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Modelling the impact of agricultural management on soil carbon stocks at the regional scale: the role of lateral fluxes

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Abstract

Agricultural management has received increased attention over the last decades due to its central role in carbon (C) sequestration and greenhouse gas mitigation. Yet, regardless of the large body of literature on the effects of soil erosion by tillage and water on soil organic carbon (SOC) stocks in agricultural landscapes, the significance of soil redistribution for the overall C budget and the C sequestration potential of land management options remains poorly quantified. In this study, we explore the role of lateral SOC fluxes in regional scale modelling of SOC stocks under three different agricultural management practices in central Belgium: conventional tillage (CT), reduced tillage (RT) and reduced tillage with additional carbon input (RT+i). We assessed each management scenario twice: using a conventional approach that did not account for lateral fluxes and an alternative approach that included soil erosion-induced lateral SOC fluxes. The results show that accounting for lateral fluxes increased C sequestration rates by 2.7, 2.5 and 1.5 g C m^{-2} yr^{-1} for CT, RT and RT+i, respectively, relative to the conventional approach. Soil redistribution also led to a reduction of SOC concentration in the plough layer and increased the spatial variability of SOC stocks, suggesting that C sequestration studies relying on changes in the plough layer may underestimate the soil's C sequestration potential due to the effects of soil erosion. Additionally, lateral C export from cropland was in the same of order of magnitude as C sequestration; hence, the fate of C exported from cropland into other land uses is crucial to determine the ultimate impact of management and erosion on the landscape C balance. Consequently, soil management strategies targeting C sequestration will be most effective when accompanied by measures that reduce soil erosion given that erosion loss can balance potential C uptake, particularly in sloping areas.

Keywords: agricultural management, carbon sequestration, soil erosion, soil organic matter, spatial modelling

Introduction

Currently, one-third of the world's land suitable for cultivation is under agricultural use, representing 12% of the world's land area (FAO, 2013), but the expected rise in the global demand for food, fibre and biofuel is likely to increase the pressure on the soil system (Koch *et al.*, 2013). This rise in soil pressure coexists with a renewed interest in the capacity of agricultural soils to (i) sequester a large fraction of the historically lost soil organic carbon (SOC), (ii) contribute to mitigate part of the annual greenhouse gas emissions to the atmosphere, (iii) reduce soil degradation (Smith *et al.*, 1998, 2005; West & Marland, 2003; Lal, 2004), (iv) influence the temperature in the lower atmosphere through effects on latent heat and albedo (Luyssaert *et al.*, 2014) and (v) preserve the soil resource for the provision of food and buffering against other global environmental threats (Mcbratney *et al.*, 2012, 2014). Agricultural management has therefore become a central issue to improve soil functionality and sequester atmospheric carbon, and a detailed quantification of its effect on SOC stocks is crucial towards predicting their impact on the global carbon cycle (Quinton *et al.*, 2010). Conservation agriculture practices consisting of changes in tillage intensity and depth, crop rotations and the use of cover crops are presented as a way to improve soil quality and increase SOC stocks (FAO, 2006). A large number of field studies have assessed the potential of these measures to sequester carbon (e.g. Ogle *et al.*, 2005; D'haene *et al.*, 2009), but the large uncertainties associated to the reported results (Baker *et al.*, 2007), and the fact that these studies have been mostly limited to the plot and field scales, question the suitability of deriving the impact of these measures at the regional scale as has been performed in the past (Smith *et al.*, 1998; Sperow *et al.*, 2003).

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Over the last two decades, SOC models (e.g. CENTURY, Roth-C, DNDC, CESAR, etc.) have been applied from small regional studies (Dendoncker *et al.*, 2008; Álvaro-Fuentes *et al.*, 2012), to national (Sleutel *et al.*, 2006; Ogle *et al.*, 2010; Van Wesemael *et al.*, 2010), European (Vleeshouwers & Verhagen, 2002; Lugato *et al.*, 2014) and the global scale (Jones *et al.*, 2005) to assess the impact of management practices on SOC stocks. These studies were a step forward in our ability to quantify changes in SOC stocks due to land management and are useful tools to evaluate the impact of agricultural policies, such as the impact of the 'greening the Common Agricultural Policy' by the European Comission (2013). Nevertheless, these studies are still limited as they (i) are restricted to the plough layer and (ii) consider the landscape as a series of nonconnected units defined by similar land use, soil type or even based on administrative borders. As a consequence, landscapes have been considered as a union of components with no flux transfers between them limiting their representation of landscape processes (e.g. SOC redistribution through soil erosion).

This static view of the landscape is challenged by recent research that showed that soil erosion and the associated redistribution and export of soil organic carbon, particularly on agricultural land, constitute important controls on changes in soil carbon storage (Sanderman & Chappell, 2013) as well as the overall landscape carbon (C) budget (Trumbore & Czimczik, 2008; Quinton *et al.*, 2010). These controls are related to the fact that soil redistribution (i) affects the spatial and vertical, that is with depth, variability of both SOC quality and storage (e.g. Gregorich *et al.*, 1998; Van Oost *et al.*, 2005; Berhe *et al.*, 2008; De Gryze *et al.*, 2008; Doetterl *et al.*, 2012), (ii) modifies the exchange rate of CO_2 between soil and atmosphere (Stallard, 1998) and (iii) can deliver considerable amounts of particulate organic matter and nutrients to water courses (Lal, 1995; Sanderman & Chappell, 2013). Although these studies have increased our understanding of the interactions between erosion processes and carbon cycling, an evaluation of the significance of soil erosion for the SOC budget at coarse scales is still lacking.

One of the positive impacts of conservation agriculture is the reduction of soil erosion rates due to a decrease in the tillage intensity and an increase in residue cover that reduces run-off (Montgomery, 2007). In addition, changes in soil erosion rates will have implications for C sequestration. Although soil erosion is considered to be one of the causes of SOC stock decrease in croplands, the SOC-depleted topsoil also presents an increased potential to sequester additional CO_2 (i.e. dynamic replacement, Harden *et al.*, 1999; Van Oost *et al.*, 2007). To date, the combined effects of soil erosion-induced changes in SOC storage and C sequestration in response to conservation practices remain mostly unassessed and should be tackled by integrating geomorphic and biological processes together with environmental conditions (Kirkels *et al.*, 2014). Assessing the impact of soil redistribution on SOC stocks under different management practices over long (i.e. years to decades) temporal and large spatial scales will require the application of spatially distributed models simulating the interaction between SOC dynamics and soil redistribution, the application of which has up to now been limited to small-sized watersheds without taking the fate of the buried C in depositional profiles or SOC cycling throughout the soil profile into account (Van Oost *et al.*, 2005; Yadav & Malanson, 2009; Yadav *et al.*, 2009; Dlugoß *et al.*, 2012; Nadeu *et al.*, 2014).

