RESEARCH ARTICLE

Soil erosion has mixed effects on the environmental impacts of wheat production in a large, semi-arid Mediterranean agricultural basin

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Abstract

Soil erosion poses a significant threat to agricultural production worldwide, with a still-debated impact on the current increase in atmospheric CO_2. Whether erosion acts as a net carbon (C) source or sink also depends on how it influences greenhouse gas (GHG) emissions via its impact on crop yield and nutrient loss. These effects on the environmental impacts of crops remain to be considered. To fill this gap, we combined watershed-scale erosion modeling with life cycle assessment to evaluate the influence of soil erosion on environmental impacts of wheat production in the Ebro River basin in Spain. This study is the very first to address the full GHG balance of erosion including its impact on soil fertility and its feedback on crop yields. Two scenarios were simulated from 1860 to 2005: an eroded basin involving conventional agricultural practices, and a non-eroded basin involving conservation practices such as no-till. Life cycle assessment followed a cradle-to-farm-gate approach with a focus on recent decades (1985–2005). The mean simulated soil erosion of the eroded basin was 2.6 t ha^{-1} year^{-1} compared to the non-eroded basin. Simulated soils in both eroded and non-eroded basins lost organic C over time, with the former emitting an additional 55 kg CO_2 ha^{-1} year^{-1}. This net C source represented only 3% of the overall life cycle GHG emissions of wheat grain, while the emissions related to the increase of fertilizer inputs to compensate for N and P losses contributed a similar percentage. Wheat yield was the most influential parameter, being up to 61% higher when implementing conservation practices. Even at the basin scale, erosion did not emerge as a net C sink and increased GHG emissions of wheat by 7–70%. Nonetheless, controlling erosion through soil conservation practices is strongly recommended to preserve soils, increase crop yields, and mitigate GHG emissions.

Keywords Agricultural soils · Ebro River basin · Greenhouse gas emissions · Life cycle assessment · Soil erosion · Wheat · Erosion modeling · RUSLE · Soil carbon dynamics

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1 Introduction

Anthropogenic greenhouse gas (GHG) emissions need to be rapidly curbed to mitigate climate change and achieve the targets set in the 2015 Paris Agreement (Smith et al. 2016). Carbon (C) sequestration in soils has been identified as an important land-based mitigation option, which is at the basis of the international "4 per 1000" initiative aiming to increase C storage in agricultural soils in order to compensate for anthropogenic emissions (Minasny et al. 2017). However, adding C to soils also requires adding nutrients to avoid stoichiometric imbalances (van Groenigen et al. 2017). Storage can be enhanced by increasing C and nutrient inputs to soils via practices such as organic

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matter amendments and establishing cover crops or agro-forestry systems (Guenet et al. 2021). However, soil erosion must also be decreased to preserve soil C and nutrient stocks. Consequently, flows due to soil erosion processes should be considered when evaluating soil C sequestration potential. According to the IPCC's recent report on climate change and land (IPCC 2022) one-quarter of Earth's ice-free land area is degraded, and erosion is the most widespread cause. Currently, agricultural soils have the highest soil erosion rates, which are accelerated by the removal of vegetation and ground cover and by soil tillage (IPCC 2022; Van Oost et al. 2007). Soil erosion on agricultural land has substantial impacts on soil fertility and food security (Lal 2009).

The role of erosion (mainly due to water) as a sink or source of atmospheric CO_2 is currently highly debated in the literature (Borrelli et al. 2017; Lal 2009; Van Oost et al. 2007). Erosion moves soil along the cascade of hillslopes, floodplains, and rivers. This redistribution may lead to C sequestration in deposition areas such as colluvial and alluvial soils due to (1) the replacement of eroded C by new litter input (photosynthesis) on eroded land and (2) the burial of eroded C in deeper soil layers, thus reducing soil C respiration. However, soil decomposition can be intensified by soil erosion during sediment transport and after deposition due to aggregate breakdown, thus increasing CO_2 emissions. Despite this uncertainty in erosion as a C sink or source, the studies cited showed that the effects of erosion on C emissions must be addressed in C-cycle research.

Mechanistic modeling is a promising approach to estimating the impacts of erosion and its drivers, and extensive efforts have been made in recent years to develop soil erosion models (Barot et al. 2015). Nonetheless, these models have been generally developed to represent fine-scale processes and may not be suitable for larger scales. Large-scale soil erosion is usually estimated using the Universal Soil Loss Equation (RUSLE) model (Naipal et al. 2018; Panagos et al. 2015). The structure of RUSLE also facilitates connecting erosion processes to C-cycle processes on land. For instance, RUSLE has been coupled with soil organic matter models such as CENTURY (Lugato et al. 2018) and

to land-surface models (Naipal et al. 2020). However, these modeling approaches still need to represent erosion-induced feedback on plant productivity, which may lead to over-predictions of C input to the soil in eroding regions.

Life cycle assessment (LCA) is a holistic approach widely used to estimate the environmental impacts of a given practice or product (Hellweg and Milà i Canals 2014). It provides a suitable framework for understanding the multi-faceted effects of soil erosion on GHG emissions and other environmental issues. It has been used to evaluate the impacts of land use on soil erosion (Núñez et al. 2013; van Zelm et al. 2017) or those of tillage practices and erosion control on the GHG balance of crops (Martin-Gorriz et al. 2020), but not on the reciprocal issue of how erosion influences environmental impacts of crops. This gap is understandable since most LCAs of agricultural crops focus on the field or farm scale (Poore and Nemecek 2018), whereas the GHG balance of erosion flows must be represented at the basin scale to estimate removal and redeposition rates, for instance using the modeling approaches previously described.

This study is the very first to address the full GHG balance of erosion including its impact on soil fertility and its feedback on crop yields. Here, we combined LCA and erosion modeling to estimate the consequences of soil erosion on the GHG balance of wheat crops at the scale of the Ebro River basin (85 550 km^2) in Spain. This large basin was selected because it experiences severe erosion (Fig. 1) (Cantero-Martínez et al. 2007) and cropland covers one-third of its area. A mechanistic model of soil and C erosion, CE-DYNAM, combining RUSLE and the C emulator of a land-surface model (Naipal et al. 2020), was used to predict net erosion (erosion losses minus deposition) from agricultural land on the hillslopes of the Ebro River basin. Two crop management scenarios were compared: conservation agriculture (CA, to control erosion) and conventional management (CT, using conventional tillage). The two scenarios make it possible to single out the effect of erosion on the environmental impacts of wheat, using the large body of data available in the literature on CA and CT management. This literature also confirms that no-tillage (as part of CA) abates soil erosion by 80 to 90% compared to CT (IPCC

Fig. 1 Evidence of soil erosion on agricultural land in the Ebro River basin (credit: Carlos Cantero-Martinez).

2022), which justifies the association between CA and the no-erosion scenario. Differences in yield were considered explicitly using two main sources of information: field-scale observations or a model derived from a global meta-analysis that compared CA and CT historically (Cantero-Martínez et al. 2007; Su et al. 2021a).

