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1 Mixed effect of soil erosion on the environmental performance of
2 wheat crops in a large agricultural watershed

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22

23 **ABSTRACT**

24

25 Soil erosion poses a major threat to agricultural production worldwide, with a still debated impact on
26 the current build-up of atmospheric CO₂. Whether erosion acts as a net carbon (C) source or sink,
27 depends also on how it affects the greenhouse gas (GHG) emissions via its impact on crop yield and
28 nutrient loss. Accounting for these effects on the environmental performance of agricultural crops
29 remains to be done.

30 Here we combined watershed-scale erosion modelling with life-cycle assessment to evaluate the
31 impact of soil erosion on the environmental impacts of wheat production in the Ebro valley in Spain
32 under rain feed conditions. Two scenarios were simulated over the 1860-2005 time slice: an eroded
33 water-shed corresponding to conventional agricultural practices, and a non-eroded control involving
34 conservation practices such as no-till. Life-cycle assessment followed a cradle-to-farm-gate approach,
35 with impacts expressed per kg of grains produced, and a focus on recent decades (1985 to 2005).

36 Soil erosion losses averaged to 2.6 tons ha⁻¹ yr⁻¹, resulting in a net annual loss of 7 kg N and 5.5 kg
37 P₂O₅, respectively, compared to the non-eroded control. Soils in both watersheds simulations (eroded
38 and non-eroded) lost organic C over time, with the eroded configuration emitting an extra 55 kg CO₂
39 ha⁻¹ yr⁻¹. This net C source in the non-eroded simulation only made up 3% of the overall life-cycle
40 GHG emissions of wheat grains, while the compensation for nutrient losses contributed a similar
41 fraction. Yield difference between the conventional and conservation practices proved to be the most
42 sensitive parameter, reaching up to 60% increase when implementing conservation practices. Even at
43 catchment scale, erosion induce by conventional practices did not emerge a net C sink, and increased
44 the GHG emissions of wheat by 7 to 70%. Erosion control through soil conservation practices is still
45 strongly recommended from the combined perspectives of soil preservation, crop yield enhancement
46 and the mitigation of GHG emissions from agriculture.

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55 **Keywords:** Agricultural soils, Ebro valley, Greenhouse gas emissions, Life-cycle assessment, Soil
56 erosion, Wheat, Erosion modelling, RUSLE, soil carbon dynamics

57

58

59 **1 Introduction**

60 Anthropogenic emissions of greenhouse gases (GHGs) need to be rapidly curbed to mitigate
61 climate change and achieve the targets laid out in the 2015 Paris agreement (Smith et al. 2016).
62 Carbon sequestration in soils has been identified as an important land based mitigation option, which
63 is at the basis of the international “4 per 1000” initiative aiming to increase carbon storage in
64 agricultural soils in order to partially compensate anthropogenic emissions (Minasny et al. 2017). It
65 can be achieved through an increase C and nutrient inputs to soils via organic matter amendment, the
66 establishment of cover crops or agroforestry systems, among others (Guenet et al., 2021). However, a
67 reduction in soil erosion is also necessary to preserve soil C and nutrients stocks. As a consequence,
68 lateral fluxes due to soil erosion processes should be factored in when evaluating soil C sequestration
69 potentials. According to the recent special report of IPCC on Climate Change and land (IPCC, 2019) a
70 quarter of all lands are degraded and erosion is the most widespread cause for degradation. Currently,
71 agricultural soils feature the highest soil erosion rates, which are accelerated by the removal of
72 vegetation and ground cover, and by soil tillage practices (IPCC, 2019; Van Oost et al., 2007). Soil
73 erosion on agricultural land has substantial impacts on soil fertility and food security (Lal 2009).

74 The role of erosion (mainly due to water) as a sink or source of atmospheric CO₂ is currently
75 highly debated in the literature (Van Oost et al. 2007; Lal 2009; Borrelli et al. 2017). Erosion causes a
76 movement of material along the cascade of hillslopes, floodplains and rivers. This redistribution of soil
77 material may lead, on the one hand, to C sequestration in deposition areas such as colluvial and
78 alluvial soils due to (1) the replacement of eroded carbon by new photosynthetic substrate on eroded
79 land, (2) burial of eroded C in deeper soil layers, hereby reducing soil carbon respiration. On the other
80 hand, soil decomposition can be intensified by soil erosion during sediment transport and after
81 deposition due to aggregate breakdown, and thus lead to higher CO₂ emissions. Despite this
82 uncertainty in erosion being a C sink or source, the above-mentioned studies showed that the effect of
83 erosion on the C emissions cannot be ignored in C cycle research.

84 To estimate the impacts of erosion and its drivers, mechanistic modelling is a promising way and
85 large efforts were done in the recent years to develop soil erosion models (Barot et al. 2015).
86 Nevertheless, these approaches are generally developed to represent processes occurring at a fine scale
87 and might not be adapted to larger scales. Large scale soil erosion is usually estimated based on the
88 Universal Soil Loss Equation Model (RUSLE) (Panagos et al. 2015; Naipal et al. 2018). The RUSLE
89 model structure also facilitates the linkage of erosion processes with the C cycle processes on land.
90 For example, different versions of the RUSLE model have been linked to soil organic matter models
91 such as CENTURY (Lugato et al., 2018), and to land surface models (Naipal et al., 2020). However,
92 these modelling approaches do not yet include the erosion-induced feedback on plant productivity,
93 which may lead to an overestimation of the C input to the soil in eroding regions.

94 Life-cycle-assessment (LCA), is a widely used, holistic approach to estimate the environmental
95 impacts of a given practice or product (Hellweg and Milà i Canals, 2014). It provides a suitable
96 framework to understand the multi-faceted effects of soil erosion on GHG emissions and other
97 environmental issues. It has been used to evaluate the impacts of land use on soil erosion (Núñez et al.
98 2013; van Zelm et al. 2017), or the effect of tillage practices and erosion control on the GHG balance
99 of crops (Martin-Gorriz et al. 2020), but not on the reciprocal issue of how erosion affects the
100 environmental performance of crops. This is understandable since most LCA of agricultural crops are
101 done at the plot or farm scale (Poore and Nemecek 2018), whereas to estimate the GHG balance of
102 erosion fluxes it must be captured at the catchment scale to estimate removal and redeposition, using
103 for instance the modelling approach described above.

