Article

Atmospheric Nitrogen Deposition to a Southeast Tibetan Forest Ecosystem

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Abstract: With atmospheric reactive nitrogen (Nr) emissions increasing globally, research into Nr deposition has attracted increasing attention, especially in remote environments. These ecosystems are very sensitive to global change, especially enhanced Nr deposition. Forest environments, in particular, are highlighted because of their important ecological function. We quantified atmospheric Nr concentrations and deposition over four years of continuous monitoring in a southeast Tibetan boreal forest ecosystem, an ecosystem in which forest biomass and carbon density are high around the world. Average annual bulk Nr deposition was $3.00 \text{ kg N ha}^{-1} \text{ y}^{-1}$, with those of reduced and oxidized species estimated at 1.60 and $1.40 \text{ kg N ha}^{-1} \text{ y}^{-1}$, respectively. Bulk deposition of both NH_4^+ and NO_3^- were controlled by precipitation amount: both Nr deposition and precipitation were highest in summer and lowest in winter. Dry deposition of NH_3 and NO_2 were 1.18 and $0.05 \text{ kg N ha}^{-1} \text{ y}^{-1}$, respectively. Atmospheric NH_3 concentrations were in the range $1.15\text{--}3.53 \text{ mg N L}^{-1}$, highest in summer and lowest in winter. In contrast, no clear trend in seasonal NO_2 concentrations was observed. Monthly NO_2 concentrations were $0.79\text{--}1.13 \text{ mg N L}^{-1}$. Total Nr deposition (bulk plus dry) was $4.23 (3.00 + 1.23) \text{ kg N ha}^{-1} \text{ y}^{-1}$ in the forest. Reduced nitrogen was the dominant species. In conclusion, Nr deposition was in the range at which forest net productivity and carbon sequestration are sensitive to any variation in nitrogen input, so quantification of Nr deposition should continue and with greater detail.

Keywords: Southeast Tibet; forest ecosystem; nitrogen deposition; reduced nitrogen; oxidized nitrogen

1. Introduction

With rapid economic growth (e.g., agricultural, industrial, urbanization), atmospheric reactive nitrogen (Nr) deposition has increased significantly and has caused a series of ecological and environmental problems [1,2], especially in China [3] and many developing countries in recent years [4]. In order to evaluate the potential effects caused by additional nitrogen inputs via atmospheric deposition, Nr deposition has been measured in different environments around the world [5–7], including forest ecosystems [8–10].

Nr deposition to forests can result in both harmful and beneficial effects. Generally, additional Nr inputs to already N-rich ecosystems result in soil acidification, nutrient imbalance and a decrease in plant diversity [11,12]. However, positive effects, such as additional carbon sequestration, can be caused by enhanced N deposition in N-limited environments [13]. In China, tropical and subtropical forests are N-rich ecosystems, but temperate and boreal forests are N-limited [14]. Due to the high

population density and intensive human activity in tropical and subtropical regions, Nr deposition is relatively high in these areas [15]. Therefore, the impacts of increasing Nr deposition and its impacts have been quantified in south China, where these forests are found [16,17]. In contrast, quantification of fluxes and impacts of Nr are rare in boreal forests because Nr deposition and human disturbance have been traditionally considered as low. However, Nr pollution has no bounds [18]. It has been reported that tree growth rates have accelerated in boreal forests, even where human activity is limited [19], and carbon sequestration was significantly enhanced in forests when Nr deposition was between 3 and 11 kg N ha⁻¹ yr⁻¹ [20]. In conclusion, although no substantial deleterious effects of Nr pollution have been observed in low N deposition regions, quantification of Nr deposition fluxes in boreal forest ecosystems is crucial because both net productivity and carbon sequestration are sensitive to additional nitrogen inputs.

The Qinghai-Tibet Plateau is considered to be an N-limited ecosystem, sensitive to global change, such as increased Nr deposition [21,22]. To date, quantification of Nr deposition using long-term in situ monitoring is rare in this region because of limited access and technical resources. Sporadic research has revealed that Nr deposition is relatively low but has increased in Nyingchi City, the main city in the region, during recent years [23]. In fact, large spatial variation in Nr deposition exists in the plateau, as high as 18 kg N ha⁻¹ y⁻¹ in urban areas and as low as 0.44 kg N ha⁻¹ y⁻¹ in remote areas [24,25], but none of the previous research has focused on forest ecosystems in the plateau. Understanding Nr deposition is essential, especially in the southeast Tibetan boreal forest ecosystem, which has one of the highest levels of carbon storage and carbon density in forest ecosystems around the world [26]. We, therefore, established an in situ monitoring site and continuously measured Nr deposition for four years in a boreal forest ecosystem at Sejila Mountain in the Qinghai-Tibet Plateau, with the aim of: (1) measuring current atmospheric Nr concentrations and Nr deposition to the forest; (2) quantifying the dominant Nr deposition species. The results will help us to predict the positive or negative effects that would occur in a southeast Tibetan boreal forest ecosystem from enhanced Nr deposition.

