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Drying-rewetting cycles alter carbon and nitrogen mineralization in litter-amended alpine wetland soil

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ABSTRACT

Wetting-drying cycles can influence decomposition of litter and soil organic carbon (SOC) and their mineralization, but such effects have seldom been explored in alpine wetland soils. We conducted a 120-day incubation experiment with alpine wetland soils to which we added litter or not. These soil samples were assigned to two constant moisture treatments (60% or 100% soil water-holding capacity, WHC) or to a wetting-drying treatment that cycled between 60% and 100% WHC. Drying-rewetting cycles significantly accelerated carbon (C) mineralization and nitrogen (N) immobilization compared to soil under saturated soil moisture conditions. Litter addition greatly increased C mineralization and N immobilization, but the intensity of mineralization was regulated by soil moisture through microbial biomass. A significantly negative relationship between C and N mineralization became more pronounced when litter was added. Thus, drying-rewetting cycles can alter C and N mineralization, and this effect can strongly depend on litter in alpine wetlands. This indicates that future climate change could affect C stocks in alpine wetland soil through altering moisture and litter production.

1. Introduction

The importance of wetlands for global carbon (C) cycling is highlighted by the very large amount of C stored in wetland soils. This C storage corresponds to 20–30% of the terrestrial soil organic C (SOC, 2300 Pg) pool, although wetlands only cover 5–8% of the land surface (Mitsch and Gosselink, 2007; IPCC, 2013). The large amount of C accumulated in wetlands is mainly ascribed to slow SOC decomposition in anaerobic environments caused by waterlogging. This C can be released back to the atmosphere via increase in microbial decomposition caused by climate change, e.g., increased temperature and changed precipitation regime (Reichstein et al., 2013). Future climate change has been predicted to increase frequencies of extreme climate events, such as intense episodes of precipitation and prolonged droughts in many regions (IPCC, 2013; Reichstein et al., 2013). This could accelerate the dynamics of soil moisture change and drying-rewetting cycles, especially in wetlands (Seneviratne et al., 2010). During drying-rewetting cycles, litter and the surface soil in wetlands may shift between anaerobic and aerobic conditions, which may further affect microbial decomposition of litter and SOC (Moyano et al., 2013; Zhu and Cheng, 2013).

The surface soil often dries gradually due to evapotranspiration and then rapidly rewets as a result of precipitation (Fierer and Schimel, 2002; Zhu and Cheng, 2013). The drying of the organic horizon in wetlands is faster and more intense than that of the mineral horizon due to the absence of capillary rise in the organic horizon. Furthermore, the C in the surface layer is not associated with soil minerals that can protect SOC from microbial decomposition. A rewetting event after a prolonged drought can increase microbial activity (Prieme and Christensen, 2001; Jarvis et al., 2007; Borken and Matzner, 2009) and thus CO_2 emission from soil (Huxman et al., 2004; Rey et al., 2005; Borken and Matzner, 2009; Inglima et al., 2009; Moyano et al., 2013). Consequently, C and nitrogen (N) mineralization are strongly accelerated (Borken and Matzner, 2009). At the same time, such changes in soil moisture increase litter-derived C incorporation into microbial biomass and litter-derived CO_2 emissions (Benesch et al., 2015). The contribution of litter-derived CO_2 during the wet period is approximately three-fold higher than during the dry period (Ataka et al., 2014). Increased soil moisture accelerates SOC decomposition caused by exudates or labile C inputs through litter decomposition (Dijkstra and Cheng, 2007; Zhu and Cheng, 2013). Therefore, drying-rewetting cycles influence available nutrients and microbial activities mainly by altering soil moisture (Jarvis et al., 2007; Bengtson et al., 2012; Bernal and Mitsch, 2013). Additionally, drying-rewetting cycles can modify SOC decomposition by breaking up aggregates and releasing inaccessible C, depending on soil

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drying intensity (Zhu and Cheng, 2013). Generally, moderate hydrophobicity can improve aggregate stability, protecting SOM from decomposition. Drying-rewetting cycles may decrease the hydrophobicity and activate SOC decomposition (Borken and Matzner, 2009), but it remains unclear how drying-rewetting cycles affect litter decomposition and SOC mineralization in alpine wetlands.