With this study, we aim to quantify the contribution of soil erosion to the regional scale SOC budget and its impact both on the C sequestration potential of agricultural management practices and on the lateral export of C that is at risk of being mineralized. We present a modified version of a coupled erosion and SOC turnover model that allows to assess the interactions between SOC dynamics and soil erosion in response to management changes at the regional scale, and use it (i) to quantify the importance of erosion-induced lateral carbon fluxes on SOC stocks under a selection of management practices and (ii) to describe and assess the impact of lateral fluxes on the spatial and vertical SOC distribution. Finally, we discuss the implications for C sequestration.

Materials and methods

To assess the significance of erosion-induced lateral SOC fluxes for regional SOC budgets in relation to soil management, we applied a spatially and depth-explicit coupled soil erosion and SOC turnover model for a selection of relevant management practices. Two different approaches were compared: a commonly used approach that considers no lateral SOC fluxes in the landscape ('conventional' approach) and a new approach which takes into account lateral SOC fluxes (export as the difference between erosion and deposition within fields) resulting from soil erosion processes ('lateral flux' approach) (Fig. 1).

Study site

The study was performed in a 250-km^2 area in central Belgium that comprises three distinct agricultural regions with differences in climate and soil characteristics: the sandy loam region, the silt loam region and the stony silt loam Condroz region (Fig. 2). Cropland extends over more than half of the area and the terrain is characterized by gentle slopes in the north (<3% on average) that become steeper in the south

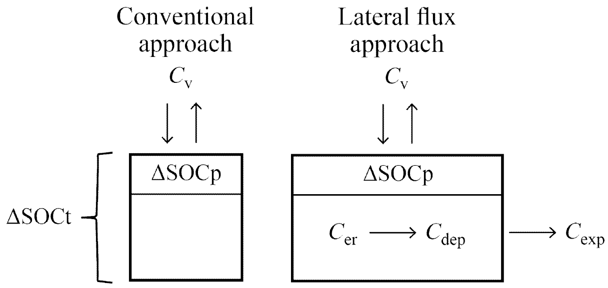

Fig. 1 Representation of the two approaches used in this study. The conventional approach accounts for SOC changes resulting from the balance between vertical inputs (crop and manure) and outputs (mineralization). The lateral flux approach includes the former and accounts also for SOC redistribution through tillage and water erosion resulting in three additional lateral fluxes: eroded carbon, deposited carbon and exported carbon.

(Condroz region). Mean annual precipitation in the area is 850 mm, and the average annual temperature oscillates between 9.4 and 9.8 °C between regions (Meersmans *et al.*, 2011). Previous studies have shown the importance of soil redistribution on the spatial and vertical variability of SOC concentration and quality in eroding cropland hillslopes (Doetterl *et al.*, 2012; Wiaux *et al.*, 2014).

Modelling approach and implementation at the regional scale

To obtain an improved simulation of SOC dynamics and erosion interactions in response to management at the regional scale, we modified the existing SPEROS-C model (Van Oost *et al.*, 2005). This model combines a soil carbon dynamics model, the Introductory Carbon Balance Model (ICBM, Andrén & Kätterer, 1997) and the spatially distributed soil erosion model SPEROS (Van Oost *et al.*, 2003) (Fig. 3). It has been successfully used to simulate erosion-induced carbon fluxes at the field and microcatchment scale (Van Oost *et al.*, 2005; Dlugoß *et al.*, 2012). Here, we describe the basic features of the model and the modifications made to improve the representation of soil management and to adapt it for a regional-scale application.

SOC redistribution and profile evolution. The sediment and SOC erosion component of the model simulates soil detachment, transport, deposition and export by water and tillage erosion processes. To facilitate the implementation of the erosion model at the regional scale, we use the Revised Soil Loss Equation (Renard *et al.*, 1997) to estimate the potential water-induced soil detachment (E_{pot}; kg m^{-2} yr^{-1}) (Equation 1):

$$E_{pot} = R * K * L * S * C * P \tag{1}$$

where R is the rainfall erosivity (MJ mm ha^{-1} h^{-1} yr^{-1}), K soil erodibility (kg h MJ^{-1} mm^{-1}), LS slope steep and length (−), and C and P (−) are the cover management and support practices. The local detachment rate is considered to equal the potential rate unless the local transport capacity is exceeded. The local transport capacity (T_c; kg m^{-1} yr^{-1}) is estimated as (Equation 2):

$$T_c = k_{tc} * E_{pot} \tag{2}$$

where k_{tc} is the transport capacity coefficient (m). If the sediment inflow exceeds the local transport capacity, the amount of material transported equals the transport capacity while the remainder is deposited. The simulation of tillage-induced soil erosion and deposition is based on a diffusion type equation (Van Oost *et al.*, 2000). The detachment of SOC by erosion (C_{ero}, kg m^{-2} yr^{-1}) is then estimated as (Equation 3):

$$C_{ero} = SOC_{SL} * ER * E_{net} * BD * D_{SL^{-1}} \tag{3}$$

where SOC_{SL} is the SOC content of the surface layer (kg m^{-2}), ER is an enrichment factor (−), E_{net} is the local detachment rate (kg m^{-2} yr^{-1}), BD is the soil bulk density (kg m^{-3}) and D_{SL} is the depth of the surface layer (m).