104 Here, we applied the LCA methodology to better understand the environmental impacts of wheat
105 production as affected by soil erosion or not at the scale of the Ebro watershed in Spain. This large
106 catchment (85 550 km² in size) was selected because it has been exposed to erosion (Cantero-Martínez
107 et al. 2007), and cropland accounts for one third of its area. The main objective was to assess the
108 impacts of soil erosion processes induced by rainfall and runoff on the environmental performance of
109 the production of wheat (a common crop in the area), by combining plot-scale LCA with catchment-
110 scale erosion modelling. The mechanistic model of soil and C erosion, CE-DYNAM, a combination of
111 the soil erosion model RUSLE with a carbon emulator of a land surface model (Naipal et al., 2020)
112 provided an estimate of net erosion (erosion losses minus deposition) from agricultural land on the
113 hillslopes of the Ebro basin. Two crop management scenarios were compared: conservation agriculture
114 (aimed to control erosion), referred to as CA, and conventional management (referred to as CT for
115 conventional tillage). Yield differences were factored in using two main sources of information: plot-
116 scale observations versus a model derived from a global meta-analysis comparing CA and CT.
117 historically (Cantero-Martínez et al. 2007; Su et al. 2021), and cropland accounts for one third of its
118 area

119

120 **2 Materials and Methods**

121

122 **2.1 The Ebro watershed**

123 The watershed of the Ebro river basin extends over the autonomous communities of Cantabria,
124 north of Castilla y Leon in the provinces of Burgos and Soria, south of Pais Vasco in the provinces of
125 Alava, La Rioja, Navarra, Aragón, Cataluña and Castellón province in the Comunidad Valenciana.
126 The climate is continental-Mediterranean with hot and dry summers followed by cold winters. Main
127 rainfall events are observed in spring (April and May) and autumn (September and October), and
128 summer droughts occur between these periods. The Ebro basin is a closed sedimentary basin where
129 climate, tectonics and soil erosion played an important role in its formation. Currently, the Ebro basin

130 is characterized by tow mountain ranges, the Pyrennees in the north and Iberian System mountain
131 ranges in the south, that surround a large alluvial plain. Therefore, soil erosion in this basin has been
132 especially affected by the steep slopes, extensive agriculture and arid /temperate climate.

133

134 **2.2 Erosion modelling**

135 Carbon erosion and deposition flows over the Ebro watershed were estimated with the novel large-
136 scale Carbon Erosion Dynamics model Version 1 (CE-DYNAM-v1) (Naipal et al. 2020). CE-
137 DYNAM consists of a sediment budget scheme coupled to an emulator of the soil carbon scheme of
138 the global Land Surface Model (LSM) ORCHIDEE. CE-DYNAM as a whole has been originally
139 calibrated for the Rhine catchment, but its model structure and parameters are designed in a way to be
140 applicable for other river catchments globally. The soil erosion module of CE-DYNAM is based on
141 the Adjusted Revised Soil Loss Equation (RUSLE) model (Naipal et al. 2015), where the yearly
142 average soil erosion removal is calculated as a product of the topographical factor (S), the rainfall
143 erosivity factor (R), the land cover and management factor (Cm), and the soil erodibility factor (K).
144 The Cm factor includes the effects of vegetation cover and crop residues on soil erosion for tilled soil
145 (conventional agriculture). The effect of conservation practices on erosion are not taken into account
146 here. The simulated yearly soil erosion rate (E) represents the potential gross removal of soil from
147 eroding hillslopes.

148

$$E = S * R * K * Cm$$

149

150 The daily gross carbon removal rate by soil erosion (Ce) can then be calculated as following:

$$Ce = \frac{\frac{SOC}{365} * E * 100}{BD_{top} * dz * 10^6} * CER$$

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

155 A fraction (f) of the eroded soil gets deposited at the foot of the hillslopes (colluvial deposits).

156 This fraction is a function of the slope gradient and the vegetation type.

157

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

158 Where a_f and b_f are constant parameters that relate f to the average topographical slope (θ) of a grid
159 cell depending on the type of land cover. These parameters have been derived from the study of Naipal
160 et al. (2020). θ_{max} is the maximum topographical slope of the watershed. The rest of the eroded flux is

161 transported to the floodplains and water network. In this study we will only consider the soil erosion
162 and deposition, with resulting C dynamics on hillslopes.

163 CE-DYNAM has three SOC pools, labelled active, slow and passive, respectively. These
164 pools differ by their C residence time, which is shortest for the active pool and longest for the passive
165 one. Changes in each SOC pool on the hillslopes are a function of the C input to the soil by litter
166 decomposition (I), the C decomposition rate (k), which is dependent on the soil physical parameters
167 such as temperature, and the net erosion rate (k_E), which is 1 minus the gross soil erosion (E) times the
168 deposition rate (f). The I and k are landcover type specific and derived from the ORCHIDEE LSM on
169 which forms the basis of the SOC scheme of CE-DYNAM.

170

$$\frac{dSOC(t)}{dt} = I(t) - (k + k_E) * SOC(t)$$

171 For a more detailed description of the model see Naipal *et al.* (2020).

172

173 CE-DYNAM was run for the period 1850-2005, with and without soil erosion dynamics, with
174 a daily time-step at a spatial resolution of 5 arcminutes (~ 8km). First the model was run to a steady
175 state based on the environmental conditions of the period 1850-1860, and afterwards transient
176 simulations were performed with changing climate and land use conditions. The overall change in
177 SOC stocks over the period 1850-2005 is then a result of climatic changes, increasing atmospheric
178 CO₂ concentrations, land use change and soil erosion. The resulting net soil C emissions due to
179 erosion can be calculated as the difference in the cumulative SOC change over the period 1850-2005
180 between the erosion and no-erosion simulations. Here we consider the C fluxes provided by CE-
181 DYNAM when erosion is activated as a surrogate of conventional agriculture practices whereas
182 simulations performed when erosion is not activated are surrogate for conservation agriculture
183 practices that limit erosion fluxes.