2. Experiments

2.1. Sampling Location

The Nr deposition monitoring site was established at the National Field Scientific Observation Station of Alpine Forest Ecosystem (29°38' N, 94°42' E, 3950 m a.s.l.) (Figure 1). This is part of the Chinese Forest Ecosystem Research Network (CFERN) and National Ecosystem Research Network of China (CEORN). The annual average precipitation and average temperature were approximately 1000 mm and −0.73 °C, respectively. The station is surrounded by local primary fir forest, and there is no Nr pollution source nearby, except for one state road (#318). The dominant vegetation species is *Abies georgei* var. *Smithii*. Most of these trees are more than one hundred years old. The undergrowth is *Sorbus* (*Sorbus rehderiana* var. *rehderiana*), *Rosa* (*Rosa omeiensis*, *Rosa sikangensis*), and *Lonicera* (*Lonicera saccata*; *Lonicera angustifolia*) as well as other herbaceous plants.

2.2. Precipitation and Gaseous Nr Measurements

Precipitation was collected with simple precipitation gauges from January 2016 to December 2019. This method has been widely used in previous studies [27]. Briefly, a gauge (SDM6, Tianjin Weather Equipment Inc., Tianjin, China) was placed on the roof at the observation station where there were no surrounding obstacles. The gauge was continuously open, and precipitation (rainwater and snow) collected by hand once a week. (The samples will therefore contain wet and some dry deposition). Collected samples were transferred to 50 mL pre-cleaned plastic bottles and stored at −18 °C until analyzed. After each sampling, deionized water was used to wash the steel funnel and glass bottles of the rain gauge to eliminate contamination.

Figure 1. Location of the Nyingchi City and nitrogen deposition monitoring site.

Atmospheric NH_3 and NO_2 only were measured using passive samplers. This is commonly accepted as providing an approximate measure of atmospheric Nr concentrations [28]. NH_3 was captured using adapted low-cost, high absorption (ALPHA; designed by the Center for Ecology and Hydrology, Edinburgh, UK) passive samplers; NO_2 was collected using Gradko diffusion tubes (Gradko International Limited, Winchester, UK). ALPHA passive samplers and Gradko diffusion tubes were fixed beside the rain gauge at the height of 2 m above the roof and exchanged with new samplers or diffusion tubes on the first day of each month. There were three duplicate ALPHA samplers and diffusion tubes. Both NH_3 and NO_2 samples were measured within 48 h of sampling.

2.3. Analytical Procedures

Precipitation samples were analyzed as in a previous study [28]. Briefly, samples were thawed and filtered through a syringe filter (0.45 μm , Tengda Inc., Tianjin, China). NH_4^+ and NO_3^- in the filtered solutions were measured by AA3 continuous-flow analysis (Bran + Luebbe GmbH, Norderstedt, Germany). The detection limits were 0.01 mg N L^{-1} .

Gaseous NH_3 captured by ALPHA samplers was extracted with 10 mL-high-purity water and measured by AA3 continuous-flow analyzer as above. Gaseous NO_2 captured by Gradko diffusion tubes was extracted with a solution containing sulfanilamide, H_3PO_4 and N-1-Naphthylethylen-diamine, and the concentration of NO_2^- determined by colorimetry at a wavelength of 542 nm. A blank sample was measured at the same time to eliminate errors.

2.4. Calculations and Statistical Analysis

Bulk Nr deposition (N_b) was calculated as the Nr concentration (N_c) multiplied by precipitation amount (P) in each precipitation event using:

$$N_b = N_c \times P \times 10^{-2}$$

NH_4^+ and NO_3^- were calculated independently. The units of N_b , N_c and P were kg N ha^{-1} , mg N L^{-1} and mm, respectively. 10^{-2} is a unit conversion factor.

Monthly bulk Nr deposition (N_s) was calculated by summing all bulk deposition fluxes collected in a month, as follows:

$$N_s = \sum_0^i N_b$$

where i represents the number of samples in the month.