A key feature of the SPEROS-C model is the three-dimensional representation of the soil landscape where the model keeps track of the evolution of the SOC profile in response to

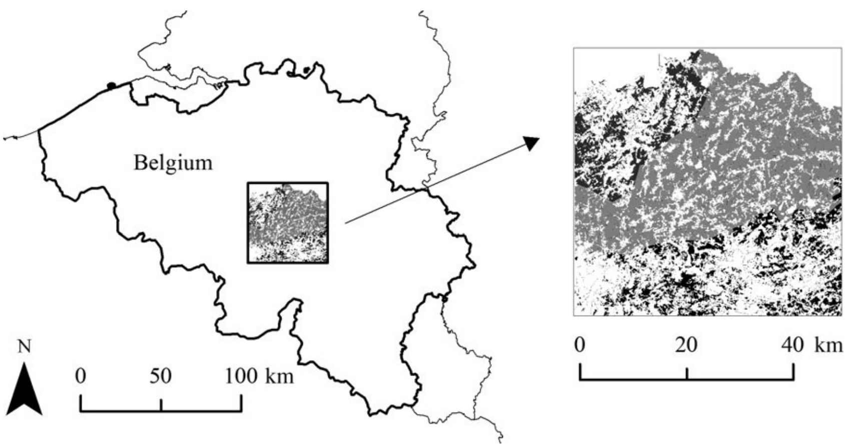

Fig. 2 Location of the study area in Belgium (left) and cropland extension within the study area in three agricultural regions: sandy loam (top left), silt loam (middle) and stony silt loam (bottom).

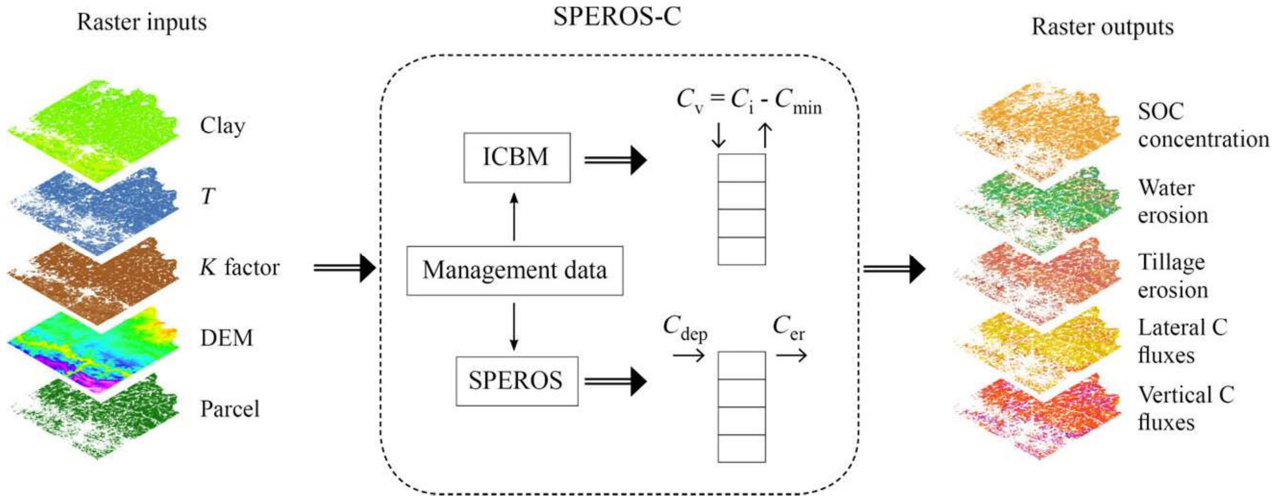

Fig. 3 Schematic representation of the SPEROS-C model, the raster inputs and the resulting spatial outputs.

erosion and deposition. The spatial variability of SOC is represented by a uniform grid of cells (20 × 20 m) while multiple soil layers of constant thickness characterize the vertical SOC profile for each cell in the grid. For this study, we represent the 1-m soil profile using four layers of 0.25 m. Given that the thickness of each soil layer is kept constant during the simulations, and assuming that the bulk density remains constant, a fraction of SOC from the first subsoil layer is transferred to the surface layer, in proportion to the erosion height. The same procedure is applied for the other subsoil layers while a constant SOC content boundary condition at 1-m depth is used for eroding profiles. Using a routing algorithm (Van Oost *et al.*, 2000) and the erosion equations described above (Equations 1–3), eroded sediment and SOC are redistributed over the landscape. The location and rate of deposition are then used to simulate the accumulation of SOC in a depositional profile whereby the vertical SOC transfer is proportional to the deposition height. The depth of a depositional profile is dynamic and equals the sum of 1 m (i.e. the initial soil profile depth) and the cumulative deposition height.

Soil carbon. The SOC dynamics component of the model is represented by the ICBM model. ICBM describes SOC dynamics using two SOC pools with a different turnover time, the young (*Y*) and the old (*O*) pool. These pools are regulated through two differential equations (Equations 4 and 5) that consider four carbon (C) fluxes: C input to soil (crop residues, roots, rhizodeposition and manure), humification from the young to the old pool and mineralization from both pools. These fluxes are modified by temperature and clay content. Turnover rates for the young and old pool decrease exponentially with depth, and an exponential root density function is used to represent C input into the soil profile. Net vertical carbon fluxes (C_v) result from the difference between C input to soil and mineralized young and old C (Equations 4 and 5).

$$\frac{dY}{dt} = i - k_y * r * Y \tag{4}$$

$$\frac{dO}{dt} = Y * h * k_y - k_o * r * O \tag{5}$$

where i is the input from crops (i_c) and manure (i_m) (kg m^{-2} yr^{-2}), h is the humification coefficient, r is a climate effect coefficient, and k_y and k_o are turnover rates for Y and O SOC, pools respectively (yr^{-1}). The humification coefficient depends on soil clay content and carbon input quality (Kätterer & Andrén, 1999) (Equation 6), and the climate effect coefficient, r, is temperature dependent through a correction factor r_T (Equation 7):

$$h = \frac{(i_c * h_c + i_m * h_m)}{i} \exp^{0.0112^{(cl-36.5)}} \tag{6}$$

$$r_T = 2.07^{\frac{(T-5.4)}{10}} \tag{7}$$

Carbon inputs from crop residues and roots are derived from the crop-specific aboveground dry biomass (AGBM), which is calculated as the crop yield divided by the crop harvest index (HI):

$$\mathrm{AGBM} = \frac{\mathrm{Yield}}{\mathrm{HI}} \tag{8}$$

Assuming that 45% of AGBM consists of carbon, the above and belowground carbon inputs are calculated as:

$$i_c = 0.45\,[(\mathrm{Res} * \mathrm{AGBM}) + (p * \frac{R}{S} * \mathrm{AGBM})] \tag{9}$$

where i_c stands for C input from crops for a specific depth layer, *Res* is the fraction of AGBM that is left as residue in the field, *R/S* the root to shoot ratio of a specific crop and *p* is the fraction of C input from roots received for the specific depth layer. Turnover times decrease exponentially with depth (z (m)) (Rosenbloom *et al.*, 2001):

$$k_{tz} = k_{t0} \exp(u * z) \tag{10}$$

where k_{tz} and k_{t0} are turnover rates at depth z and 0, respectively, and u is a dimensionless exponent that needs to be calibrated through an inverse modelling approach. An exponential root density profile is used to model C input into the soil profile (Equation 11), while manure input is distributed in the plough layer only.

$$\Phi(z) = \begin{cases} 1 & z \leq z_r \\ \exp(-c(z - z_r)), & z > z_r \end{cases} \quad (11)$$

where z_r is a reference depth, set to 0.25 m, and c is a constant factor.

