Effects of Simulated Nitrogen Deposition on Leaf-Litter Decomposition and Nutrient Release of A Cold Temperate Coniferous Forest in Jiaozi Snow Mountain National Nature Reserve, Southwest China

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Abstract

Purpose

Litter decomposition is a key process of nutrient cycling in terrestrial ecosystems, an important part of the global carbon budget, and deeply affected by global atmospheric nitrogen deposition. However, the effects of different forms of N addition on litter decomposition and nutrient release are unclear in a cold temperate coniferous forest in a subtropical Chinese plateau.

Methods

Three N sources $(NH_4)_2SO_4$, $NaNO_3$, and NH_4NO_3 were used in the gradient N deposition method. Each N source was divided into four treatments, from low to high, they were CK (control 0 kg $N \cdot hm^{-2} \cdot a^{-1}$), low N (low-N 5 kg $N \cdot hm^{-2} \cdot a^{-1}$), medium n (medium-N 15 kg $N \cdot hm^{-2} \cdot a^{-1}$), high N (high-30 kg $N \cdot hm^{-2} \cdot a^{-1}$), and each treatment repeated three times.

Results

After two years, the litter decomposition rates of low and medium ammonium nitrate treatments were the fastest as compared to the control, while high and low ammonium nitrate treatments were the slowest. Under the same nitrogen deposition conditions, the litter decomposition rates of low nitrogen treatments were higher than high nitrogen treatments. The order of litter decomposition rates was ammonium nitrate > ammonium sulfate > sodium nitrate. Nitrogen deposition decreased the amount of C in litter leaves but increased N and P levels slightly. Phosphorus changes over time were more complex than C and N over time.

Conclusions

These results showed that high nitrogen deposition in the future could increase litter decomposition rates and delay the nutrient release, which may be beneficial to improve soil carbon sequestration.

Declarations

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Introduction

Human activities have altered the global climate and environment over the past 20 years, which includes an increase in atmospheric nitrogen deposition (Lv et al. 2013). Atmospheric nitrogen deposition, particularly nitrogen oxides, comes primarily from fossil fuel emissions and ammonia from fertilizer production and use. Nitrogen deposition continues to increase in the world's industrial areas and has accelerated in many developing regions (Galloway et al. 2003; Matson et al. 1999, 2002). As such, atmospheric nitrogen deposition constitutes a global problem; nitrogen release has increased from 35 TgN/yr(1850) to 104 TgN/yr and will likely continue to increase in the future (Galloway et al. 2008), perhaps reaching 114 TgN/yr in 2100 (Wang et al. 2017). China currently undergoes severe atmospheric nitrogen deposition. From 1980–2100, the annual growth rate was 0.41 kgN/$hm^{2,}$ and that amount increased by 60% from the 1980s (13.2 kgN/hm^2) into the early 21st century (21.1 kgN/hm^2) (Liu et al. 2013). Elevated atmospheric nitrogen deposition is a prevailing problem that affects nearly every functioning aspect and composition of forest ecosystems (Aber et al.2003), such as N dynamics, N runoff, soil enzyme activities, and litter decomposition (Chen and Mulder 2007).

Litter decomposition plays an important role in carbon and nutrient cycles in forest ecosystems (Melillo et al. 1982) because litter decomposition releases carbon dioxide (CO_2) and soil nutrients (Berg and Mcclaugherty 2008; Wood et al. 2006). Abiotic and biological factors such as climate, soil physical and chemical properties, litter chemical properties, nitrogen (n) availability, and soil biological activities impact litter decomposition (Berg 2000; Lavelle et al. 1993; Swift et al. 1979). As an important part of global change, nitrogen deposition is expected to change the decomposition process of litter, and ultimately affects the carbon storage and nutrient status of ecosystems (Hobbie 2008; Knorr et al. 2005; Tu et al. 2014).

In the 1980s, studies on the effects of nitrogen deposition on forest litter decomposition were conducted in Europe and North America (Emmett et al. 1998). In general, the effects of nitrogen deposition on forest litter decomposition were divided into three categories: promotion, inhibition, and no effect. For example, Hobbie and Vitousek found that simulated nitrogen deposition promoted litter decomposition (Hobbie and Vitousek 2000). A Vestgarden study reported that nitrogen addition promoted litter decomposition (Vestgarden 2001), and Anderson and Hetherington (1999) also reached a similar conclusion. Kuperman (1999) analyzed factors that impacted litter decomposition and concluded that nitrogen deposition was the most important factor to accelerate litter decomposition. Some scholars also found that nitrogen input inhibited forest litter decomposition. For example, in the simulated nitrogen deposition experiment of Korean pine and a broad-leaved mixed forest, Magill and Aber (1998) found that increased nitrogen

deposition inhibited forest litter decomposition. In addition, some studies have shown that nitrogen input initially promotes litter decomposition but inhibits it in later stages. For example, Zhang et al. found in simulated nitrogen deposition of fir litter decomposition that nitrogen addition initially promoted decomposition (0–15 days) but inhibited it after many weeks (47 days, Zhang and Wang 2012).