The monthly Nr concentration (N_c) in precipitation was calculated by dividing N_s by precipitation amount (P_s) in the month:

$$N_c = N_s/P_s$$

$$P_s = \sum_0^i P$$

where i represents the number of precipitation events in the month.

Dry deposition was calculated as the atmospheric Nr concentration multiplied by the deposition velocity. Monthly atmospheric Nr concentrations were obtained from the measured NH_3 and NO_2 concentrations and deposition velocities in forests from Flechard et al. [10]; these have been widely used to estimate Nr deposition in forest environments [29,30]. Deposition velocities of NH_3 and NO_2 were 1.6 cm s^{-1} and 0.16 cm s^{-1} , respectively.

We defined March to May as spring, June to August as summer, September to November as autumn, and December, January and February as winter.

Variance analysis was performed by SPSS 19.0. The Duncan test was used to test the significance between monthly precipitation amounts, NH_4^+ deposition, NO_3^- deposition, and atmospheric NH_3 and NO_2 concentrations. Figures were drawn with originPro8.

3. Results

3.1. Variation in Monthly Precipitation and NO_3^- and NH_4^+ Deposition

Precipitation varied throughout the year (Figure 2a), first increasing and then decreasing. It was highest in July (183.4 mm) and lowest in December (3.35 mm). No significant differences between amounts of precipitation in July and August were observed, based on the Duncan test. Precipitation in January, February, November and December were also similar. Variation in both NO_3^- -N and NH_4^+ -N deposition in different months showed the same pattern as precipitation. Bulk Nr deposition mainly occurred from March to October, accounting for 96.2 percent of total annual NO_3^- -N deposition and 97.2 percent of annual NH_4^+ -N deposition. Monthly bulk deposition of NO_3^- -N and NH_4^+ -N ranged from 0.01 – $0.23 \text{ kg N ha}^{-1}$ and 0.01 – $0.28 \text{ kg N ha}^{-1}$, respectively (Figure 2b,c).

3.2. Variations in the Monthly Volume Weighted Mean NO_3^- and NH_4^+ Concentrations and Their Relationship

Monthly volume-weighted mean concentrations of both NO_3^- and NH_4^+ showed similar patterns of high in winter and low in summer, with little variation from March to September. Monthly concentrations of NO_3^- and NH_4^+ were in the range of 0.12 – 0.39 and 0.16 – 0.38 mg L^{-1} , respectively (Figure 3a,b). In order to understand the predominant Nr deposition forms in different months, monthly concentrations of both NO_3^- and NH_4^+ were used in a linear fitting procedure. The result revealed that monthly NO_3^- and NH_4^+ concentrations in precipitation are positively significantly correlated ($R^2 = 0.951$) (Figure 3c), and the dominant Nr form shifted from NH_4^+ to NO_3^- as the bulk Nr concentration increased.

Figure 2. Variation of monthly precipitation amount (a), NO_3^- (b) and NH_4^+ (c) deposition. Different letters above the error bar means that the difference reaches a significant level. Correspondingly, the same letters exists above the error bar that means the difference is insignificant.

3.3. Variation in the Monthly Atmospheric NO_2 and NH_3 Concentrations

Monthly mean concentrations of atmospheric NO_2 and NH_3 were in the ranges of 0.79–1.13 and 1.15–3.53 mg N L^{-1} , respectively (Figure 4). There was no obvious trend of monthly atmospheric NO_2 concentrations throughout the year. In contrast, atmospheric NH_3 concentrations increased from January to May and then decreased from August to December. The highest NH_3 concentrations occurred from March to August, corresponding to the plant growing season.

3.4. Annual Trends in Atmospheric Nr Concentrations and Bulk Deposition

Atmospheric Nr concentrations increased from 2016 to 2019, being 3.11, 2.79, 3.52 and 3.62 $\mu\text{g N m}^{-3}$ in 2016, 2017, 2018 and 2019, respectively (Figure 5a). NH_3 was the dominant Nr species during the experimental period, with concentrations of 2.20, 1.98, 2.55 and 2.63 $\mu\text{g N m}^{-3}$ in 2016, 2017, 2018 and 2019, respectively.

Figure 3. Variation in the monthly volume-weighted mean NO_3^- (a) and NH_4^+ (b) concentrations and their relationship (c). In (c), the dashed line shows when the NO_3^- concentration is equal to the NH_4^+ concentration in precipitation. Black squares above the dashed line mean that NH_4^+ -N was the predominant N deposition form. Black squares represent the monthly volume weight mean N concentration; the red line represents the fitting line.