Model implementation. To implement the model at the regional scale, we introduced changes to account for spatial patterns and temporal input variability due to crop rotations and management. The study area was classified into three land use classes: cropland, stream network and other land use. Tillage erosion and water erosion redistributed soil and carbon between cropland cells, while water erosion fluxes could export a fraction of the mobilized sediment and carbon out of croplands and into other land use classes or the river system (Fig. 4). Therefore, the stream network and 'other land use' classes were only receptors of sediment and carbon from cropland. SOC dynamics were only modelled on cropland while the models kept track of the exported sediment and carbon.

Scenario description

Studies assessing the effect of management on SOC stocks often compare scenarios that combine a varying amount and type of C inputs to soil (e.g. farmyard manure, straw, compost) and different degrees of soil physical disturbance (e.g. tillage depth and intensities).

In this study, we used two approaches to assess the differences in SOC stocks and SOC fluxes between three management practices. This resulted in six different scenarios. The first approach, termed 'conventional', consisted in a spatial application of the SOC dynamics module in SPEROS-C (the ICBM model) over the study area, while excluding lateral fluxes. For the second approach, termed 'lateral flux', both the soil erosion and the SOC dynamics module of the SPEROS-C model were used. The three management scenarios consisted in the following:

conventional tillage with 10% residue return to the soil (CT), reduced tillage with 10% residue return to the soil (RT) and reduced tillage with 50% residue return to the soil (RT+i). Each of these scenarios was run with the 'conventional' approach and the 'lateral flux' approach on a 2-years winter wheat–maize crop rotation, which is typical for the study area.

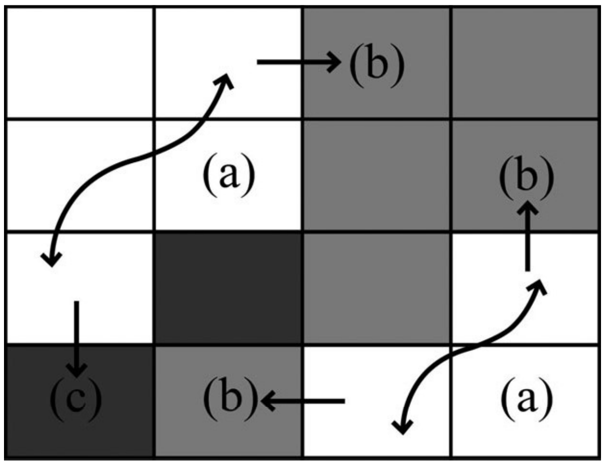

Fig. 4 Simulated C fluxes between arable fields (white), other land uses (light grey) and the fluvial system (dark grey) for the scenarios using the lateral flux approach. Represented: (a) soil redistribution fluxes within fields by tillage and water erosion: (b) SOC export fluxes to other land uses and (c) to the fluvial system.

Input data and model parameterization

Soil erosion parameters. A digital elevation model with a 20-m resolution was used (Goidts *et al.*, 2009). A parcel raster was created classifying pixels according to three land use classes: cropland, river system and other land use, based on a land use map by Meersmans *et al.* (2011). River pixels were defined from a run-off raster as those pixels with a contributing upland area of at least 0.4 km^2. The transport coefficients for tillage were set to deliver tillage erosion rates consistent with those measured in the study area (Van Oost *et al.*, 2009), obtaining average tillage erosion rates of 3.8 and 1.9 Mg ha^{-1} yr^{-1} for CT and RT, respectively. A typical plough layer depth of 0.25 m was used (Goidts *et al.*, 2009) for both CT and RT, and overall, four depth layers of 0.25 m each were modelled. The sediment transport coefficient (k_{tc}) for water erosion was calibrated at 55 m for CT and RT, resulting in an erosion rate of 5.2 Mg ha^{-1} yr^{-1} that matched average erosion rates for the study area (Van Oost *et al.*, 2009), while the k_{tc} was reduced to 25 m for the RT+i scenario to account for increased soil residue cover. A temporally and spatially constant rainfall erosivity factor (R factor) of 0.087 MJ mm m^{-2} h^{-1} yr^{-1}, corresponding to the annual rainfall, was applied throughout the study area after De Moor & Verstraeten (2008) while the soil erodibility (*K* factor) map was taken from Meersmans *et al.* (2011). RUSLE's crop-specific cover factors were taken from Verstraeten *et al.* (2006) (Table 1) and were reduced by 75% to account for increased residue cover in the RT+i scenarios. These changes lead to a decrease of 40% in the soil water erosion rate for the RT+i scenario relative to CT, which is in agreement with results from other studies under temperate climate (Shipitalo & Edwards, 1998; Alliaume *et al.*, 2014). In this study, we assumed no selectivity in the removal or transport of SOC and set the value for *ER* (Equation 3) to unity. Based on a 4-year monitoring programme, Wang *et al.* (2010) show that there is only a slight enrichment (ER = 1.3), indicating that most erosion occurs under the form of aggregated soil material and not individual soil particles. In our uncertainty assessment, we employ a very large range of soil erosion rates (range between 2.6 and 10.1 Mg ha^{-1} yr^{-1}, Table 4). This range is much larger than the potential contribution of enrichment.

SOC dynamics. Using the default ICBM settings, SOC turnover rates in topsoil were set to 0.8 yr^{-1} for young SOC and

Table 1 Crop input parameters

	Yield (kg m^{-2})*	Harvest index *'†	Root: Shoot†'‡	C factor §	*AGBM*¶ (kg m^{-2})
Winter wheat	0.78	0.60	0.41	0.28	1.30
Maize	0.99	0.50	0.17	0.45	1.98

References: *Gobin (2010), †Bolinder *et al.* (1999), ‡Prince *et al.* (2001), §Verstraeten *et al.* (2006) ¶see Equation 8 in this study.