Previous studies suggested that the mechanism of nitrogen deposition promoting forest litter decomposition may proceed as follows: (1) in the short term, nitrogen input alleviates the nitrogen limitation of microbial activities, which is conducive to microorganism growth and promotes litter decomposition (Carreiroet al. 2000; Norris et al. 2013); (2) nitrogen increases lead to nitrogen absorption and accumulation increases in plants, which increases nitrogen levels in the forest litter and changes the amounts of other elements as well. Litter with high levels of nitrogen promote litter decomposition (Ågrenet al. 2001; Magill et al. 2000; Mo et al. 2004; Vestgarden 2001); (3) nitrogen input promotes plant biomass and litter input, thus changing the forest surface vegetation composition and litter microenvironment. This increases the abundance of some nitrogen indicator species, which results in litter decomposing readily and a higher litter decomposition rate (Berg and Matzner 1997; Shaver et al. 2001; Suding et al. 2005); (4) nitrogen input increases cellulase and lignin enzyme activities, which promotes litter decomposition (Fenn et al. 1998; Fog 1988; Hobbie 2000; Saiya-Cork et al. 2002; Waldrop et al. 2004). The mechanism of nitrogen deposition inhibiting forest litter decomposition my proceed as follows: (1) nitrogen input changes the community structure and microorganism diversity, reduces the biomass and microorganism activity, and inhibits the litter decomposition rate (Aber et al. 1989; Berg et al. 2001; Carreiroet al. 2000; Compton et al. 2004; Cox et al. 2001); (2) nitrogen input increases the lignin and cellulose levels in the litter, forms stable compounds and refractory humic compounds during decomposition, and reduces the litter decomposition rate (Aertset al. 2006;Berg et al. 2001; Manning et al. 2008); (3) nitrogen input inhibits basidiomycetes albicans or Fusarium fulvum growth and limits lignin production degrading enzymes, thus delaying litter decomposition (Berg and Matzner 1997; Koide and Wu 2003;Thirukkumaranand Parkinson 2000); (4) nitrogen input limits C or P on microbial degradation, which results in the lack of energy for litter decomposition and slows the litter decomposition rate (Kuperman 1999; Xu et al. 2006); (5) nitrogen input inhibits soil animal activities or stops them altogether (Hobbie2000). The no-effect pathway may proceed as follows: (1) during litter decomposition, the impacts of nitrogen input offset (Gundersen 1998;Hobbie 2000) (2) the forest ecosystem itself is not limited by nitrogen, or the forest has strong adaptability to nitrogen input, so that nitrogen input has no effect on litter decomposition (Gundersen 1998; Kwabiah et al. 1999) (3) the quality of litter carbon was insufficient to induce a decomposition response to nitrogen input (Hobbie 2000).

Although the effects of nitrogen addition on litter decomposition have been studied, most previous studies have focused on tropical, subtropical, and temperate regions (Ge et al. 2013; Mo et al. 2004; Zhu et al. 2016). In contrast, relatively few studies have explored the effects of nitrogen addition on boreal coniferous forest decomposition in the subtropical plateau of China. Furthermore, numerous studies have investigated the effect of a single nitrogen source on litter decomposition in forest ecosystems (Green et al. 2006;Sinsabaugh et al. 2002; Wang et al. 2010). However, a single nitrogen source may not accurately reflect the ecological effect of atmospheric nitrogen deposition on litter decomposition in forest

ecosystems, which underscores the importance of identifying the effects of different forms of nitrogen addition on litter decomposition.

In this study, we simulated different forms of nitrogen deposition to explore their effects on the litter decomposition rate and nutrient release of boreal coniferous forests of Jiaozixueshan National Nature Reserve in the subtropical plateau of Southwest China. The objectives of this study were to determine differences in litter decomposition of boreal coniferous forests in response to different forms of nitrogen addition. We tested the following hypotheses: (1) Nitrogen addition promotes litter decomposition of boreal coniferous forests; (2) Litter decomposition is faster when adding NH_4NO_3 rather than $(NH_4)_2SO_4$ or $NaNO_3$; (3) nitrogen addition delays release of nitrogen and phosphorus from litter.

Materials And Methods

The study area

The research site was located in the jiaozixueshan National Nature Reserve, Kunming, Yunnan province, Southwest China, with an area of 16193 hm^2 between 102° 48'–102° 58' E and 26° 00'– 26° 11' N. The cold temperate coniferous forest below the glacier snow line, which provided the most complete protection in the jiaozixueshan nature reserve, was taken as the study area. The experimental site was located in the field experimental research base of the Institute of Ecology and Environment, Yunnan University. Jiaozishan National Nature Reserve rises from its lowest elevation of 1200 m in the Pudu River Valley to its highest elevation (4344.1 m) in the Xueling Mountains, for an elevation change of 3144.1 m. It is the remnant of the Gong Wang Mountain system and the highest mountain in North Central Yunnan. It is known as "the first mountain in Central Yunnan". The nature reserve has developed the most complete climate, soil, vegetation, natural vertical band spectrum, and the most abundant vegetation types in the eastern Yunnan Plateau. It typifies vegetation in the eastern Yunnan Plateau, and also the best-preserved area of the original vegetation in the central Yunnan region. This includes the largest area of Alpine Forest in China, the lowest latitude and altitude of the Chinese Alpine Forest, and the easternmost and lowest altitude of Abiesfargesii in China. There are 1611 species of wild vascular plants belonging to 154 families, 507 genera, and 293 species of wild vertebrates recorded in the reserve. The annual average temperature ranges from 0–13 °C, and annual precipitation averages between 916–1390 mm. The base soil of the reserve is red soil.

Experimental design

In order to eliminate the influence of species composition and microenvironment heterogeneity on the test results, a randomized block design was adopted for this study. The species composition, community structure, and habitat in the plant community were relatively uniform, the terrain was relatively gentle, and the community area was large enough to make enough buffer zones around the test area as sample plots. In the selected sample plots (*Abiesgeorgei*Orr), a total of 30 20 m × 20 m quadrat groups were established. The 20 m × 20 m quadrats of each group were at the same level, and the spacing of each