184

185 2.3 Input data for CE-DYNAM

186 The input datasets for CE-DYNAM as described below are the same as in Naipal *et al.*, (2020).
187 The land cover fractions are derived from the historical 0.25° maps of Peng *et al.*, (2017). Daily
188 precipitation data for the period 1850–2005 is derived from the Inter-Sectoral Impact Model
189 Intercomparison Project (ISIMIP), product ISIMIP2b Frieler *et al.*, (2017). These data are based on
190 model output of the Coupled Model Intercomparison Project. Phase 5 (CMIP5 output of IPSL-
191 CM5A-LR (Taylor et al. 2012), which are bias corrected using observational datasets and the method
192 of Hempel et al. (2013) and made available at a resolution of 0.5 °. Data on soil bulk density and other
193 soil parameters to calculate the soil erodibility factor (K), available at the resolution of 1 km, have
194 been taken from the Global Soil Dataset for use in Earth System Models (GSDE) (Shangguan et al.
195 2014). Finally, the slope steepness factor (S), was originally estimated at the resolution of 1 km, based

196 on the GTOPO30 digital elevation model. All datasets that were not available at a 5 arcminute
197 resolution were regridded to this specific spatial model resolution before calculating the erosion rates.
198

199 2.4 Life-Cycle Assessment

200

201 2.3.1 Goals and scope

202 The objectives of the LCA were two-fold: to appraise the contribution of soil erosion processes to
203 the GHG balance of wheat, and to examine the effect of erosion control practices on the environmental
204 performance of this crop in the context of the Ebro basin. To single out the effect of soil erosion, two
205 sets of simulations were carried out with the catchment-scale model CE-DYNAM: a scenario with
206 minimum erosion (reflecting erosion control practices) and a scenario with standard, default erosion
207 (reflecting conventional crop management practices thus far). These simulations correspond to
208 “eroded” and “non-eroded” watersheds, respectively, associated with CT and CA management
209 practices.

210 To compensate for the loss of crop productivity associated with the losses of soil material and
211 nutrients, fertilizer input rates were adjusted in the eroded case based on the estimated rates of N and P
212 losses. These were estimated from the net C losses output by the erosion model assuming C:N and C:P
213 ratios of 10 and 38, respectively, in eroded material (Xu et al., 2013). Nevertheless, we did not
214 represent the effect of soil erosion on physical and hydraulic parameters (e.g. porosity, water holding
215 capacity).

216 We used cradle-to-farm-gate system boundaries and the functional unit for this system was the
217 production of one kilogram of wheat. The time-frame considered in the inventory extended from the
218 harvest of the previous crop to the harvest of wheat.

219 2.3.2 Life-cycle inventory (LCI)

220

221 LCA was carried out with the SimaPro software (SimaPro 8.0.3, Pré Sustainability, LE
222 Amersfoort, NL), and two LCI databases: AGRIBALYSE (Colomb et al., 2015), which focuses on
223 agricultural products in France, and EcoInvent (v3.6; Ecoinvent, Zurich) for background processes.

224 Wheat production was adapted to the Ebro valley conditions based on the management and yield
225 data of Cantero-Martínez et al. (2007), which reports on various tillage systems at 3 field sites in the
226 Ebro river valley. The no-till system was used for the non-eroded watershed modelling while a
227 cropping system involving mouldboard plough was associated with the eroded watershed. The
228 “Agramunt” site was chosen since it involved wheat crops every other year from 2000 to 2005. Crops
229 received organic fertilizer inputs in this trial. However, since inorganic fertilizer types predominate in

230 the Ebro catchment, organic input rates were converted into equivalent mineral fertiliser inputs based
231 on the fertilising values of manure and slurry (Table 1).

232 Wheat yields are subject to large inter-annual and spatial variations across the Ebro basin
233 (Cantero-Martinez et al., 2007; Plaza Bonilla et al., 2018). Since yield plays an important role in the
234 outcome of LCAs, we used various sources of information and scenarios to determine crop yields for
235 the conservation and standard tillage practices.

236 In the first yield scenario, the wheat yield in conventional agriculture (eroded watershed) was
237 calculated from the average of the wheat yields of each province and autonomous community in the
238 Ebro catchment area in Spain from 1996 to 2006. Yield data were taken from the annual report of the
239 Ministry of Agriculture, Fisheries and Food of the Spanish Government. The average wheat yield in
240 the eroded watershed analysis was 3283 kg grains ha⁻¹. Then, to assess the yield in conservation
241 agriculture, we used the results of a model built from a meta-analysis comparing the yields of
242 conventional (CT) and conservation agriculture (CA) wheat worldwide (Su et al., 2021), using
243 machine-learning and a global data-base. The average yield change ratio between CA and CT
244 $\left(\frac{Yield_{CA}-Yield_{CT}}{Yield_{CT}}\right)$ from 1996 to 2006 was estimated at -0.010 over the Ebro watershed area. This led to
245 an estimate of 3250 kg grains ha⁻¹ for the yield of wheat under conservation agriculture practices. This
246 scenario has the advantage of relying on catchment-scale data and a global model. Nevertheless, yields
247 vary widely depending on climate, soil conditions and crop management characteristics. In some
248 particular cases, conservation agriculture achieves higher yields than its conventional counterpart, as
249 reported by Lampurlanés *et al.*, (2016) in the Ebro basin. We therefore developed a second yield
250 scenario based on these observations One of the main reasons for this is that CA not only reduces
251 erosion but also improves the water-holding capacity of the soil. The average wheat yield considered
252 in this scenario was averaged over 7 years at the studied region and was 2314 kg ha⁻¹ for the no-till
253 system (non-eroded watershed) and 1436 kg ha⁻¹ for the CT system (eroded watershed). Lampurlanés
254 *et al.*, (2016) in the Ebro basin. We therefore developed a second yield scenario based on these
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256 the water-holding capacity of the soil. The average wheat yield considered in this scenario was
257 averaged over 7 years at the studied region and was 2314 kg ha⁻¹ for the no-till system (non-eroded
258 watershed) and 1436 kg ha⁻¹ for the CT system (eroded watershed).

