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Consequences of afforestation for soil nitrogen dynamics in central China

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ABSTRACT

The effects of afforestation are of great importance for terrestrial nitrogen (N) cycling. However, the consequences of afforestation for soil nitrogen (N) dynamics remain poorly quantified. We investigated soil net N mineralization and nitrification rates as well as the inorganic N (NH_4^+-N and NO_3^--N) concentration in the top soil (0–10 cm) in a woodland, shrubland and adjacent cropland in the Danjiangkou Reservoir region of central China using the in situ closed-top tube incubation technique over one year. Afforestation significantly decreased the soil net N mineralization rate and soil inorganic N concentration but increased the soil NH_4^+-N concentration and soil ammonification rate. The major form of soil inorganic N was NO_3^--N in the cropland versus NH_4^+-N in the woodland. The soil net N mineralization and nitrification rates were more sensitive to soil moisture than to soil temperature and were positively correlated with soil moisture. In contrast, the soil net N mineralization and nitrification rates were negatively related to SOC and the C:N ratio. These results suggest that afforestation could decrease N mineralization and availability due to increasing the recalcitrant C input and plant N uptake, which might in turn cause progressive N limitation over the long term.

1. Introduction

Soil nitrogen (N) accounts for approximately 88% of the global plant N demand (Schlesinger, 1997) and has an impact on net primary productivity (Elser et al., 2007; LeBauer and Treseder, 2008) and soil respiration (e.g., Janssens et al., 2010) in terrestrial ecosystems. Any change in soil N availability can greatly affect plant growth and productivity and, thus, impact ecosystem functions (Friedlingstein et al., 2001; Cheng et al., 2010; Esser et al., 2011). Land use change, such as afforestation has been considered to enhance soil organic matter (SOM) accumulation and hence alter the soil N availability for plant growth (Zeng et al., 2009; Mao and Zeng, 2010; Pospisilova et al., 2011). This effect could potentially influence the net primary productivity of terrestrial ecosystems (Scanlon et al., 2008; do Carmo et al., 2012).

The consequences of afforestation for soil N dynamics have been extensively studied, but they remain controversial. For instance, afforestation has been reported to either cause significant alteration in soil N mineralization (Yan et al., 2008; Pandey et al., 2010) or to have a negligible effect on soil N mineralization (Goodale and Aber, 2001; Zeng et al., 2009). These inconsistent results are unsurprising because N dynamics primarily consists of complex processes which are regulated by multiple factors (Liu et al., 2010; Laughlin, 2011; Thamdrup, 2012). Afforestation affects soil N dynamics by altering the species composition and net primary production and causing environmental changes (Guo and Gifford, 2002; Cheng et al., 2011). To understand how afforestation will impact soil N dynamics, it is crucial to clarify the effects of both biotic (e.g., the species composition and microbial C and N) and abiotic (e.g., soil temperature, moisture, and pH) factors on soil N dynamics following land use change.

Afforestation can influence soil N dynamics primarily by altering the quantity and quality of SOM (Guo and Gifford, 2002; Potthast et al., 2010; Papini et al., 2011; Cheng et al., 2013; Wieder et al., 2013). Soil N transformation (i.e., mineralization and nitrification) is a microbial process regulated by SOM contents, and changes in SOM inputs resulting from afforestation therefore potentially lead to differences in soil N mineralization and nitrification (Usman et al., 2000; Templer et al., 2005). For instance, Templer et al. (2005) demonstrated that increase in SOM enhances soil N mineralization, while increased N mineralization is associated with higher microbial biomass. In addition to biotic factors, abiotic factors (e.g., soil temperature, moisture, and pH) also exert important influences on soil N dynamics following land use change (Wang et al., 2006; Shan et al., 2011). For example, several studies have demonstrated that soil moisture plays a key role in regulating soil N

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mineralization in grassland ecosystems (Zhang et al., 2008; Liu et al., 2010). Furthermore, an increased soil pH appears to stimulate soil N mineralization (Li et al., 2007) and, especially, soil N nitrification (Yao et al., 2011) because a low soil pH limits nitrifiers due to aluminum toxicity (Matsumoto, 2000; Kochian et al., 2005). However, changes in SOM inputs and soil properties following land use change can affect soil N dynamics differently among various terrestrial ecosystems (Uri et al., 2008; Tripathi and Singh, 2009; Evans et al., 2011). Therefore, additional field studies on the response of soil N dynamics to land use change are necessary, particularly regarding soil N dynamics in afforested soil, where ecosystem function is relatively fragile.

The Danjiangkou Reservoir, with a drainage area of 95,200 km^2, is a water source for the central route of China's South-to-North Water Transfer Project (Zhang et al., 2009). Water quality and riparian ecosystem function are of great importance to both the government and the public (Chen et al., 2007; Zhu et al., 2008). Human activities such as deforestation and tillage around the reservoir have resulted in soil erosion, water pollution and soil C and N losses (Zhu et al., 2010; Liu et al., 2012). In recent decades, afforestation has been conducted to restore and protect the Danjiangkou Reservoir riparian ecosystem (Liu et al., 2012). Because of the relatively poor soil fertility in the region, the afforested lands exhibit low vegetation coverage and, hence, are often a mosaic of large open areas where no litter or root formation occurs (Zhu et al., 2008; Liu et al., 2011; Cheng et al., 2013). Previous research in the region around the Danjiangkou Reservoir has indicated that the spatial and temporal distribution of reservoir water quality are closely correlated with land uses and changes in vegetation cover (Li et al., 2009); however, little is known about the effects of land use change on N dynamics.

