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Importance of land use patterns for erosion-induced carbon fluxes in a Mediterranean catchment

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ABSTRACT

Land use and land cover (LULC) have a strong influence on the intensity of soil erosion processes and the consequent loss of soil organic carbon (C) and nutrients from soils. Yet, at the landscape scale most studies quantifying the effect of soil erosion on soil C dynamics have focused on homogeneous (mainly agricultural) areas. Here we study the effect of LULC patterns on erosion-induced lateral C fluxes and the net ecosystem C balance at the catchment level. The SPEROS-C model, a soil erosion model coupled with a C dynamics model, was applied to 12 catchments ranging from 8 to 430 ha in SE Spain and calibrated with field-based data on sediment yield and soil C concentration. Four LULC classes were considered: forest, shrubland, pasture and agriculture. Agricultural areas were the most dynamic sites accounting for 70% of the eroded soil but only for 45% of the total eroded C. The remaining percentage of eroded C derived mainly from relatively C rich forest soils which were the dominant LULC class in terms of spatial extent. The impact of soil erosion on annual ecosystem C fluxes was highest for agricultural soils due to lower C input and soil C stocks. During the study period (28 years), 26% of the eroded C remained within the catchments' hillslopes deposited in 6% of the hillslope area, and not homogeneously distributed over the landscape. These results indicate that assessment of the role of soil erosion on soil C dynamics and ecosystem C fluxes should be undertaken from a landscape perspective, including the effects of LULC on the redistribution of erosion-induced C fluxes.

1. Introduction

Population migration from rural areas driven by socioeconomic factors has led to a large scale abandonment of agricultural fields in the mountainous Mediterranean areas of Spain over the last century (García-Ruiz et al., 1996). This process has often been accompanied by implementation of soil conservation measures to reduce soil erosion rates (reforestation and construction of sediment retention structures), resulting in a significant increase in vegetation cover during a short-time period (García-Ruiz et al., 1996; Boix-Fayos et al., 2008). While the impact of these human-induced processes on catchment hydrological and sediment dynamics has been widely studied (e.g. Boix-Fayos et al., 2007; García-Ruiz and Lana-Renault, 2011), their effect on ecosystem carbon (C) stocks and fluxes at the landscape scale remains poorly quantified. This is surprising given the large number of field plot studies assessing the effect of vegetation cover type, land use conversions and fires on soil C and C fluxes (Albaladejo et al., 2013). Results from these studies have underlined the influence of land use and land cover types on the intensity of soil erosion processes and the consequent loss of soil C and nutrients (Pardini et al., 2003; Ruiz-Colmenero et al., 2013). Thus, at the landscape level, with a mosaic of land use and land cover types, the effect of soil erosion on soil C can be expected to present a high spatial variability as additional factors become involved controlling the intensity of erosion and depositional processes (Lü et al., 2007; Fiener et al., 2008; Yadav and Malanson, 2009, 2013; Sanderman and Chappell, 2013).

Landscape assessment of the impact of soil erosion on soil C stocks and its significance for ecosystem C budgets requires the quantification of lateral (e.g. transfers of soil organic carbon particles) and vertical (e.g. C mineralization) fluxes associated with soil detachment, and sediment transport and deposition. The assertions that a fraction of the transported and deposited C by soil erosion processes could be mineralized (Schlesinger, 1990), and that C lost from eroding slopes could be partially replaced by fresh C input (Stallard, 1998; Harden et al., 1999) opened the debate on the role that erosion-induced C fluxes could play in the C cycle. Since then, most studies on the subject have focused on agricultural

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catchments that were homogeneous in terms of land use and soil characteristics (e.g. Dlugoß et al., 2012; Doetterl et al., 2012; Jacinthe et al., 2004; Van Oost et al., 2005). However, to better assess the role of soil erosion in the C budget there is a need to characterize the role of spatial and temporal variability of LULC types and other environmental conditions (i.e. slope, lithology, climate) on lateral and vertical C fluxes (Berhe et al., 2007; Van Oost et al., 2007).

Erosion-induced C fluxes are a component of the net ecosystem carbon balance (NECB). The NECB includes input and output of fluxes between terrestrial and atmospheric components (vertical fluxes), as well as input and output by advection (lateral fluxes) of atmospheric CO_2, dissolved organic and inorganic carbon, and particulate organic carbon (Chapin III et al., 2006). In this balance, the lateral redistribution of organic carbon over the landscape triggers two fluxes: the direct removal or deposition of soil particles and the indirect change in the rates of C input to soil from net primary productivity and of C mineralization to the atmosphere. In other words, the lateral C flux derived from eroding sites can be part of a steady-state flux where the input from vegetation (C replacement) balances erosional losses (Stallard, 1998; Harden et al., 1999). Improving our knowledge on the effect of soil erosion on C mineralization and replacement requires a good understanding of the factors and processes controlling their rate of disturbance such as erosion-induced lateral C fluxes.

