is a slow, extended process to achieve improvement in soil quality after afforestation of Mongolian pine in the study area. Copyright © 2009 John Wiley & Sons, Ltd.

**KEY WORDS:** soil N transformation; ion exchange resin bag method; Pinus sylvestris var. mongolica Litv.; Keerqin sandy lands; China

**INTRODUCTION**

Desertification and dust storms have gained increasing attention from many national governments and people (Verón et al., 2006; Xu, 2006). Desertification in large areas in northern China has jeopardized the survival of local people, economic development, regional and national ecological wealth, with impacts on the entire country (Jiang et al., 2002; Wang et al., 2006). Ecological engineering using sand fixation and afforestation have been considered an effective method for desertification and sand–dust storm prevention (Jiang et al., 2002; Yang et al., 2005). Located at the ecotone of agropastoral systems in northern China, the Keerqin sandy lands are highly vulnerable to environmental change and have been severely impacted by desertification. Combating desertification in the Keerqin sandy lands is of national importance as a sustainable development issue in China. Projects of afforestation for sand fixation were initiated in Keerqin sandy lands in the early 1950s. In the earliest projects, Mongolian pine (Pinus sylvestris var. mongolica Litv.) trees were planted in Zhanggutai, the southeastern part of Keerqin sandy lands. Observations showed that at early growth stage the plantations had adapted well compared with the performance of this species in its native region of Honghuaijir (Jiao, 1989; Jiang et al., 2002). Subsequently, Mongolian pine trees have been widely planted since the late 1960s, with Mongolian pine plantations in northern China extending over 500 000 ha (Chen et al., 2006b).

Unfortunately, since the early 1990s, the Mongolian pine forests planted earlier have faced serious problems, with unexpectedly slow growth, abnormal development, and even mortality over large areas (Zeng et al., 1996).
Our overall research goal was to evaluate the possible impacts of climate, water, nutrients, disease, insect pest factors, and land management practices on this decline syndrome.

Nitrogen (N) is an important macronutrient for plants (Vitousek et al., 1982; Mooney et al., 1987; Attiwill and Adams, 1993). Its availability greatly affects plant growth in many terrestrial ecosystems (Chapin et al., 1986; Vitousek and Howarth, 1991; Hooper and Johnson, 1999). Soil N accounts for more than 90% of N sources for plants (Kaye et al., 2003). Available soil N mainly comes from the mineralization of organic N (Binkley and Hart, 1989; Vestgarden and Kjønaas, 2003). Soil N availability directly influences the productivity of terrestrial ecosystems (Reich et al., 1997); in addition, N availability impacts plant biodiversity, community succession, and long-term ecosystem resilience (Schlesinger, 1997; Heitkamp et al., 2008). Although the classical N mineralization paradigm of N cycling may be incomplete, and many studies have shown that plants can use amino acids and other organic N forms (Schimel and Bennett, 2004), soil N mineralization is still an important criterion for evaluating forest N cycling (Loreau, 1994; Larsen, 1995; Schlesinger, 1997; Chen et al., 2006b).

In the study region, human population has increased since the 1970s, intensifying the impacts of human activities on the local forest ecosystems, including overgrazing and leaf litter raking (Jiang et al., 2002). In this study, the ion exchange resin bag method was used to study soil N availability, with the objectives of determining the impacts of stand age and management practices on soil N availability and its transformation rate in Mongolian pine plantations.

MATERIALS AND METHODS

Study Area

The study area is located in the southeast Keerqin sandy lands (Figure 1), in the sub-humid arid climatic zone. Average annual precipitation is 450 mm, with an annual potential evaporation of 1300–1800 mm, and average annual temperature, 6.2°C. The annual accumulated temperature of > 10°C is 2890–2950°C, and the annual frost-free period is 145–150 days. The major soil type in this area is aeolian sandy soil, with low organic matter (SOM), N, and phosphorus (P) contents (Chen et al., 2002). Herbaceous plants dominate the vegetation. Overgrazing and leaf litter raking are the most common human disturbances to the forest ecosystem with 30–70% losses of litter from the Mongolian pine plantations reported (Chen et al., 2004, 2005).