In this study, we examined soil properties, inorganic N concentrations, and soil net N mineralization, nitrification and ammonification rates in woodland, shrubland and adjacent cropland soils. We hypothesized that afforestation (with woodland and shrubland plantations) would significantly affect soil N dynamics due to changes in SOM inputs and soil environmental factors. To test this hypothesis, we specifically focused on (1) how 15 years of afforestation has potentially impacted soil inorganic N concentrations and N transformation rates and (2) how the associated changes in soil properties (i.e., SOC, C:N ratios, microbial biomass C and N, soil temperature and moisture, and pH) determine soil N dynamics.

2. Materials and methods

2.1. Study site description

This study was conducted at the Wulongchi Experimental Station (32°45′N, 111°13′E) in the Danjiangkou Reservoir region (Cheng et al., 2013). The elevation at the station is approximately 280–400 m. The Danjiangkou Reservoir lies within the northern subtropical zone, and the region exhibits a subtropical monsoon climate, with a mean annual temperature of 15.7 °C and monthly average temperatures of 27.3 °C in July and 4.2 °C in January. The annual precipitation is 749.3 mm, 70–80% of which falls between April and October. The soil is yellow brown consisting of 11% sand, 41% silt, and 48% clay in the top 30 cm (Zhu et al., 2010). Approximately 15 years ago, following a reorganization of land usage in the region, a large cropland area was converted to a woodland plantation of coniferous trees (*Platycladus orientalis* (Linn.) *Franco*) and a shrubland plantation (*Robinia pseudoacacia* and *Amorpha fruticosa*) (Zhu et al., 2010). Although a detailed record of the cropland cultivation history adjacent to the site is not available, farmers have reported that they typically cultivated wheat and maize. This maize and wheat cultivation was managed through conventional agricultural practices, including plowing to a depth of 0.4 m, mineral fertilization and chemical weeding. The aboveground biomass of wheat straw and maize was removed through harvesting.

2.2. Field measurements

In October 2010, three stands of each approximately 75 ha (500 m × 1500 m), including a woodland, an adjacent shrubland, and an adjacent cropland (i.e. the control), were delimited in this study. The distances between the three stands are approximately 1 km. A comprehensive survey of soil and vegetation was conducted to ensure the comparability (e.g., similar soil types and topographies) of the sampling plots (Cheng et al., 2013). Within each stand, we randomly set 5 sub-plots (2 m × 2 m) for each land use type. Soil net N mineralization, nitrification and ammonification rates were measured eight times from October 2010 to October 2011 using the in situ closed-top tube incubation technique (Raison et al., 1987). In each sub-plot, a pair of 5 cm-diameter PVC tubes was driven 10 cm into the ground soil at the beginning of each incubation period. Within each tube, the changes in the inorganic-N content during the incubation period represented the net N mineralized from organic sources (Rhoades and Coleman, 1999). One tube was immediately removed for determination of the initial NH_4^+-N and NO_3^--N concentrations and soil properties. The other tube was left in the field to undergo incubation prior to being retrieved. All of the collected soil samples were temporarily stored in a refrigerator at 4 °C prior to analysis.

2.3. Laboratory analyses

The soil samples collected from the same sub-plot were mixed thoroughly after all visible plant material and stones were physically moved using forceps. Soil NH_4^+-N and NO_3^--N were extracted with a 2 M KCl solution (soil:solution, 1:5) and then filtered through a 0.45 μm filter. The extracted solutions were measured via colorimetric techniques at 645 nm and 420 nm to determine NH_4^+-N and NO_3^--N concentrations, respectively. Soil microbial biomass C (MBC) and N (MBN) were determined using the chloroform fumigation direct extraction method (Vance et al., 1987). Both fumigated (24 h) and non-fumigated soil samples were extracted by incubation with 0.5 M K_2SO_4 (soil:solution, 1:4) for 1 h on a reciprocal shaker, followed by filtering. The total organic C and total N contents of the extract solutions were immediately analyzed using a multi N/C 2100 analyzer (Analytik Jena, Germany) to determine MBC and MBN, respectively. The correction factor was estimated as 0.45 (Beck et al., 1997) for MBC and 0.54 (Brookes et al., 1985) for MBN. Soil moisture was measured after being oven-dried at 105 °C for 24 h to constant weight.

Additional soil samples were air dried for the measurement soil pH, soil total N and soil organic carbon (SOC). The soil pH was determined in a water suspension (soil:water, 1:2.5) using a pH meter equipped with glass electrode. Soil samples were treated with 1 N HCl for 24 h at room temperature to remove any inorganic C present for the measurement of SOC (Cheng et al., 2006). Non-HCl-treated soil subsamples were analyzed to quantify soil total N. Both the HCl-treated and non-HCl-treated soil samples were subjected to measurements using an element analyzer (Flash EA 1112, Italy). Soil bulk density was determined via the core method. Within the 0–10 cm layer, soil temperature was measured with a temperature probe, at the same time soil moisture (% volumetric) was measured through time domain reflectometry during each incubation period. Air temperature and precipitation over study period were recorded at a nearby meteorological station, approximately 5 km from the study site, in the town of Wulong Chi.