2 Materials and methods

2.1 The Ebro River basin

The Ebro River basin is an extensive area covering more than 8 Mha from 200 to 3000 m asl. The dominant climate is continental-Mediterranean with hot and dry summers followed by cold winters. The mean annual rainfall depends on the elevation and location along the east-west line of the basin, ranging from 200 to 1800 mm. In all locations, rainfall events occur in spring (April and May) and autumn (September and October). Rainfall intensity may reach 80–100 mm h^{-1}, which favors soil erosion, especially in areas with low precipitation and sparse plant cover. Summer droughts occur between the two wet periods. Mean monthly temperatures range from −5 °C (with extremes down to −20 °C in the Pyrenees Mountains) in January to more than 30 °C in July and August, with extremes exceeding 40 °C in the lowlands at the center of the basin (García-Ruiz et al. 2000; Garrido and Garcia 1992; Llasat 2001; Martín-Vide and Lopez-Bustins 2006; Peñarrocha et al. 2002; Romero et al. 1998).

The Ebro River basin is a closed sedimentary basin whose formation was strongly influenced by climate, tectonics, and soil erosion. It is a large alluvial plain surrounded by low mountain ranges to the north (Pyrenees) and south (Iberian System). Flat zones dominate its central area (200–400 m asl; 40% of the basin), where slopes range from 0 to 5%. The surrounding area (400–1400 m asl; 40% of the basin) has slopes ranging from 0 to 20%. The remaining area (800–3000 m asl; 20% of the basin) has slopes ranging from 0 to 40% (including high mountain peaks). Wheat and barley, the main crops in this basin, are cultivated below 1400 m. The land above 1400 m asl is occupied by pastures, meadows, and high forests, where erosion is low because of the continuous soil cover.

2.2 Erosion modeling

C erosion and deposition flows in the Ebro River basin were predicted using the novel large-scale and spatially explicit carbon erosion dynamics model, version 1 (CE-DYNAM-v1) (Naipal et al. 2020). CE-DYNAM consists of a sediment budget submodel coupled with an emulator of the soil C submodel of the global land surface model (LSM). ORCHIDEE CE-DYNAM was originally

calibrated for the Rhine River basin, but its model structure and parameters make it applicable to other river basins globally. It considers hillslopes and floodplains with soil removal occurring in hillslopes and redeposition occurring in both hillslopes and floodplains. Hillslopes and floodplains are delineated following the gridded global dataset of Pelletier et al. (2016) which maps soils, intact regolith, and sedimentary deposit thicknesses at 5 arcmin resolution. The soil erosion submodel of CE-DYNAM uses RUSLE (Naipal et al. 2015), which calculates the annual mean soil erosion rate (E, expressed in g soil m^{-2} year^{-1}) following Eq. (1) as a product of a topographical factor (S, dimensionless), a rainfall erosivity factor (R, MJ mm m^{-2} h^{-1} year^{-1}), a soil erodibility factor (K, g h MJ^{-1} mm^{-1}), and a land cover and -management factor (Cm, dimensionless):

$$E = S \times R \times K \times Cm \qquad (1)$$

The Cm factor includes the effects of vegetation cover and crop residues on soil erosion for tilled soil (conventional agriculture). The effects of conservation practices on erosion are not considered. E represents the potential gross removal of soil from eroding hillslopes.

The daily gross C removal rate by soil erosion (Ce, g C m^{-2} d^{-1}) is then calculated as follows (Eq. 2):

$$Ce = \frac{\frac{SOC \times E}{365} \times 100}{BD_{top} \times dz \times 10^6} \times CER \qquad (2)$$

where BD_{top} is the bulk density of the surface layer (g cm^{-3}), CER is the enrichment ratio (i.e., the ratio of the C content in the eroded soil to that of the source soil material, set to 1), and dz is the soil layer thickness (m). CE-DYNAM simulates the dynamics of soil organic C (SOC, g C m^{-2}) down to a depth of 2 m.

A fraction (f) of the eroded soil is deposited at the foot of the hillslopes (colluvial deposits), depending on the slope gradient and the vegetation type. It is described by Eq. (3):

$$f = 1 - a_f e^{\left(\frac{b_f \theta}{\theta_{max}} \right)} \qquad (3)$$

where a_f and b_f (derived from Naipal et al. (2020)) are constant parameters that relate f to the mean topographical slope (θ) of a grid cell in degree depending on the type of land cover, and θ_{max} is the maximum topographical slope of the watershed (in degree).

The rest of the eroded soil is transported to the floodplains and water network.

CE-DYNAM has three SOC pools—active, slow, and passive—which differ in their C residence time, which is shortest for the active pool and longest for the passive one.

Changes in each SOC pool on the hillslopes are calculated as follows (Eq. 4):

$$\frac{dSOC(t)}{dt} = I(t) - \left(k + k_E\right)SOC(t) \tag{4}$$

where I is the C input to the soil by litter decomposition; k is the C decomposition rate, which depends on soil physical parameters such as temperature; and k_E is the net erosion rate, which equals 1 minus $E \times f$.

Parameters I and k are specific to the land cover type and derived from the ORCHIDEE LSM, forming the basis of the SOC submodel of CE-DYNAM. See Naipal et al. (2020) for a more detailed description of CE-DYNAM.

CE-DYNAM was run from 1850 to 2005, with and without soil erosion, with a daily time-step at a spatial resolution of 5 arcminutes (~ 8 km). This resolution was adequate for capturing erosion flows at basin to continental scales (Fendrich et al. 2022). First, the model was run to a steady state based on the environmental conditions of the period 1850–1860, and then transient simulations were performed with changing climate and land-use conditions. The overall change in SOC stocks over 1850–2005 thus resulted from climate change, increasing atmospheric CO_2 concentrations, land-use change, and soil erosion. The resulting net soil C emissions due to erosion were then calculated as the cumulative SOC change over the period 1850–2005 of the eroded basin minus that of the non-eroded basin. We considered that the C flows predicted by CE-DYNAM when erosion was activated represented CT practices, while those predicted when erosion was not activated represented CA practices that limit erosion.

2.3 Input data for CE-DYNAM

The input datasets for CE-DYNAM were the same as those used by Naipal et al. (2020). Land cover proportions were derived from historical 0.25° maps of Peng et al. (2017). Daily rainfall data for the period 1850–2005 were derived from the Inter-Sectoral Impact Model Intercomparison Project (ISIMIP) product ISIMIP2b (Frieler et al. 2017). These data were based on the model output of the Coupled Model Intercomparison Project Phase 5 (CMIP5 output of IPSL-CM5A-LR) (Taylor et al. 2012), which were bias-corrected using observational data sets and the method of Hempel et al. (2013) and made available at a resolution of 0.50°. Data on soil bulk density and other soil parameters used to calculate the soil erodibility factor (K), available at the resolution of 1 km, were taken from the Global Soil Dataset for use in Earth System Models (GSDE) (Shangguan et al. 2014). Finally, the slope steepness factor (*S*) was initially estimated at the resolution of 1 km, based on the GTOPO30 digital elevation model. Datasets that were not available at

a 5 arcminute resolution were regridded to this resolution using bilinear interpolation. To evaluate model predictions, we extracted data on SOC stocks from the GSDE product (Shangguan et al. 2014) when crops covered at least 30% of the area, using the land cover maps of Peng et al. (2017).