Study Plots

Study plots were selected in Zhanggutai Town of Zhangwu County, Liaoning Province, southeast Keerqin sandy lands (42°43’N, 122°22’E); Daqinggou Ecological Station, Institute of Applied Ecology, Chinese Academy of

Figure 1. Map of research region (southeastern Keerqin sandy lands), three sampling sites (Daqinggou, Zhanggutai, and Ganqika), and native habitat (Honghuaiji) of sandy Pinus sylvestris var. mongolica Litv.
Sciences, Ganqika Town, Keerqinzuo county, Inner Mongolia Autonomous Region (42°58′N, 122°21′E); and Daqinggou National Nature Reserve (42°45′–42°48′N, 122°13′–122°15′E). These three areas have very similar climate, soil types, and vegetation. Eight Mongolian pine plantations and non-forested (grassland) plots with visually healthy ecosystem were selected from these three areas with different age classes and management practices. Each plot had an area of 30 × 20 m² (Table I). Age classes of the study plots were grassland; young (15 years), middle-aged (28–30 years), and close-to-mature (47 years) Mongolian pine plantations. The management measures for the study plots were non-grazing (plots were enclosed by fences) and free-grazing (plots were grazed by cattle, sheep, or goat) (Table I). The basic stand characteristics of the experimental plots, such as tree density, mean diameter at breast height (DBH), basal area of tree, mean height, etc., are described in Table I. The soil bulk density, pH, and nutrient concentrations of the experimental plots are described in Table II.

Study Methods

In this study, we used ion exchange resin (IER) bag to evaluate nitrification and nitrogen mineralization (Binkley and Matson, 1983; Mo et al., 2003). Because IER bags were in contact with soil solution and exposed to the same soil environment conditions as tree roots, analysis of IER extracts provided a measure of the relative availability of NH₄⁺-N and NO₃⁻-N in soil solution during the incubation period. The content of NH₄⁺-N and NO₃⁻-N in the resin bags represents the net results of the two processes: nitrogen mineralization and nitrogen uptake by plants and microbes (Kjønaas, 1999). We refer to the content of NH₄⁺-N and NO₃⁻-N in the resin bags as an index of N availability.

Numerous 4 × 4 cm² sized nylon bags (1 × 1 mm² grid) were prepared for the experiments. About 10 g resins (Amberlite IRC resin, which is a 1:1-5 mixture resin of IR-120 and IRA-400) were put in each resin bag, which was then sealed. Water contents of the resins were measured. The 30 × 20 m² plots were divided into six subplots (10 × 10 m²), with two resin bags buried in each subplot on 16 May 2004. The resin bags were buried at two different depths, 4 and 11 cm to measure the average NH₄⁺-N and NO₃⁻-N amount in 0–7.5 and 7.5–15 cm soil layer, respectively. The resin bags were removed on 1 July 2004, the plant roots and soils adhering to the bags were removed, and the resin bags were transferred and stored in the laboratory refrigerator (4°C). We used 2 mol L⁻¹ KCl solutions to extract the NH₄⁺-N and NO₃⁻-N, and the colorimetry method to measure NH₄⁺-N and NO₃⁻-N concentrations (Liu et al., 1996). The experiments were repeated twice (from 1 July to 15 August and from 15 August to 1 October). In our study, we defined the period from 15 May to 1 July, from 1 July to 15 August, and from 15 August to 1 October as spring, summer, and fall, respectively.

Stand characteristics were measured in May 2004. Samples were collected on 1st August 2004 for measuring soil properties, with samples collected from five locations in each subplot and composited into a single sample (0–15 cm). As a result, six composited soil samples were collected for each site. All soil samples were analyzed according to methods of Liu et al. (1996).

Statistical Methods

Soil NH₄⁺-N content refers to the absorbed NH₄⁺-N amount per day by a unit of dry mass cation resin (IR-120) (µg d⁻¹ g⁻¹ dry resin); soil NO₃⁻-N content refers to the absorbed NO₃⁻-N amount per day by a unit dry mass anion resin (IRA-400). Mineral N equals to the total of NH₄⁺-N and NO₃⁻-N; relative nitrification index (RNI) is expressed as NO₃⁻-N amount divided by mineral N amount (Lavoie and Bradley, 2003). The distribution of soil NO₃⁻-N content did not follow the normal distribution pattern, so data were transformed using the log-transformation method. After transformation, all the data followed normal distribution patterns. No significant difference was detected between different soil layers (with the depth of 0–7.5 and 7.5–15 cm) for soil NH₄⁺-N, NO₃⁻-N, mineral N, and RNI in each plot. Therefore, the data for NH₄⁺-N and NO₃⁻-N content in each subplot were the mean values found in the two layers, with six replicates from each site.

