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Natural recovery rates of moss biocrusts after severe disturbance in a semiarid climate of the Chinese Loess Plateau

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ABSTRACT

Biocrusts are vulnerable to large scale disturbances including trampling activities, and their recovery rates are highly variable with estimates fluctuating by more than one order of magnitude, from < 5 years (very fast) to > 250 years (very slow). Also, the development of microbial abundance and community diversity of biocrusts after disturbances is poorly understood. In a semiarid climate of the Chinese Loess Plateau, we conducted a recovery experiment on well-developed moss biocrusts after a severe disturbance, during which the entire upper 3 cm of the surface was removed. In the following nine years, the general characteristics and microbial community of the recovering and undisturbed biocrusts as well as a substrate with no crust (bare sand) were periodically determined. Through linear or logistic extrapolation of the observed recovery rates, the recovery time of the biocrusts after disturbance was estimated by both biocrust characteristics and microbial community. Recovery time yielded the following estimates: a) coverage within 3 years, b) thickness within 8 years, c) biomass within 9–13 years, and d) cultivable microbial density within 11–13 years. More importantly, the recovery time of the disturbed moss-biocrusts estimated by the number of bacteria and fungi was ~10 and 20 years, respectively, and that estimated by the bacterial and fungal community diversity was 12–14 and 12–16 years, respectively. In conclusion, moss biocrusts would take 15–20 years to achieve full recovery, which was shorter than many previously published estimates that regarded biocrusts and especially moss-dominated biocrusts to have a long recovery time of hundreds of years. However, it should be also kept in mind that very fast recovery (< 5 years) of biocrusts was less reliable because such estimations are mostly based on visual cover only rather than on the multi-variables of the recovering biocrusts.

1. Introduction

Biocrusts are photoautotrophic communities formed by cyanobacteria, mosses, soil lichens, green algae, fungi and bacteria under a variety of climatic conditions (Belnap et al., 2016; Bowker et al., 2018). They are extensively developed and widely distributed in arid and semiarid regions in the world, and they have been regarded as an important component of land cover (Kidron et al., 2015; Xiao et al., 2011). Biocrusts can impact many ecosystem services, such as affecting water redistribution (Kidron, 1999; Rodríguez-Caballero et al., 2012), preventing wind erosion (Belnap et al., 2014; Bu et al., 2015; Ma et al., 2017), increasing soil C and N (Chamizo et al., 2012; Kidron et al., 2015), facilitating (Funk et al., 2014; Godínez-Alvarez et al., 2012) or inhibiting (Deines et al., 2007; Su et al., 2007) vascular plant establishment and growth, and promoting soil biodiversity (Castillo-Monroy et al., 2011; Maier et al., 2014). Particularly, moss biocrusts attract more attention because they usually are more impactful on various ecological processes than cyanobacteria, green algae, or lichen biocrusts due to their greater biomass and larger thickness, especially by stabilizing the soil surface and changing the soil water regime (Jia et al., 2008; Xiao and Hu, 2017; Xiao et al., 2016). It is widely accepted that biocrusts are ecosystem engineers and play an important role in soil processes and functioning (Bowker et al., 2010; Weber et al., 2016), and that their rehabilitation are promising measures for combating land

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degradation and desertification (Assouline et al., 2015; Rossi et al., 2017).

Biocrusts are quite fragile and are highly vulnerable to soil surface disturbance (Dojani et al., 2011; Langhans et al., 2010; Yang et al., 2018). Once disturbed, they may sharply degrade in coverage and biomass, especially in species richness and community diversity (Gómez et al., 2012; Steven et al., 2015; Williams et al., 2018). Results from previous research have suggested that it might take a very long time for biocrust to recover after disturbance, with estimated recovery of moss-dominated crusts being > 250 years (Belnap, 1993; Belnap and Eldridge, 2003; Weber et al., 2016). It is also generally accepted that recovery of biocrusts to pre-disturbance states once they were heavily disturbed is a long process (Weber et al., 2016; Williams et al., 2018), especially for moss biocrusts. Nevertheless, some other quantitative measurements implied that the recovery time of biocrusts is substantially shorter, between 6 and 9 years for cyanobacterial crusts and 17–22 years for moss-dominated crusts (Kidron et al., 2008), pointing at very large variations in the estimates of the natural recovery rates.

The large variation reported for the recovery rates of biocrusts was partially explained by the differences in biocrust type, climatic zones, severity and timing of disturbances, and soil types (Weber et al., 2016; Williams et al., 2018), but also by the different methods applied and the type of the measured variables (e.g., coverage, biomass, nutrients) (Concostrina-Zubiri et al., 2014; Weber et al., 2016; Xiao et al., 2014). Most importantly, the recovery rates of biocrusts in these studies were mostly estimated based on morphological assessments (such as crust cover), and sometimes by the crust biomass (such as chlorophyll content), but did not include the genomic/sequencing assessments (Kidron et al., 2008; Maestre et al., 2006; Xiao et al., 2014). At present, it is commonly believed that biocrust recovery rates should ideally be evaluated by multiple lines of evidence. The soil microbial abundance and community diversity are possibly very important indicators of biocrust recovery (Liu et al., 2017b; Xiao and Veste, 2017), but they were less considered and poorly understood in biocrust recovery assessment (Bates et al., 2010; Kuske et al., 2012; Liu et al., 2017a; Liu et al., 2013; Nejidat et al., 2016).