2.4 Life cycle assessment

2.4.1 Goals and scope

The LCA had two objectives: assess the contribution of soil erosion to the GHG balance of wheat and examine the effects of erosion-control practices on the environmental impacts of this wheat production in the Ebro River basin. To focus on the effects of soil erosion, two scenarios were simulated with CE-DYNAM: minimum erosion ("non-eroded," reflecting CA) or standard erosion ("eroded," reflecting CT).

To compensate for the loss of crop productivity associated with erosion losses, fertilizer input rates were adjusted in the eroded scenario based on estimated rates of N and P losses. These rates were estimated from the net C losses predicted by CE-DYNAM, assuming C:N and C:P ratios of 10 and 38, respectively, in eroded material (Xu et al. 2013). We did not represent the effects of soil erosion on physical or hydraulic soil properties (e.g., porosity or water-holding capacity).

We used cradle-to-farm-gate system boundaries, and the functional unit for this system was the production of 1 kg of wheat. The time frame considered in the life cycle inventories (LCI) extended from the harvest of the previous crop to the harvest of wheat.

2.4.2 Life cycle inventory

LCA was performed using SimaPro® software (version 8.0.3, Pré Sustainability, Amersfoort, NL) and two LCI databases: AGRIBALYSE (Colomb et al. 2015), which focuses on agricultural products in France, and ecoinvent (v3.6; ecoinvent, Zurich). AGRIBALYSE was used to model direct emissions of reactive N and P using generic models (Colomb et al. 2015), and ecoinvent, which covers Europe, to model background processes.

Foreground processes related to wheat production in the Ebro River basin were based on management and yield data for tillage systems at three field sites (Cantero-Martínez et al. 2007). A no-till system was used to represent the non-eroded watershed, while a cropping system that included moldboard plowing was used to represent the eroded watershed. The "Agramunt" site was chosen since it grew wheat crops every other year from 2000 to 2005. Crops received organic fertilizer inputs in this study. However, since inorganic fertilizers predominate in the Ebro River basin, the

Table 1 Crop management data for wheat production on the non-eroded and eroded watersheds (corresponding to conventional tillage (CT) and conservation agriculture (CA) practices, respectively), for yield scenario no. 1. The higher N and P inputs for CT compensate for the net loss of nutrients through erosion. *a.i.* active ingredient.

	Non-eroded watershed (CA)	Eroded watershed (CT)
Crop rotation	Barley from 1990 to 2000, then wheat every other year until 2005	
Yield	3250 kg grain DM ha^{-1}	3283 kg grain DM ha^{-1}
Tillage	No-till	1 moldboard plowing (30–35 cm) 1–2 cultivator passes (15 cm) 1 cultivator pass before sowing
Sowing	No-till disc drill: 500 seeds m^{-2}	
Herbicide inputs	Glyphosate (0.685 kg a.i. ha^{-1}) MCPA (0.644 kg a.i. ha^{-1}) Prosulfocarb (4 kg a.i. ha^{-1})	Prosulfocarb (4 kg a.i. ha^{-1})
Fertilizer inputs (inorganic)	134 kg N ha^{-1} 81.5 kg P$_2$O$_5$ ha^{-1} 142 kg K$_2$O ha^{-1}	(134 + 6) kg N ha^{-1} (81.5 + 4) kg P$_2$O$_5$ ha^{-1} 142 kg K$_2$O ha^{-1}
Disease and pest management	No treatment	
Harvest	Medium-sized standard combine harvester. Straw is chopped and spread on the field.	

organic input rates were converted into equivalent mineral fertilizer inputs based on manure and slurry fertilizing values (Table 1).

Wheat yields vary significantly over the years and spatially across the Ebro River basin (Cantero-Martinez et al. 2007; Plaza-Bonilla et al. 2018). Since yield strongly influences LCA results, we used multiple sources of information and scenarios to estimate crop yields for the CA and CT practices.

In yield scenario no. 1, the wheat yield under CT (3283 kg grain ha^{-1}) was calculated as the mean of wheat yields across each province and autonomous community in the Ebro River basin in Spain from 1996 to 2006. Yield data were taken from the annual report of the Ministry of Agriculture, Fisheries and Food of the Spanish Government. The yield under CA was estimated using a model built from a meta-analysis that compared the yields of CT and CA wheat worldwide (Su et al. 2021a), using machine learning (and a random forest algorithm), and a global database (Su et al. 2021b). The model estimated their proportional difference in mean yield $\left(\frac{YieldCA - YieldCT}{YieldCT} \right)$ from 1996 to 2006 as −0.010 over the Ebro River basin, which led to an estimate of 3250 kg grain ha^{-1} for the wheat yield under CA. Scenario no. 1 had the advantage of relying on watershed-scale data and a global model. Nonetheless, yields vary widely as a function of climate, soil conditions, and crop management characteristics. In the Ebro River basin, CA can achieve higher yields than CT (Lampurlanés et al. 2016), since it also increases the water-holding capacity of soils (Skaalsveen et al. 2019). Therefore, we developed yield scenario no. 2 based on the local observations of Lampurlanés et al. (2016). The mean wheat yields in this scenario were averaged over 7 years: 2314 kg ha^{-1} under CA (non-eroded) and 1436 kg ha^{-1} under CT (eroded).

The net effect of erosion on changes in soil C at the field scale was predicted by the CE-DYNAM simulations, averaged over the period 1985–2005. This erosion-induced C sink/source term for hillslopes aggregated the following processes and flows: C replacement on eroding soils by new litter input (photosynthesis), C burial in deposition sites, increased decomposition of deposited SOC, and the C flow that leaves the hillslopes of the Ebro River basin. The floodplains were excluded from the analysis, and it was assumed that the C flow leaving the hillslopes was entirely respired and lost. Wind erosion was not considered in this study mainly because CE-DYNAM does not simulate this process. Moreover, wind erosion models require parameters such as aerodynamic roughness length, which are difficult to estimate at the scale of the Ebro River basin (Kardous et al. 2005). In addition, the erosion flows predicted by CE-DYNAM should be considered a simplification of reality, since CA does not fully abate soil erosion (IPCC 2022). Nonetheless, CE-DYNAM has already been evaluated on other basins, with good performance indicators (Fendrich et al. 2022; Naipal et al. 2020), and is based on a robust approach to simulate erosion (Naipal et al. 2018; Panagos et al. 2015); thus, we considered this approach suitable for our purpose.

2.4.3 Impact characterization

Environmental impacts were estimated using the IMPACT World+ characterization method (IMPact Assessment of Chemical Toxics), as integrated into SimaPro® software (Bulle et al. 2019). This method estimates impacts in 16 midpoint categories: human toxicity (cancer and non-cancer), particulate matter formation, ionizing radiation, ozone layer depletion, photochemical oxidant formation, freshwater

ecotoxicity, terrestrial acidification, freshwater acidification, marine eutrophication, global warming, non-renewable energy, land occupation, mineral extraction, and water scarcity.