Bulk N_r deposition was 2.91, 3.14, 2.85 and 3.09 kg N ha⁻¹ y⁻¹ in 2016, 2017, 2018 and 2019, respectively (Figure 5b). NH_4^+ -N was the dominant species, its deposition being 1.56, 1.70, 1.51 and 1.62 kg N ha⁻¹ y⁻¹ in 2016, 2017, 2018 and 2019, respectively.

Annual dry deposition of NH_3 and NO_2 was calculated from their deposition velocities multiplied by their concentrations. In 2016, 2017, 2018 and 2019, the deposition of NH_3 was 1.11, 1.00, 1.29 and 1.33 kg N ha⁻¹ y⁻¹, respectively, and of NO_2 was 0.05, 0.04, 0.05 and 0.05 kg N ha⁻¹ y⁻¹, respectively.

The average total annual N_r deposition (bulk plus dry) was 4.23 (3.00 + 1.23) kg N ha⁻¹ y⁻¹. Reduced nitrogen was dominant, accounting for 65.7 percent of the total deposition. The percentage of reduced nitrogen was very high in the dry deposition, being 96.1 percent of the annual dry N_r deposition.

Figure 4. Variation in monthly atmospheric NO₂ (a) and NH₃ (b) concentrations. Different letters above the error bar means that the difference reaches a significant level. Correspondingly, the same letters exists above the error bar that means the difference is insignificant.

Figure 5. Annual trends in atmospheric Nr concentrations (a) and bulk Nr deposition (b).

4. Discussion

4.1. Bulk Nr Deposition and Monthly Precipitation

Bulk Nr deposition was determined from precipitation amount and Nr concentrations in precipitation [28]. We found that Nr deposition was mainly controlled by precipitation, increasing with precipitation. It has been reported that Nr deposition is logarithmically related to precipitation amount because atmospheric NH_4^+ and NO_3^- are scavenged by rainfall, and scavenging decreases as rainfall intensity, duration and amount increases [31]. At our site, the relationship between precipitation amount and Nr deposition fitted both linear and logarithmic equations. However, with a linear relationship for both NH_4^+ and NO_3^- giving the better fit (Figure 6a,b). This could be because atmospheric NH_4^+ and NO_3^- concentrations were both so low as to be almost unaffected by rainfall intensity, duration and amount. Although we could not measure gaseous/particulate NH_4^+ and NO_3^- concentrations because of limited technical resources, the low NH_3 and NO_2 concentrations support this idea to some extent (Figure 4a,b). Overall, precipitation amount was the dominant factor determining Nr deposition, both Nr deposition and precipitation showing similar patterns of monthly variation.

Figure 6. Linear and logarithmic relationships between precipitation and NO_3^- (a) and NH_4^+ (b) concentrations. Black squares represent the Nr concentration in each precipitation sample; the red line represents the fitting line.

4.2. Variation of Monthly Bulk Nr Concentrations and Atmospheric Nr Concentrations

Bulk Nr concentrations are usually inversely correlated with precipitation amount [28]. We found this, with high concentrations of both NH_4^+ and NO_3^- in November to February when precipitation amounts were relatively low (Figure 2a and (Figure 3a,b). This trend could have been amplified by the collection of some dry deposition (e.g., dust) in the continuously open precipitation gauges, which would have had the biggest effect when precipitation was low. Monthly variation of NH_4^+ concentrations in precipitation was comparatively less than those of both precipitation amount and atmospheric NH_3 concentrations from January to September. Both atmospheric NH_3 concentrations and precipitation amounts were high in Spring and Summer (Figure 2a and (Figure 3b). However, bulk NH_4^+ concentrations were not low, i.e., not diluted by large amounts of precipitation, because of high atmospheric NH_3 concentrations. However, the dominant Nr species in precipitation changed from

NH_4^+ to NO_3^- as bulk Nr concentrations increased (Figure 3c). Low monthly bulk Nr concentrations mainly occurred in spring and summer.

NH_3 emissions are usually the largest in summer due to intensive agricultural activities and higher temperatures [32]. Atmospheric NH_3 concentrations were highest in summer at our site (Figure 4b) and in a range of different ecosystems, as shown by in situ monitoring [23,30]. Ambient NO_2 is mainly attributed to industrial sources (e.g., fossil fuel burning; automobile exhausts; power plants), and so is highest in the most populated areas [33,34]. Although some research has shown that NO_2 exhibits seasonal variation [29,35] because of fossil fuel burning in winter, there were no NO_2 pollution sources near our site other than the state road (#318), and so NO_2 concentrations showed little seasonal variation.