test quadrat was at least 10 meters to prevent mutual interference. A 15 m × 15 m central area was placed in the set quadrats. The project studied and analyzed the effects of NH_4^+ and NO_3^- inputs and their interaction (through the NH_4NO_3 input analysis of their interaction) on soil carbon cycling microorganisms and organic carbon components in a cold temperate coniferous forest, combined with domestic and foreign nitrogen deposition simulation experiments, three nitrogen sources $(NH_4)_2SO_4$ (NHS), $NaNO_3$ (NAN), NH_4NO_3 (NHN) (analytical purity) were established, and the gradient N deposition method was used. Each N source was divided into four treatments, from low to high, they were CK (control 0 kg $N{\cdot}hm^{-2}{\cdot}a^{-1}$), low N (low-N 5 kg $N{\cdot}hm^{-2}{\cdot}a^{-1}$), medium N (medium-n 15 kg $N{\cdot}hm^{-2}{\cdot}a^{-1}$), and high N (high-30 kg $N{\cdot}hm^{-2}{\cdot}a^{-1}$). Each treatment was repeated three times. Based on the randomized block test design, each treatment was set in the established 20 m × 20 m quadrat block. In October 2018, before sampling the forest floor, newly fallen, naturally withered, aspen needles were collected from the ground of the research plots, and those needles were collected from the forest, and dried naturally for later use. Litter decomposition was conducted by the litter-bag method. The sample weight in each decomposition bag was 10 g. Twenty bags of varying decomposition materials were selected, dried to constant weight at 80 °C, and weighed to determine the moisture content. The samples in each bag were crushed and filtered using a 2 mm sieve and marked as the initial samples for analysis. In January 2019, the decomposition bags were randomly placed in the corresponding sample plots. Seventy-five decomposition bags in each plot were randomly buried at six different points for a total of 2250 bags.

Method of simulating N-type settlement

After establishing the sample plot, the simulated N settlement treatment began and lasted for two years. During the entire test period, the annual dosage was divided into 12 equal parts and sprayed in solution form at the beginning of each month. According to the treatment level, the $(NH_4)_2SO_4$, $NaNO_3$, and NH_4NO_3 needed for each quadrat were dissolved in 20 L water and sprayed evenly in the woodland. The control plot was sprayed with the same amount of water to reduce the impact of additional water on the forest biogeochemical cycle.

Sample collection

Litter bags were randomly retrieved from each plot at 3-month intervals. From January 2019 to December 2020, the litter bags were collected eight times. Each time, three bags of each decomposition material were randomly collected from each sample plot and returned to the laboratory. After sample collection, they were air-dried to remove sediment and debris stuck to the outside of the litter bags. The litter bags were cut to remove the decomposition materials. Invasive roots and soils were removed with tweezers and brushes, placed in paper bags, and numbered. The litter was dried to constant weight at 80 °C and weighed to calculate the mass loss rate of litter leaves during decomposition. Each litter bag was crushed and sieved for analysis. In the later decomposition stage, the large mass loss required the use and combination of three sample bags from the same decomposition material for analysis. The total

nitrogen, phosphorus, potassium, calcium, and magnesium levels were measured, and the nutrient release rate and residual rate were calculated.

Laboratory analysis

Concentrations of C and N in tree organs, above-ground herbs, underground herbs, standing litters, and soils were determined using a Vario EL □ Elemental Analyzer (Elementar Scientific Instruments, Germany). Phosphorus concentrations in tree organs, above-ground herbs, underground herbs, standing litters, and soils were determined by the $HClO_4$-H_2SO_4 colorimetric method.

Data analysis

The calculation method of mass remaining (MR) in each stage is as follows:

$$MR = (w1 / w0) \times 100 \quad (1)$$

where w1 is the mass of litter in this stage (g), and w0 was the initial litter mass (g).

Calculating the nutrients remaining (NR) in each stage is as follows:

$$NR = (C1 / w1) / (C0 / w0) \times 100 \quad (2)$$

where C1 is the nutrient content of litter in this stage ($mg \cdot g^{-1}$), C0 is the initial nutrient content ($mg \cdot g^{-1}$), w1 is the mass of litter in this stage (g), and w0 is the mass of initial litter (g).

Spss25.0 software was used to fit the Olson negative exponential decay model (Olson 1963):

$$y = a \times e^{-kt}. \quad (3)$$

where y is MR (%); a is the fitting parameter; K is the annual decomposition coefficient; t is time (a). Litter decomposition T50% = -ln (1 - 0.50) / K; T95% = -ln (1 - 0.95) / K.

The calculation for the net nutrient release (NNR) of different litter materials after decomposition for 2 years is as follows:

$$NNR = 100 - NR \quad (4)$$

when NNR is positive, it means that the element undergoes net release after decomposition; in contrast, when NNR is negative, it means levels of the element declined.

Repeated measurement analysis of variance analyzed the mass and nutrients remaining of litter decomposition materials. When interactions between time and N treatment were significant, a single factor analysis of variance was used for each sampling index, and multiple comparisons were conducted by the S-N-K method. Significance levels were set at $P < 0.05$ for all statistical analyses.

Results

Effects of different nitrogen deposition forms on the dynamic change of leaf mass loss during litter decomposition

After two years of litter decomposition, the leaf mass residue of different treatments showed a gradual downward trend, especially during the first 18 months of decomposition in which the mass residue rate decreased rapidly, then gradually slowed (Fig. 1).

Table 1 shows the fitting results of the Olson index decay model. Those results show that the correlation coefficients (R^2) of the Olson index decay model showed a significant correlation ($P < 0.001$), indicating a good fit of leaf litter dry matter residue rate and time. The decomposition coefficient of the control was 0.450, and it took 1.54 and 6.66 years to decompose 50% and 95% of the litter, respectively. Compared with the control, ammonium sulfate low nitrogen treatment, ammonium sulfate high nitrogen treatment, ammonium nitrate medium nitrogen treatment, ammonium nitrate high nitrogen treatment, ammonium nitrate low nitrogen treatment, ammonium nitrate medium nitrogen treatment and ammonium nitrate high nitrogen treatment all promoted forest leaf litter decomposition. These treatments increased the decomposition rates by 18.44%, 8.44%, 4.22%, 0.22%, 41.11%, 3.43% and 10.22%, respectively, while ammonium sulfate medium nitrogen treatment and ammonium nitrate low nitrogen treatment slowed down the decomposition rate by 1.78% and 2.89%, respectively. Nitrogen deposition has a certain promoting effect on litter decomposition in a cold temperate coniferous forest.