3 Results and discussion

3.1 Evaluation of the erosion model

Erosion had a strong influence on the total predicted SOC stock averaged for the period 1995–2005 in crop fields to a depth of 30 cm, which equaled 86.5 t or 11.4 C ha^{-1} without or with erosion, respectively (Fig. 2a). This large difference was due in part to the fact that the maximum erosion rate occurred during the entire simulation (1860–2005), even though it had been calibrated from modern agricultural techniques. Thus, CE-DYNAM likely over-predicted erosion rates during the first several decades of the simulation since the mechanization of agriculture, responsible for higher erosion rates, started in this region in the 1940s. The soil C data extracted from the GSDE product resulted in a mean stock of 67.3 t C ha^{-1} to a depth of 30 cm. This stock lay between those predicted by the model with or without erosion. However, CE-DYNAM simulated extreme cases that considered no erosion or erosion rates calibrated from CT without considering erosion-control measures that can be used locally in fields. We also compared the predicted erosion rates over the Ebro River basin to results from other studies (Lizaga et al. 2018; Quine et al. 1994). For instance, CE-DYNAM predicted a lower gross soil erosion rate (6.6–18.2 t ha^{-1} year^{-1}) than that estimated by Lizaga et al. (2018) using a

^{137}Cs tracer (25.4 t ha^{-1} year^{-1}) (Fig. 2b). However, these authors estimated the gross soil erosion rate for only for one watershed of the Ebro River basin (23 km^2, in Navarra province), which has a Mediterranean-continental climate with an Atlantic influence and receives more rainfall than the central Ebro River basin does. This watershed is also generally steeper than the central Ebro River basin. Differences between the two areas studied can thus explain the model's underpredictions. However, our assumption of a steady state from 1850 to 1860 in CE-DYNAM simulations may also have contributed to the underprediction of erosion. It is important to consider past changes in climate and land use to be able to better predict current soil erosion. However, the lack of knowledge about previous land use and land management makes this task difficult. Furthermore, SOC was derived from predictions of ORCHIDEE simulations at a coarse resolution, while plant cover was derived from a global map, which may also have contributed to the differences. Finally, net soil erosion rates predicted by CE-DYNAM were also compared to those derived from observations of Lizaga et al. (2018) and Quine et al. (1994). CE-DYNAM predicted net soil erosion rates of 1.7–4.7 t ha^{-1} year^{-1} (equivalent to 43.0–95.5 kg C ha^{-1} year^{-1}), which were lower than those derived from Lizaga et al. (2018) (8.3 t ha^{-1} year^{-1}). This pattern was expected since CE-DYNAM had already underpredicted gross soil erosion rates. Quine et al. (1994), also using a ^{137}Cs tracer method, estimated net soil erosion rates of 16–25 t ha^{-1} year^{-1} for a small sub-basin of the Ebro River basin of 0.52 km^2. The small area of these sub-basins compared to the entire Ebro River basin may make them less representative. Finally, model predictions seemed reasonable since CA practices were not

Fig. 2 Evaluation of the erosion model for (**a**) soil organic carbon (SOC) stocks at 30 cm vs. data from the Global Soil Dataset for use in Earth System Models (GSDE) (Shangguan et al. 2014), (**b**) gross soil erosion rate vs. observations of Lizaga et al. (2018), and (**c**) net soil erosion rate vs. observations of Lizaga et al. (2018) (light gray) and Quine et al. (1994) (dark gray). In **a**, data were filtered to represent only the cropland area, while data in **b** and **c** represented the entire basin. Error bars represent 1 standard deviation across the simulation pixels.
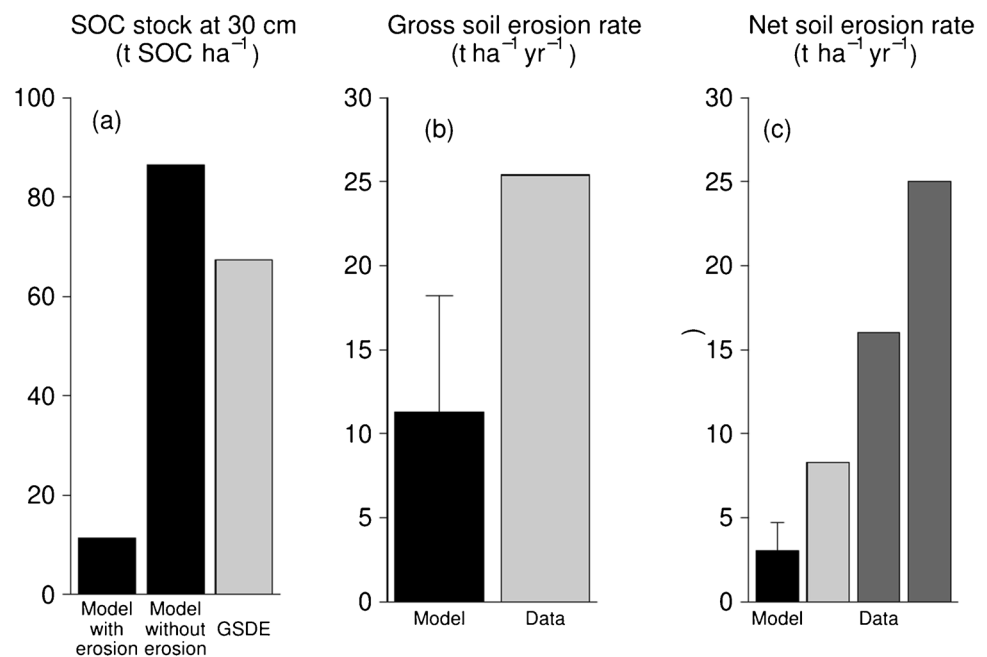
represented in CE-DYNAM, which can simulate only no erosion or maximum erosion in each grid cell. CE-DYNAM may underpredict net soil erosion rates, but the lack of data for the entire Ebro River basin makes evaluating model predictions difficult.

3.2 Contribution of soil erosion to the GHG balance of wheat

Cropland soils emerged as net sources of CO_2, with an estimated emission of 215 and 249 kg CO_2 ha^{-1} year^{-1} for the non-eroded and eroded watersheds, respectively. Despite the large effect of erosion on soil C stocks (Fig. 2), the losses

experienced since 1860 in the eroded basin were still more than compensated by the rates of C burial at sites experiencing net deposition. Indeed, erosion drastically influenced the spatial distribution of SOC, with higher SOC stocks in the floodplain and lower SOC stocks on hillslopes. Ultimately, this sink effect was no longer active in the non-eroded basin.

