Quantifying the erosion effect on current carbon budget of European agricultural soils at high spatial resolution

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Abstract

The idea of offsetting anthropogenic CO$_2$ emissions by increasing global soil organic carbon (SOC), as recently proposed by French authorities ahead of COP21 in the ‘four per mil’ initiative, is notable. However, a high uncertainty still exists on land C balance components. In particular, the role of erosion in the global C cycle is not totally disentangled, leading to disagreement whether this process induces lands to be a source or sink of CO$_2$. To investigate this issue, we coupled soil erosion into a biogeochemistry model, running at 1 km$^2$ resolution across the agricultural soils of the European Union (EU). Based on data-driven assumptions, the simulation took into account also soil deposition within grid cells and the potential C export to riverine systems, in a way to be conservative in a mass balance. We estimated that 143 of 187 Mha have C erosion rates <0.05 Mg C ha$^{-1}$ yr$^{-1}$, although some hot-spot areas showed eroded SOC >0.45 Mg C ha$^{-1}$ yr$^{-1}$. In comparison with a baseline without erosion, the model suggested an erosion-induced sink of atmospheric C consistent with previous empirical-based studies. Integrating all C fluxes for the EU agricultural soils, we estimated a net C loss or gain of $-2.28$ and $+0.79$ Tg yr$^{-1}$ of CO$_2$eq, respectively, depending on the value for the short-term enhancement of soil C mineralization due to soil disruption and displacement/transport with erosion. We concluded that erosion fluxes were in the same order of current carbon gains from improved management. Even if erosion could potentially induce a sink for atmospheric CO$_2$, strong agricultural policies are needed to prevent or reduce soil erosion, in order to maintain soil health and productivity.

Keywords: coupled erosion-SOC models, erosion, European Union, lateral C fluxes, soil organic carbon

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Introduction

Reducing uncertainties about the magnitude and direction of greenhouse gas (GHG) fluxes from the land-use sector is a key requisite for implementing more targeted GHG mitigation policies that include soil management activities (Ogle et al., 2014). Recent regional model simulations suggest that European croplands, as a whole, are currently either a small carbon (C) sink or a small source, but not a large source of CO$_2$ (Eglin et al., 2010). Moreover, empirical-based estimates of soil C stocks from long-term soil monitoring networks in the EU are often contrasting, reporting no changes (Chapman et al., 2013; Reynolds et al., 2013), small increases (Reinkeveld et al., 2009) or soil organic carbon (SOC) decreases (Goidts & van Wesemael, 2007; Capriel, 2013; Heikkinen et al., 2013). While European agricultural soil C stocks as a whole are likely to be not far from an equilibrium state, significant uncertainties still exist, in part due to the omission of important components of the overall C balance. In particular, soil erosion due to agricultural activities can result in significant lateral C fluxes (Chappell et al., 2014) which are often neglected in SOC models, as well as in measurement-based inventory approaches based on repeated soil sampling.

In addition to the transport of C from eroding sites, erosion introduces several feedbacks on SOC turnover that influence the soil C balance and hence net C flux to or from the atmosphere. Firstly, the physical removal of SOC (and the associated mineral soil) causes a depletion of topsoil carbon stocks, which may be partially compensated by increased storage of incoming fixed carbon (as plant residues), referred as dynamic replacement (Harden et al., 1999). Secondly, the SOC in the topsoil lost via erosion is generally substituted by more recalcitrant subsoil C pools, leading to soil imbalance among vertical C fluxes components (respiration and fixation). The subsequent interactions involving SOC transported in sediments, mineralization and deposition are so complex that it is still uncertain whether the
The net effects of erosion processes represent a source or a sink of CO₂ (Van Oost et al., 2007; Lal & Pimentel, 2008; Kuhn et al., 2009).

A few studies have coupled erosion/transport models and SOC models of different complexity (Van Oost et al., 2005) to disentangle all these interactions, but the scale has generally been limited to individual landscapes and watersheds or small regions (Nadeu et al., 2015). However, at large regional/continental scales, such as for the EU, the impact of soil erosion on overall SOC balance is still unknown due to the lack of high-resolution soil erosion rates as input for process-based biogeochemical models.

