



Impacts of afforestation on plant diversity, soil properties, and soil organic carbon storage in a semi-arid grassland of northwestern China



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ABSTRACT

Grassland afforestation, occurring in many parts of the world, can modify the nature and transformations of soil organic carbon and associated soil properties, which in turn can affect plant diversity and ecosystem function. Afforestation area has grown rapidly over the last few decades in the semi-arid grasslands of the Qilian Mountains in northwestern China in an effort to restore mountain vegetation. However, ecological consequences of this land use change are poorly known. We investigated the effects of grassland afforestation on plant diversity, soil properties, and soil organic carbon and nitrogen storage at the soil depth of 0–70 cm. Our results showed that afforestation decreased percent cover and aboveground biomass, and increased plant diversity of herbaceous community. Afforestation also decreased soil bulk density and pH, and increased soil water content. Generally, afforestation favored an increase in soil organic carbon, total nitrogen, and organic carbon storage, and resulted in a significant increase in total phosphorus in the surface soil (0–5 cm), although a slight decrease ($P > 0.05$) was observed in the subsoil. In addition, afforestation significantly increased soil C:N ratio in the upper soil. Results of this study demonstrate the potential for afforestation to increase soil organic carbon and nitrogen storage in semi-arid grasslands of Qilian Mountains. This has important implications for C sequestration in this area.

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1. Introduction

Land-use and land-cover changes have attracted increasing scientific interest in the past decades in relation to their contribution to global change and potential impacts on carbon (C) dioxide sequestration, soil quality, ecosystem function, and long-term sustainability (DeFries et al., 2004; Laganieri et al., 2010; Sauer et al., 2012; Deng et al., 2014a, 2014b; Bárcena et al., 2014; Deng and Shanguan, 2016). Grassland afforestation, mainly with coniferous trees, has expanded rapidly in the last decades (Otto and Simpson, 2005; Rudel et al., 2005; Chen et al., 2008; Wei et al., 2009; Hewitt et al., 2012; Vassallo et al., 2013); the reasons for this land-use change included an increasing demand for timber production, and a growing need to control soil erosion, restore vegetation, and mitigate CO₂ emissions (Chen et al., 2007; Chen et al., 2008; Hewitt et al., 2012; Vassallo et al., 2013). Rapid expansion of plantation areas highlights the need to understand ecological consequences of this land-use change for the maintenance of long-term nutrient availability, sustainable productivity, and C sequestration (Berthrong et al., 2012).

Conversion of native grasslands to forest plantations modifies primary production, ecosystem structure (Vassallo et al., 2013), the quantity and quality of litter inputs, root turnover (including exudates), and

microclimatic conditions such as moisture and temperature (Nosetto et al., 2005; Laganieri et al., 2010). Shifts in plant species can lead to changes in soil properties and C stocks, which in turn have the potential to affect biomass production and ecosystem function (Jackson et al., 2002; Foster et al., 2003; Jobbágy and Jackson, 2003; Chen et al., 2008; Wei et al., 2009; Wang et al., 2016). Moreover, the “nutrient pumping” effect, observed following the conversion of grasslands to forests, results in the redistribution of nutrients, with decreasing concentrations at intermediate depths and increasing at the soil surface (Jobbágy and Jackson, 2004; Farley and Kelly, 2004).

Current understanding of ecological consequences of grassland afforestation includes plot-scale changes in plant diversity and community structure, soil organic carbon (SOC) and total nitrogen (TN) stocks, soil moisture, acidity, soil nutrient status, and microbial community structure (Jobbágy and Jackson, 2003; Farley and Kelly, 2004; Alrababah et al., 2007; Chen et al., 2008; Berthrong et al., 2009a; Berthrong et al., 2009b; Wei et al., 2009; Berthrong et al., 2012; Hewitt et al., 2012; Deng et al., 2016). However, extensive uncertainty remains at region scales, especially for arid and semi-arid regions (Hu et al., 2008; Zhang et al., 2013), and the results vary greatly with climate, forest age and type, and soil type and management practices (Jobbágy and Jackson, 2003; Farley and Kelly, 2004; Chen et al., 2008; Hu et al., 2008; Berthrong et al., 2009b; Wei et al., 2009; Wei et al., 2010; Deng et al., 2014a, 2014b; Bárcena et al., 2014). For example, afforestation in a temperate grassland of Inner Mongolia resulted in an initial loss of

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total soil nitrogen (TN) during the first few years, but a gradual recovery with increasing stand age (Hu et al., 2008). Similar trends for soil N stocks were observed in Ecuadorian grasslands (Farley and Kelly, 2004). However, a global meta-analysis showed that SOC were reduced with afforestation of grasslands but not significantly ($P > 0.05$) (Shi et al., 2013). Jackson et al. (2002) found a significant and negative relationship between precipitation and changes in SOC and TN when grasslands were invaded by woody vegetation, with drier sites gaining, and wetter sites losing both SOC and TN. Soils frequently become more acidic with afforestation (Jobbágy and Jackson, 2003; Farley and Kelly, 2004; Berthrong et al., 2009a); however, detailed studies were conducted mainly in New Zealand, America and Australia (Berthrong et al., 2009a), and these effects remain uncertain in other regions. Additionally, many studies have focused on the effects of grassland afforestation in the topsoil (e.g., 0–20 cm) due to historical practices and the ease of sampling, and the responses in deep soil layers are still poorly understood (Chang et al., 2012). However, increasing evidence suggests that the SOC contents in the subsoil are also sensitive to changes in land use and management (Chang et al., 2012; Shi et al., 2013; Deng et al., 2014a, 2014b). Thus, data are needed from different regions and from the subsoil to increase the understanding of the ecological consequences of grassland afforestation.