The Tibetan Plateau contains a vast area of wetlands (roughly 5.1×10^3 km^2) (Xing et al., 2009). The Zoigê Wetlands, located in the northeastern part of the Tibetan Plateau, represent the largest peatland in China (Zhang et al., 2011), which contains about 0.5 Pg C, roughly 88% of the C stocks in the Tibetan wetlands (Xing et al., 2009). These alpine wetlands are predicted to undergo more frequent and intense drying-rewetting cycles due to climate change (Huntington, 2006; IPCC, 2013; Liao, 2013). To clarify effects of drying-rewetting cycles caused by climate change, we conducted a 120-day incubation experiment with litter additions to soils collected in the Zoigê alpine wetlands. We hypothesized (1) that drying-rewetting cycles greatly increase litter and SOC decomposition compared to saturated moisture conditions, and (2) that litter reduces N mineralization through increase in microbial N immobilization, and the this process can be intensified by drying-rewetting cycles.

2. Materials and methods

2.1. Research site and soil/litter collection

Both soil and litter were collected from a wetland located at the Zoigê Natural Reserve (33°35′N, 102°57′E, 3440 m a.s.l.) on the eastern Tibetan Plateau, China. The wetland is characterized by a subalpine, continental and monsoonal climate, with mean annual temperature of 0.6 °C and mean annual precipitation from 500 to 657 mm (Ding et al., 2004; Yu et al., 2010). In this wetland, total vegetation cover is over 90%. The dominant species are *Carex muliensis* Hand-Mazz. and *Carex lasiocarpa* Ehrh., which contribute >60% of total vegetation cover. Other common species include *Potentilla anserina* L., *Kobresia tibetica* Maxim., and *Caltha palustris* L. The soil is classified as humus marsh soil (Chai et al., 1965), corresponding to histosols based on FAO soil taxonomy (WRB, 2014).

On the soil surface, there was a litter layer <2 cm thick that was little decomposed. We collected the litter of the two dominant species (*C. muliensis* and *C. lasiocarpa*) from five randomly selected sites at least 20 m apart. The litter was brought to a laboratory, washed, air dried, and cut to about 2 cm long. Litter N content was 18.2 ± 0.9 g kg^{-1} (mean ± SE, n = 5), organic C content was 454 ± 6 g kg^{-1} and C:N ratio was 25.0 ± 2.0. In the Zoigê Wetlands, the surface soil, including the litter layer, experiences frequent drying-rewetting cycles (Liao, 2013).

We collected the soil from 0 to 10 cm depth in the H horizon at the same sites after litter removal. The soil was immediately brought to the laboratory and passed through a 4-mm sieve. The soil contained 70.4 ± 15.9 g kg^{-1} total organic C, 293 ± 27 mg kg^{-1} dissolved organic carbon (DOC), 25.7 ± 6.9 mg kg^{-1} microbial biomass carbon (MBC) and 5.6 ± 0.6 g kg^{-1} total N. The soil had a C:N ratio of 12.6 ± 1.3, pH of 6.9 ± 0.1 and a bulk density of 0.3 ± 0.03 g cm^{-3}, and contained 9.8% clay, 80.9% silt, and 9.3% sand.

2.2. Incubation experiment

The sieved soil was homogenized and randomly separated into two portions. In one portion, moisture was adjusted to 60% of the soil water-holding capacity (WHC), and in the other it was adjusted to 100% of the soil WHC. The two portions were then pre-incubated for five days at room temperature (about 20 °C) in the dark before litter addition. We chose 60% WHC as the dry treatment because it favors soil microbes and is common in the Zoige Wetlands.