To estimate the magnitude of the lateral C fluxes induced by soil erosion at these large scales, we coupled a new high-resolution (100 m²) dataset of soil erosion (Panagos et al., 2015a) based on RUSLE, with the biogeochemical CENTURY model running for European agricultural soils on a grid of 1 km (Lugato et al., 2014a). Although the fate of transported SOC in sediments could not be tracked at this scale, this study gives an unprecedented spatial estimation of SOC eroded from agricultural fields and its effect on SOC balance.

Materials and methods

Overview

The methodology integrates a high-resolution erosion spatial layer based on the Revised Universal Soil Loss Equation (RUSLE) into the SOC modelling application at the EU level, based on CENTURY biogeochemical model (Fig. 1).

CENTURY is a process-based model designed to simulate carbon (C) and nitrogen (N) dynamics in natural or cultivated systems, using a monthly time step. The soil organic matter submodel includes three SOC pools, namely active, slow and passive, along with two fresh residue pools, structural and metabolic, each with a different decomposition rate. Soil temperature and moisture, soil texture and cultivation practices act as modifying factors on potential decomposition rate constants. The model also simulates the soil water balance, using a weekly time step, and a suite of simple plant growth models are included to simulate biomass carbon and nitrogen dynamics of crops, grasses and trees.

Using this model, Lugato et al. (2014a) created a consistent SOC stock baseline for the European agricultural soils (including arable, grassland and orchard land use), covering a total of about 187 Mha. The erosion map of Panagos et al. (2015a) was based on the revision and improvement of all factors of RUSLE equation, leading to a final high-resolution product of the soil loss by water erosion in Europe. Originally, these two model applications were developed independently but using common datasets including, in particular, official EU data for land use (CLC, http://www.eea.europa.eu/publications/COR0-landcover) and management in agricultural areas (EUROSAT, http://ec.europa.eu/eurostat) and soils (ESDAC, http://esdac.jrc.ec.europa.eu/). Due to the complexity of the two modelling applications, we refer to the original papers for a detailed explanation on erosion and SOC modelling; in the next section, we describe in more depth how SOC simulations were coupled with soil erosion, which was not included in the original European SOC simulations by Lugato et al. (2014a).

SOC balance and C fluxes calculation at grid cell level

To introduce the erosion component into the CENTURY simulations, we calculated the baseline (2000–2010) SOC balance...
and associated C fluxes for each of the 1.87M grid cells, considering both the eroding and depositional areas within the cell and the combined fluxes.

For the average eroding area (ER scenario), the balance is as follows:

\[ d\text{SOC} \approx C_{i\text{fix}} + C_{i\text{sub}} - C_{H} - C_{\text{erod}} - \text{DOC} \]  

(1)

where \(d\text{SOC}\) is the SOC stock change in the fixed profile 0–30 cm, \(C_{i\text{fix}}\) is the C input through remaining NPP (after C exportation by harvest, but including roots and manure), \(C_{H}\) is the heterotrophic respiration, \(C_{i\text{sub}}\) is the incoming SOC from deeper layer (below 30 cm), \(C_{\text{erod}}\) is the lateral C flux by sediments transport and DOC the C exported as dissolved organic carbon.

For the depositional areas (DEP scenario) the balance is as follows:

\[ d\text{SOC} \approx C_{i\text{fix}} + C_{\text{dep}} - C_{H} - C_{\text{bur}} - \text{DOC} \]  

(2)

where \(C_{\text{dep}}\) is the C deposited coming from eroding areas and \(C_{\text{bur}}\) is the C that is moved out (i.e. buried) of the simulated profile (0–30 cm) as a consequence of soil deposition.

However, the net changes in SOC storage do not directly equate to the net CO\(_2\) atmospheric flux (Nadeu et al., 2015) in the grid cell due to the combination of net vertical C fluxes \((C_{i\text{fix}} - C_H)\) of eroding and depositional areas and the mineralized fraction of SOC displaced and transported by erosion. Therefore, the net CO\(_2\) flux (fCO\(_2\)) in the grid cell (ERD scenario) is given by:

\[ f\text{CO}_2 = \left[ C_{i\text{fix}} - C_H \right] - C_{\text{min}} \]  

(3)

where \(\left[ C_{i\text{fix}} - C_H \right]\) is the net vertical C flux and \(C_{\text{min}}\) represents the fraction of eroded SOC mineralized during the transport (for consistency with the previous equations, positive and negative values of fCO\(_2\) indicate a net CO\(_2\) gain and loss, respectively).