4.3. Reduced and Oxidized Nitrogen Deposition in Bulk and Dry Deposition

Du et al. reported that NH_4^+ dominated bulk Nr deposition in forest ecosystems in China, with a mean bulk deposition of $9.4 \text{ kg N ha}^{-1} \text{ y}^{-1}$ for $\text{NH}_4^+\text{-N}$ and $3.9 \text{ kg N ha}^{-1} \text{ y}^{-1}$ for $\text{NO}_3^-\text{-N}$ [36]. Our results revealed that both reduced and oxidized nitrogen deposition in this Southeast Tibetan forest ecosystem were much lower than the average bulk Nr deposition in China, with values of $1.60 \text{ kg N ha}^{-1} \text{ y}^{-1}$ for reduced nitrogen and $1.40 \text{ kg N ha}^{-1} \text{ y}^{-1}$ for oxidized nitrogen, with reduced nitrogen dominant.

Based on the NitroEurope network, Flechard et al. found that mean concentrations of NH_3 and NO_2 were 1.03 and $2.15 \mu\text{g N m}^{-3}$ in forest ecosystems in Europe [10]. We measured NH_3 and NO_2 concentrations of 2.34 and $0.92 \mu\text{g N m}^{-3}$, respectively. Atmospheric NH_3 concentrations in southeast Tibet were, therefore, higher than the average in Europe, but those of NO_2 lower. This is not surprising because the Tibetan forest is in a remote area, surrounded by dense primordial forest. Low NO_2 concentrations can be attributed to the fact that atmospheric Nr sources are mostly natural.

High atmospheric NH_3 concentrations can be explained by fog events that frequently occur in the research area, resulting in high NH_4^+ concentrations [37]. Average atmospheric NH_3 and NO_2 concentrations across China are 6.1 and $6.8 \mu\text{g N m}^{-3}$ [28], almost 2.6 and 7.4 times than those in our research area, respectively.

Clearly, the atmosphere in the southeast Tibetan forest is relatively “clean”, especially for NO_2 . However, ammonia is likely to be more ecologically and biologically toxic than oxidized species [38,39], and the annual deposition of reduced nitrogen was higher than that of oxidized nitrogen. Therefore, this southeast boreal forest ecosystem could be threatened by Nr deposition dominated by reduced species.

4.4. Comparison of Nr Deposition and Potential Ecological Effects with Other Regions

It is interesting to compare the current Nr deposition to this southeast Tibetan forest ecosystem with bulk deposition to other forest ecosystems around the world published in the last decade (Table 1). Clearly, bulk Nr deposition to our site is very low. It has been reported that external Nr inputs can increase long-term carbon sinks in boreal, temperate and tropical forests [20,40]. Thomas reported that forest carbon sequestration was significantly enhanced when Nr deposition was between 3 and $11 \text{ kg N ha}^{-1} \text{ y}^{-1}$ [20]. It had been reported that the critical nitrogen load of deciduous conifers in cold-temperate and temperate mountains was in the range $5\text{--}10$ (or $10\text{--}15$) $\text{kg N ha}^{-1} \text{ y}^{-1}$ [11,41]. Considering the low annual Nr deposition to this Tibetan forest ecosystem, which is lower than the Nr deposition critical load, the changes of forest net productivity should be given more attention.

Table 1. Comparison of bulk/wet Nr deposition in a range of forest ecosystems.

