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Land degradation impact on soil carbon losses through water erosion and CO_2 emissions

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ABSTRACT

Worldwide concerns with global change and its effects on our future environment require an improved understanding of the impact of land cover changes on the global C cycle. Overgrazing causes a reduction in plant cover with accepted consequences on soil infiltration and soil erosion, yet the impact on the loss of soil organic carbon (SOC) and its associated processes remain unaccounted for. In this study performed in South Africa, our main objective was to evaluate the impact of plant cover reduction on (i) SOC erosion by water in both particulate (POC) and dissolved (DOC) forms, and (ii) soil CO_2 emissions to the atmosphere. The study performed under sandy-loam Acrisols investigated three proportions of soil surface coverage by plants (Cov), from 100% (Cov100) for the "non-degraded" treatment to 25–50% (Cov50) and 0–5% (Cov5). POC and DOC losses were evaluated using an artificial rainfall of 30 mm h^{-1} applied for a period of 30 min on bounded 1×1 m² microplots (n = 3 per treatment). CO_2 emissions from undisturbed soil samples (n = 9) were evaluated continuously at the laboratory over a 6-month period. At the "non-degraded" treatment of Cov100, plant-C inputs to the soil profile were 1950 ± 180 gC m^{-2} y^{-1} and SOC stocks in the 0–0.02 m layer were 300.6 ± 16.2 gC m^{-2}. While soil-C inputs by plants significantly ($P < 0.05$ level) decreased by 38.5 ± 3.5% at Cov50 and by 75.4 ± 6.9% at Cov5, SOC, the losses by water erosion of 0.75 gC m^{-2} at Cov100 increased from 66% at Cov50 (i.e. 3.76 ± 1.8 gC m^{-2}) to a staggering 213% at Cov5 (i.e. 7.08 ± 2.9 gC m^{-2}). These losses were for the most part in particulate form (from 88.0% for Cov100 to 98.7% for Cov5). Plant cover reduction significantly decreased both the cumulative C–CO_2 emissions (by 68% at Cov50 and 69% at Cov5) and the mineralization rate of the soil organic matter (from 0.039 gC–CO_2 gC^{-1} at Cov100 to 0.031 gC–CO_2 gC^{-1} at Cov5). These results are expected to increase our understanding of the impact of land degradation on the global C cycle. Further in-situ research studies, however, need to investigate whether or not grassland degradation induces net C-emissions to the atmosphere.

1. Introduction

General interest in carbon (C) budgets has increased following the signing of the Kyoto Protocol on climate change and this has accelerated in recent years because soil carbon is recognized to be an important controlling factor of numerous ecosystem functions (Tate et al., 2000).

Grasslands, savannas and shrublands represent about 30% of the earth's surface area (De Fries et al., 2000) and are thus a key component of the global C cycle (Grace et al., 2006). In most semi-arid regions of the world, especially in Africa, overgrazing plays a major role in land degradation (Daily, 1995). Yet, the expected direct or indirect consequences on the biomass and soil C pools have been much less publicized than deforestation in rain forests. Overgrazing occurs when plants are exposed to intensive grazing for extended periods of time, or are without sufficient recovery periods (Harris, 2010; Moretto et al., 2001; Todd and Hoffman, 1999). Overgrazing is expected to directly decrease the amount of soil carbon, but usually only by a small amount, since grasses allocate most of their photosynthesised organic C to roots rather than to shoots (Sims and Singh, 1978). Because much of the carbon in the soil decomposes slowly (Fontaine et al., 2007), the impact of decreased rooting on SOC stocks might take years after overgrazing starts. There are also indirect soil C depletion mechanisms at work. First, fire, be it man-caused or natural in origin, is shown to cause long-term changes in soil organic matter. In a South African grassland of KwaZulu-Natal, Fynn et al. (2003) showed that burning causes a decrease in above-ground litter inputs, but increases turnover of root material below the surface. Surprisingly, the suppression of fire is likely to favor tree encroachment into grasslands (Houghton et al., 1999; Van Auken, 2000) with either positive or negative consequences on soil C stocks. Operating over an estimated area of 220 million hectares in the United States, Houghton et al. (1999) estimated that tree encroachment is responsible for 18 to 34% of the total estimated C

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sink in the whole country. These results were further contested by Jackson et al. (2002) who suggested that when shrubs invade grasslands, carbon inputs to the topsoil decrease because woody species allocate more of their total C production above-ground than below-ground and what they do send to roots is distributed deeper than the first meter of the soil.