259 The net effect of erosion on soil C variations at the field scale was derived from the transient CE-
260 DYNAM simulations, averaged over the 1985-2005 time slice. This erosion-induced C sink/source
261 term for hillslopes compounded the following processes and fluxes: C replacement on eroding soils by
262 new litter input (photosynthesis), C burial in deposition sites, increased decomposition of deposited
263 SOC and the lateral C flux that leaves the hillslopes of the Ebro basin. The floodplains were excluded
264 from the analysis, and it was assumed that the C flux leaving the hillslopes was entirely respired and
265 lost.

266

267 2.3.3 Impact characterization

268 Environmental impact assessment was carried out with the IMPACT World+ characterization
269 method (IMPact Assessment of Chemical Toxics), as integrated in the SimaPro software (Bulle et al.
270 2019). This method allows analysing the results with 15 midpoint categories which are linked to 4
271 damage categories. The following midpoint categories were included: human toxicity (cancer and non-
272 cancer), respiratory inorganics, ionizing radiation, ozone layer depletion, respiratory organics, aquatic
273 ecotoxicity, terrestrial acidification, land occupation, aquatic acidification, aquatic eutrophication,
274 global warming, non-renewable energy and mineral extraction.

275

276 **3 Results and discussion**

277 **3.1 Evaluation of the erosion model**

278 We show that erosion has an important impact on the total SOC stock averaged for the period
279 1995-2005 in crop fields at 30cm depth (Fig. 2a) with a total SOC stock of 86.5 t C ha⁻¹ when erosion
280 was set to zero and 11.4 t C ha⁻¹ when erosion was considered. It must be noted here that the big
281 differences between the two simulations is partially due to the fact that maximized erosion is
282 considered during the entire simulations (1860-2005) whereas it was calibrated on modern agriculture
283 techniques. Thus, we probably overestimate the erosion fluxes for the first decades of the simulation
284 since mechanization of agriculture responsible for higher erosion rates started in this region in the
285 1940's. To evaluate the model we extracted data on SOC stocks from the GSDE product (Shangguan
286 et al. 2014) when plant cover was at least 30% crop using the land cover maps from Peng *et al.*,
287 (2017). We calculated over the studied crop regions an average SOC stock at 30cm depth of 67.3 t C
288 ha⁻¹. This lies in between the values predicted by the model with or without erosion, knowing that the
289 model simulations are extreme cases considering no erosion at all or erosion rates calibrated over
290 conventional agriculture without taking into account the erosion control measures that can be locally
291 deployed in the fields. We also compared our simulated erosion rates over the Ebro basin with
292 findings of other studies (Quine et al. 1994; Lizaga et al. 2018). In Fig. 2b, we compared the gross
293 erosion rate predicted by the CE-DYNAM model and the rate estimated by Lizaga *et al.*, (2018) using
294 ¹³⁷Cs tracer. Our model underestimates the gross erosion rate compared to the rate derived from Lizaga
295 *et al.*, (2018). The model predicts a gross soil erosion rate ranging from 6.6 to 18.2 t ha⁻¹ yr⁻¹
296 (equivalent to a range from 167.8 to 371.7 kg C ha⁻¹ yr⁻¹), whereas Lizaga *et al.*, (2018) estimated a
297 gross soil erosion rate over their studied region to be 25.4 t ha⁻¹ yr⁻¹. It is important to note that Lizaga
298 *et al.*, (2018) did not estimate the soil gross erosion rate over the full Ebro basin region but over a sub
299 basin of 23 km² located in Navarra province that is a Mediterranean-continental climate suffering of
300 Atlantic influence receiving more rainfall than in the central Ebro valley area. This sub basin is also
301 generally steeper than the central Ebro valley area. The model-observation errors may therefore be

302 explained the differences between the two studied regions. However, also the fact that we assume a
303 model steady state in the period 1850-1860 for CE-DYNAM may contribute to the underestimation in
304 erosion. It is important to take into account the past changes in climate and land use as possible in
305 order to be able to better predict the current state of soil erosion but the absence of precise knowledge
306 on previous land use and land management makes this task very complex. Furthermore, the SOC is
307 derived from a simulation performed by the coarse resolution ORCHIDEE LSM and the plant cover is
308 derived from a global map, which may also contribute to the discrepancies. Finally, net soil erosion
309 rate from CE-DYNAM can also be compared to observations derived from Lizaga *et al.*, (2018) and
310 from Quine *et al.* (1994). In this case the model predicts a net soil erosion rate ranging from 1.7 to 4.7
311 $\text{t ha}^{-1} \text{yr}^{-1}$ (equivalent to a range from 43 to 95.5 $\text{kg C ha}^{-1} \text{yr}^{-1}$) which is lower than the rate derived
312 from Lizaga *et al.*, (2018) who estimated a net soil erosion rate of 8.3 $\text{t ha}^{-1} \text{yr}^{-1}$. This could be
313 expected as the gross erosion rate is already underestimated by the CE-DYNAM model. Quine *et al.*
314 (1994), also using the ^{137}Cs tracer method, estimated a rate ranging from 16 to 25 $\text{t ha}^{-1} \text{yr}^{-1}$ for a small
315 sub basin of 0.52 km^2 . The representativeness of such sub basins compared to the entire Ebro basin is
316 therefore not guaranteed. Finally, since conservation agriculture practices are not implemented in the
317 CE-DYNAM model that can run with no erosion or with maximized erosion for every grid cells, the
318 performances of the model are reasonable. Net erosion rates might be under estimated by CE-
319 DYNAM but the absence of data over the entire Ebro basin makes the evaluation of the model not
320 straightforward.