Per kg of wheat grain harvested, these soil C sources represented only 5% of life cycle GHG emissions of wheat production for yield scenario no. 1 (Fig. 3). Their value is consistent with the range of 0–1700 kg CO_2 ha^{-1} year^{-1} reported by Goglio et al. (2018) in their survey of the life cycle GHG balances of wheat crops worldwide, although some studies based on direct field observations of changes in SOC indicated larger

Fig. 3 Relative life cycle impacts of producing 1 kg of wheat in non-eroded (dashed line) and eroded (solid line) watersheds of the Ebro River basin for (**a**) yield scenarios no. 1 (i.e., eroded yield 1% higher) and (**b**) no. 2 (i.e., non-eroded yield 61% higher). For a given impact category, the system with the highest impact equals 100%.
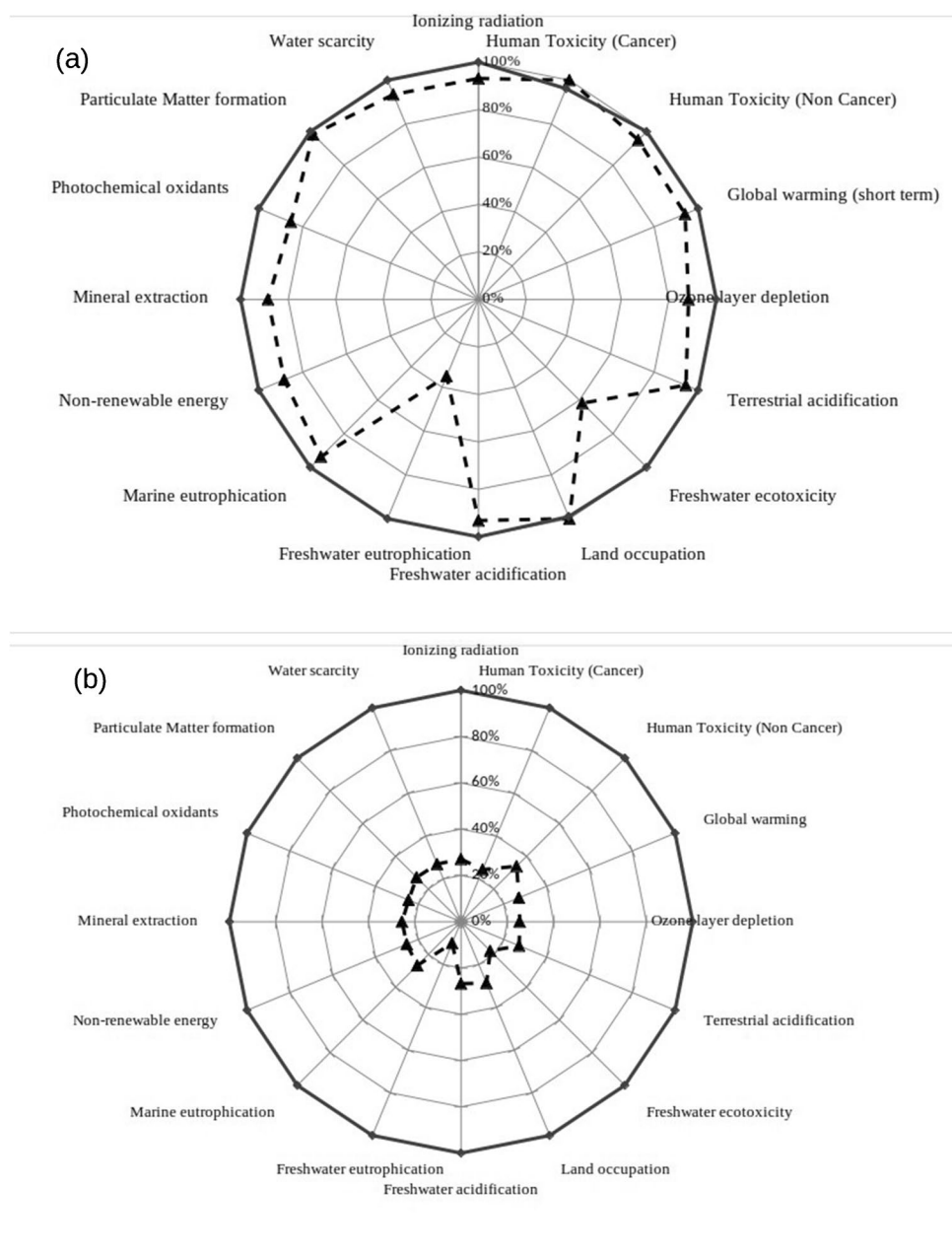
sinks or sources (-3.5 to 5.0 Mg CO_2 ha^{-1} year^{-1}). In general, CA favors soil C sequestration, and Álvaro-Fuentes et al. (2011) estimated that it could store 200–500 kg C ha^{-1} year^{-1} (i.e., 730–1830 kg CO_2 ha^{-1} year^{-1}), whereas CT systems were simply neutral. The fact that CE-DYNAM erosion simulations began before 1985 (the first year of the 20-year sequence used to calculate changes in SOC) and that the SOC contents of CA and CT systems differed significantly at that time (reflecting the "legacy" of past erosion processes) mitigated the sink capacity of CA. Factoring in the past (1860–1985) differences in SOC between the two systems simulated by CE-DYNAM would amount to an additional sink of 480 kg C ha^{-1} year^{-1} for the CA system. This sink would offset 40% of the life cycle GHG emissions of the CA systems (i.e., 1210 kg C ha^{-1} year^{-1}).

Along with potential soil C sink terms, N_2O emissions (which made up 85% of direct field emissions of GHG) and

GHG emissions (kg CO₂-eq. kg⁻¹ grain)
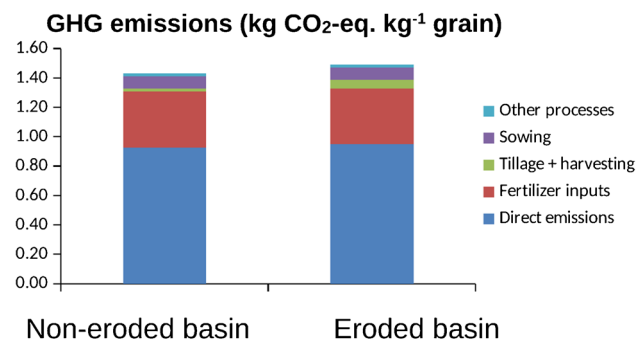
Fig. 4 Contribution analysis for the global warming impact of producing 1 kg of wheat in non-eroded (CA) and eroded (CT) sections of the Ebro River basin for yield scenario no. 1 (i.e., eroded yield 1% higher). Direct emissions include field emissions of CO2 and N2O.

the use of inorganic fertilizers strongly influenced the GHG balance of wheat (Fig. 4). N fertilizers contributed the largest impact, followed by P and potassium fertilizers. The contribution of these inputs to the total GHG balance (35%) was similar to that (30%) reported by Poore and Nemecek (2018). From the perspective of fertilizer use, another consequence of erosion is the need to compensate for nutrient losses. This adjustment increases off-farm emissions due to the production and transportation of additional inputs and field emissions of N_2O derived from fertilizer N. Compared to the non-eroded basin, both sources of emissions increased by 3% for the eroded basin, resulting in 6% higher GHG emissions overall (Table 2).