SOC loss through soil water erosion and leaching is another matter of concern in degraded lands. Indeed, several feedbacks exist between plant cover reduction and soil infiltration by water and soil water erosion (Mapfumo et al., 1999; Valone et al., 2002) that may dramatically decrease SOC stocks (Chaplot et al., 2005, 2009; Kuhn et al., 2009). The decrease in soil surface coverage by vegetation, together with trampling, increases soil surface crusting, which in turn, decreases soil infiltration by water and increases the erosion of fertile C-enriched surface soil horizons (Gamougoun et al., 1984; Podwojewski et al., 2011; Trimble and Mendel, 1995). There is a considerable amount of literature regarding soil erosion and grassland degradation, but literature focusing on the interactions between land degradation and SOC losses is widely missing.

From the few available studies on soil erosion impact on SOC stocks in overgrazed grasslands, we learn that soil truncation can significantly reduce SOC stocks as much as 45% in the case of a 0.1 m truncation, as shown by Charley and Cowling (1968) in Australia. While Charley and Cowling (1968) were probably one of the first to quantify the erosion impact of SOC stocks with irreversible consequences on plant production, little is known about the impact of grassland degradation on soil C stocks and C losses through water erosion and organic matter decomposition. Moreover, while many studies provide experimental evidence of the linkage between land management and SOC mineralization and associated soil CO_2 emissions (Jacinthe et al., 2002; Jackson et al., 2002), little information is available on the way grassland degradation affects soil respiration. The changes in gas fluxes, especially CO_2 at the soil–atmosphere interface, need to be known as a first step towards the better understanding of grassland degradation in the global C cycle.

In the study of a degraded grassland, our main objective was to investigate the impact of surface coverage by plants on particulate (POC) and dissolved (DOC) SOC losses by water erosion and soil CO_2 emissions to the atmosphere.

2. Materials and methods

2.1. Description of the study site

2.1.1. Main characteristics of the study area

The study site was located in Potshini, 8 km south of Bergville, a town in KwaZulu-Natal, South Africa (Fig. 1). It was a degraded natural grazing area with clear evidence of land degradation, such as bare soil, isolated tall grass tussocks and the presence of pedestals and exposed roots, while other patches are entirely covered by vegetation. The area is a typical open communal grazing land with no fertilization, where the common practice is to burn the grass in order to regenerate it and to prevent the rangeland from becoming closed stands of woody species (Harrison and Shackleton, 1999).

The climate of the area is temperate, with cold dry winters and rainy summers. The mean thirty-year annual precipitation at the Bergville meteorological station, 10 km North of the study area (Long: 29.38°; Lat: −28.81°), is 684 mm, the potential evaporation is 1600 mm and the mean annual temperature is 13 °C (Schulze, 1997). Summer rainfall occurs between October and March and frosts are common in winter. In the area, a 30-minute rainfall has a 2-year return period rainfall intensity of 49 mm h^{-1}, with a 90% occurrence of this intensity being between 37 and 61 mm h^{-1}. The 10-year return period rainfall intensity is 76 mm h^{-1} and for a 100-year period, it is 115 mm h^{-1}.