321

322 **3.2 Contribution of soil erosion to the GHG balance of wheat**

323 Based on the CE-DYNAM simulations of soil organic C variations over the recent (1985-2005)
324 time slice, cropland soils emerged as net sources of CO_2 , with an estimated magnitude of 215 and 249
325 $\text{kg CO}_2 \text{ha}^{-1} \text{yr}^{-1}$ for the non-eroded and eroded watersheds, respectively. It is notable that despite the
326 large effect of erosion on soil C stocks (Fig. 2), the losses experienced since the 1860 starting point in
327 the eroded watershed were still more than compensated by the rates of C burial in deposition sites.
328 Indeed, erosion drastically affect the spatial distribution of SOC with higher SOC stocks in floodplain
329 and lower SOC stocks in hillslopes. Ultimately, this sink effect was no longer active in the non-eroded
330 configuration.

331 Figure 3 puts these soil C sources in perspective by reporting the life-cycle GHG emissions of
332 wheat production for yield scenario #1, per kg of wheat grains harvested. Overall, they only accounted
333 for 5% of life-cycle GHG emissions. Their magnitude fits with the 0 to 1700 $\text{kg CO}_2 \text{ha}^{-1} \text{yr}^{-1}$ range
334 reported by Goglio *et al.* (2018) in their survey of the life-cycle GHG balances of wheat crops
335 worldwide, although some studies based on direct field observations of SOC variations pointed to
336 larger sink or source strengths (from -3.5 to 5 $\text{Mg CO}_2 \text{ha}^{-1} \text{yr}^{-1}$). Conservation agriculture in general is
337 assumed to favour soil C sequestration, and Alvaro-Fuentes *et al.* (2011) estimated it could store

338 between 200 and 500 kg C ha⁻¹ yr⁻¹ (730 to 1 830 kg CO₂ ha⁻¹ yr⁻¹) whereas conventional systems were
339 simply neutral. The fact that erosion simulations started prior to 1985 (the first year of the 20-year
340 sequence used here for SOC variation calculations) and that CA and CT systems had largely different
341 SOC contents at that time (reflecting the “legacy” of past erosion processes) certainly mitigated the
342 sink capacity of CA. Factoring in the past (1860 to 1985) differences in SOC simulated with CE-
343 DYNAM between the two systems would amount to attributing an extra 480 kg C ha⁻¹ yr⁻¹ sink term to
344 the CA system. This would offset 40% of the life-cycle GHG emissions of the CA systems, which are
345 equivalent to 1210 kg C ha⁻¹ yr⁻¹. Along with potential soil C sink terms, Figure 4 highlights the
346 importance of N₂O emissions (which made up 85% of direct field emissions of GHG), and the use of
347 chemical fertilizers on the GHG balance of wheat. Nitrogen fertilizers bore the largest impact,
348 followed by phosphorus and potassium fertilizers. The 35% share of these inputs in the overall carbon
349 footprint was similar to the 30% ratio reported by Poore and Nemecek (2018). From a fertilizer use
350 perspective, another consequence of erosion is the need to compensate for nutrient losses. This
351 adjustment has a two-fold impact: a rise in off-farm emissions due to the manufacturing and
352 transportation of extra inputs, and an increase in the field emissions of N₂O derived from fertilizer N.
353 Compared to the non-eroded scenario, the eroded case resulted in a 5% relative increase for both
354 terms, and thus a 10% relative increase overall (Figure 3).

355 Wheat production with CT practices required more frequent use of agricultural machinery
356 (ploughing, fertilization) inducing a slight increase in CO₂ emissions. Indeed, diesel consumption was
357 3-times higher with CT than with CA, but contributed only a minor fraction (under 10%) of the
358 emissions. Also, erosion per se lead to higher impacts for some of the impact categories due to the
359 pollution of surface water through runoff of nutrients, organic matter and chemicals. This was
360 particularly apparent for freshwater eutrophication and ecotoxicity, which were 25% lower under CA
361 (Figure 3 and Table 2). Conversely, the loss of soil with CT mitigated to a minor extent (5%) the
362 impact of CA wheat on human toxicity (non cancer) because a reduced exposure of human consumers
363 to contaminated soils.

364 Here, we considered that only mineral forms of fertilizers inputs whereas organic ones may impact
365 the productivity and the SOC dynamics of cropland. For instance, Larney & Janzen, (1997) showed
366 that organic fertilizers were more effective at supplying P to crops in eroded areas than the input of
367 phosphates, which were immobilized by calcium carbonates. Comparing inorganic and organic
368 fertilizers to maintain crop productivity on eroded lands could bring interesting insight into the
369 environmental performance of cereals in eroded catchments.

370

371 **3.3 Erosion and crop yields**

372 Figure 3 compares the environmental impacts of wheat production under CA and CT using a first
373 scenario in which yields differed by only 1% between both managements. Under this assumption of

374 similar yields, soil erosion *per se* caused higher emissions to the air and water compartments in
375 general, and resulted in 10 to 25% larger impacts for all categories except human toxicity. With the
376 second scenario based on cropping systems trials comparing CA and CT, differences between the two
377 managements and their associated erosion configurations were much larger, reaching 60 to 80% (Fig.
378 3b), again with the exception of human toxicity (non cancer). This magnitude could be expected from
379 the 61% relative yield difference between CA and CT wheat (Table 1), and the differences in
380 environmental impacts reported in the previous sections, whereby CA outperformed CT because of
381 soil erosion processes.

382 Overall, experimental trials on the effect of soil erosion rates on crop yields are inconsistent, but a
383 recent meta-analysis pointed out that they were negative when erosion depth dropped under 10 cm
384 (Zhang et al. 2021). In the Ebro valley, the soil losses simulated by CE-DYNAM in the eroded
385 watershed (averaging 1.5 t ha⁻¹ yr⁻¹) indicate a rapid depletion of the topsoil and point at an erosion
386 depth larger than this threshold (see also picture on Figure 1). This adverse effect on crop productivity
387 is backed by the field data used for yield scenario 2, with the additional benefits of soil conservation
388 practices for the CA/non eroded watershed configuration. These benefits are known to be larger under
389 semi-arid conditions (Su et al., 2021), even though the model we used only pointed to a marginal
390 effect. The CA system also featured less intensive management practices than its conventional
391 counterpart, related to the absence of tillage and reduced fertilizer inputs. No-till is known to trade-off
392 with other practices such as weed management, and collecting management data representative of the
393 area is paramount to produce reliable results.

