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Effect of breakdown and dispersion of soil aggregates by erosion on soil CO_2 emission

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ABSTRACT

Soil erosion is a serious problem around the world, in addition to soil loss, erosion is also considered to have a significant impact on SOC dynamics and CO_2 release to the atmosphere. There is disagreement on the overall effect of soil erosion on CO_2 emission at the landscape level. We measured the proportion of various aggregate size fractions at different slope positions which were affected by erosion. The cumulative CO_2 emitted from bulk soil samples and different aggregate sizes fractions from five slope positions in a landscape was examined in a 128-d incubation study to estimate how the breakdown and dispersion of soil aggregates by erosion affect C loss to the atmosphere via CO_2 emission. The proportion of coarse size aggregate fractions (>0.25 mm) at eroded slope positions was greater than that at depositional positions. Bulk soil samples at the summit emitted the greatest cumulative CO_2-C (0.49 ± 0.04 g C kg^{-1} soil) among all the slope positions. During the initial 22 days of incubation, the CO_2 emission rate from coarse size aggregate fractions (0.024 ± 0.009 g C kg^{-1} soil d^{-1}) was six times higher than that from small size aggregate fractions (0.0038 ± 0.0011 g C kg^{-1} soil d^{-1}) at the depositional toe-slope position. The CO_2 emission rate from coarse size aggregate fractions at depositional slope positions (toe-slope and foot-slope) was significantly greater than that at eroded slope positions (summit, shoulder-slope and back-slope). We concluded that the breakdown and dispersion of aggregates by erosion impacts both aggregate size distribution and CO_2 emission from the aggregates at different slope positions, thus, affects the C loss to the atmosphere.

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1. Introduction

Soil organic carbon (SOC) is an important component of the global carbon cycle. Growing concern about increasing concentration of CO_2 in the atmosphere has aroused interest in SOC sequestration as a viable option for greenhouse effect mitigation (Polyakov and Lal, 2004a). A matter of concern impacting SOC dynamics including CO_2 release is soil erosion which is a wide spread problem around the world (Lal, 2003; Liu et al., 2003; Van Oost et al., 2005).

Erosion influences SOC in two ways: 1) redistribution of SOC within the watershed or ecosystem, and 2) loss of C to the atmosphere by mineralization (Polyakov and Lal, 2004a). There is disagreement on the overall effect of soil redistribution imparted by erosion on C sequestration at the landscape level (Van Oost et al., 2007). The C mineralization rate at depositional positions is greater than that at eroded positions (Lal, 2005), and some studies suggest that erosion results in a net flux of up to 1 Gt C per year to the atmosphere due to increased rates of mineralization (Lal, 2003). In contrast, another study suggests that soil erosion does not significantly increase carbon mineralization (Van Hemelryck et al., 2010), and furthermore, erosion is favorable for C sequestration at depositional slope positions due to SOC being buried by sediment and protected from mineralization (Lomander et al., 1998; Fierer et al., 2003; Polyakov and Lal, 2004b; Chan et al., 2007).

Soil redistribution by erosion may increase the sensitivity of SOC mineralization rate to environmental variables. While there is a general consensus of opinion among researchers about the impact of erosion on SOC on upland locations, the fate of SOC in redistributed sediment at the depositional positions is a subject of debate (Lal, 2005; Vandenbygaart et al., 2012). There are two possible contrasting outcomes of the deposited SOC (Polyakov and Lal, 2008). First, burial and subsequent accumulation of SOC by sediment in the depositional sites may occur (Avnimelech and Mchenry, 1984; Stallard, 1998; Smith et al., 2001). Second, SOC mineralization and the loss to the atmosphere in the form of CO_2 and CH_4 may occur. It has been reported that shortly after an erosion event, the SOC mineralization rate in disturbed sediment in depositional slope positions is substantial (Jacinthe et al., 2002; West and Wali, 2002).

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Soil aggregation and organic carbon accumulation are two intrinsically linked phenomena (Barreto et al., 2009). Particulate organic material (POM) is the most physically mobile fraction of SOC. Preferential transport of POM due to its low density is one of the major causes of SOC enrichment during an erosion event (Ghadiri and Rose, 1991). Besides physical movement of soil and SOC within it, erosion processes have a profound effect on the structure of transported sediments (Wan and El-Swaify, 1997). Aggregates which protect SOC are degraded by raindrop impact and mechanical shaking during translocation by erosion in runoff exposing the SOC to mineralization (Wei et al., 2014). Erosion disintegrates soil aggregates, altering their air and water regimes, and microbiological activity which in turn have an impact on the CO_2 emission from soil.

Previous incubation studies showed that soil disturbance and aggregate breakdown can significantly affect CO_2 emission (Franzluebbers, 1999; Jacinthe and Lal, 2001). Polyakov and Lal (2008) conducted laboratory rainfall simulation and incubation studies to determine the effect of breakdown and dispersion of soil aggregates by erosion on the CO_2 emission. During the initial 20 days of incubation the amount of CO_2 released from initial coarse size aggregate fractions was nine times greater than that from small size aggregate fractions due to the greater initial amount of SOC in coarse aggregates and its exposure to the environment imparted by the aggregate breakdown.

Very few studies have been conducted to evaluate the effect of soil erosion on distribution of aggregate fractions at various slope positions in a landscape and the effect of breakdown and dispersion of soil aggregates by erosion on soil CO_2 emissions. Thus, the objectives of this paper were to: (1) estimate the difference in the proportions of aggregate fractions at different slope positions affected by erosion, and (2) estimate how the breakdown and dispersion of soil aggregates by erosion affects CO_2 emission and the net effect of erosion on C loss to the atmosphere.