Part of the C eroded was assumed to directly move out from the grid cell and be potentially transferred to the riverine system (C\(_{\text{out}}\)) not generating any CO\(_2\) flux, as subsequent fluxes from this exported C were considered outside of our system boundary.

**Model implementation and assumptions**

The CENTURY model does not directly estimate the soil loss by erosion, but soil erosion rates can be specified as an input, which allows the model to account for the lateral C fluxes associated with the sediment transport (Fig. 1). To do that, we made the following steps and assumptions:

1. we resampled the 100 m\(^2\) resolution soil erosion rates (Mg soil ha\(^{-1}\) yr\(^{-1}\)) to the 1 km\(^2\) grid of the SOC modelling. As the RUSLE provides an average annual erosion rate from water, a unique event was implemented into the CENTURY model each year, during the autumn. A sensitivity analysis confirmed the negligible effect of cumulative eroded C with the implementation of a single annual erosion event vs. separate monthly erosion events (Figure S1). When erosion occurs, a fraction of SOC stock in the topsoil is lost in proportion to the erosion rate: \(C_{\text{erod}} = \text{SOC} \times \text{[erosion/(bulk density \times depth)]} \times \text{ER}\)

2. to be conservative with the mass balance, we assumed that each (1 km\(^2\)) grid cell retained a proportion of eroded C. The partition was based on Van Oost et al. (2007), who found that 53–95% of eroded SOC was retained in the catchment and redeposited in a limited area (14–35%) within the same catchment. Taking the central values, we assumed that 25% of our grid cells were depositional areas, which received 70% of eroded soil. The remaining 30% is accounted for as leaving the grid cell as potential sediments and C (C\(_{\text{out}}\)) discharged to riverine systems;

3. we kept a fixed soil profile of 0–30 cm, where the SOC budget was calculated. We did not account for any enrichment of C in deposited sediments, as was performed in a similar study at regional scale (Nadeu et al., 2015). The uncertainty related to this assumption was assessed through a sensitivity analysis (Table S1).

As the C mineralized during sediment transport (C\(_{\text{min}}\)) has a direct impact on the net C flux (Eqn 3), we ran two scenarios considering 10 and 30% C mineralization rates during sediment transport to depositional areas. The mineralization of eroded SOC is still a source of controversy between studies suggesting a disturbance effect from erosion (i.e. aggregate breakup) with increased mineralization (Lal, 2003) and, others, suggesting a minor influence of soil displacement and transport on SOC turnover in sediments (Kirkels et al., 2014). The mineralization rate enhancement factors span the range often used in similar studies (Zhang et al., 2014; Chappell et al., 2015) and, hence, represent an important uncertainty which has not been included in previous simulation model-based approaches.

The replacement of eroded soil in the fixed profile (0–30 cm) comes from the subsoil layers (SSL) ‘recruitment’, characterized by a SOC composition defined quantitatively and qualitatively as a partition among the three Century’s SOC pools (active, slow and passive). These pools are functionally defined on the basis of mean residence time, as opposed to measured fractions, and thus cannot be constrained by measurements applicable at the EU scale. The most rationale approach was to adopt the calibration of Harden et al. (1999), which implicitly related the subsoil composition to the topsoil composition; accordingly, the subsoil SOC pools at the time \(t = 0\) (that is 1900, as explained in point 4) were composed as a fraction of the top soil pools:

\[
\text{SSL}(t0) = (0.2 \times \text{Active surface}) + (0.4 \times \text{Slow surface}) + (0.8 \times \text{Passive surface})
\]

However, to investigate the effect of SOC subsoil pool initialization adopted, we ran an additional sensitivity analysis (Table S2). The lower horizon pools were decremented at each time step using the following relationships for each of the three SOC pools (i):
we derived the long-term SOC pool partition from previous simulations, in which an equilibrium period of more than 2000 years was carried out (Lugato et al., 2014a). Starting in 1900, the erosion process was implemented in the model, keeping the climate, soil and topographic factors (R, K, LS respectively, Fig. 1) constant. While we considered those factors quite invariable on a centennial scale, the C-factor associated with the crop type (Panagos et al., 2015b; Table S3) was dynamically varied with crop rotations and land-use changes.

The simulated land use was based on the Corine Land Cover 1990, 2000 and 2006, supplemented with EUROSTAT statistics to build up crop rotations and implement consistent agronomic inputs (fertilization, irrigation etc.). Before 1990, we assumed the same land use but with different agro-techniques characterized by lower productivity crops, lower rates of mineral nitrogen and different rotation schemes (see Lugato et al., 2014a, for further details).