Wheat production with CT practices required more frequent use of agricultural machinery (e.g., plowing, fertilization), which increased CO_2 emissions slightly. Indeed, diesel consumption was 3 times as high under CT than under CA, but contributed only a small percentage ($< 10\%$) of the emissions. Erosion also increased impacts for certain impact categories due to pollution of surface water through runoff of nutrients, organic matter, and chemicals. This was particularly apparent for freshwater eutrophication and ecotoxicity, which were 25% lower under CA (Fig. 3, Table 2). Conversely, the loss of soil under CT mitigated to a minor extent (5%) the impact of CA wheat on human toxicity (non-cancer) due to reduced exposure of humans to contaminated soils.

We assumed input of only inorganic fertilizers, although organic fertilizers may influence the productivity and SOC dynamics of cropland. For instance, Larney and Janzen (1997) showed that organic fertilizers supplied P to crops more effectively in eroded areas than inorganic phosphate fertilizers, which were immobilized by calcium carbonate. Comparing

Table 2 Midpoint life cycle impacts of producing 1 kg of wheat in the non-eroded (CA) and eroded (CT) sections of the Ebro River basin for yield scenario no.1 (i.e., eroded yield 1% higher). Key to units (see Bulle et al. (2019) for more details): *CTUh, CTUe* comparative toxic units for humans and ecosystems, respectively, *NMVOC* non-methane volatile organic compounds.

Impact category	Unit	Non-eroded watershed (CA)	Eroded watershed (CT)
Human toxicity (cancer)	CTUh	2.87×10^{-9}	2.76×10^{-9}
Human toxicity (non-cancer)	CTUh	3.23×10^{-8}	3.40×10^{-8}
Particulate matter formation	kg PM$_{2.5}$ eq.	4.12×10^{-4}	4.19×10^{-4}
Ionizing radiation	Bq C-14 eq.	1.8	1.9
Ozone layer depletion	kg CFC-11 eq.	5.37×10^{-8}	6.08×10^{-8}
Photochemical oxidant formation	kg NMVOC eq.	3.10×10^{-3}	3.63×10^{-3}
Freshwater ecotoxicity	CTUe	59.2	95.8
Terrestrial acidification	kg SO$_2$ eq.	1.18×10^{-5}	1.25×10^{-5}
Freshwater acidification	kg SO$_2$ eq.	1.09×10^{-8}	1.25×10^{-8}
Freshwater eutrophication	kg PO$_4$ P-lim eq.	1.13×10^{-4}	3.20×10^{-4}
Marine eutrophication	kg N eq.	1.63×10^{-4}	1.74×10^{-4}
Global warming	kg CO$_2$ eq.	1.42	1.51
Non-renewable energy	MJ primary	5.56	6.29
Land occupation	m^{2}org.arable	0.206	0.204
Mineral extraction	kg deprived resource	1.00×10^{-2}	1.13×10^{-2}
Water scarcity	m^3 world eq.	24.4	26.1

inorganic and organic fertilizers to maintain crop productivity on eroded land could bring further insight into the environmental impacts of cereals in eroded watersheds.

3.3 Erosion and crop yields

When the wheat yield under CT was only 1% higher than that under CA (yield scenario no. 1), soil erosion generally caused higher emissions to the air and water, and resulted in 10–25% higher impacts under CT for all categories except human toxicity (Fig. 3a). In contrast, since yield scenario no. 2 was based on cropping system experiments that compared CA and CT, differences in impact between them and their associated erosion configurations were much larger, reaching 60–80% higher under CT (Fig. 3b). This magnitude of the difference was expected due to the 61% higher yield of CA wheat compared to that of CT wheat (Table 1) and the differences in environmental impacts observed previously, in which CA outperformed CT due to the latter's soil erosion.

Overall, experiments on the effects of soil erosion on crop yields have been inconsistent, but a recent meta-analysis highlighted that they were negative except when "the remaining A horizon depth was greater than 25 cm or when the erosion depth was less than 5 cm" (Zhang et al. 2021). In the Ebro River basin, the soil losses predicted by CE-DYNAM in the eroded watershed (mean = 1.5 t ha^{-1} year^{-1}) indicated rapid depletion of the topsoil and a depth of erosion greater than 10 cm year^{-1} (Fig. 1). This unfavorable effect on crop productivity is backed by the field data used for yield scenario no. 2, with the additional benefits of conservation practices for the non-eroded basin configuration. These benefits are known to be greater under semi-arid conditions (Su et al. 2021a), even though the model derived by Su et al. predicted only a marginal effect. The discrepancy between the yields estimated from the local field trial (in scenario no. 1) and the model trained on a global data set (collating ca. 4000 datapoints from such trials worldwide) in scenario no. 2 questions the relevance of comparing these two approaches. While most agricultural LCAs rely on local field data (Poore and Nemecek 2018), practitioners have also pointed out the need to produce regional averages (Hellweg and Milà i Canals 2014). Such data would be particularly suitable to represent a large river basin. However, it is difficult to test the reliability of the global model of Su et al. (2021a) at such an intermediate scale and to replace its results directly with local field data. Scaling up agroecosystem models that better represent the Ebro River basin would be a good option for deriving regional estimates, similar to the approach taken by Plaza-Bonilla et al. (2018). CA also had less intensive management practices than CT due to the absence of tillage and lower fertilizer inputs. No-till is known to incur trade-offs with other practices such as weed

management, and management data representative of the study area need to be collected to produce reliable results.

Finally, CE-DYNAM erosion predictions did not consider feedback between erosion and primary productivity. CE-DYNAM simulates only C, but erosion also removes N and P in reality, which influences factors such as soil physical structure and water infiltration capacity. Thus, CE-DYNAM predictions of higher SOC loss due to erosion over the 1985–2005 period may be underestimated because simulated erosion does not influence primary production and thus C inputs into the soil. Therefore, these predictions of the influence of erosion on SOC should be considered conservative.

3.4 The overall GHG balance of erosion

We demonstrated that soil erosion increased the GHG emissions of wheat through its impact on SOC dynamics, nutrient losses, inorganic fertilizer use, crop management practices and, most importantly, crop yields. The difference ranged from 0.1 to 2.5 kg eq. CO_2 kg^{-1} of wheat grain produced, depending on the yield scenario. This value was higher than the global mean of 1.1 kg eq. CO_2 kg^{-1} reported by Poore and Nemecek (2018) for wheat, and the range of 0.9–1.1 kg eq. CO_2 kg^{-1} for wheat in France in the AGRIBALYSE database (Colomb et al. 2015). Compared to the non-eroded basin, erosion alone increased on-field emissions of CO_2 (related to SOC losses) and indirect emissions from inorganic fertilizer use, while diesel consumption was higher under CT than under CA. This resulted in a compounded increase of 7% in the GHG emissions of wheat production, not considering the yield effects. On the other hand, eroded soil buries SOC and represents a net sink (Van Oost et al. 2007). CE-DYNAM predicted this sink to be 6.7 g C m^{-2} year^{-1}, corresponding to 5.5% of the wheat's life cycle GHG emissions. Factoring in this sink to obtain the overall GHG balance of the eroded scenario, the effect of C burying by eroded soil is similar to the GHG emissions due to other erosion-related processes (7%). To consider the uncertainties in modeling erosion, the LCA results estimated with the baseline model configuration were compared to those estimated with configurations that minimized or maximized soil erosion in the Ebro River basin. This uncertainty analysis yielded relative differences of less than 5% for all indicators except human toxicity (differences of 5–10%), freshwater eutrophication (20–40%), and freshwater ecotoxicity (10–20%). Thus, the uncertainty in erosion modeling had little influence on the life cycle impacts overall.