0.006 yr^{-1} for old SOC, while humification coefficients were set to 0.125 yr^{-1} for crop residues and 0.31 yr^{-1} for manure (Kätterer & Andrén, 1999). No differences in the turnover rates for SOC under CT and RT were made; this reflects the absence of a clear effect of tillage practices on SOC dynamics in the study area (Angers & Eriksen-Hamel, 2008; D'haene *et al.*, 2009). Current mean annual air temperature for the three agricultural regions comprised within the study area as well as the spatial variability of the clay content was derived from Meersmans *et al.* (2011). Crop input data were taken from literature (listed in Table 1). The percentage of residue left on the field was 10% (typical of the area, 20% in the case of maize) which translated into an average input of 2.5 Mg C ha^{-1} for the 2-yr winter wheat and maize rotation.

The model parameters related to SOC turnover at depth as well as the amount of C entering the surface layer via manure is not well constrained in literature. To this end, the coefficients regulating SOC decomposition in depth as well as manure inputs (unknown for the study area) were parameterized using inversed modelling (Dlugoß *et al.*, 2012) for the conventional approach CT scenario by varying SOC coefficients until the model reproduced the observed layer-specific SOC concentrations of reference profiles in the region not affected by soil redistribution (Doetterl, 2013, see Model evaluation section). The inverse modelling resulted in the following: (i) an annual C input through manure of 0.06 kg m^{-2} and (ii) 45% of the root dry matter in the first metre allocated in the plough layer. The optimal calibrated SOC (%) profile had an RSME of 0.02%.

Model implementation and output analyses

A model spin-up was performed using the CT scenario over a period of 250 years, without soil erosion for the conventional approach and including soil erosion for the lateral flux approach. After spin-up, the three management practices, CT, RT and RT+i, were run over a period of 100 years, which is the time it takes SOC changes to reach a new equilibrium (Foereid & Høgh-Jensen, 2004).

We evaluated the model output in terms of (i) changes in SOC inventories for the 1-m soil profile (*dSOCt*), (ii) changes in SOC inventories for the plough layer (*dSOCp*), (iii) net soil-atmosphere exchange (C_v), which is defined by the difference between local C input ($i_c + i_m$) and SOC mineralization, whereby positive values indicate a net uptake into soils and negative value a net release from soils to the atmosphere, and (iv) carbon export from croplands (C_{exp}). Note that C_v is calculated as $dY/dt + dO/dt$ (see Equations 4 and 5) and can be interpreted as the sequestration/release of atmospheric CO2 as it represents the vertical soil-atmosphere C flux, while *dSOC* represents the effect of both sequestration/release of atmospheric CO_2 and lateral C fluxes (i.e. SOC erosion losses or additional input due to SOC deposition).

Model evaluation

The model performance was evaluated by comparing modelled SOC stocks for the four layers at eroding and depositional sites against a set of observational eroding and depositional SOC profiles located within the study area. These observational profiles were randomly sampled from agricultural fields in the Belgian loam belt down to 1 m depth using a closed tube sampling system and classified according to their topographical position (Doetterl, 2013). To establish a comparison with our model output data, we used 32 eroding and 32 depositional profiles and grouped the original 5 cm resolution samples in four depth intervals (0–25, 25–50, 50–75 and 75–100 cm) assigning a SOC value for each depth interval that represented the average of all grouped samples. In addition, we tested boundaries to the model results by modifying two key parameters: one in the soil erosion component (k_{tc} factor) and the other relative to SOC dynamics (changes in the proportion of Y and O pools). For the k_{tc} factor, the values were chosen based on reported sediment delivery ratios (SDR) and erosion rates in the loam belt. A k_{tc} of 25 m was chosen as the lower boundary and a k_{tc} of 250 m as the upper. This gave a range of SDR values from 5% to 15% and erosion rates of 2.5–10 Mg ha^{-1} yr^{-1}. For SOC dynamics, default ICBM parameters give a 5% of Y carbon in topsoil. Changes in the proportion between Y and O carbon in topsoil will be the result of changing C inputs and k_y and k_o. Therefore, ICBM was recalibrated for these parameters to obtain a 2.5% and 7.5% of Y carbon in the total carbon pool representing the low and the high boundaries, respectively, of our model evaluation. These four values were initially combined into four scenarios and run for CT with the 'lateral' approach. The resulting combinations yielding the lower (CT low) and higher (CT high) values for C_v and C_{exp} (2.5% of Y and k_{tc} of 25 and 7.5% of Y with a k_{tc} of 250) were considered to be the low and high thresholds for our result uncertainty.

Results

SOC stocks

SOC stocks ranged between 30 and 44 Mg ha^{-1} yr^{-1} in the plough layer and 63 and 74 Mg ha^{-1} yr^{-1} for the entire soil profile at the end of the simulation (Table 2). SOC concentration in the plough layer (0–0.25 m) was in all cases higher for the RT+i scenario than for RT and

CT while scenarios including lateral fluxes had systematically lower SOC stocks than those following the conventional approach (Table 2).

Spatial and vertical distribution of SOC

The coefficients of variation (CV) for SOC concentration in the plough layer ranged from 4% to 12% for the whole study area. The lowest values corresponded to the conventional approach while the CV was more than doubled when including lateral SOC fluxes in the simulations (Table 2) resulting in a higher SOC spatial variability linked to soil redistribution patterns (Fig. 5). In addition, including lateral fluxes led to a higher proportion of the SOC allocated in the subsoil (below 0.25 m), as demonstrated by a lower stratification ratio between topsoil SOC and subsoil SOC in Table 2. The difference in the stratification ratio between the two modelled approaches ranged between 3% and 5%, for all management practices.

SOC stock changes and C fluxes

Lateral flux simulations showed an increase in *SOCt* stocks in cropland between 0.2 and 0.4 g C m^{-2} yr^{-1} relative to scenarios using the conventional approach (Table 3). This represented an average annual increase of 0.01% SOC for the 100-year simulations (Table 3, Fig. 6). Changes in *SOCp* for the lateral approach differed between scenarios showing a SOC loss for CT, no change for RT and SOC accumulation in the RT+i scenario. Simulated C sequestration rates (C_v) were higher in all scenarios when lateral fluxes were included than those using the conventional approach (Table 3). Lateral C exports from croplands were in the same order of magnitude as the C sequestration (2.5–2.7 g m^{-2} yr^{-1}) for CT and RT scenarios, but represented only 10% of sequestered SOC for RT+i (Table 3). The majority of the SOC exported beyond cropland field borders remained in other land uses classes while the SOC delivery ratio to the river network was 7% for the CT and RT scenarios and 6% for the RT+i scenario.