Results

Geographical distribution of eroded SOC

Incorporating the erosion process into the high-resolution (1 km² grid cell) SOC simulation with CENTURY produced a detailed geographical distribution of average eroded SOC rates in the EU (Fig. 2). The highest erosion rates were in southern Spain, central and southern Italy, Romania and western UK and were closely correlated with the soil erosion patterns (Figure S2). However, around 86% of the European agricultural areas showed SOC erosion losses <0.1 Mg C ha⁻¹ yr⁻¹ and 143 Mha (including depositional areas) with rates <0.05 Mg C ha⁻¹ yr⁻¹ (Fig. 3).

Simulated SOC losses by erosion were high in the orchard land use (Fig. 3), probably because of their dominant presence in hilly areas. Although the C-factor of grasslands (0.09) is much lower than for annual crops (0.3 on average, Table S3), the distribution of SOC loss in grassland tended towards higher values compared with cropland. This could be due to the much higher SOC stocks in the 0–30 cm layer (150.3 and 64.5 t C ha⁻¹ for grassland and cropland, respectively), which sustains SOC losses even with lower erosion rates.

SOC balance and net C fluxes at the EU level

The average eroding area showed a negative SOC stock change (−0.139 Mg C ha⁻¹ yr⁻¹), driven by the negative balance between the vertical fluxes (Ci_fix − Ci_sub) (Table 1). The eroded C was only 2% of vertical fluxes, but higher than SOC in subsoil recruitment (Ci_sub). Cumulative for the EU level, the eroded SOC totalled 0.010 Pg C yr⁻¹ which represents a very small amount compared to the total SOC stock of 14.9 Pg C in the top 0–30 cm (Figure S3) and about 1% of annual net primary production.

In the average depositional area, the C fluxes were of the same order of those in eroding areas. The SOC deposited (Ci_depo) increased with the reduction of the mineralization rate transport (0.098 and 0.125 Mg C ha⁻¹ yr⁻¹ for 30 and 10% mineralization rates, respectively). The SOC stock change was higher in eroding areas, although those soils are subjected to an increase of the soil profile depth as a consequence of the deposition event. In parallel, SOC moving down from the 30 cm layer due to surface deposition had an average rate of 0.154 to 0.167 Mg C ha⁻¹ yr⁻¹.

Combining the C fluxes, we calculated the net CO₂ flux for the average ERD grid cell (Eqn 3, Table 1). The vertical C flux balance was between −0.084 and −0.086 Mg C ha⁻¹ yr⁻¹ (i.e. a net C loss to the atmosphere), but still less negative than a hypothetical simulation without erosion (−0.092 Mg C ha⁻¹ yr⁻¹). Adding the term related to SOC mineralization during transport (Cmin: 0.011 and 0.004 Mg C ha⁻¹ yr⁻¹), the overall balance was very similar to the no-erosion (NE) scenario, but with a different net atmospheric exchange direction. In fact, fCO₂ increased from −0.095 to −0.090 Mg C ha⁻¹ yr⁻¹, assuming 30 and 10% enhanced C mineralization during sediment transport. This implies a net loss (−0.003 Mg C ha⁻¹ yr⁻¹) and net gain (0.002 Mg C ha⁻¹ yr⁻¹) for the former and latter scenario, respectively, in comparison with a baseline without erosion in which the net CO₂ flux (fCO₂) was −0.092 Mg C ha⁻¹ yr⁻¹ (Table 1).

With the output available we calculated the erosion-induced vertical C exchange ratio (Fig. 4), as the difference of the net vertical fluxes (Ci_fix − Ci_sub) between eroding and non-eroding simulations, divided by the eroded C (Cerod). The Figure 4 depicts the spatial distri-
bution of this variable, representing the sink capacity of soils per unit of SOC eroded. In areas with highest C erosion rates (Fig. 2), the values almost ranged between 0.1 and 0.5, while a lower sink capacity resulted in central and eastern part of Europe.