The results of repeated ANOVA measurements showed significant differences in leaf litter mass loss measured at different time points (MS = 12498.93, F = 3811.62, DF = 8, P = 0.000), and there were also significant differences among different groups (MS = 444.504, F = 21.47, DF = 9, P = 0.000). The decomposition rate of litter leaves occurred most rapidly under low and medium ammonium nitrate treatments; the slowest rates occurred using control, high and low ammonium nitrate treatments. Under the same nitrogen deposition, the decomposition rates of litter leaves with low nitrogen treatments were higher than those using high nitrogen treatments. Combined with the decomposition coefficient (k), 50% decomposition years, 95% decomposition years and average litter residual mass rates (41.78% for ammonium sulfate, 48.56% for ammonium nitrate and 38.78% for ammonium nitrate), decomposition rates were determined: ammonium nitrate > ammonium sulfate > sodium nitrate.

Effects of different nitrogen deposition forms on C, N and P levels in leaves during litter decomposition

In general, C levels in litter leaves decreased gradually during decomposition (Fig. 2). After two years, the average amount of leaf carbon was 289.80 g/kg under ammonium sulfate treatment, 314.55 g/kg under sodium nitrate treatment and 231.63 g/kg under ammonium nitrate treatment. Compared with initial C amounts, the amount of carbon leaves dropped by 40.73% (ammonium sulfate), 36.28% (sodium nitrate) and 52.97% (ammonium nitrate).

The results of repeated ANOVA measurements showed the amount of C in leaves differed significantly depending on the nitrogen treatment and a function of time (MS = 201685.38, F = 1222.44, DF = 8, p = 0.000). The results of several ANOVA measurements showed significant differences in C levels in litter leaves under different forms and levels of nitrogen deposition at different decomposition time points, and there were also significant differences in C content of litter leaves under different forms and levels of nitrogen deposition (MS = 31352.95, F = 31.30, DF = 9, p = 0.000). Compared to the control, low and medium ammonium nitrate treatments significantly reduced the carbon content of litter leaves, while low and high sodium nitrate treatments significantly increased the amount of carbon in the leaves. There was no significant difference in the carbon content of litter leaves under other nitrogen levels.

In general, the N content of litter leaves gradually increased as the decomposition time increased (Fig. 3). After two years, average amounts of N in the leaves were 18.41 g/kg (ammonium sulfate), 17.35 g/kg (sodium nitrate), and 19.79 g/kg (ammonium nitrate), which represents increases of 50.07% (ammonium sulfate), 11% (sodium nitrate), and 52. 14% (ammonium nitrate) as compared with initial N levels.

Nitrogen levels in litter leaves varied significantly at different decomposition time points (MS = 120.67, F = 118.25, DF = 8, p = 0.000), and the N content of litter leaves between different nitrogen forms and levels also showed significant differences (MS = 24.18, F = 10.61, DF = 9, p = 0.000). Nitrogen levels in litter leaves using a low nitrogen ammonium nitrate treatment increased the most. The nitrogen treatment of sodium nitrate had the least significant impact on nitrogen levels.

Changes in P levels were more complicated than for C and N as a function of decomposition time (Fig. 4). Phosphorus amounts in litter leaves for all three nitrogen treatments decreased from 0–3 months, increased from 3–9 months, decreased from 9–15 months, increased from 15–18 months, and decreased up to 24 months. After two years, average P amounts P in leaves of the three N forms were 1.25 g/kg (ammonium sulfate), 1.23 g/kg (sodium nitrate) and 1.41 g/kg (ammonium nitrate), which represents increases of 4.28%, 10.44%, and 21.71%, respectively, as compared to initial P levels. Phosphorus levels in litter leaves differed significantly, based on their treatment type, amount, and time (MS = 0.808, F = 134.17, DF = 8, p = 0.000), but there was no significant difference between different nitrogen forms and levels (MS = 0.145, F = 1.84, DF = 9, p = 0.123).

Effects of different forms of nitrogen deposition on the release of C, N, and P in leaves during decomposition

The release of C in litter leaves was basically the same, rapidly releasing from 0–6 months, then stabilizing (Fig. 5). After two years, the average C residue rates were 24.76 % (ammonium sulfate), 30.47% (sodium nitrate), and 18.34 % (ammonium nitrate), respectively.

There were significant differences in the C residue rates of litter leaves at different decomposition time points (MS = 18369.54, F = 4416.44, DF = 8, p = 0.000), and there were also significant differences in the C residue rates using different nitrogen sources and amounts (MS = 1369.43, F = 67.50, DF = 9, p =

0.000). Compared with the control, the residue rate of leaf C was the lowest in the low ammonium nitrate treatment and highest in the low nitrogen sodium nitrate treatment.

Using different nitrogen sources and amounts, the release of N from litter leaves was basically the same, increasing from 0–9 months, then decreasing from 12–24 months (Fig. 6). After two years, the average residue rates of N were 63.97% (ammonium sulfate), 67.91% (sodium nitrate), and 60.02% (ammonium nitrate), respectively.

There were significant differences in P residue rates of litter leaves using different nitrogen sources and amounts at different time points (Fig. 7). Compared with the control, the P residue rates of litter leaves were the highest using a high treatment of ammonium sulfate, and the lowest under the low treatment of ammonium nitrate. After two years, the average residual rates of P in the leaves were 43.22% (ammonium sulfate), 54.32% (sodium nitrate), and 46.60% (ammonium nitrate).

There were significant differences in the P residue rates of litter leaves at different decomposition time points (MS = 13451.09, F = 485.09, DF = 8, p = 0.000), as well as using different nitrogen sources and amounts (MS = 739.41, F = 4.98, DF = 9, p = 0.001). Compared with the control, the P residual rate of litter leaves was the lowest under the high treatment of ammonium sulfate and the highest under the medium treatment of sodium nitrate.