To avoid confusing the influences of stand age or management practice on soil N availability, Tukey’s multiple comparison method following ANOVA was used to identify significant differences among stand ages under non-grazing and free-grazing management practices. A T-test was used to compare the differences between non-grazing and free-grazing ecosystems with three pairs of plots (ZWL0 vs. SWL0, DSL28 vs. DSM28, and ZSL30 vs. ZSL28).
Table I. Site names and vegetation characteristics of study plots in Keerqin Sandy Lands, China

<table>
<thead>
<tr>
<th>Plot codes</th>
<th>Sites</th>
<th>Stage (year old)</th>
<th>Density (trees ha(^{-1}))</th>
<th>Diameter at breast height (cm)</th>
<th>Basal area of tree (m(^2)ha(^{-1}))</th>
<th>Height (m)</th>
<th>Management practices</th>
</tr>
</thead>
<tbody>
<tr>
<td>ZWL0</td>
<td>Zhanggutai</td>
<td>Grassland (0)</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>Free-grazing</td>
</tr>
<tr>
<td>SWL0</td>
<td>Ganqika</td>
<td>Grassland (0)</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>Non-grazing &gt; 10 years</td>
</tr>
<tr>
<td>SSM15</td>
<td>Ganqika</td>
<td>Young (15)</td>
<td>600</td>
<td>5.81 ± 0.35a(^*)</td>
<td>1.59 ± 0.10a</td>
<td>3.03 ± 0.14a</td>
<td>Non-grazing for 5 years</td>
</tr>
<tr>
<td>DSL28</td>
<td>Daqinggou</td>
<td>Middle-aged (28)</td>
<td>750</td>
<td>12-00 ± 0.78b</td>
<td>8.48 ± 0.55b</td>
<td>7.56 ± 0.31b</td>
<td>Non-grazing &gt; 10 years</td>
</tr>
<tr>
<td>DSM28</td>
<td>Daqinggou</td>
<td>Middle-aged (28)</td>
<td>1000</td>
<td>13-32 ± 0.86b</td>
<td>13.94 ± 0.90c</td>
<td>8.72 ± 0.21b</td>
<td>Non-grazing &gt; 10 years</td>
</tr>
<tr>
<td>ZSL30</td>
<td>Zhanggutai</td>
<td>Middle-aged (30)</td>
<td>825</td>
<td>14-15 ± 0.63b</td>
<td>12.97 ± 0.82c</td>
<td>7.93 ± 0.13b</td>
<td>Free-grazing</td>
</tr>
<tr>
<td>ZSM30</td>
<td>Zhanggutai</td>
<td>Middle-aged (30)</td>
<td>850</td>
<td>14.35 ± 0.80b</td>
<td>13.75 ± 0.78c</td>
<td>8.01 ± 0.20b</td>
<td>Non-grazing for 4 years</td>
</tr>
<tr>
<td>ZSL47</td>
<td>Zhanggutai</td>
<td>Close-to-mature (47)</td>
<td>725</td>
<td>19-28 ± 1.98c</td>
<td>21.17 ± 2.18d</td>
<td>13.46 ± 0.86c</td>
<td>Free-grazing</td>
</tr>
</tbody>
</table>