Discussion

The CENTURY model has previously been used with erosion inputs to simulate SOC redistribution and dynamic in landform segments (Pennock & Frick, 2001; Liu et al., 2003) and small watersheds (Yadav & Malanson, 2009; Lacoste et al., 2015). Other approaches have included coupled dynamic soil erosion/distribution model operating at catchment scale with SOC models of moderate complexity, to understand the SOC cycle at a landscape-scale level (SPEROS-C, Van Oost et al., 2005). Recently the latter model was modified and applied at a regional level, in three agricultural regions of 250 km² in central Belgium (Nadeu et al., 2015).

In this context, the successful application of landscape SOC modelling requires spatially detailed information of SOC vertical distribution and landscape geomorphology, as input to accurately simulate vertical
and lateral transport of soil (and SOC) over the specified land area. It is inevitable that by increasing the scale of investigation more assumptions need to be made, because those inputs at the level of individual land form units are not available for an area the size of 187 Mha, which is the agricultural area we are currently simulating. Therefore, in our application, the grid cells are non-connected units, and thus, they do not receive or transfer C from one to another. Moreover, we simulated only the agricultural land use, not accounting for potential lateral C input from other land uses. Despite these assumptions, this is the first attempt to integrate erosion and depositional processes at continental scale, being conservative in a C mass balance.

Our analysis is also distinctive in term of extent and spatial resolution, at least at EU level. One of the strength of this application is that the model mechanistically simulates the SOC balance in cropland, orchard and grassland varying dynamically the C-factor (and hence the erosion) according to crops or land-use change.
Although the comparison with other studies applying different methodologies, assumptions and spatial extent/resolution should be performed with care, the assessment of C fluxes magnitude and direction against existing values could increase the model confidence or highlight the components that need to be improved. For instance, Ciais et al. (2010) using a combination of models and inventory method, corrected for erosion flux, estimated a net loss in cropland of EU-25 of 0.083 and 0.13 Mg C ha\(^{-1}\) yr\(^{-1}\), respectively. These values are consistent with our simulation under ERD scenario, in which estimated losses were 0.095–0.090 Mg C ha\(^{-1}\) yr\(^{-1}\).

In our modelling framework, the SOC stock change integrates all vertical and lateral fluxes components and is a driving variable for calculating the eroded SOC. The comparison of simulated SOC stocks against those calculated from a highly standardized soil sampling (LUCAS survey, http://ec.europa.eu/eurostat/web/lucas/overview), involving about 20 000 samples...
collected in 2009 across the EU (Figures S4 and S5), produced a good fit with low model uncertainty. The cumulative stock was, also, comparable with previous pan-European model run (Smith et al., 2005 and Figure S3).

A global distribution of cropland SOC erosion was performed by Van Oost et al. (2007), using a combination of caesium-137 and carbon inventory measurements and spatially explicit models. The authors estimated a cropland SOC erosion rate of 0.16 Mg C ha\(^{-1}\) yr\(^{-1}\), more than double that our flux (0.068 Mg C ha\(^{-1}\) yr\(^{-1}\)). Despite the different methodology and underlying assumptions, part of the inconsistency may be related to tillage erosion rates, not represented in this study, which are in the same order of magnitude as interrill and rill erosion on European arable land (Van Oost et al., 2009). Moreover, in their global analysis, the authors identified the SOC eroded and its replacement at the erosion sites as the two most important controls on sink magnitude, with a derived ratio of vertical to later flux of 0.26. Our map of erosion-induced vertical C exchange demonstrated that CENTURY is able to simulate these feedbacks, predicting values coherent with the above-mentioned study (Fig. 4 and Figure S6).

The net vertical C fluxes were less negative in eroded areas than those in none-erosion scenario, corroborating the idea that the dynamic replacement can partially offset the C loss by erosion (Harden et al., 1999). With the 30% mineralization hypothesis of transported C, the net CO\(_2\) flux was \(-0.095\) against \(-0.092\) Mg C ha\(^{-1}\) yr\(^{-1}\) without erosion. Converted to CO\(_{2eq}\), the erosion across all agricultural soils of the EU contributed 2.28 Tg yr\(^{-1}\) (1Tg = 1Mt) of additional CO\(_2\) emissions. This is not totally negligible considering that policy-oriented scenarios in European arable soils were estimated to sequester around 12.6–42 Tg yr\(^{-1}\) of CO\(_{2eq}\) (Lugato et al., 2014b). Moreover, using the lower estimate for C mineralization during sediment transport, the agricultural soils would be a CO\(_2\) atmospheric sink of 0.79 Tg yr\(^{-1}\) of CO\(_{2eq}\), with respect to a baseline without erosion. This sequestration capacity might be even higher, according to recent studies showing negligible C mineralization upon transportation (Wang et al., 2010).