Discussion

Effects of nitrogen deposition on litter decomposition

The effect of nitrogen deposition on litter decomposition is still controversial, and two aspects of endogenous litter quality and exogenous resource supply are proposed to explain the direction and magnitude of nitrogen deposition (Chen et al. 2012; Zechmeister-Boltenstern et al. 2015). In our study, after two years of litter decomposition, leaf mass residues gradually declined, (especially over the first 18 months), and the mass residue rates decreased rapidly before gradually slowing. This may be due to soluble components such as sugars and amino acids readily decompose early on when the litter decomposition rate is faster, while the relative amount of lignin increases in later stages and slows or stops decomposition (Magill and Aber 1998). Berg et al. (2010) proposed that climate and substrate-chemistry variables explained the decomposition of pine needle litter in later stages. Nitrogen in the matrix stimulated early decomposition but inhibited decomposition over time (Berg and Meentemeyer 2002; Johansson et al. 1995). In the early stage of litter development, nitrogen mainly enters the litter from the soil, which helps meet microorganism nitrogen demands (Staaf and Berg 1977, 1982). The nutrient status of forest land and its impact on microbial processes have been preserved for a long time, even in the case of reduced nitrogen deposition, attributed to the mobilization of large amounts of organic nitrogen stored in the system (Dörr et al. 2010). Guendehou et al.(2014) studied the changes and decomposition of chemical components in the leaves of five dominant tree species in tropical forests and concluded that the chemical quality of litter was the main factor in the decomposition process, including acidolysis chemicals, lignin, and initial concentration. Many studies have shown that nitrogen has a

negative effect on the decomposition at the later stage of decomposition. Prescott believes that the addition of nitrogen often leads to more stable organic matter and increased humification (Berg et al. 2010; Berg and Matzner 1997; Berg and Meentemeyer 2002;Hyvönen et al. 2007). In addition, some studies have shown that a linear matrix quality affects litter decomposition rates (Cornwell et al. 2008; Zhang et al. 2008), and climate factors play an indirect role in many cases (Wardle et al. 2009).

Nitrogen addition directly affects litter decomposition by improving the availability of soil N and changing the input quantity and quality of litter. It also indirectly affects decomposition by affecting microbial community composition changes and soil enzyme activities (Manning et al. 2008). In our study, the three forms of nitrogen deposition generally promoted litter decomposition in cold temperate coniferous forests and confirmed our first hypothesis. Those results suggested a litter decomposition rate increase upon nitrogen addition might be due to the priming effect of external nitrogen addition on microbial community activity (Lv et al. 2013). According to that priming theory, the positive priming effect should be most obvious in the soil with low nutrient availability, and most forest ecosystems are limited by nitrogen (Craine et al. 2007; Lebauer and Treseder 2008). The increase of microbial metabolic activity promoted litter decomposition (Wang et al. 2011). Recent studies showed a variety of bacteria and fungi coexist and may interact during litter decomposition (Johnston et al. 2016; Purahong et al. 2015; Urbanová et al., 2015). Some studies also showed a significant positive correlation between the litter decomposition rate and soil N availability, with higher nitrogen levels leading to faster litter decomposition. So nitrogen deposition increases nitrogen available in the soil, a small amount of available nitrogen increases the amount of litter decomposition, and excessive nitrogen inhibits litter decomposition (Li et al. 2016; Lin et al. 2017; Shen et al. 2018).

Consistent with our second hypothesis, the effect of ammonium nitrate on litter decomposition rate was more significant than that of ammonium sulfate and sodium nitrate. This was also consistent with previous studies (Lv et al. 2013; Wang et al. 2011). There are two reasons for this. First, when all N requirements are met, especially when mixed N is added to the soil, the presence of a large number of micro decomposer communities specifically utilizing a single form of N will greatly accelerate litter decomposition. Secondly, the balance between inorganic nitrogen and organic nitrogen of soil microorganisms may be disturbed by adding a single form of nitrogen to the soil subsystem, resulting in less stimulation to the metabolic activity of soil microbial decomposers (Guo et al. 2011;Lv et al. 2013;Sinsabaugh et al. 2002; Wang et al. 2011).

Effects of nitrogen deposition on the dynamic change of C, N and P during litter decomposition

The release of nutrient elements during forest litter decomposition was affected by nutrient elements, the decomposition stage, and the litter quality. There are three modes: direct release, leaching, and enrichment release (Mo et al. 2006). In this study, the amount of carbon in the litter decreased gradually, which agreed with Shen's results (Shen et al. 2019). Munasinghe and Herath(2014) suggested the release of C was consistent with the litter decomposition rate change. The effect of N deposition on C levels becomes very complex, and determined not only by the species and activity of microorganisms in the

ecosystem but also by climate, plant species and environmental conditions (Wen et al. 2017). In the present study, nitrogen levels in the litter increased over time. The N residual rate of different nitrogen forms and levels of litter at different decomposition time points differed significantly. Generally speaking, nitrogen concentrations in litter increased during decomposition, especially in the early stages (Aerts 1997). The results of the meta-analysis by Lu et al. showed that nitrogen addition increased N nitrogen in the litter by 24% and underground by 35% (Lu et al. 2011). Some studies also showed that long-term N deposition increased the amount of nitrogen in the litter, and litter with high N levelss met the demand of microbial N decomposers that resulted in greater N release (Shen et al. 2019). In this study, the N release was greater than accumulation during leaf decomposition using different nitrogen sources and amounts, the amount of N released from litter remained unchanged, increasing from 0–9 months and decreasing from 12–24 months. It has been suggested the forest needs fixed nitrogen before reaching a critical decomposition concentration. Shen et al. (2019) considered that N release from litter was closely related to C / N, indicating that there was a critical C / N value, and N release occurred when that ratio was less than the critical value. Parton et al. (2007) showed that the critical value was 40, while Moore et al. (2006) proposed a critical value of 55. In this study, changes in P over time were more complex than C and N levels. After two years, the average residue rates of phosphorus in leaves were 43.22% (ammonium sulfate), 54.32% (sodium nitrate), and 46.60% (ammonium nitrate), respectively. Most studies on the effect of N deposition on litter P planning showed that low N promoted P return, while high N inhibited P return and increased P fixation in litter (Shen et al. 2019). Long-term N input promoted the migration of soil soluble P (HPO_4^{2-} and $H_2PO_4^-$) to inactive phosphate pool, which was difficult for plants to absorb (Chen et al. 2016). The addition of N promoted phosphatase release from soil microorganisms and roots, promoted the mineralization of soil P, and accelerated P release during organic matter decomposition in tropical forests. The results showed that P fixation only occurred in litter with low N deposition (Cleveland and Townsend 2006). Under medium-high N deposition, P levels of litter decreased followed by mineralization, especially with high N deposition (Kuperman 1999). Tresederand Vitousek (2001) nitrogen application experiment in the Hawaiian rainforest shows that nitrogen input can cause plant tissues to produce many extracellular enzyme phosphatases, resulting in the decomposition of organic matter to produce more phosphate. Cleveland and Townsend (2006) simulated nitrogen depositions in a tropical rain forest showed that nitrogen addition accelerated the release of phosphorus during decomposition. Elser et al. (2007) found that the increase of nitrogen input may lead to the lack of phosphorus in all ecosystems, and when nitrogen deposition causes nitrogen saturation in the ecosystem, it induces phosphorus to limit healthy ecosystem development (Gress et al. 2007). These studies showed that when an ecosystem reaches nitrogen saturation, the input of nitrogen continues. It is possible to curb the turnover of phosphorus in an ecosystem.

