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Microbial responses to the erosional redistribution of soil organic carbon in arable fields

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ABSTRACT

Quantifying the potential for eroding agricultural soils to act as sinks or sources of atmospheric carbon relies on accounting for the pools and fluxes of soil organic carbon (SOC) and nutrients, e.g. nitrogen (N), affected by erosion. Herein, we report the outcomes of an experiment where a C_4 maize (*Zea mays*) crop ($\delta^{13}C = -12.1‰$) was cultivated and incorporated for 2 years to introduce a 'pulse' of ^{13}C-enriched SOC to a C_3 arable soil ($\delta^{13}C = -27.4‰$). Soils were sampled at eroding (top slope and upper slope) and depositional (lower slope and slope foot) positions of an accelerated erosion pathway that were confirmed using ^{137}Cs measurements. The sand particle-sized fraction (63–2000 μm) was predominant and increased in the depositional slope positions due to selective loss of fine particles and preferential deposition of the coarsest fraction of transported sediment. There was a significant isometric relationship between the percentage SOC and total N: top slope > upper slope > lower slope, with similar values in the slope foot to the top slope. The $\delta^{15}N$ values of the soils were enriched (7.3‰) at the slope foot, compared with the other slope positions (average 6.3‰), suggesting increased denitrification rates. The $\delta^{13}C$ values of the soil microbial biomass C extracted from surface soils (0–5 cm) at each slope position showed that the proportion of maize C being incorporated into the soil microbial biomass declined in the downslope direction from 54% (top slope) to 43% (upper slope) to 18% (lower slope) in inverse proportion to the size of the soil microbial biomass, and increased to 41% at the slope foot. This suggests dynamic replacement of the SOC with crop C in the eroding slope positions and dilution of the transported C by C3-SOC in the depositional slope positions. This paper is evidence that erosional distribution of soil carbon leads to differential microbial utilisation of SOC between eroding and depositional sites.

1. Introduction

Terrestrial ecosystems hold the potential to capture and store substantially more carbon (C) in soil organic matter (SOM) through changes in management that are also of benefit to the multitude of ecosystem services that soils provide (Dungait et al., 2012a). There is an opportunity to store more C in agricultural soils by replenishing the 55–78 Pg soil organic carbon (SOC) lost from the global terrestrial C pool after land use change from native vegetation to crop production (Lal, 2004; Smith et al., 2008). Full exploitation of this potential relies on accounting for the pools and fluxes of SOC affected by erosion (Smith et al., 2010). The magnitude of the error associated with the calculations of the global erosion flux in agricultural soils is very large (0.5 ± 1.5 Pg a^{-1}; Quinton et al., 2010) and will be compounded by the effects of climate change wherein soil erosion rates are expected to increase by 1.7% for each 1% increase in total precipitation (Nearing et al., 2004). Soil erosion is the most widespread form of soil degradation, accounting for up to 70% of C loss from cultivated soils (Gregorich et al., 1998), and is a source of CO_2 that contributes 0.8–1.2 Pg C a^{-1} to the atmospheric C pool (Lal, 2008). 'Critical' concentrations of less than 1–2% SOC in soil are considered to impair soil function leading to lower crop yields

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(Huber et al., 2008). The attrition, transport and deposition of eroded soils affect not only the physical distribution of SOC, but must drastically alter the process of C mineralization (Gregorich et al., 1998). The soil microbial biomass is generally considered an important biotic driver of soil C efflux (Paterson et al., 2008). Clearly, more accurate quantification of the biological processes associated with soil erosion is required to constrain estimates of the potential of agricultural soils to act as net sources or sinks of atmospheric C.

Temperate agricultural soils are affected by tillage and water erosion at around the same order of magnitude (Van Oost et al., 2006). Tillage erosion transports soil over short distances with less effect on the total SOC stock than water erosion (Zhang et al., 2006). Transported SOC is concentrated in depositional zones where it is buried, thereby reducing the potential for its decomposition (Schimel et al., 1985). The sources of the transported SOC are a mixture of the crop biomass and previously buried SOC exposed by ploughing. Areas from which SOC is eroded are also sinks because of the enhanced physical protection of new plant (crop) derived C as fresh mineral facies from the subsoil are exposed (Harden et al., 1999). When both sinks are considered as a net flux, tillage erosion actually creates a sink for SOC of 0.12 Pg C a^{-1} (Van Oost et al., 2007), although this stock of C may be vulnerable to future loss by management, land use change and a warming climate (VandenBygaart et al., 2012). However, much as understanding the physical C sink capacity of eroded slopes is important, in productive agriculture the quality of the SOC must be considered as a component of SOM that promotes a range of soil quality parameters required for sustainable crop production (Dungait et al., 2012b). Eroded soil usually has reduced productivity because of poor nutrient and water holding capacity even with addition of organic or inorganic fertilisers (Lal and Pimental, 2008). Renwick et al. (2004) stressed the importance of understanding the pathways of C displaced by erosion and quantifying the magnitude of erosion-induced emissions of greenhouse gases, including N_2O, requiring that other nutrient cycle dynamics affected by erosion are also considered.