394 Finally, an important point to mention is that our estimation of erosion from CE-DYNAM does
395 not consider the feedbacks between erosion and primary productivity. CE-DYNAM is a C-only model
396 and the loss of soil due to erosion only remove C whereas in reality soil erosion also remove N and P
397 and affect soil physical structure, water infiltration etc. Therefore, the higher SOC loss due to erosion
398 over the 1985-2005 period given by CE-DYNAM can be underestimated because primary production
399 and therefore C inputs into the soil is not affected by erosion. Our estimation of erosion effect on SOC
400 should be therefore considered as an optimistic estimation likely underestimated.

401

402 **3.4 The full GHG balance of erosion**

403

404 Here we demonstrated that soil erosion increased the GHG emissions of wheat through its impact
405 on SOC dynamics, nutrient losses, chemical fertilizer use and crop management practices and, most
406 importantly, crop yields. The difference ranged between 0.1 and 2.5 kg eq. CO₂ kg⁻¹ of wheat grains
407 produced depending on the yield scenario. This is quite significant considering the world average of
408 1.1 kg eq. CO₂ kg⁻¹ reported by Poore and Nemecek (2018) for this crop, and the average value of 0.89
409 1.1 kg eq. CO₂ kg⁻¹ reported by the AgriBalyse database in France (Colomb et al., 2015). Compared to

410 the control watershed, erosion on its own increased on-field emissions of CO₂ (related to SOC losses),
411 indirect emissions from chemical fertilizer use while diesel consumption was higher with conventional
412 crop management than with soil conservation practices. This resulted in a compounded increase of 7%
413 in the GHG emissions of wheat production, yields effects notwithstanding. On the other hand, erosion
414 leads to the burying of soil organic carbon and represents a net sink (Van Oost et al. 2007). Using the
415 simulations done by CE-DYNAM we calculated this sink to be 6.7 g C m⁻² yr⁻¹, which corresponds to
416 5.5% of the wheat's life-cycle emissions. Factoring in this sink to obtain the "full" GHG balance of
417 the eroded scenario, we find that the effect of C burying soil is commensurate with the GHG emissions
418 due to other erosion-related processes (which a share of 7%).

419 Whether erosion is a net sink or a source of C is still debated (see Doetterl et al. 2015 for review)
420 but its effects on the life-cycle impacts of crops had hitherto not been taken into account. We show
421 that in the Ebro watershed, even though a fraction of soil C ends up buried in river sediments through
422 erosion, this fraction is on a part with the GHG emissions directly caused by erosion, and small
423 compared to the overall life-cycle emissions of wheat, making erosion carbon-neutral at best.

424

425 **4 Conclusion**

426 Soil erosion had mixed effects on agricultural GHG emissions over the Ebro watershed. From a
427 life-cycle perspective, taking into account all the processes involved in wheat production in this area,
428 erosion was both a source of emissions and a sink through the burial of C in river sediments. These
429 two terms being of similar magnitude, erosion *per se* appeared as a carbon-neutral process. The major
430 differences between the eroded and non-eroded watersheds occurred with the scenario assuming a
431 large yield gap between the conventional and conservation-orientated crop management practices. In
432 this case the zero-erosion control clearly outperformed its eroded counterpart, with 60 to 80% lower
433 life-cycle impacts per kg of wheat produced. Notwithstanding yield differences, soils in the eroded
434 watershed had lost a large amount of carbon by the beginning of the time period investigated here
435 (1985 to 2005), leading to an overall loss of c. 0.5 t C ha⁻¹ yr⁻¹ relative to the un-eroded control,
436 potentially offsetting 40% of the wheat's C footprint. Factoring in this legacy of past land use changes
437 and management practices is not an easy task in retrospect. However, our modelling and assessment
438 shows that it is important to control erosion and to adopt conservation practices to preserve soil
439 structure, crop productivity, and mitigate the environmental impacts of crops in the context of climate
440 change.

441

442 **Declarations**

443

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- 455 • Authors' contributions:
456 C.R., B.Ga and N.G. ran the LCA models, V.N. ran the CE-DYNAM model, B.Ga, B. Gu and N. G.
457 designed the study, C.C-M provided data. All the authors contributed to the model analysis and to the
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459

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461

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465

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575 Table 1. Crop management data for wheat production on the non-eroded and eroded watersheds
 576 (corresponding to conventional tillage – CT – and conservation agriculture – CA – practices,
 577 respectively), for yield scenario #1. The extra N and P inputs for CT compared to CA compensate for
 578 the net loss of organic matter through erosion.

579

| | Non-eroded watershed (CA) | Eroded watershed (CT) |
|--------------------------------------|--|---|
| Crop rotation | Barley from 1990 to 2000 then wheat one year in two until 2005 | |
| Yield in scenario #1 | 3250 kg grain DM ha ⁻¹ | 3283 kg grain DM ha ⁻¹ |
| Yield in scenario #2 | 2314 kg grain DM ha ⁻¹ | 1436 kg grain DM ha ⁻¹ |
| Tillage | No-till | 1 mouldboard ploughing (30-35 cm) 1/2 cultivator passes (15cm) 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 P2O5 ha ⁻¹ 142 kg K2O ha ⁻¹ | (134 + 6) kg N ha ⁻¹ (81.5 + 4) kg P2O5 ha ⁻¹ 142 kg K2O ha ⁻¹ |
| Disease and pest management | No treatment | |