\(^*\)Mean ± 1 SE, \(n > 30\); the same characters indicate no significant difference \((p > 0.05)\), and different ones indicate significant difference \((p < 0.05)\).
<table>
<thead>
<tr>
<th>Plots</th>
<th>Bulk density (g cm(^{-3}))</th>
<th>pH</th>
<th>Organic carbon (g kg(^{-1}))</th>
<th>Total N (g kg(^{-1}))</th>
<th>Total P (g kg(^{-1}))</th>
<th>C: N ratio</th>
<th>N: P ratio</th>
</tr>
</thead>
<tbody>
<tr>
<td>ZWL0</td>
<td>1.64 ± 0.01a</td>
<td>6.79 ± 0.06a</td>
<td>3.72 ± 0.12a</td>
<td>0.328 ± 0.070a</td>
<td>0.086 ± 0.011a</td>
<td>11.34 ± 3.21a</td>
<td>3.81 ± 0.98a</td>
</tr>
<tr>
<td>SWL0</td>
<td>1.68 ± 0.01b</td>
<td>6.75 ± 0.05a</td>
<td>4.55 ± 0.10b</td>
<td>0.518 ± 0.057b</td>
<td>0.269 ± 0.012b</td>
<td>8.78 ± 2.13b</td>
<td>1.93 ± 0.53b</td>
</tr>
<tr>
<td>SSM15</td>
<td>1.61 ± 0.02a</td>
<td>6.78 ± 0.05a</td>
<td>3.92 ± 0.18a</td>
<td>0.519 ± 0.058b</td>
<td>0.080 ± 0.014a</td>
<td>7.55 ± 1.98b</td>
<td>6.49 ± 1.87c</td>
</tr>
<tr>
<td>DSL28</td>
<td>1.57 ± 0.03a</td>
<td>6.61 ± 0.06a</td>
<td>3.62 ± 0.15c</td>
<td>0.730 ± 0.071d</td>
<td>0.169 ± 0.014c</td>
<td>4.96 ± 1.09c</td>
<td>4.32 ± 1.06c</td>
</tr>
<tr>
<td>DSM28</td>
<td>1.52 ± 0.01c</td>
<td>6.48 ± 0.05a</td>
<td>3.40 ± 0.12c</td>
<td>0.770 ± 0.076d</td>
<td>0.136 ± 0.009c</td>
<td>4.42 ± 1.21c</td>
<td>5.66 ± 1.56c</td>
</tr>
<tr>
<td>ZSL30</td>
<td>1.65 ± 0.01a</td>
<td>6.67 ± 0.05a</td>
<td>3.97 ± 0.08a</td>
<td>0.415 ± 0.006c</td>
<td>0.111 ± 0.002a</td>
<td>9.57 ± 2.17b</td>
<td>3.74 ± 0.98a</td>
</tr>
<tr>
<td>ZSM30</td>
<td>1.62 ± 0.01a</td>
<td>6.37 ± 0.07a</td>
<td>3.90 ± 0.14a</td>
<td>0.395 ± 0.038c</td>
<td>0.101 ± 0.007a</td>
<td>9.87 ± 1.94b</td>
<td>3.91 ± 0.98a</td>
</tr>
<tr>
<td>ZSL47</td>
<td>1.67 ± 0.03b</td>
<td>6.09 ± 0.11b</td>
<td>7.37 ± 0.50d</td>
<td>0.533 ± 0.008c</td>
<td>0.141 ± 0.001a</td>
<td>13.83 ± 3.65ad</td>
<td>3.78 ± 0.76a</td>
</tr>
</tbody>
</table>

*Mean ± 1 SE, n = 6; the same characters indicate no significant difference, and different ones indicate significant difference.*
ZSM30). The Tukey’s multiple comparison method was also used to identify seasonal differences for each stand age or each management practice. In addition, a Pearson’s correlation was used to test the relationship among soil N availability and soil properties. All statistical analyses were performed using SPSS 10.0 software (SPSS Inc., 2001), with probability \( p < 0.05 \) indicating significant differences between two values.

### RESULTS

The effect of vegetation development on soil N transformation was evaluated by comparing mean values during the growing season for non-grazing and for free-grazing management, separately. For the non-grazing ecosystems, soil N transformations did not differ significantly across stand ages (Table III). In contrast, the free-grazing ecosystems, soil \( \text{NH}_4^+ - \text{N} \), \( \text{NO}_3^- - \text{N} \), and mineral N in close-to-mature forest were significantly higher than those in grassland and middle-aged forest, while there were no significant differences between grassland and middle-aged forest. In addition, soil RNI in both middle-aged and close-to-mature stands was higher than that in grassland (Table III).

The effect of grazing management on soil N transformations was evaluated by comparing values both within season and across the entire growing season (Figure 2). All measures of N transformations were significantly higher in the free-grazing than the non-grazing plots. These patterns held up in all seasons except for the index of soil RNI during the summer season (Figure 2).

Seasonal patterns in soil N transformations differed somewhat with stand development and grazing treatment (Figure 3). Most consistent was the trend toward lower N transformation rates in summer than spring and fall for the grassland and young pine stands. Seasonal patterns in the older pine plantations were more complex with little consistency across N transformation variables, ages, or grazing treatments (Figure 3).