By creating favorable microhabitats in dry environments, biocrusts usually harbor a larger number and higher diversity of soil microorganisms than uncrusted soils (Castillo-Monroy et al., 2011; Steven et al., 2014; Zaady et al., 2010). The soil bacteria and fungi in biocrusts are significant contributors to primary productivity in dryland ecosystems, and they are also central to C and N cycling, as well as degrading organic matter (Maier et al., 2014; Steven et al., 2014). Generally, soil microbial communities provide soil ecosystem services and play a key role in soil organic matter turnover, C sequestration, and even hydrology (Brussaard et al., 2007; Chilton et al., 2017; Nannipieri et al., 2003). The loss of soil microbial diversity will likely reduce multifunctionality, negatively impacting ecosystem services such as soil quality and functions (Chilton et al., 2017; Delgado-Baquerizo et al., 2016). Thus, the microbial abundance and community diversity was generally used to identify the successional stages of biocrusts in natural development processes (Lan et al., 2013; Yu et al., 2012; Zhang et al., 2018), or in recovery processes after disturbance (Kuske et al., 2012; Steven et al., 2015; Zaady et al., 2010), with a greater abundance and higher community diversity of bacteria and fungi associated with more stable and developed successional biocrust stages (Lan et al., 2013; Nejidat et al., 2016; Yu et al., 2012).

In this study, we conducted a recovery experiment on well-developed moss biocrusts after a severe disturbance in a semiarid climate of the Chinese Loess Plateau. The severe disturbance was simulated through a total removal of topsoil layer including the biocrusts. In the following nine years, the general characteristics (biocrust coverage, thickness, and different measurements of biomass) and microbial community (abundance and diversity of bacterial and fungal community) of the recovering and undisturbed biocrusts as well as bare sand were periodically determined. The objectives of this study were to identify the differences of general characteristics and microbial community between the recovering and undisturbed biocrusts through periodical measurements during the first nine years after the disturbance, and subsequently to estimate biocrust recovery (i.e., of disturbed biocrusts naturally recovered to the same states of undisturbed biocrusts) rate based on these differences.

2. Materials and methods

2.1. Study area

The study was conducted in a ~7 km^2 watershed named Liudaogou (38°46′–38°51′ N, 110°21′–110°23′ E, 1081–1274 m above sea level) on the northern Loess Plateau, which has a semiarid climate with 409 mm mean annual precipitation (MAP) (~80% occurs in the summer) and 1337 mm average annual water evaporation (Xiao et al., 2016). The mean annual temperature is 8.4 °C, with mean monthly temperature ranging from −9.7 °C in winter (Dec.–Feb.) to 23.7 °C in summer (Jun.–Aug.) (Xiao et al., 2016). Natural biocrusts dominated by mosses *Bryum arcticum* (R. Brown) B.S.G. and *Didymodon vinealis* (Brid.) Zander are extensively developed on fallow lands, shrublands, and grasslands of the watershed, with coverage reaching 70–80% (Xiao et al., 2010).

A representative sparse shrubland with a 5-degree northeast facing slope and well-developed natural biocrusts was selected as the experimental site. The soil on the experimental site was an aeolian sandy soil (entisols in soil taxonomy of the United States Department of Agriculture), and its texture was loamy sand (soil texture classification system of the United States Department of Agriculture) with 81% sand, 14% silt, and 5% clay. The saturated water content (v/v), field capacity (v/v), wilting point (v/v), and steady-state infiltration rate of the top 10 cm soil were 1.56 g cm^{-3}, 21.4%, 12.6%, 1.7%, and 0.37 cm min^{-1}, respectively; while its pH value, organic matter, available nitrogen, and available phosphorus contents were 8.5, 0.13%, 11.1 mg kg^{-1}, and 0.52 mg kg^{-1}, respectively.

2.2. Experimental design and sample collection

Three treatments with four replicates were established, including (i) disturbed (scalped) moss-dominated biocrusts, (ii) undisturbed (intact) biocrusts, and (iii) bare, non-crusted sand. The disturbance included the total removal of the upper 3 cm of the surface, including the biocrusts.

The three treatments were established randomly on the experimental site in 12 plots, 5 × 3 m each, on Jun. 26, 2005. The bare, non-crusted plots, which were subjected to severe wind erosion (and therefore remained non-crusted) were demarcated at a distance of 5–10 m from the other plots. During the course of the experiment, all three plot types were monitored under natural conditions without further management practices.

In the following nine years, the biocrust characteristics including cover, thickness, moss density, and biomass were frequently measured (8–18 times). More importantly, four sampling campaigns were carried out immediately at the onset of the experiment i.e., at time 0 (Jun. 26, 2005), and after 2.2 years (Sep. 21, 2007), 6.1 years (Jul. 26, 2011), and 9.2 years (Sep. 23, 2014). The annual rainfall recorded during the experiment (2005–2014) ranged from 279.8 mm to 669.5 mm, averaging 427.2 mm (Fig. 1). For crust measurements, 20 mm of the topsoil with and without biocrusts were randomly sampled from 12 sub-sampling points using Petri dishes (90 mm diameter × 20 mm height) for each sample. Half of the samples were used for the measurement of general characteristics. Meanwhile, another half of the samples were homogenized and sieved (< 2 mm) to remove roots; then they were packed on dry ice and stored at −20 °C until further processing of their microbial community (only for the samples taken at 9.2 years after the disturbance).