The Qilian Mountains, located in the northern margin of the Tibetan Plateau, are the source of several key inland rivers in northwestern China, including the Heihe, Shiyang, and Shule. The mountains were designated as a National Nature Reserve in 1988 for their key role in maintaining regional ecological security. Forests, dominated by Qinghai spruce (*Picea crassifolia*), and grasslands are the main landscape types in this area (Wang et al., 2001). However, forest cover decreased from 22.4% in 1949 to 12.4% during the 1990s due to deforestation associated with increasing demand for timber production, and with global warming (Wang and Cheng, 1999). Loss of forest cover affected hydrological processes, and had important consequences for sustainable development in the region (Wang and Cheng, 1999; He et al., 2012). Since the 1970s, the area of afforestation has been increasing in an effort to restore mountain vegetation, and many semi-arid grasslands were converted to *P. crassifolia* plantation forests (He et al., 2012). Although this land use has grown rapidly over the past four decades, little is known about the effects of change in vegetation cover on plant diversity, soil properties, and soil C stocks in these ecosystems. Thus, the objectives of the present study were to investigate the effects of grassland afforestation on: (1) plant composition and diversity of herbaceous community; (2) soil properties; and (3) soil C and N storage.

2. Methods

2.1. Study area

The study site was located in the Dahuang Mountain Forest Reserve (100°22'E, 38°43'N, 2919 m a.s.l.) in the Qilian Mountains. The area is situated approximately 45 km southeast of Shandan County, Gansu Province, in northwestern China. The site has a semiarid and cold temperate climate, with a mean annual temperature (MAT) of about 1 °C and mean annual precipitation (MAP) of about 400 mm, falling mainly between July and September. The main parent material is calcareous rock, which is overlaid by a relatively thin soil layer (<1 m deep) (Jiang et al., 2013). Native vegetation patterns are closely related to topographic aspects, and represent a mosaic of grassland, forest, and small areas of scrubland. Forests, dominated by the *P. crassifolia*, are distributed on shaded, north-facing slopes; grasslands are mainly found on sunny, south-facing, and semi-shaded, east- or west-facing slopes. Since the 1970s, most of the grasslands on east-facing and west-facing slopes have been converted into *P. crassifolia* plantation forests, and the forest cover has increased from 24.6% to 52.8%. Differences in topographical aspects and vegetation patterns induced divergent soils properties. Soils are classified according to the FAO classification system as

Haplic Kastanozems on sunny and semi-shaded slopes, and Haplic Phaeozems on shaded slopes (IUSS Working Group, WRB, 2014).

2.2. Experimental design, soil sampling, and vegetation survey

In early August 2014, two study sites were selected, one on west-facing and one on east-facing slope (Table 1). At each site, native grasslands and *P. crassifolia* plantation forests occurred directly adjacent to each other. Site conditions (e.g. topography and vegetation patterns) and management practices of the selected grasslands and plantations were typical in the region. Three replicate sample plots of 30 × 30 m² were randomly located in each grassland and forest for a total of 15 sample plots (Table 1). Geographic coordinates and elevations of each plot were obtained using a global positioning system (GPS) with differential correction.

Within each plot, five randomly-located soil profiles were excavated (after removing the surface litter layer), and soil samples were collected at depths of 0–5, 5–15, 15–30, 30–50, and 50–70 cm. In addition, undisturbed soil cores were obtained from each layer for the measurements of bulk density using a standard container with the volume of 100 cm³. In forest plots, tree height, diameter at breast height (DBH), tree crown area, and the number of trees per plot were measured. We measured canopy height at 25 points in each plot, using a telescopic measuring rod (5 cm precision). We also measured leaf area index (LAI) using CI-110 with a fisheye lens (Juarez et al., 2009) at 1.5 m above ground every 3 m along the diagonal of each plot to evaluate the light conditions. Ten quadrats of 1 × 1 m² were randomly located to investigate species composition and percent cover of the herbaceous community (main understory vegetation); subsequently, the herbaceous layer was harvested at 2–3 cm above ground and oven-dried at 65 °C to a constant weight to determine aboveground biomass of understory vegetation. Vegetation cover for the herbaceous layer was visually estimated by two experienced observers. For grassland plots, species composition, percent cover, and aboveground biomass of the plant community were also investigated separately in ten quadrats of 1 × 1 m².

Importance values (IV, Eq. (1)) for herbs were calculated using the following equations (Zhao et al., 2009):

$$IV_i = \frac{(RA_i + RC_i + RF_i + RB_i + RH_i)}{5} \quad (1)$$

where IV_i , RA_i , RC_i , RF_i , RB_i and RH_i were the importance values (%), relative density (%), relative coverage (%), relative frequency (%), relative biomass (%), and relative height (%) of species i , respectively.

Plant diversity analysis was conducted for the herbaceous plant community. Species richness index (R), Shannon index (H), and Pielou evenness index (E) were calculated as biodiversity indicators at the quadrat level. Species richness (R) was calculated as the number of species identified in each quadrat, while Shannon index (H, Eq. (2)) and Pielou evenness index (E, Eq. (3)) were calculated using the following equations (Spellerberg and Fedor, 2003; Deng et al., 2014a, 2014b):

$$H = -\sum_{i=1}^n (P_i \ln P_i) \quad (2)$$

$$E = \frac{H}{\ln S} \quad (3)$$

Where S is the total species numbers of the herbaceous community, and P_i is the proportional density of species i (number of individuals of species i divided by the total number of individuals of all species).

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