Model evaluation

We tested the ability of the model to reproduce SOC profiles at eroding and depositional areas. For eroding profiles, the model slightly underestimated SOC concentration for the topsoil and lower half of the profile (0–0.25 m, 0.50–1 m) while it overestimated SOC concentration in the 0.25–0.50 m layer (Fig. 7). The overall fit for eroding profile was better than that for the depositional profile, with RMSE's of 0.13% for the eroding and 0.17% for the depositional profile (Table 4). In the case of the latter, the model underestimated topsoil SOC concentration and overestimated SOC concentrations for the rest of the profile layers. The fit of the depositional profile improved for the CT low scenario that presented a higher SOC concentration in the plough layer (Table 4).

Discussion

Significance of lateral SOC fluxes

Including lateral fluxes due to soil redistribution processes into the simulations led to a net lateral export of 1.2–2.3 g C m^{-2} yr^{-1} from cropland over the 100-year study period (Table 3). It is important to note that despite the lateral SOC export, cropland represented a net carbon sink with net C sequestration rates (C_v) in the order of 1.5–2.7 g C m^{-2} yr^{-1}. These apparently conflicting findings result from the fact that SOC removal by erosion induces disequilibrium between SOC content and C input. This disequilibrium leads to a carbon sink as long as continued production supplies new photosynthetically derived C to soils. In other words, as Stallard (1998) and Harden *et al.* (1999)

Table 2 Average SOC output values for the six scenarios at the end of the 100-year simulation period

	*SOCt** (Mg ha^{-1})	*SOCp*† (Mg ha^{-1})	SOC stratification‡	CV *SOCp*§ (%)
CT conv	65.1	34.9	1.2	3.5
CT lat	63.1	30.8	1.0	11.7
RT conv	65.1	35.0	1.2	3.5
RT lat	63.2	31.2	1.0	10.7
RT+i conv	74.8	44.6	1.5	3.5
RT+i lat	73.5	41.5	1.3	7.6

*Average SOC stocks for the full profile.
†Average SOC concentration in the plough layer.
‡Stratification ratio between SOC in the plough layer and in the subsoil.
§Coefficient of variation of SOC concentration in the plough layer for the study area.

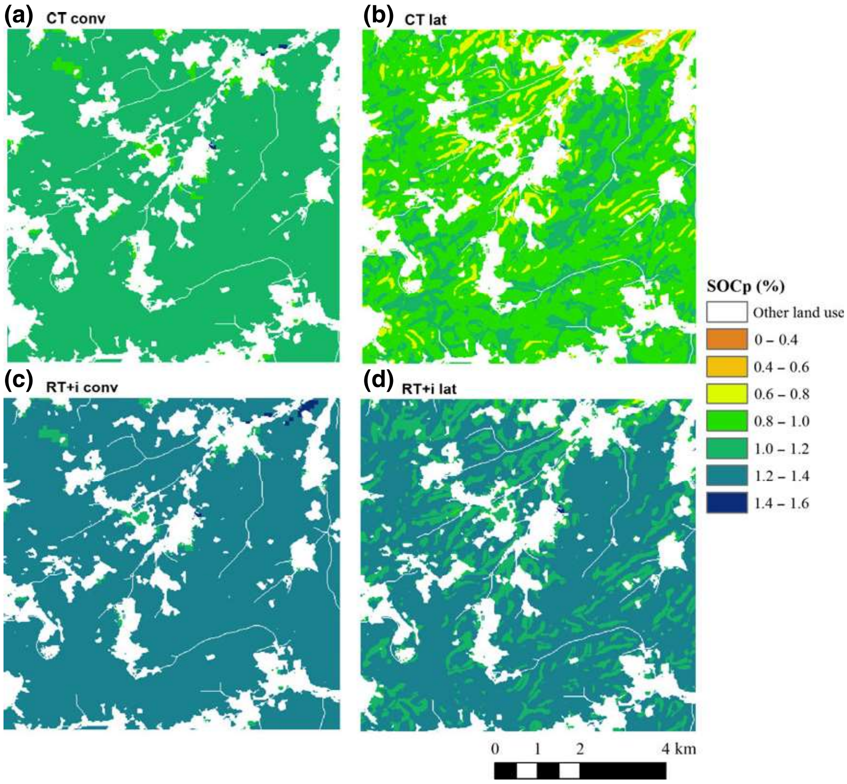

Fig. 5 SOC in the plough layer (*SOCp*) (%) for the following: (a) CT conv, (b) CT lat, (c) RT+i conv and (d) RT+i lat in a subset of the study area covering about 40 km^2. Areas with higher *SOCp* for (a) and (c) correspond to areas with higher clay content. In the case of (b) and (d), *SOCp* patterns are dominated by soil redistribution.

Table 3 Average annual changes in SOCt, SOCp and annual net C_v and C_{exp} from cropland area for each management scenario using the conventional approach and the lateral flux approach. All values correspond to an average of a 100-year simulation

	Conventional approach (g m^{-2} yr^{-1})			Lateral flux approach (g m^{-2} yr^{-1})			
Scenario	*dSOCt*	*dSOCp*	C_v*	*dSOCt*	*dSOCp*	C_v*	C_{exp}†
CT	0	0	0	+0.4	−0.3	+2.7	2.3
RT	0	0	0	+0.2	+0.0	+2.5	2.3
RT+i	+9.7	+9.7	+9.7	+10.0	+9.7	+11.2	1.2

*Net vertical flux; positive values indicate net C sequestration and negative values a net C loss.
†C export from cropland by water erosion. Note that vertical fluxes apply to cropland only and not to exported C.

proposed, eroded carbon is dynamically replaced. The replacement of eroded carbon simulated by our model is fully consistent with conceptual (Stallard, 1998; Berhe *et al.*, 2007), empirical (e.g. Quine & Van Oost, 2007; Van Oost *et al.*, 2007) and modelling (e.g. Manies *et al.*, 2001; Liu *et al.*, 2003) studies. The SPEROS-C model represents this process in a mechanistic manner by adding an additional loss term for the SOC content of the surface layer. Given continued C inputs from plant material, a fraction of this additional loss will be replenished. The continuous replacement of eroded SOC resulted in higher C sequestration rates for CT and RT+i relative to scenarios that did not include lateral fluxes (Table 3), although *SOCp* remained lower for those including lateral fluxes (Table 2).