580 ,

581 Table 2: Life-cycle impacts of the production of one kg of wheat in the non-eroded (CA) and eroded
582 (CT) Ebro watersheds, for yield scenario #1, expressed at mid-point level.
583 Key to units (see Bulle et al. 2019 for more details): CTUh, CTUe: comparative toxic units for humans
584 and ecosystems, respectively; NMVOC: non-methane volatile organic compounds.
585

| Impact category | Unit | Non-eroded watershed (CA) | Eroded watershed (CT) |
|----------------------------------|------------------------------|----------------------------------|------------------------------|
| Human toxicity (cancer) | CTUh | $2.87 \cdot 10^{-9}$ | $2.76 \cdot 10^{-9}$ |
| Human toxicity (non-cancer) | CTUh | $3.23 \cdot 10^{-8}$ | $3.40 \cdot 10^{-8}$ |
| Particulate matter formation | kg PM2.5 eq. | $4.12 \cdot 10^{-4}$ | $4.19 \cdot 10^{-4}$ |
| Ionizing radiation | Bq C-14 eq. | 1.8 | 1.9 |
| Ozone layer depletion | kg CFC-11 eq. | $5.37 \cdot 10^{-8}$ | $6.08 \cdot 10^{-8}$ |
| Photochemical oxidants formation | kg NMVOC eq. | $3.10 \cdot 10^{-3}$ | $3.63 \cdot 10^{-3}$ |
| Freshwater ecotoxicity | CTUe | 59.2 | 95.8 |
| Terrestrial acidification | kg SO ₂ eq. | $1.18 \cdot 10^{-5}$ | $1.25 \cdot 10^{-5}$ |
| Freshwater acidification | kg SO ₂ eq. | $1.09 \cdot 10^{-8}$ | $1.25 \cdot 10^{-8}$ |
| Freshwater eutrophication | kg PO ₄ P-lim eq. | $1.13 \cdot 10^{-4}$ | $3.20 \cdot 10^{-4}$ |
| Marine eutrophication | kg N eq. | $1.63 \cdot 10^{-4}$ | $1.74 \cdot 10^{-4}$ |
| 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 | $1.00 \cdot 10^{-2}$ | $1.13 \cdot 10^{-2}$ |
| Water scarcity | m ³ world eq. | 24.4 | 26.1 |

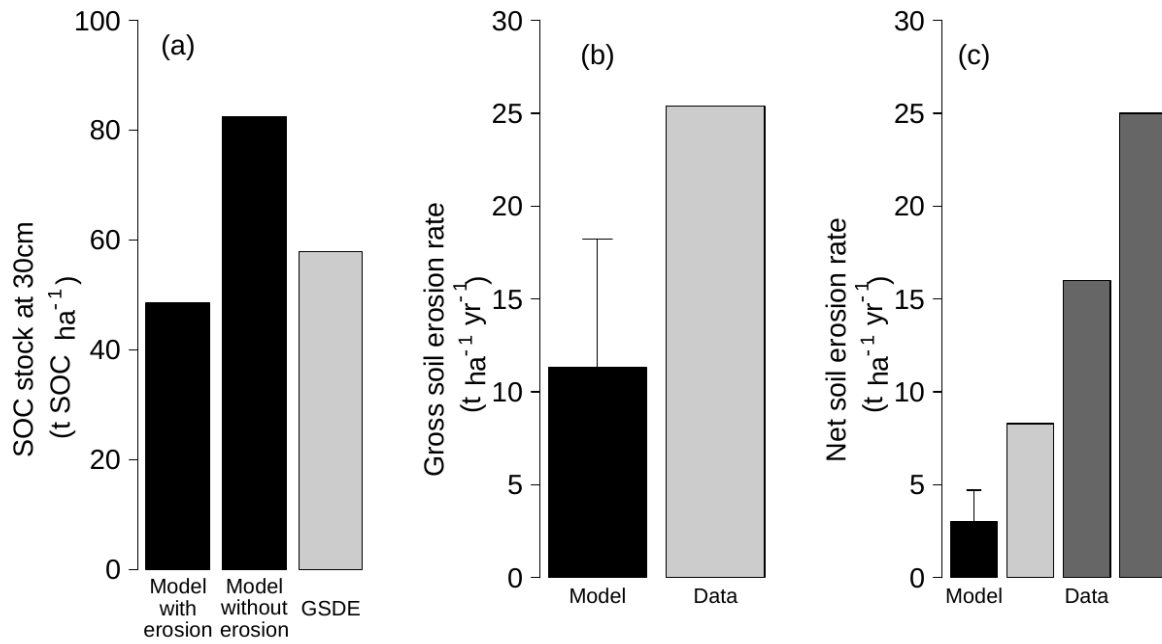
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590 Figure 1: Evidence of soil erosion processes on agricultural land in the Ebro valley (credit: Julien
591 Tournebize, INRAE).