Conclusions

These results showed that nitrogen deposition promoted litter decomposition in cold temperate coniferous forests. The order of the litter decomposition rates was ammonium nitrate > ammonium sulfate > sodium nitrate. Under the same nitrogen deposition condition, the decomposition rate of low

nitrogen treatments exceeded high nitrogen treatments. Nitrogen deposition decreased levels of C in the litter leaves, while the N content increased gradually over time. There were also significant differences in N levels using different nitrogen sources and amounts on the litter. N levels in litter increased the most in low ammonium nitrate treatment and increased the least in the sodium nitrate treatment. The change of P over time was more complex. The P content of litter leaves using different nitrogen sources and amounts differed significantly over time, but there was no significant difference among different nitrogen sources and amounts. The amounts of C, N and P released from the litter leaves were greater than that enriched. Litter decomposition is a long process, usually taking several years or even decades to complete. The two-year decomposition experiment time was short, more long-term decomposition experiments are needed to accurately predict the ecosystem carbon balance and the ability to respond to environmental changes.

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Tables

Table 1 The Olson exponential models for the average decomposition rate of litterfall leaf

Treatment	Olson equation	K/(t/a)	50% Decomposition time/a	95% Decomposition time/a	R^2	*P* value
N0		0.450	1.54	6.66	0.902	*P*< 0.001
Low-NHS[a]		0.533	1.30	5.62	0.904	*P*< 0.001
Medium-NHS[b]		0.442	1.56	6.78	0.947	*P*< 0.001
High-NHS[c]		0.488	1.42	6.14	0.940	*P*< 0.001
Low-NAN[d]		0.437	1.57	6.86	0.904	*P*< 0.001
Medium-NAN[e]		0.469	1.48	6.39	0.971	*P*< 0.001
High-NAN[f]		0.451	1.54	6.64	0.942	*P*< 0.001
Low-NHN[g]		0.635	1.09	4.72	0.898	*P*< 0.001
Medium-NHN[h]		0.466	1.49	6.42	0.972	*P*< 0.001
High-NHN[i]		0.496	1.40	6.04	0.938	*P*< 0.001

[a]Low-$(NH_4)_2SO_4$; [b]Medium-$(NH_4)_2SO_4$; [c]High-$(NH_4)_2SO_4$; [d]Low-$NaNO_3$; [e]Medium-$NaNO_3$; [f]High-$NaNO_3$; [g] Low-NH_4NO_3; [h]Medium-NH_4NO_3; [i]High-NH_4NO_3

Figures

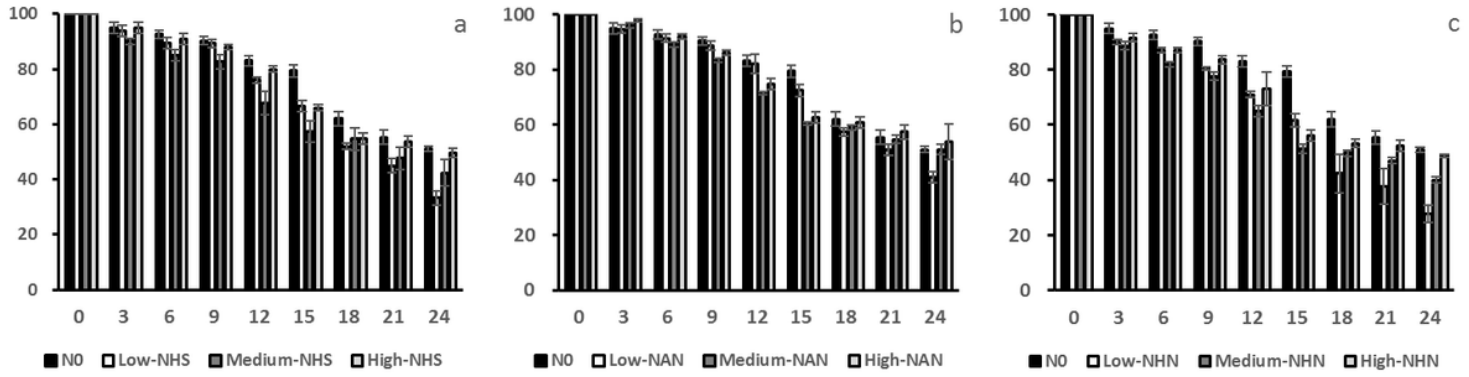

Figure 1

Effects of three forms of nitrogen deposition on dry matter residue rate in litterfall leaf decomposition (n = 3); (a, three levels of (NH4) 2SO4 addition; b, three levels of NaNO3 addition; c, three levels of NH4NO3 addition)