Our model simulations clearly indicate that C sequestration and SOC change can be very different (with differences as large as 3 g C m^{-2} yr^{-1} for the plough layer) when lateral fluxes are accounted for. C sequestration rates do not equal the predicted or observed change in the SOC inventory, as the overall C budget is also controlled by the exported SOC, that is *dSOCt* = C_v-C_{exp}. This implies that using differences in SOC stocks as a measure for net C sequestration of

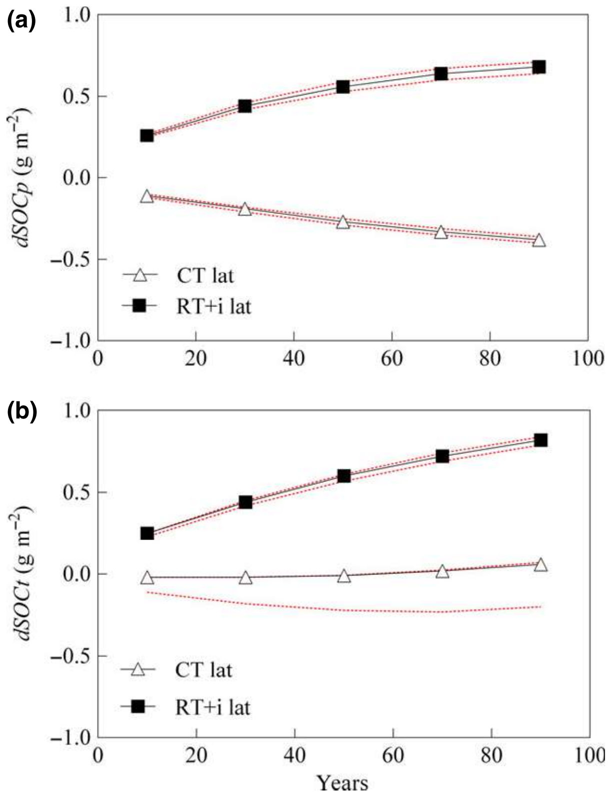

Fig. 6 Temporal evolution of SOC stocks changes in g C m^{-2} yr^{-1} for CT lat and RT+i lat in a) the plough layer (*dSOCp*) and b) the full profile (*dSOCt*). Dashed lines represent the maximum and minimum boundaries obtained with the 'low' and 'high' scenarios.

specific management practices results in an underestimation of potential sequestration rates for eroding landscapes. Furthermore, observations of declining SOC stocks in the plough layer could, without proper understanding of lateral C fluxes, lead to the erroneous interpretation that the landscape under consideration was acting as a carbon source. This is illustrated by the results of the CT scenario, where the *SOCp* shows a decline of 0.3 g C m^{-2} yr^{-1} but where the cropland soils are representing a small, but net sink for atmospheric carbon (i.e. C_v of 2.7 g C m^{-2} yr^{-1}) (Table 3) on average over the simulated period. In a modelling study in an agricultural watershed in Illinois, USA, Yadav *et al.* (2009) estimated that soil erosion could reduce potential cropland C sequestration rates by as much as 0.6 Mg C ha^{-1} annually when accounting for lateral erosion and depositional fluxes. However, the loss rate may have been overestimated as they did not include SOC replacement in their simulation.

An important observation from this study is that the exported SOC by water erosion from the cropland field (C_{exp}) is an important component of the overall C budget. The model estimates that c. 1.2–2.3 g m^{-2} yr^{-1} of SOC was on average exported from cropland. As the export of SOC is in the same order of magnitude as the net C sequestration (Table 3), the fate of this exported SOC will play a key role in the overall C budget at the regional scale. In fact, taking results for CT (Table 3), a 20% mineralization rate for exported C (Polyakov & Lal, 2004) would reduce C sequestration rates by 17%,

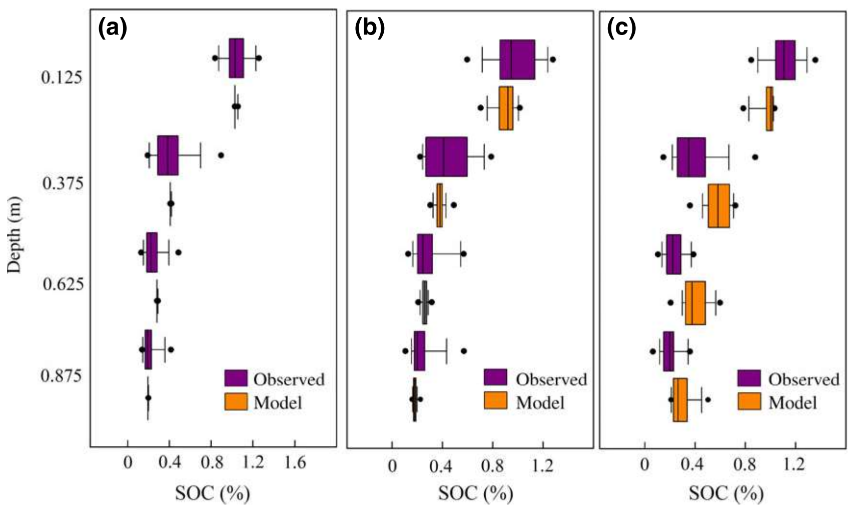

Fig. 7 SOC concentration in the four modelled layers. Comparison between observations (observed) and simulations (model) for the following: (a) noneroding observed profiles (N = 75) vs. CT conv scenario (N = 75) (used for calibration); (b) eroding observed profiles (N = 32) vs. random eroding profiles from CT lat (N = 120); and (c) depositional observed profiles (N = 32) vs. random depositional profiles (N = 48) from the CT lat scenario. Upper and lower ends of the boxplot respresent the 25th and 75th percentiles.