Another component of the total C budget is the fate of C exported from cropland into riverine transport, which we quantified as 0.015 Mg C ha\(^{-1}\) yr\(^{-1}\). Unfortunately, we could not find continental studies isolating the agricultural contribution on riverine particulate organic carbon delivery. Ludwig et al. (1996) using a large set of measurements made in world rivers, estimated the continental particulate organic carbon fluxes to oceans in Europe as between 0.1 and 2 tonnes km\(^{-2}\) yr\(^{-1}\) (equal to 0.001–0.02 Mg C ha\(^{-1}\) yr\(^{-1}\)). This flux comprises sources from all land use and processes (e.g. river bank erosion, landslides); therefore, we cannot make a direct comparison but only highlight that the order of magnitude was the same.

The sensitivity analyses we conducted on some parameters, that we were not able to constrain in our modelling framework, revealed a robustness of model results. For instance, the decrease by 50% of the initial subsoil pool (SSL) composition resulted in a fCO\(_2\) change of about 5% (\(-0.086\) vs \(-0.090\) Mg C ha\(^{-1}\) yr\(^{-1}\) in the 10% mineralization scenario, Table S1). Moreover, the model behaved in a predictable way, because lower C ‘recruitment’ from subsoil induced a larger dynamic replacement, leading to a net sink of 3.73 Tg yr\(^{-1}\) of CO\(_{2eq}\) compared to the no-erosion baseline.

Our results are promising but new improvements could increase the accuracy of the estimations. One field of investigation is related to the enrichment factor of eroded SOC, which might arise due to selective transportation depending on rainfall intensity (Nie et al., 2015) or varying in time with the specific water erosion processes (Fiener et al., 2015). When we increased by 100% the enrichment factor (from 1 to 2), fCO\(_2\) changed by about 3% (\(-0.088\) vs \(-0.090\) Mg C ha\(^{-1}\) yr\(^{-1}\) in the 10% mineralization scenario, Table S1). In fact higher SOC displacement by erosion resulted in higher erosion-induced vertical C exchange (Figure S6), leading to a net sink of 2.51 Tg yr\(^{-1}\) of CO\(_{2eq}\) with respect to the no-erosion baseline.

Selective transportation may also change the textural composition and bulk density of eroding and depositional sites, but there is no physical or empirical model to dynamically account for this process into a biogeochemical model at this scale. The most challenging step is the creation of a spatial-explicit erosion/transport/deposition model working at continental scale that can be coupled with a SOC biogeochemical model. While technically feasible, more effort should be made at national and regional levels to make the necessary input data available.

Although the agricultural soils may act as a source or sink of CO\(_2\) depending on the assumptions, erosion seems to induce net C fluxes in the same order of current carbon gains from improved management (Borrelli et al., 2016). Ultimately, we strongly support the idea that erosion should be prevented or reduced by agricultural policies and sustainable management practices, in order to maintain the soils in stable and good conditions (Lal, 2014) and increase their resilience to human and natural perturbations.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Figure S1. Sensitivity analysis on cumulative eroded SOC (g m⁻²) over a period of 100 years, simulating one erosion event per year instead of monthly events.

Figure S2. Map of soil loss by water erosion in Europe.

Figure S3. Trend of cumulative SOC stock in the 0–30 cm layer at the EU level, during the period 1980–2030.

Figure S4. Comparison of SOC stocks (in the 0–30 cm soil profile) estimated with CENTURY (cmt) and measured in LUCAs sampling points (lcs).

Figure S5. 2σ confidence interval errors of simulated SOC stocks.

Figure S6. Erosion-induced vertical C exchange (Mg C ha⁻¹ yr⁻¹), calculated as the difference by the net vertical fluxes (C_i - C_o) between eroding and non-eroding simulations, plotted against the corresponding lateral C fluxes by erosion grouped in classes.

Table S1. Sensitivity analysis scenario on fCO₂ flux, increasing by 100% (from 1 to 2) the enrichment factor (EF).

Table S2. Sensitivity analysis scenario on fCO₂ flux, reducing by 50% the SOC pools composition of the subsoil layer (SLL).

Table S3. C-factor implemented dynamically in CENTURY according to crop rotations and land use changes.