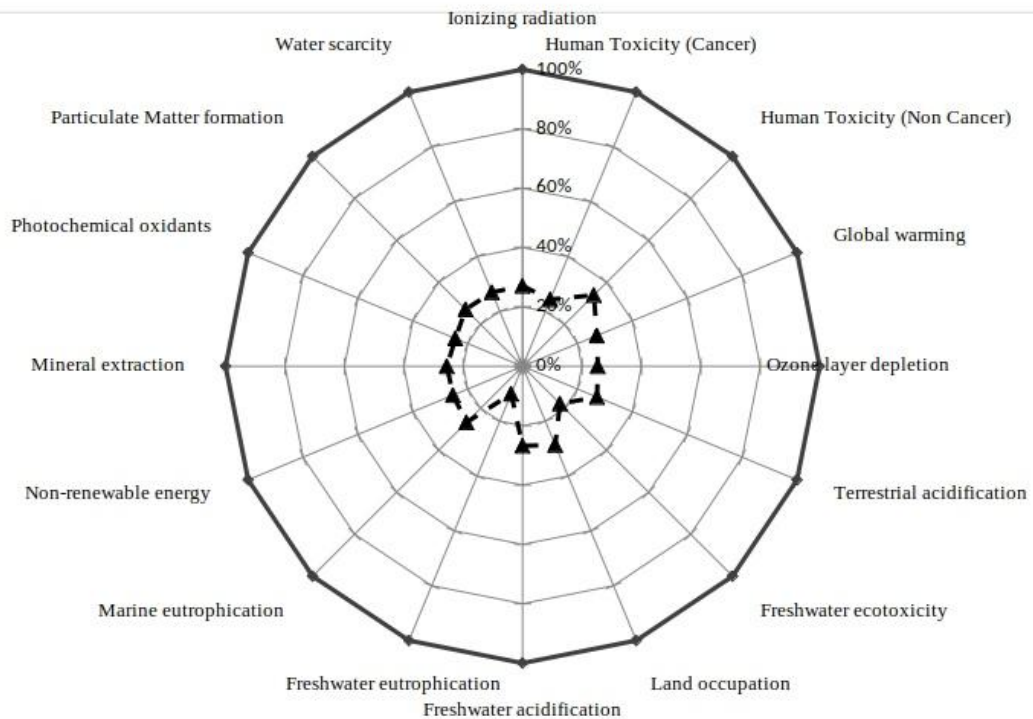
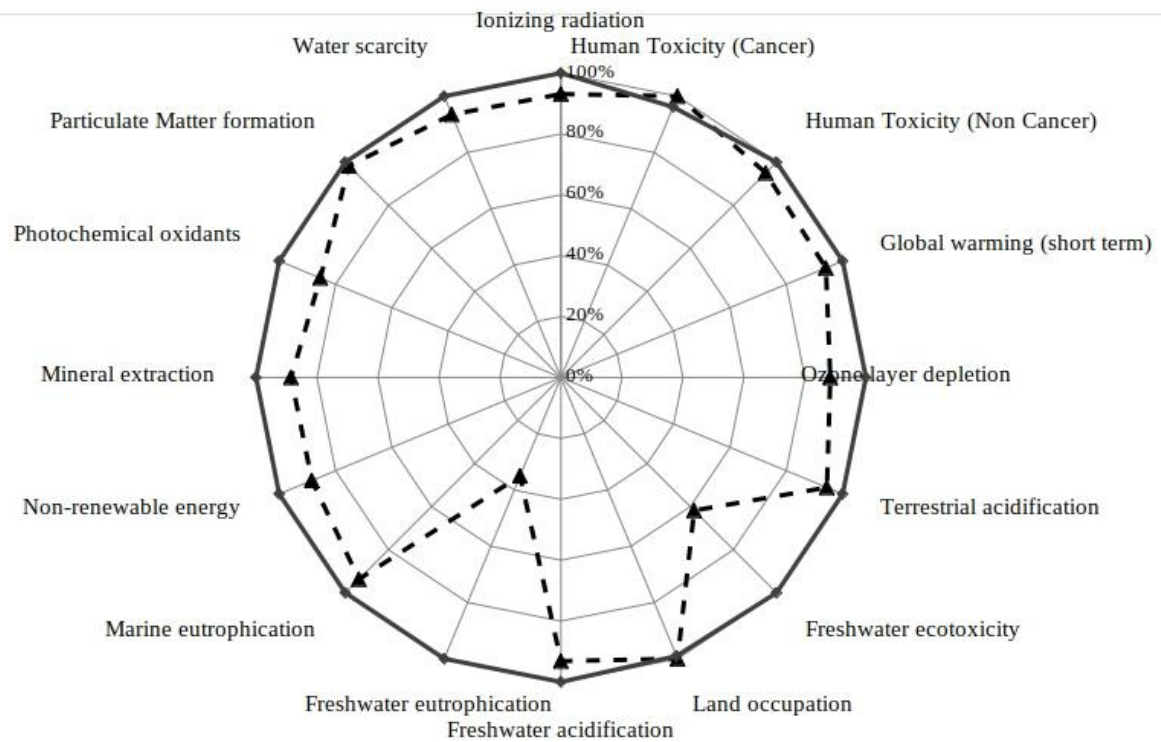
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594 Figure 2: Evaluation of the erosion model for SOC stocks at 30 cm against data from the GSDE
 595 product (Shangguan et al. 2014) (a), for the gross soil erosion rate against observations from Lizaga *et*
 596 *al.*, (2018) (b), for the net soil erosion rate against observations from observations from Lizaga *et al.*,
 597 (2018) in light gray and from Quine *et al.*, (1994) in deep gray (c).

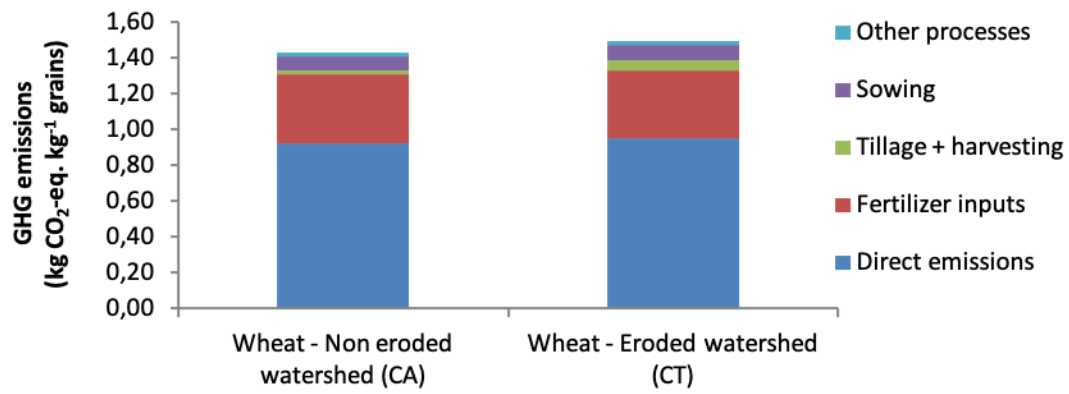
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601

602 **Figure 3:** Relative life-cycle impacts of the production of 1 kg of wheat on the non-eroded (dashed
 603 line) and eroded (solid line) Ebro watersheds, for yield scenarios #1 (top) and #2 (bottom). The system
 604 with the highest impact is given the 100% value for the various impact categories.



605

606

607 Figure 4: Contribution analysis for the global warming impact of the production of one kg of wheat in

608 the non-eroded (CA) and eroded (CT) Ebro watershed for yield scenario #1. Direct emissions

609 encompass field exchanges of CO₂ and N₂O.

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