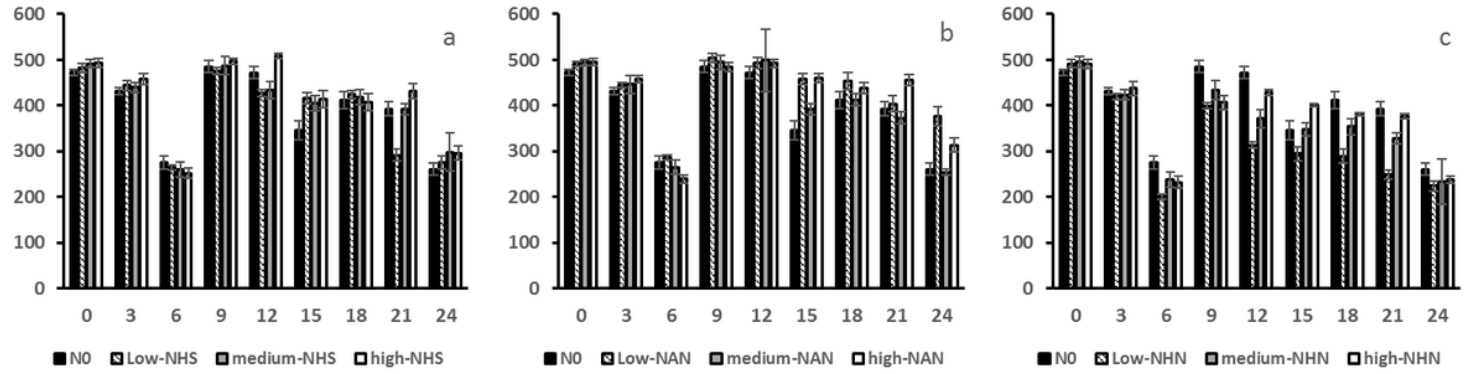

Figure 2

Effects of three forms of nitrogen deposition on carbon levels in litterfall leaf decomposition (n = 3); (a, three levels of (NH4) 2SO4 addition; b, three levels of NaNO3 addition; c, three levels of NH4NO3 addition)

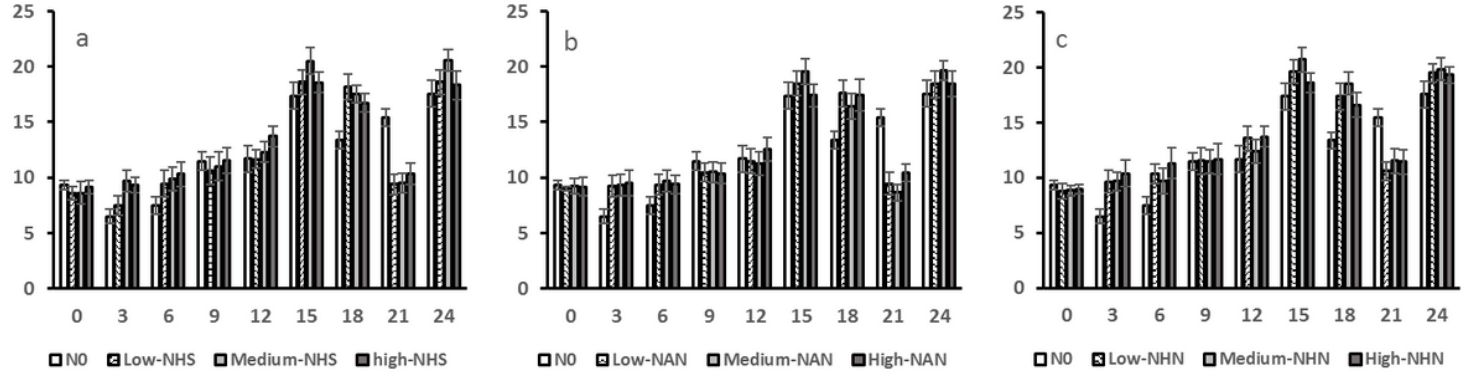

Figure 3

Effects of three forms of nitrogen deposition on nitrogen levels in litterfall leaf decomposition (n = 3); (a, three levels of (NH4) 2SO4 addition; b, three levels of NaNO3 addition; c, three levels of NH4NO3 addition)

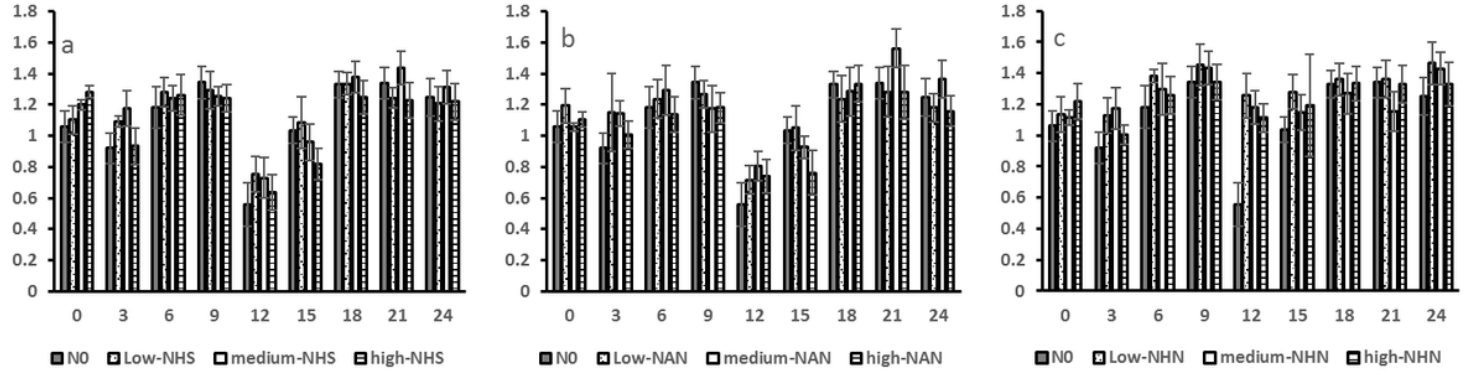

Figure 4

Effects of three forms of nitrogen deposition on phosphorus levels in litterfall leaf decomposition (n = 3); (a, three levels of (NH4) 2SO4 addition; b, three levels of NaNO3 addition; c, three levels of NH4NO3 addition)