Table 4 Results from the sensitivity analysis for the 'low' and 'high' scenarios. The RMSE resulting from the comparison of SOC concentration at different depths for eroding and depositional profiles is also reported

Scenario	Fraction Y* (%)	k_{tc}† (m)	Mean erosion rate (Mg ha^{-1} yr^{-1})	*SOCp*‡ (%)	RMSE erosion§ (%)	RMSE deposition¶ (%)	C_v (g m^{-2} yr^{-1})	C_{exp} (g m^{-2} yr^{-1})
CT lat	5%	55	5.2	0.91	0.13	0.17	2.7	2.3
CT low	2.5%	25	2.6	0.95	0.09	0.13	1.6	1.2
CT high	7.5%	250	10.1	0.86	0.16	0.34	5.1	4.7

*Fraction of young carbon in the SOC pool.
†k_{tc} value (see Equation X).
‡SOC concentration in the plough layer at the end of the simulation period.
§RMSE (in % of SOC) resulting from the comparison between observational and modelled erosion profiles.
¶RMSE (in % of SOC) resulting from the comparison between observational and modelled deposition profiles.

while a mineralization of 50% of the exported C would reduce it by 42%. On the contrary, effective burial of exported C would contribute to increasing C sequestration rates in eroding landscapes by the same magnitudes. These values point to the need to (i) include SOC export by erosion processes in assessments of the C sequestration potential of management options, (ii) improve our capabilities to assess the sink/source behaviour of eroded SOC beyond the cropland boundaries and (iii), as a precaution, recommend management practices based on maximizing C_v and minimizing C_{exp}, as the latter could be mineralized and would turn a limited sequestration into an overall emission.

SOC changes and profile depth

The inclusion of lateral fluxes had a different impact on *SOCp* and *SOCt*. In the case of the CT lat scenario, average *SOCp* was reduced by 0.01% annually while *SOCt* increased annually by the same amount (Table 3, Fig. 6). On the contrary, no differences between *SOCp* and *SOCt* were found for CT in the conventional approach (Table 3). These results indicate that changes in SOC stocks based on the plough layer only can overestimate SOC losses in eroding landscapes. In addition, the results also show that *SOCp* was 25% lower for CT lat than RT+i lat and yet the difference between both scenarios was reduced to 12% for *SOCt*. Differences in the SOC stock distribution in depth can be further observed by comparing stratification ratios between the different scenarios (Table 2) and are consistent with studies showing how the higher SOC stocks accumulating in the subsoil under conventional tillage could balance part of the effects of increased topsoil SOC concentration under conservation tillage practices (Yang & Wander, 1999; Baker *et al.*, 2007; Angers & Eriksen-Hamel, 2008). Although these observations stress the importance of sampling below the plough layer in areas where soil redistribution takes place, changes in the plough layer are still at the base of studies deriving national estimates of potential C sequestration in soils due to management practices (Smith *et al.*, 1998; Sperow *et al.*, 2003).

SOC stock variability

In the previous section, changes in cropland SOC stocks have been expressed in terms of an average rate per unit area for a certain depth, as is commonly performed (Sleutel *et al.*, 2006; Dendoncker *et al.*, 2008). Yet, the vertical and spatial variability of SOC can be crucial for the assessment of SOC stock changes (Meersmans *et al.*, 2009; Yadav *et al.*, 2009). By incorporating the impact of soil redistribution on SOC stocks, we observed that accounting for lateral fluxes reduced SOC stocks, increased their spatial heterogeneity and lead to an increase in the average proportion of SOC allocated in the subsoil relative to the topsoil (Table 2). This was due to a reduced SOC concentration in the plough layer at eroding sites and an increase in subsoil SOC at depositional sites, as has been observed in field studies along soil toposequences (Berhe *et al.*, 2008; Doetterl *et al.*, 2012; Wiaux *et al.*, 2014). These changes in the spatial configuration of SOC stocks can complicate the assessment of SOC changes through management for carbon credit programs, given the difficulty to discriminate between management and erosion-induced SOC changes (Yadav *et al.*, 2009).

Model performance and limitations

The estimates of C sequestration, changes in SOC inventories, SOC erosion and export reported here critically depend on the performance of the coupled erosion and SOC dynamics model. In terms of model outputs, the simulated erosion and export rates are consistent with existing regional assessments (Goor, 2007) and the RT+i scenario using the conventional approach resulted in a C sequestration rate that averaged

Table 5 Carbon sequestration rates in studies using similar management scenarios and a conventional modelling approach

Treatment	Value (Mg ha^{-1} yr^{-1})	Period	References
Reduced tillage and 50% straw incorporation	0.097	100 years	This study
Reduced tillage	0	100 years	This study
All cereal straw left in the field	0.04*	30 years	Lugato *et al.* (2014)
Reduced tillage	0.03–0.2*	30 years	Lugato *et al.* (2014)
50% straw left in field	0.133†	4 years	Sleutel *et al.* (2006)
Reduced tillage	0.007†	4 years	Sleutel *et al.* (2006)

*Median value for Europe.
†Data for Flanders, Belgium.

0.1 Mg ha^{-1} yr^{-1}, which is consistent with the reported C sequestration potential of similar agricultural management practices (Table 5).

We acknowledge that the ICBM model is relatively simple, and some processes involved in SOC dynamics were not accounted for in this study. A test of the model sensitivity based on changes in the most uncertain input parameters on SOC stocks and fluxes showed that changes in the distribution of SOC between the young and old pool had a small effect on SOC stocks with changes up to 1%. On the other hand, changes in a key parameter for soil redistribution that translated into an annual soil erosion rate of 10 Mg ha^{-1} (double of the original value) doubled C sequestration rates for the CT lat scenario, while reducing the erosion rate by half had the opposite effect. Nevertheless, doubling the erosion rate reduced the goodness of fit between observed and modelled SOC stocks at eroding and depositional sites, indicating that this scenario was not the most adequate for the study area.

Our model approach provided model estimates of SOC stock and profile distributions that are consistent with observations at eroding and depositional sites (Fig. 7). The model sensitivity analysis indicated that the absolute values of the predicted lateral fluxes of C (e.g. C_{exp}) as well as the net sequestration rates (C_v) are uncertain. However, the model results are sufficiently realistic and robust when used to compare the conventional C sequestration approach with an approach that explicitly accounts for lateral C fluxes. We therefore argue that our approach provides a reasonable basis to estimate the significance of erosion-induced lateral fluxes for C sequestration assessments.

Further research should aim to include the feedback between soil redistribution and soil properties. This would allow accounting for (i) spatial variability in C inputs, through changes in crop yields, litter distribution and quality, and (ii) changes in soil properties over time as a result of soil redistribution, including soil bulk density, soil temperature and moisture. This will, however, require the collection of new observations along erosional gradients.

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