2. Materials and methods

2.1. Study area

The present laboratory incubation study was conducted on soil samples taken from the same field site and slope positions as our previous study where CO_2 flux was measured in situ (Wei et al., 2014). The study area was a small (9.59 ha) watershed located in the southern part of the black soil region in Northeast China, (44° 43′ N, 125° 52′ E). The average annual precipitation was 534 mm of which more than 70% occurred in June, July and August. The soil was a Mollisol soil (Black soil) with a clay loam texture. The bulk density of the surface soil (the top 30 cm of soil was considered to be surface soil where soil property was uniform due to cultivation) ranged from 1.20 to 1.48 g cm^{-3}. The pH value ranged from 6.03 to 6.58. The sand, silt and clay content in the study watershed ranged from 37.6 to 47.4%, 19.0 to 23.9%, and 33.7 to 38.5%, respectively (Table 1). Corn was the dominant crop for many years prior to conducting the present study.

The study agricultural field was representative of approximately 40% of the cultivated Black soil in Northeast China, with slope gradient ranged from 0° to 8°. Over one hundred years of cultivation with little soil conservation measures has resulted in serious erosion in this area (Yang et al., 2011). The eroded soil and SOC from the eroded slope positions in our study was estimated to be 6.2 t ha^{-1} yr^{-1} and 83.9 kg ha^{-1} yr^{-1}, respectively, and the deposited soil and SOC at the depositional slope positions was estimated to be 1.7 t ha^{-1} yr^{-1} and 22.7 kg ha^{-1} yr^{-1}, respectively (Fang, 2005).

2.2. Soil sampling and processing

We selected three transects on the north slope of the watershed, the three transects were spaced at 132 cm (two corn rows) and parallel to both the corn rows and the slope gradient. Five representative slope positions, summit, shoulder-slope, back-slope, foot-slope and toe-slope were selected on each of the three transects, approximately 12 m, 40 m, 130 m, 200 m and 230 m, respectively, from the northern boundary of the watershed. The slope gradient at the five slope positions were 2.60 ± 1.12°, 3.52 ± 0.46°, 2.23 ± 0.58°, 1.16 ± 0.67° and 1.10 ± 0.58°, respectively. Soil samples were collected in September, 2011 by digging a pit at each of the 15 sites (five slope positions and three transects) and one bulk soil sample was collected at each of the depths of 0–5 cm, 5–10 cm, 10–20 cm, 20–30 cm at each site, totally 60 bulk soil samples (15 sites × 4 depths) were collected. One sample for bulk density determination was also taken at each of these depths by pushing a 5 cm high × 5 cm diameter cutting ring into the vertical face. We collected soil samples at different depths to get the SOC content for the different depths. The 60 bulk samples were manually broken and air-dried immediately after collecting and then measured. Total SOC content of each of the 60 air-dried bulk samples was determined by dry combustion using a FlashEA1112 elemental analyzer (ThermoFinnigan, Milan, Italy). All soil samples were free of carbonates and so total C was assumed to SOC. The SOC content of the 0–30 cm layer was calculated as the average of the SOC content for the 0–5, 5–10, 10–20 and 20–30 cm layers weighted according to the layer thickness and bulk density. The surface 30 cm soil was considered to be uniform as a result of tillage by moldboard plow, and this was the reason why we obtained a composite sample for 0–30 cm depth soil to analyze as aggregate classes although erosion would just justify only the surface 10 cm soil in the measured year.

Parts of the bulk samples from the various depths were composited in proportion to the thickness and bulk density of each depth layer to form a single sample representative of the 0–30 cm layer for site. The composited bulk samples for the 0–30 cm depth were separated into three aggregate size fractions (coarse, >0.25 mm; medium, 0.25–0.053 mm; small, <0.053 mm) by wet sieving with 0.25 and 0.053 mm screen openings and the three fractions were then air-dried. Wet sieving is a well established method in soil aggregate analysis, it has been widely used in previous studies to isolate and get water-stable aggregate fractions of various sizes (Dagesse, 2013; Matkin and Smart, 1987; Michael et al., 1993; Pojasok and Kay, 1990; Polyakov and Lal, 2008). Care was taken to prevent breaking the aggregates during the wet sieving. During the air-drying process, the aggregate samples were gently broken when the soil moisture content was about 20% to prevent consolidation. We did not do sand corrections for aggregate fractions for two reasons. Firstly, sand particles are smaller than much of the macro-aggregates bound together by organic matter, and are legitimate components of coarse size aggregate fractions (Yang

Table 1
Soil characteristics of 0–30 cm depth at various slope positions.

Slope position	Slope gradient (°)	pH	Bulk density (g cm^{-3})	Soil texture (%)		
				Clay (<0.002 mm)	Silt (0.002–0.02 mm)	Sand (0.02–2 mm)
Summit	2.60 ± 1.12	6.44 ± 0.26	1.27 ± 0.01	37.4 ± 0.67	22.5 ± 0.62	40.1 ± 0.51
Shoulder	3.52 ± 0.46	6.58 ± 0.19	1.32 ± 0.01	38.5 ± 0.33	23.9 ± 1.12	37.6 ± 0.78
Back	2.23 ± 0.58	6.31 ± 0.22	1.33 ± 0.01	36.6 ± 0.11	20.0 ± 0.41	43.4 ± 0.31
Foot	1.16 ± 0.67	6.17 ± 0.41	1.37 ± 0.04	34.6 ± 0.22	19.1 ± 0.39	46.3 ± 0.50
Toe	1.10 ± 0.58	6.03 ± 0.15	1.40 ± 0.03	33.7 ± 0.33	19.0 ± 0.43	47.4 ± 0.75

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