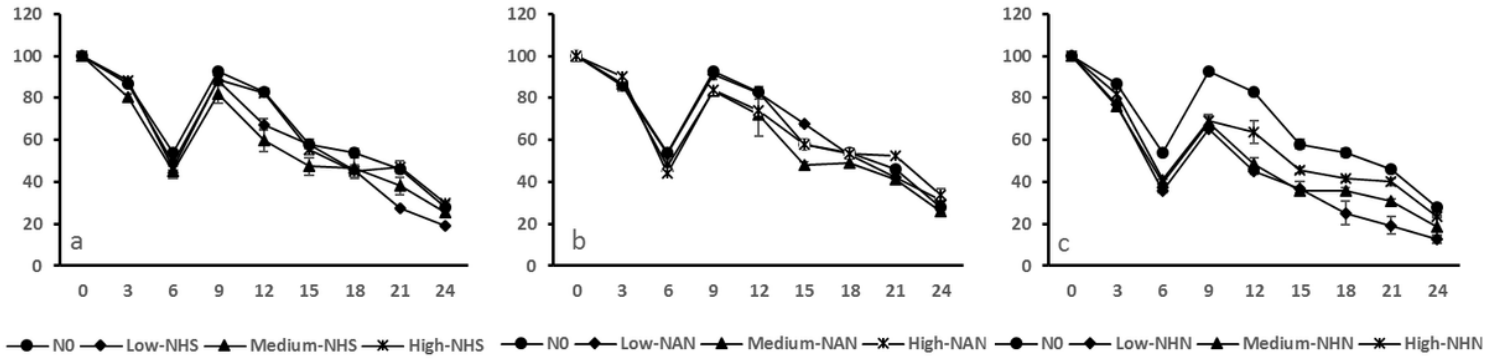

Figure 5

Effects of three forms of nitrogen deposition on C nutrient residues in litterfall leaf decomposition (n = 3); (a, three levels of (NH4) 2SO4 addition; b, three levels of NaNO3 addition; c, three levels of NH4NO3 addition)

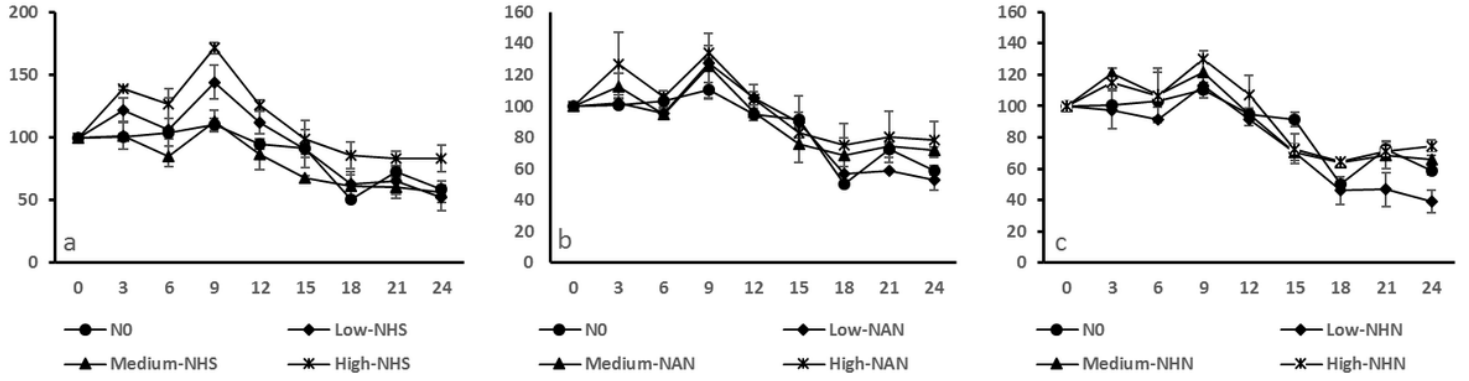

Figure 6

Effects of three forms of nitrogen deposition on N nutrient residues in litterfall leaf decomposition (n = 3); (a, three levels of (NH4) 2SO4 addition; b, three levels of NaNO3 addition; c, three levels of NH4NO3 addition)

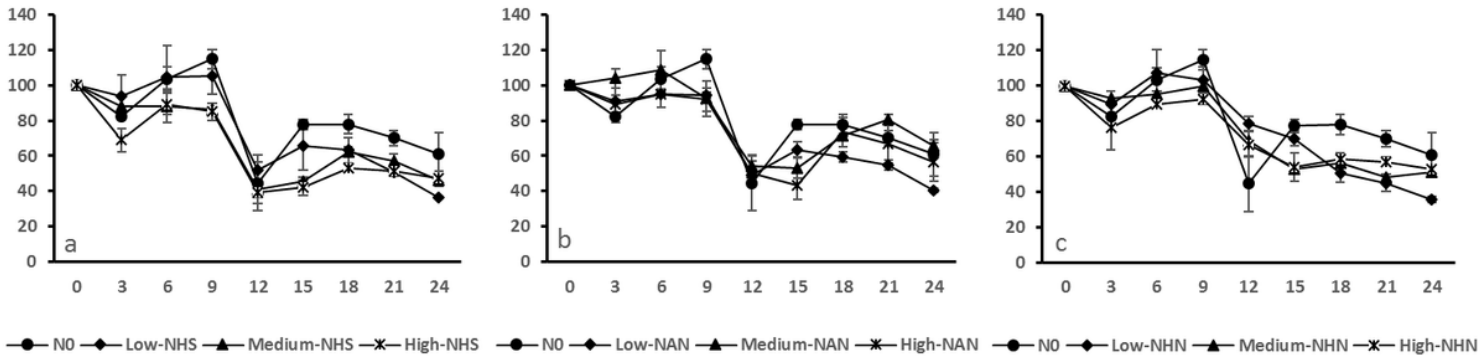

Figure 7

Effects of three forms of nitrogen deposition on N nutrient residues in litterfall leaf decomposition (n = 3); (a, three levels of (NH4) 2SO4 addition; b, three levels of NaNO3 addition; c, three levels of NH4NO3 addition)

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- Data.xlsx