Whether erosion is a net sink or a source of C is still debated (see Doetterl et al. (2015) for a review), but its effects on the life cycle impacts of crops had not been considered until now. We show that in the Ebro River basin, even though some soil C ends up buried in river sediments through erosion, this fraction is similar to the GHG emissions directly caused by erosion and is much smaller than the overall life cycle emissions of wheat, making erosion C-neutral at best.

4 Conclusion

This study aimed to address the full GHG balance of erosion, including its impact on soil fertility and its feedback on crop yields. It is the very first one to encompass watershed-scale processes in the life cycle assessment of wheat production and to address trade-offs between soil and nutrient losses on the one hand and C burial in sediments on the other. Soil erosion had mixed effects on agricultural GHG emissions over the Ebro River basin since it was both a source of emissions and a C sink. Because these two flows had a similar magnitude, erosion appeared as a C-neutral process, a robust conclusion despite the uncertainties involved in modeling erosion and C flows. The effect of mitigating soil erosion through conservation-oriented crop management practices had a much larger effect than soil erosion on the environmental impacts of wheat due to the benefits on wheat yields. With such management, the non-eroded watershed outperformed its eroded counterpart, with 60–80% lower life cycle impacts per kg of wheat produced. These differences were again little influenced by uncertainties on erosion fluxes, except for freshwater ecotoxicity and eutrophication. Regardless of the difference in yield, soils in the eroded watershed had lost a large amount of C by the beginning of the period investigated (1985–2005), potentially offsetting 40% of wheat's GHG balance. Factoring in this legacy of past land-use changes and management practices is a complex task in retrospect. However, this model-based assessment shows that it is essential to control erosion and adopt conservation practices to preserve soil structure and crop productivity, as well as to mitigate the environmental impacts of crops in the context of climate change. The latter may indeed enhance water-related erosion processes in the future, especially in the Mediterranean area (IPCC 2022).

Data availability   Erosion simulation data and LCA input and output data are available upon request from the authors.

Code availability   Model data can be accessed from the Zenodo repository: https://doi.org/https://doi.org/10.5281/zenodo.2642452

Declarations

Ethics approval   Not applicable.

Consent to participate   Not applicable.

Consent for publication   Not applicable.

Competing interests   The authors declare no competing interests.

References

Álvaro-Fuentes J, Easter M, Cantero-Martinez C et al (2011) Modelling soil organic carbon stocks and their changes in the northeast of Spain. Eur J Soil Sci 62:685–695. https://doi.org/10.1111/j.1365-2389.2011.01390.x

Barot S, Abbadie L, Couvet D et al (2015) Evolving away from the linear model of research: a response to Courchamp et al. Trends Ecol Evol 1–2. https://doi.org/10.1016/j.tree.2015.05.005

Borrelli P, Robinson DA, Fleischer LR et al (2017) An assessment of the global impact of 21st century land use change on soil erosion. Nat Commun. https://doi.org/10.1038/s41467-017-02142-7

Bulle C, Margni M, Patouillard L et al (2019) Impact World+: a globally regionalized life cycle impact assessment method. Int J LCA 24:1653–1674. https://doi.org/10.1007/s11367-019-01583-0

Cantero-Martínez C, Angás P, Lampurlanés J (2007) Long-term yield and water use efficiency under various tillage systems in Mediterranean rainfed conditions. Ann Appl Biol 150:293–305. https://doi.org/10.1111/j.1744-7348.2007.00142.x

Colomb V, Amar S A, Basset-Mens C et al (2015) Agribalyse, the French LCI database for agricultural products: high quality data for producers and environmental labelling. OCL Oilseeds Fats Crops Lipids 22. https://doi.org/10.1051/ocl/20140047

Doetterl S, Berhe AA, Nadeu E et al (2015) Erosion, deposition and soil carbon: a review of process-level controls, experimental tools and models to address C cycling in dynamic landscapes. Earth Sci Rev 154:102–122. https://doi.org/10.1016/j.earscirev.2015.12.005

Fendrich AN, Ciais P, Lugato E et al (2022) Matrix representation of lateral soil movements: scaling and calibrating CE-DYNAM (v2) at a continental level. Geosci Model Dev 15(20):7835–7857. https://doi.org/10.5194/gmd-15-7835-2022

Frieler K, Lange S, Piontek F et al (2017) Assessing the impacts of 1.5°C global warming – simulation protocol of the Inter-Sectoral Impact Model Intercomparison Project (ISIMIP2b). Geosci Model Dev 10:4321–4345. https://doi.org/10.5194/gmd-10-4321-2017

García-Ruiz JM, Arnáez J, White SM et al (2000) Uncertainty assessment in the predition of extreme rainfall events: an example from the Central Spanish Pyrenees. Hydrol Process 14:887–898. https://doi.org/10.1002/(SICI)1099-1085(20000415)14:5%3C887::AID-HYP976%3E3.0.CO;2-0

Garrido J, Garcia JA (1992) Periodic signals in Spanish monthly precipitation data. Theor Appl Climatol 45:97–106. https://doi.org/10.1007/BF00866398

Goglio P, Smith W, Grant B et al (2018) A comparison of methods to quantify greenhouse gas emissions of cropping systems in LCA. J Cleaner Prod 172:4010–4017. https://doi.org/10.1016/j.jclepro.2017.03.133

Guenet B, Gabrielle B, Chenu C et al (2021) Can N2O emissions offset the benefits from soil organic carbon storage? Glob Change Biol 27(2):237–256. https://doi.org/10.1111/gcb.15342

Hellweg S, Milài Canals L (2014) Emerging approaches, challenges and opportunities in life cycle assessment. Science 344:1109–1113. https://doi.org/10.1126/science.1248361

Hempel S, Frieler K, Warszawski L et al (2013) A trend-preserving bias correction: the ISI-MIP approach. Earth Syst Dyn 4:219–236. https://doi.org/10.5194/esd-4-219-2013

IPCC (2022) Climate Change and Land. An IPCC Special Report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems WG I WG II WG III IPCC Special Report on Climate Change, Camb Univ Press ISBN 9781009157988. https://doi.org/10.1017/9781009157988

Kardous M, Bergametti G, Marticorena B (2005) Wind tunnel experiments on the effects of tillage ridge features on wind erosion horizontal fluxes. Ann Geophys 23(10):3195–3206. https://doi.org/10.5194/angeo-23-3195-2005