### DISCUSSION

**Soil Properties, N Transformation, and N Availability**

There are various primary factors influencing soil N transformation and its availability including soil temperature, soil water, soil physical and chemical quality, soil animals and microbes, vegetation types, climate, natural or human induced disturbances, and so on (Schlesinger, 1997; Chen et al., 2007; Yu et al., 2008). Among these factors, soil nutrients are one of the most important at the small scale (Chen et al., 2006a). In this study, although \( \text{NH}_4^+ - \text{N} \), \( \text{NO}_3^- - \text{N} \), and mineral N significantly correlated with both soil pH and organic carbon content, there were no significant correlations with soil total N, total P, C:N ratio, N:P ratio (Table IV).
Soil pH has a key influence on soil nutrient availability. Although variability in pH is very small among the eight research sites, it did decline significantly with stand development (Table II). Many studies have clearly shown that low pH in coniferous forests can inhibit nitrification (Rudebeck and Persson, 1998; Ste-Marie and Paré, 1999); nevertheless soil NO$_3^-$-N in the close-to-mature forest was the highest among all the plots. Low pH normally inhibits nitrification at pH values around 4–5 (Rudebeck and Persson, 1998; Ste-Marie and Paré, 1999). In our study, the soil pH was around 6–7. Apparently the sandy soils in the study area are fairly well buffered against

![Figure 2](image-url)
Figure 3. The changes of soil NH$_4^+$-N (A), NO$_3^-$-N (B), mineral N (C), and relative nitrification index (D) with the age of Mongolian pine plantations under non-grazing and free-grazing management practices, respectively. Error bars show the standard error of the mean. The different letters represent significance among the three different seasons for each age ($p < 0.05$).
acidification, compared with soils in many other forest regions (Rudebeck and Persson, 1998; Ste-Marie and Paré, 1999; Vestgarden and Kjønaas, 2003). We found that soil pH correlated negatively with stand basal area ($r = -0.90$, $n = 8$, $p < 0.01$); hence the biomass of these pine forests probably plays a key factor in relation to soil N availability and forest ecosystem function.

We were surprised to find that soil total N, soil C:N ratio, and N:P ratio were insignificantly correlated with soil N availability in our study, because many studies have shown that they are significantly related to soil N transformation rate (Fisher and Binkley, 2000). We therefore inferred that other important factors influence soil N transformation rate of Mongolian pine plantations, with one of the primary factors being grazing. In addition, there were significant correlations among soil NH$_4^+$, NO$_3^-$, and mineral N, indicating that ammonification and nitrification are closely related processes in N transformation.

### Afforestation, Soil N Availability, and Its Components

Soil N transformation is one of the most important ecological processes and can be used to evaluate vegetation restoration and ecosystem stability (Larsen, 1995). We found that only the soil available N of close-to-mature Mongolian pine plantations was significantly higher than that of grassland, while there were no significant differences among grassland, young and middle-aged Mongolian pine plantations. Therefore, the increase of available N by afforestation was a long-term process with the most prominent responses occurring after age 28–30 years (Table III). Most of the previous studies have showed that Mongolian pine plantations in the sandy lands decline with stand age, and that decline is correlated with low soil nutrient content (Jiao, 1989; Chen et al., 2002; Jiang et al., 2002). The decline of stands mainly occurs after about 30 years and some signs of decline were apparent visually shoot blight, needle loss, and even mortality in the middle-aged and older stands (Jiang et al., 2002). Therefore, our results suggest that soil N supply ability cannot meet the need of pine growth with stand development, especially for forests over 28–30 years old.

Our data also showed that soil RNI decreased from grassland to young forest, and increased from young to middle-aged forest under non-grazing stands. However, under free-grazing stands, it increased from grassland to middle-aged forest, and maintained these levels until close-to-mature forest (Table III). This indicates that the components of soil available N can change during stand development. Moreover, the dominant form of soil available N was soil NO$_3^-$-N, which generally increased with stand development, especially in the close-to-mature forest. The N requirement of plants is predominantly supplied by NH$_4^+$-N and/or NO$_3^-$-N from soil solution, with the energetic cost of uptake and assimilation generally higher for NO$_3^-$-N than for NH$_4^+$-N (Falkengren-Grerup, 1995; Bassirirad et al., 1996). Thus, the plants absorbing NH$_4^+$-N may be more competitive than those absorbing NO$_3^-$-N (Jackson et al., 1989). We therefore surmised that changes in soil available N forms of Mongolian pine plantations may be unfavorable to N absorption by the trees.