Water erosion selectively removes the most labile components of SOC (Zhang et al., 2006) including a large proportion as light fraction (e.g. <1.8 g cm^{-3}) because this material can be easily transported (Gregorich et al., 1998). The slaking of aggregates leading to the exposure of previously encapsulated SOC from eroding slope segments is a source of previously protected 'old' C (Lal, 2003). Both selective transport of the light fraction and slaking of aggregates lead to the enhanced availability of SOC to mineralisation; total microbial C increases downslope in cultivated soils where water erosion is the primary cause of soil (re)distribution (Voroney et al., 1981). Under most climate change scenarios for Western Europe, the intensity of more sporadic rainfall events is predicted to increase with longer, drier summers (IPCC, 2007). Therefore, the contribution of accelerated erosion by water to SOC erosion in temperate environments may become more important, and this will be exacerbated by the trend towards increased production of forage crops such as maize (*Zea mays*). Jaafar and Walling (2010) measured gross and net soil erosion in fields under maize stubble and observed considerably greater rates of contemporary erosion (measured using ^{7}Be, a cosmogenic isotope with a half-life ($t_{1/2}$) of 53.3 days used to explore short term erosion processes) than the longer-term gross and net soil erosion rates for the same field (measured using ^{137}Cs). ^{137}Cs is an artificial radionuclide ($t_{1/2} = 30.2$ yr) released into the atmosphere from atomic weapons testing and emissions from nuclear power stations. ^{137}Cs is highly particle reactive, $K_d \approx 10^5$, and upon reaching the soil surface is quickly and irreversibly sorbed by clay minerals and other binding sites (Zapata, 2002). Accumulated fallout inventories (Bq m^{-2}) vary regionally according to latitude and precipitation patterns. Subsequent redistribution principally occurs in sediment-associated forms. Local variations in ^{137}Cs inventories have been widely used to characterise the spatial patterns of water-borne erosion and deposition at the field scale (cf. Walling and Quine, 1990); to reveal the topographic significance of tillage erosion (Van Oost et al., 2006); and, to trace sediment sources (Rowan et al., 2012) and quantify catchment-scale sediment budgets (Walling and Zhang, 2010).

The amount of SOC lost globally by mineralisation after mobilization of eroded soil is suggested to be 20% of the eroded SOC (Lal, 2003), or 1 Pg C a^{-1} (Lal, 2008). Mineralisation rates of 2–37% have been demonstrated in eroded soil (Jacinthe et al., 2002; Van Hemelryck et al., 2009), and Doetterl et al. (2012) recently reported a 25% increase in potential soil respiration in the top soils from eroding and depositional *versus* uneroded slope positions in temperate arable soils predominantly affected by tillage erosion. The source of the mineralised SOC was speculated upon, but could not be confirmed as recent (i.e. crop derived) or legacy (i.e. previously stabilised) C because the SOC could not be provenanced. However, establishing the origin of mineralised SOC between the 'new' and 'old' sources is possible by deploying the natural abundance ^{13}C-labelling of C_4 plants (Dungait et al., 2010). This approach has been used (i) at the field scale to source the origin of maize C (a C_4 plant) input to trapped sediment in run-off buffers with C_3 vegetation (Jacinthe et al., 2009) and to investigate the effect of vegetation change on erosion dynamics in natural dryland environments (Puttock et al., 2012); and, (ii) to assess the incorporation of 'new' (C_4) or 'old' (C_3) SOC into soil microorganisms by determining the δ^{13}C value of soil microbial biomass C (Blagodatskaya et al., 2011). Using a combination of these methods, we tested the hypothesis that the majority of SOC being utilised by soil microorganisms in eroding agricultural top soils is derived from contemporary crop residues.

2. Materials and methods

2.1. Experimental plot design

An experiment allowing recent plant C inputs to be distinguished from SOC formed under the previous C_3 crops was established on the Balruddery Farm at The James Hutton Institute's Centre for Sustainable Cropping, near Dundee, in the east of Scotland, UK (see Table 1 for latitude and longitude; National Grid Reference: NO 304329). Six adjoining fields (5.3–8.0 ha; total area 42.5 ha) were planted with maize in May 2009 and 2010, and the entire crop was chopped and ploughed into the soil in the late

Table 1
Characteristics of top soil samples (0–5 cm) from four slope positions of an erosion pathway on Balruddery Farm, compared with a reference soil from an adjacent field. Figures in brackets are standard errors of the mean.

Description	Latitude and longitude	^{137}Cs (Bq m^{-2})	Status	Texture (% sand:silt:clay)	SOC (%)	TN (%)	C:N
Top slope	56°28.931′N 3°07.844′E	1764 (208)	Eroding	58:28:14	1.75 (0.03)	0.15 (0.01)	11.7
Upper slope	56°28.885′N 3°07.583′E	2033 (110)	Eroding	48:37:15	2.05 (0.04)	0.20 (0.01)	10.4
Lower slope	56°28.747′N 3°07.124′E	2329 (653)	Depositional	72:20:08	2.54 (0.26)	0.23 (0.02)	11.1
Slope foot	56°28.686′N 3°06.914′E	2723 (274)	Depositional	65:23:12	2.00 (0.27)	0.15 (0.01)	13.7
Reference site	56°29.765′N 3°07.436′E	1990 (112)	Stable	48:36:16	1.78 (0.03)	0.16 (0.01)	11.1