Here, we hypothesized that different LULC patterns result in a different magnitude and spatial distribution of the lateral C fluxes induced by soil erosion processes, with implications for the NECB. The objectives of this study were to: (i) quantify and compare the soil erosion-induced lateral C fluxes in a selection of small catchments with heterogeneous land use, soil and topography, (ii) evaluate the role of LULC on lateral C fluxes and, (iii) assess and discuss the implications of erosion-induced lateral C fluxes on ecosystem C stores and the NECB.

To simulate erosion-induced C fluxes at the catchment or regional scale, a modeling approach is helpful. We used the SPEROS-C model (Van Oost et al., 2005) which has successfully been applied in small agricultural catchments (Van Oost et al., 2005; Dlugoß et al., 2012). The representation of C turnover and input for different soil layers in the model allows SPEROS-C to simulate C dynamics in the whole soil profile under the impact of deposition of the spatially redistributed C, as well as to simulate the effect of topsoil removal on the C dynamics in eroding profiles (Van Oost et al., 2009). When applied at the catchment scale, the model can simulate spatial transfers between geomorphic positions and LULC classes.

2. Methods

2.1. Study site

The study sites are located within the Rogativa catchment (~50 km^2, 38° 08′N, 2° 13′W) in the Murcia province, in Spain (Fig. 1). The Rogativa catchment has a dry-subhumid Mediterranean climate with a mean annual temperature of 13.3 °C and 581 mm of mean annual rainfall (Boix-Fayos et al., 2007). Lithology consists of marls in the valley floors and limestones on higher elevations (IGME, 1978). Soils are classified as Regosols in their majority, combined with Leptosols at higher elevations and Cambisols on the north facing slopes under forest cover (Alías et al., 1991).

The Rogativa catchment has been subjected to hydrological-correction works to control soil erosion since the 1970s that involved construction of check-dams and afforestation. Due to these measures, the catchment is currently divided into 58 smaller catchments. For this study, 12 of these catchments were selected (Table 1, Fig. 1); those located in the headwaters constituting closed hydrological units (3,5,22,23,49) and those receiving input from only one upstream catchment separated by a check-dam (1,14,21,24,34,51,57). Catchments with more than one check-dam upstream or in the main Rogativa channel were not selected to avoid too complex sediment dynamics.

Important LULC changes were reported for the Rogativa catchment during the second half of the 20th century, indicating an increase in forest cover and a decrease of rainfed agricultural area (Boix-Fayos et al., 2007). Currently, the landscape represents a mixture of evergreen forests and shrublands, pastures, rainfed cereals, walnut plantations and vineyards (see Boix-Fayos et al., 2009 for further details). For this study, LULC was divided into four classes: forest, shrubland, pasture and grazing, and agriculture (including cereals and orchards).

2.2. Soil and sediment sampling and analyses

Soil sampling was carried out to characterize the top 10 cm of soil under each LULC class. Samples were taken from 2 locations per LULC class and per catchment during several field campaigns (between the years 2004 and 2010) (N = 120). Sediments retained behind the check-dams in each catchment were sampled at different depths by taking cores directly behind the dam, upstream. An auger was used to take the samples in 5 cm increments until bedrock was reached (0.3–1.5 m). This resulted in 5–25 samples from each sediment wedge (N = 149).

Soil samples were air-dried, gently crushed and sieved at 2 mm prior to undergoing further analytical procedures. Intact soil cores were dried at 105 °C for bulk density determination. Sediment samples were oven-dried at 60 °C, weighted for bulk density determination and afterwards crushed and sieved at 2 mm. Organic carbon content was determined by the wet oxidation method for samples taken until 2009 preheating the mixture of potassium dichromate and the concentrated sulphuric acid to 170 °C (30 min) to ensure a complete combustion (Yeomans and Bremner, 1988). For samples taken after 2009, organic carbon content was measured by dry combustion method in an elemental analyzer (FlashEA 1112 Series, Thermo Fisher Scientific). A comparison between the two methods was done on a random subset of the samples (N = 17). A linear regression between both was obtained (r^2 = 0.94) with no significant differences between average values of sample replicates (paired t-test, p > 0.05), and no correction factor between methods was applied. Clay content was determined using a Coulter LS200 (Beckman, USA) after eliminating organic matter and chemically dispersing the samples using a mixture of sodium hexametaphosphate and sodium carbonate anhydrous for 18–24 h.

2.3. SPEROS-C model

A detailed description of the SPEROS-C model can be found in Van Oost et al. (2005). Here we only describe the main model concepts and modifications applied.

2.3.1. Soil carbon dynamics

The C component is described in the ICBM model (Andrén and Kätterer, 1997) using two pools; a young and an old carbon pool. Four main fluxes control C stocks in each pool: (1) annual C input to the soil from above and belowground biomass, (2) C mineralization from the young pool, (3) C transformation from the young to the old pool, and (4) C mineralization from the old pool. In SPEROS-C, humification of soil C depends mainly on C input rates, litter humification coefficient and the clay content of soil. In this study, the litter humification coefficient and C turnover rates for each C pool were kept as defined in Andrén and Kätterer (1997) and considered homogeneous for the four LULC classes, since no data on these parameters was available for the site.