We also found that the increase in available N was generally during the summer and fall (Figure 3). However, since the height growth periods of Mongolian pine are from mid-May to early June (Jiao, 1989; Chen et al., 2006b),

<table>
<thead>
<tr>
<th>Variables</th>
<th>NH$_4^+$-N</th>
<th>NO$_3^-$-N</th>
<th>Mineral N</th>
<th>Relative nitrification index</th>
</tr>
</thead>
<tbody>
<tr>
<td>$pH$</td>
<td>-0.78**</td>
<td>-0.67*</td>
<td>-0.68*</td>
<td>-0.38NS</td>
</tr>
<tr>
<td>Organic carbon</td>
<td>0.94***</td>
<td>0.76**</td>
<td>0.77**</td>
<td>0.37NS</td>
</tr>
<tr>
<td>Total N</td>
<td>0.02NS</td>
<td>0.28NS</td>
<td>0.27NS</td>
<td>0.42NS</td>
</tr>
<tr>
<td>Total P</td>
<td>-0.02NS</td>
<td>0.14NS</td>
<td>0.13NS</td>
<td>0.51NS</td>
</tr>
<tr>
<td>C:N ratio</td>
<td>0.63NS</td>
<td>0.34NS</td>
<td>0.35NS</td>
<td>0.02NS</td>
</tr>
<tr>
<td>N:P ratio</td>
<td>-0.11NS</td>
<td>-0.13NS</td>
<td>-0.13NS</td>
<td>-0.44NS</td>
</tr>
<tr>
<td>NH$_4^+$-N</td>
<td>0.89***</td>
<td>0.90***</td>
<td>0.48NS</td>
<td>0.77**</td>
</tr>
<tr>
<td>NO$_3^-$-N</td>
<td></td>
<td>1.00***</td>
<td></td>
<td>0.75**</td>
</tr>
<tr>
<td>Mineral N</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note: NS means no significance. *$p < 0.05$; **$p < 0.01$; ***$p < 0.001$; $n = 8$. 

the increase later in the season N availability may not be useful for tree height growth. We therefore agree that the insufficiency of available N was one of the most important reasons for the decline of Mongolian pine plantations in the sandy lands (Chen et al., 2006a,b).

Human Disturbances and Soil N Transformation

Vitousek et al. (1997) summarized the influences of human activities on N cycling in terrestrial ecosystems. They pointed out that human activities had changed the status of soil N in the ecosystems. The insufficiency of soil availability was one of the limiting factors to plant growth in many terrestrial ecosystems; although due to excessive N input into some ecosystems, N pollution had brought many negative impacts.

The results of our study showed soil N transformations in non-grazing ecosystem were lower that those in free-grazing ecosystem. Moreover, the increase of NO$_3^-$-N was much greater than the increase of NH$_4^+$-N (Figure 2). This might be attributed to grazing, which can cause changes in soil structure and understory vegetation composition producing favorable conditions for soil nitrification. These include increases in soil bulk density, water content, and rates of root turnover, and decreases in soil porosity and understory cover (Shariff et al., 1994; Evans and Belnap, 1999). As described previously, plants and microbes prefer to use NH$_4^+$-N, since less energy is needed for its absorption (Nadelhoffer et al., 1984; Vitousek and Sanford, 1986; Jackson et al., 1989). The higher NO$_3^-$-N to NH$_4^+$-N ratio in the free-grazing plots could indicate that non-grazing resulted in the soil N transformation process being more favorable for the absorption by plants. However, the higher soil mineral N, which is expected with urine and feces in the grazed ecosystems results in higher soil NO$_3^-$-N. Such high NO$_3^-$-N can result in leaching and consequent depletion of other key nutrients, particularly of cations such as K. That cannot be healthy for either forest soils or ground water in the long run.

Based on these impacts, reduction, or elimination of grazing might perhaps be a better choice for fostering forest and human development in the study area. As we expected, neither of the grasslands showed high soil NO$_3^-$-N, even with grazing (Table III). We consider that either the grassland plants readily use NO$_3^-$-N as an N source or the grassland soils have a relatively high N immobilization potential. Grassland, as the typical local vegetation type, may be the best choice for ecological restoration. The influence of grazing on soil N availability after afforestation needs more careful consideration.

ACKNOWLEDGEMENTS

We thank the Liaoning Provincial Institute of Sand Fixation and Afforestation for supplying us research sites, Guangsheng Chen, Zhanyuan Yu and Qiong Zhao for their field helps, Heming Lin and Guiyan Ai for their laboratory analysis, and Greg Nagle for English improvement. We also thank two anonymous reviewers for their valuable comments on an earlier version of this manuscript. This study was funded by the National Natural Science Foundation of China under grant Nos. 30872011 & 3060047 and the National Key Technologies R and D Program of China under grant No. 2006BAD26B0201-1.

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