Soils in the area are typical acidic Acrisols (W.R.B., 2006) developed from sandstones. Slopes are characterized by deep Acrisols (~2 m) with a massive structure. Horizons are compact (soil bulk density (ρ_b) between 1.4 and 1.6), except the A horizon with $\rho_b = 0.8$ g cm^{-3}. The humiferous A horizon is generally dark reddish-brown (5YR 3/3), blocky and friable. The Bw horizon, situated below, from 0.4 to 0.9 m depth, is dark reddish-brown (2.5YR3/4), massive and clayey. A sandy saprolite is reached at a depth of about 1.7 m. Footslopes and valley bottoms exhibit features of waterlogging (Soil Survey Staff, 1999). The A horizon is dark gray (2.5YR4/1) and enriched by organic matter.

2.1.2. Characteristics of the study site

The study site is a 10 × 30 m² plot with a mean slope gradient of 8%. The plot shows a gradation of soil surface coverage by plants (Cov) from 0–5% (Cov5), 25–50% (Cov50) to 100% (Cov100) (Fig. 1). Soils are relatively homogeneous across the plot and show an average soil depth of 1.5 m. Because fires are set up regularly in the grassland, the soils did not exhibit any O horizon. The A-horizon is brown (7.5YR4/4), sandy loam (from 55 to 68% sand), with a low percentage of clay (15.0–17.7%), but a high content of fine sand (45–50%). The A horizon is 0.35 m thick and shows a fine granular structure. It is acidic (pH = 4.9–5.2) with a low exchangeable capacity (CEC = 2–4 $Cmol^{(+)}$ kg^{-1}) and a soil organic carbon content between 9 and 12 gC kg^{-1}. The sub-surface organo-mineral AB_W horizon (0.35–0.85 m) shows a similar texture, but a lower carbon and nitrogen content and a lower CEC (Table 1). Beneath lie two clayey mineral sub-surface (Bw) horizons, reddish (5YR4/6), with an apedal structure and significantly more clayey (clay = 211–224 g kg^{-1} for Bw1 and clay = 297–326 g kg^{-1} for Bw2). The average clay content in the 0–0.02 m layer was 17.7 ± 2.5% for Cov100 and decreased to 15.4 ± 1.9% for Cov5, but these differences were not statistically significant (Table 1). The concentration of cations significantly decreased from Cov5 to Cov100 at a rate of between 22% for Na^+ and 55% for K^+.

2.2. The experimental methods

Assessment of Cov impact on the depletion of soil organic carbon (SOC) stocks by erosion was performed using rainfall simulation, while estimates of CO_2 losses through organic matter decomposition were obtained via the incubation of undisturbed soil samples. Finally, information on SOC stocks and grass biomass were collected in an attempt to compare SOC losses to soil C stocks and soil C input by plant.

2.2.1. Particulate and dissolved soc by water erosion

Nine 1 m^2 runoff micro-plots were installed within the plot with three micro-plot replicates per class of Cov. The metal borders surrounding the micro-plots were inserted to a depth of 0.1 m and a collector and a 50-liter tank were placed at the bottom end to collect surface runoff and erosion SOC. Artificial rainfall was applied on dry soils at the end of the 6-month dry season to control both the rainfall and soil conditions. At the time of the study in June 2007, the grass was fully-grown and no grazing had occurred since the grass growing cycle, which started with the first rains of October 2006.

The rainfall, with a rainfall intensity of 30 mm h^{-1} and a rainfall duration of 30 min, was produced using the rainfall simulator manufactured by Capelec (1995), with an oscillating nozzle (Teejet SS 6560) situated 4 m above the soil surface. The water pressure of 40 ± 0.2 kPa generated raindrops with an average kinetic energy of 25 J m^{-2} mm^{-1}. Before each rain event, rainfall intensity and its spatial variations were determined to be lower than 5% of the average 30 mm h^{-1}, by using small bowls placed within the micro-plots at each node of a 0.2 m grid. The rainfall characteristics corresponded to a one-year return period rainfall event (Smithers and Schulze, 2002). Following Wischmeier and Smith (1978), the artificial rainfall had an erosivity of 3.2 MJ cm ha^{-1} h^{-1}.