Lal R (2009) Soils and world food security. Soil Till Res 102:1–4. https://doi.org/10.1016/j.still.2008.08.001

Lampurlanés J, Plaza-Bonilla D, Álvaro-Fuentes J et al (2016) Long-term analysis of soil water conservation and crop yield under different tillage systems in Mediterranean rainfed conditions. F Crop Res 189:59–67. https://doi.org/10.1016/j.fcr.2016.02.010

Larney FJ, Janzen HH (1997) A simulated erosion approach to assess rates of cattle manure and phosphorus fertilizer for restoring productivity to eroded soils. Agric Ecosys Envir 65:113–126. https://doi.org/10.1016/S0167-8809(97)00047-9

Lizaga I, Quijano L, Gaspar L et al (2018) Estimating soil redistribution patterns with 137Cs measurements in a Mediterranean mountain catchment affected by land abandonment. L Degrad Dev 29:105–117. https://doi.org/10.1002/ldr.2843

Llasat MC (2001) An objective classification of rainfall events on the basis of their convective features: application to rainfall intensity in the northeast of Spain. Int J Climatol 21:1385–1400. https://doi.org/10.1002/joc.692

Martin-Gorriz B, Maestre-Valero JF, Almagro M et al (2020) Carbon emissions and economic assessment of farm operations under different tillage practices in organic rainfed almond orchards in sarid Mediterranean conditions. Sci Hortic-Amsterdam 261:108978. https://doi.org/10.1016/J.SCIENTA.2019.108978

Martín-Vide J, Lopez-Bustins JA (2006) The Western Mediterranean Oscillation and rainfall in the Iberian Peninsula. Int J Climatol 26:1455–1475. https://doi.org/10.1002/joc.1388

Minasny B, Malone BP, McBratney AB et al (2017) Soil carbon 4 per mille. Geoderma 292:59–86. https://doi.org/10.1016/j.geoderma.2017.01.002

Naipal V, Reick C, Pongratz J et al (2015) Improving the global applicability of the RUSLE model - adjustment of the topographical and rainfall erosivity factors. Geosci Model Dev 8:2893–2913. https://doi.org/10.5194/gmd-8-2893-2015

Naipal V, Ciais P, Wang Y et al (2018) Global soil organic carbon removal by water erosion under climate change and land use change during AD-1850-2005. Biogeosciences 15:4459–4480. https://doi.org/10.5194/bg-15-4459-2018

Naipal V, Lauerwald R, Ciais P et al (2020) CE-DYNAM (v1), a spatially explicit, process-based carbon erosion scheme for the use in Earth system models. Geosci Model Dev 13:1201–1222. https://doi.org/10.5194/gmd-2019-110

Núñez M, Antón A, Muñoz P et al (2013) Inclusion of soil erosion impacts in life cycle assessment on a global scale: application to energy crops in Spain. Int J Life Cycle Assess 18:755–767. https://doi.org/10.1007/s11367-012-0525-5

Panagos P, Borrelli P, Meusburger K (2015) A new European slope length and steepness factor (LS-Factor) for modeling soil erosion by water. Geosciences 5:117–126. https://doi.org/10.3390/geosciences5020117

Peñarrocha D, Estrela MJ, Millán M (2002) Classification of daily rainfall patterns in a Mediterranean area with extreme intensity levels: the Valencia region. Int J Climatol 22:677–695. https://doi.org/10.1002/joc.747

Peng S, Ciais P, Maignan F et al (2017) Sensitivity of land use change emission estimates to historical land use and land cover mapping. Glob Biogeochem Cycles 31:626–643. https://doi.org/10.1002/2015GB005360

Plaza-Bonilla D, Álvaro-Fuentes J, Bareche J et al (2018) No-tillage reduces long-term yield-scaled soil nitrous oxide emissions in rainfed Mediterranean agroecosystems: a field and modelling approach. Agric Ecosys Envir 262:36–47. https://doi.org/10.1016/j.agee.2018.04.007

Poore J, Nemecek T (2018) Reducing food's environmental impacts through producers and consumers. Science 360:987–992. https://doi.org/10.1126/science.aaq0216

Quine TA, Navas A, Walling DE et al (1994) Soil erosion and redistribution on cultivated and uncultivated land near las bardenas in the central Ebro river Basin, Spain. L Degrad Dev 5:41–55. https://doi.org/10.1002/ldr.3400050106

Romero R, Guijarro JA, Ramis C et al (1998) A 30-year (1964–1993) daily rainfall data base for the Spanish Mediterranean regions: first exploratory study. Int J Climatol 18:541–560. https://doi.org/10.1002/(SICI)1097-0088(199804)18:5%3C541::AID-JOC270%3E3.0.CO;2-N

Shangguan W, Dai Y, Duan Q et al (2014) A global soil data set for earth system modeling. J Adv Model Earth Syst 6:249–263. https://doi.org/10.1002/2013MS000293

Skaalsveen K, Ingram J, Clarke LE (2019) The effect of no-till farming on the soil functions of water purification and retention in North-Western Europe: a literature review. Soil Till Res 189:98–109. https://doi.org/10.1016/j.still.2019.01.004

Smith P, Davis SJ, Creutzig F et al (2016) Biophysical and economic limits to negative CO2 emissions. Nat Clim Chang 6:42–50. https://doi.org/10.1038/nclimate2870

Su Y, Gabrielle B, Beillouin D et al (2021) High probability of yield gain through conservation agriculture in dry regions for major staple crops. Sci Rep-UK 11:3344. https://doi.org/10.1038/s41598-021-82375-1

Su Y, Gabrielle B, Makowski D (2021) A global dataset for crop production under conventional tillage and no tillage systems. Sci Data 8(1):33. https://doi.org/10.1038/s41597-021-00817-x

Taylor KE, Stouffer RJ, Meehl G (2012) An overview of CMIP5 and the experiment design. Bull Am Meteorol Soc 93:485–498. https://doi.org/10.1175/BAMS-D-11-00094.1

van Groenigen JW, van Kessel C, Hungate B et al (2017) Sequestering soil organic carbon: a nitrogen dilemma. Environ Sci Technol 51:4738–4739. https://doi.org/10.1021/acs.est.7b01427

Van Oost K, Quine T, Govers G et al (2007) The impact of agricultural soil erosion on the global carbon cycle. Science 318:626–9. https://doi.org/10.1126/science.1145724

van Zelm R, van der Velde M, Balkovic J et al (2017) Spatially explicit life cycle impact assessment for soil erosion from global crop production. Ecosyst Serv. 30:220–227. https://doi.org/10.1016/j.ecoser.2017.08.015

Xu X, Thornton PE, Post WM (2013) A global analysis of soil microbial biomass carbon, nitrogen and phosphorus in terrestrial ecosystems. Glob Ecol Biogeogr 22:737–749. https://doi.org/10.1111/geb.12029

Zhang L, Huang Y, Rong L et al (2021) Effect of soil erosion depth on crop yield based on topsoil removal method: a meta-analysis. Agron Sustain Dev 41(5):63. https://doi.org/10.1007/s13593-021-00718-8

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