The incubation experiment used a factorial design with three soil water treatments (dry, wet, and drying-rewetting cycle) crossed with two litter treatments (with and without litter addition), making in total six treatments. In the dry treatment, water content in the soil was kept to 60% WHC during the whole experiment, and in the wet treatment water content was maintained at 100% WHC. In the drying-rewetting cycle treatment, the soil was subjected to 12 cycles of wetting and drying regimes during the experimental period (Fig. 1). Within each cycle, soil water content was first maintained at 100% WHC for two days and then gradually decreased from 100% to 60% WHC during eight days of drying in a closed volume of air with 20 g $CaCl_2$. The amount of $CaCl_2$ was determined by the pre-incubation experiment.

We placed the pre-incubated soil (corresponding to 60 g dry weight) in 600-ml jars. One third of the jars were filled with the soil pre-incubated at 60% WHC, and used for the dry treatment. Another 1/3 were filled with the soil pre-incubated at 100% WHC, and used for the wet treatment. The remaining 1/3 were also filled with the soil pre-incubated at 100% WHC, but used for the drying-rewetting treatment. For each of the three water treatments, half of the jars were randomly selected for addition of air-dried litter pieces corresponding to 2 g dry weight.

The experiment started on 28 July 2012, and the soil in the jars was incubated in dark for up to 120 d at 20 °C. At the start of the experiment, four additional jars with a beaker containing 10 ml distilled H_2O and a beaker containing 10 ml 0.1 M NaOH were sealed, and served as controls to account for CO_2 trapped from the air. In each treatment, jars were each provided with a vial containing 10 ml 0.1 M NaOH to trap CO_2 from respiration and a vial containing $CaCl_2$ to absorb water vapor, and were also sealed. NaOH solution and $CaCl_2$ were replaced periodically, well before saturation occurred (Fig. 1). In this way, CO_2 released from the soil was measured at each dry and wet period.

2.3. Chemical analyses

C and N contents of soil and litter were analyzed with a stable isotope-ratio mass spectrometer (MAT-253,Thermo Fisher, USA), and DOC in fresh soil was determined with a TOC analyzer (Analytikjena Corp., Germany). Soil pH was measured using a digital pH meter. Soil bulk density was estimated by dividing the weight of the dried soil by the volume of the soil (Blake and Hartge, 1986). The particle fractions of the soil samples were analyzed with a Longbench Mastersizer 2000 (Malvern Instruments, Malvern, England). Microbial biomass C (MBC) and N (MBN) were determined using a chloroform fumigation extraction method (Brookes et al., 1985; Wu et al., 1990).

To quantify CO_2 emissions from the soil, CO_2 trapped in the NaOH solution was analyzed by titrating NaOH against 0.1 M HCl (He et al., 2014). Total CO_2 emission (C mineralization) was calculated by aggregating CO_2 emissions for the complete duration of the incubation. Soil

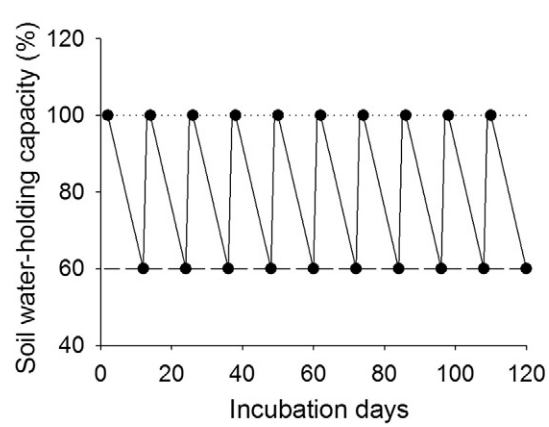

Fig. 1. Soil moisture content during the entire experimental period. CO_2 trapping was conducted (indicated as black dots) at each drying-rewetting cycle.