Location	Sampling Years	Bulk NH_4^+ Deposition ($\text{kg N ha}^{-1} \text{ y}^{-1}$)	Bulk NO_3^- Deposition ($\text{kg N ha}^{-1} \text{ y}^{-1}$)	Bulk Inorganic Nr Deposition ($\text{kg N ha}^{-1} \text{ y}^{-1}$)	References
Wuyi Mountains (Fujian Province)	2014–2015		20.2		[42]
Shitai county (Anhui Province)	2014–2015	9.24	10.5	27.7	[43]
Lingyan Mountain (Sichuan Province)	2015–2016	16.4	10.0	26.4	[44]
Tiantong Mountain (Zhejiang Province)	2011–2013	10.4	8.49	18.9	[45]
Jigong Mountain (Henan Province)	2014–2015	7.90	7.60	15.5	[29]
Shengnongjia Mountain (Hubei Province)	2015	5.24	6.65	11.98	[46]
Fuji Mountain Japan				8.04	[47]
Heihe Tianlaochi (Gansu Province)	2015	0.30	5.55	5.85	[48]
Tianmu Mountain (Zhejiang Province)	2013–2014			5.25	[49]
Sejila Mountain (Tibet Autonomous Region)	2016–2019	1.60	1.40	3.00	This Study
Switzerland	2014	1.04–10.7	1.55–6.86	2.58–17.56	[50]
17 forest sites (Swiss)	2014	1.00–10.7	1.40–8.60	2.40–18.4	[9]
East of England	2005–2010			6.6–13.8	[51]
European forests	2010–2020			>6.5	[52]
Gongga Mountain (Sichuan Province)	2008–2011	3.17–4.80	1.66–3.00		[53]
Spanish forests (Spain)	2011–2013			2.42–6.83	[34]
Loch Vale Colorado	2010–2017			2.52–4.58	[54]

4.5. Implications and Uncertainty Analysis

Although bulk Nr deposition was more precisely measured than dry Nr deposition, there are errors in the former. For example, both NO_3^- and NH_4^+ in precipitation are absorbed by the forest canopy, especially in lower Nr deposition regions [9]. Additionally, total bulk Nr deposition is underestimated when dissolved organic nitrogen (DON) is not measured, as was the case in this work. Izquieta-Rojano et al. reported that the contribution of DON to total bulk Nr deposition ranged from 34% to 56% in forest ecosystems [55].

For dry deposition, deposition velocities are influenced by several factors and vary across ecosystems [9,10]. The deposition velocity of NH_3 is often overestimated in forest ecosystems because NH_3 exchange to foliar surfaces is bidirectional, so NH_3 deposition can be overestimated. Due to poor technical resources at our research site, we could not estimate the actual Nr deposition velocities of different Nr species. We used deposition velocities of NH_3 and NO_2 to “background environments” across China to calculate Nr deposition (NH_3 : 0.47 cm s^{-1} ; NO_2 : 0.17 cm s^{-1}) [28], making the deposition of NH_3 and NO_2 0.33 and $0.05 \text{ kg N ha}^{-1} \text{ y}^{-1}$, respectively. Reduced nitrogen deposition still dominates.

Flechard et al. reported that the percentages of NH_3 in total dry reduced nitrogen deposition and NO_2 in total dry oxidized nitrogen deposition were 77.4% and 21.1% [10], respectively. Clearly, NH_3 was the dominant species in dry reduced nitrogen deposition, but NO_2 was not so important in dry oxidized nitrogen deposition. Dry deposition of NH_3 and NO_2 were 1.18 and $0.05 \text{ kg N ha}^{-1} \text{ y}^{-1}$ in our research area. We used these percentages (77.4% and 21.1%) to convert NO_2 deposition to total oxidized nitrogen deposition and NH_3 deposition to total reduced nitrogen deposition giving values of total reduced nitrogen and oxidized nitrogen of 1.52 and $0.24 \text{ kg N ha}^{-1} \text{ y}^{-1}$, respectively. Reduced nitrogen was the dominant Nr deposition species in this southeast Tibet forest ecosystem.

5. Conclusions

Based on four years of continuous monitoring, we found that Nr deposition was small in a southeast Tibet boreal forest. Average concentrations of atmospheric NH_3 and NO_2 were 2.34 and $0.90 \mu\text{g N m}^{-3}$, respectively. Bulk deposition of NH_4^+ and NO_3^- was 1.60 and $1.40 \text{ kg N ha}^{-1} \text{ y}^{-1}$, respectively. Nitrogen deposition showed a clear seasonal variation—high in summer and low in winter—dependent on the monthly variation in precipitation. Additionally, atmospheric NH_4^+ concentrations were high in summer and low in winter. Annual Nr deposition was less than the critical load ($5\text{--}10 \text{ kg N ha}^{-1} \text{ y}^{-1}$), being $4.23 \text{ kg N ha}^{-1} \text{ y}^{-1}$ and dominated by reduced nitrogen. Deposition was mainly concentrated in the months of plant growth. As a result, carbon sequestration and net productivity in this southeast Tibet boreal forest ecosystem deserve more research because these regions may benefit from slightly increased Nr deposition. More comprehensive methods, as used in Nr deposition monitoring, should be employed in these regions sensitive to global change.

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