Quantifying and modelling the impact of land consolidation and field borders on soil redistribution in agricultural landscapes (1954 – 2009)

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Abstract

Soil erosion rates in cultivated areas have intensified during the last decades leading to both on and off-site problems for farmers and rural communities. Furthermore, soil redistribution processes play an important role in sediment and carbon storage within, and exports from, cultivated catchments. This study focuses on the impact of land consolidation and changes in landscape structure on medium term soil erosion and landscape morphology within a 3.7-ha field in France. The area was consolidated in 1967 and we used the ¹³⁷Cs-technique to quantify soil erosion for the period (1954-2009). We measured the ¹³⁷Cs

inventories of 68 soil cores sampled along transects covering the entire area and especially specific linear landforms located along both present and past field borders (i.e., lynchets and undulations landforms, respectively). These results were then confronted with the outputs of a spatially-distributed ^{137}Cs conversion model that simulates and discriminates soil redistribution induced by water and tillage erosion processes. Our results showed that tillage processes dominated the soil redistribution in our study area for the last 55 years and generated about 95% (i.e., $4.50 \text{ Mg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$) of the total gross erosion in the field. Furthermore, we demonstrated that soil redistribution was largely affected by the presence of current and also former field borders, where hotspots areas of erosion and deposition ($> 20 \text{ Mg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$) were concentrated. Land consolidation contributed to the acceleration of soil erosion through the conversion of depositional areas into sediment generating areas. Although the conversion model was able to reproduce the general tendencies observed in the patterns of ^{137}Cs inventories, the model performance was relatively poor with a r^2 of 0.20. Discrepancies were identified and associated with sampling points located along the current field borders. Our data suggests that tillage erosion processes near field boundaries cannot be described as a typical diffusive process. These processes near field boundaries should be characterised and taken into account in a future version of the model to accurately simulate rates and patterns of past soil redistribution in fragmented cultivated hillslopes. We also showed that the use of an accurate DEM resulting from LIDAR data, based on present-day topography, leads to the underestimation of soil redistribution rates by the model, especially in this landscape submitted to recent and important morphological changes. Our results have important implications for the simulation of tillage erosion processes and our understanding of soil redistribution processes in complex cultivated areas. This is of particular interest to improve our knowledge and prediction of patterns of soil physical parameters, such as carbon storage or water content, particularly sensitive to surface erosion and landscape structuration.

Keywords: Soil erosion; Field border; Land consolidation; Cesium-137; Conversion model; LIDAR

1. Introduction

During the last decades, soil erosion rates in cultivated areas of western Europe have intensified and have become a problematic issue for farmers and rural communities. When it is triggered by heavy storms, soil erosion and the associated muddy floods can have disastrous and costly consequences (e.g. Pimentel et al., 1995; Evrard et al., 2007a). Additional concerns about soil erosion are related to its subsequent negative impacts, such as water pollution, decline in biodiversity and crop yields or reduction of soil water storage capacity or organic carbon sequestration (e.g. Andraski and Lowery, 1992; Berger et al., 2006; Boardman and Poesen, 2006; Papiernick et al., 2009). Because soil is a non-renewable resource at human timescales, soil protection is crucial. Quantification of erosion and deposition rates and the identification of their driving processes and their spatial variability therefore constitute a prerequisite to develop and implement soil protection strategies.

Traditionally, soil redistribution processes observed on croplands in western Europe were mainly attributed to water erosion processes. In that case, transport intensity is controlled by topographical settings such as slope, drainage area and planform curvature (Poesen, 1984; Foster, 1986; Chaplot and Le Bissonnais, 2003). However, research has shown that tillage operations result in significant soil redistribution in intensively farmed cropland (e.g. Lindstrom et al., 1992; Govers et al., 1994). Tillage erosion results from the net downslope translocation of soil, controlled by slope gradient change, during farming operations (Lindström et al., 1990; Govers et al., 1994; Montgomery et al., 1999; Van Muysen et al., 1999). As shown by Govers et al. (1994), water and tillage-induced erosion

depend on different topographical parameters resulting in a specific spatial signature in the landscape. Tillage erosion is the most intense on landscape positions where water erosion is minimal (i.e. on convexities and in the proximity of upslope field boundaries), whereas areas of tillage deposition often coincide with areas of maximal water erosion as hollows (Govers et al., 1994; 1996; Van Oost et al., 2000).

The hydrological and sedimentological connectivity across cultivated hillslopes is to a large extent controlled by the presence of field borders and associated linear elements, such as hedges, roads, furrows and grass strips, which induce landscape fragmentation (Van Oost et al., 2000; Follain et al., 2006; Szilassi et al., 2006). Vegetated borders (e.g., grass strips, hedges and grassed waterways) can reinfiltrate surface runoff and trap sediment transported by water (Van Dijk et al., 1996; Caubel et al., 2003; Evrard et al., 2008). In contrast, concave anthropogenic features (e.g., furrows) provide preferential drainage pathways, thereby increasing hydrological and sedimentological connectivity across the landscape. Linear landscape elements with a compacted surface (e.g. roads, land tracks) have a limited infiltration capacity and then enhance runoff and hydrological connectivity (Wemple et al., 1996; Forman and Alexander, 1998). In tilled fields, all types of field borders act as lines of zero-flux (Dabney et al., 1999; Van Oost et al., 2000). Consequently, tillage-induced deposition and erosion preferentially occur upslope and downslope of field borders that are oriented parallel to contour lines.

Field borders can therefore act as barriers to water and sediment fluxes generated by both water and tillage erosion (Dabney et al., 1999; Govers et al., 1999; De Alba, 2003; Van Dijk et al., 2005; Knapen et al., 2008). Interaction between erosion and deposition processes at the vicinity of field borders leads to the development of anthropogenic linear landforms of several metres width (e.g., ridges-and-furrows, headlands, and lynchets). These features are common in the agricultural landscapes of western Europe (Callot, 1980; Hooke, 1988;

Zadora-Rio, 1991). These landforms are not conserved after the removal of the field border but will instead keep evolving, and may finally lead to the formation of undulations (Houben, 2008; Chartin et al., 2011). Assessing the effect of field borders and their potential removal is then essential to understand past, present and future spatial patterns of soil redistribution and soil properties in the current global change context.

Agricultural policy and mechanisation led to the massive removal of field borders through the implementation of numerous land consolidation schemes between 1960 and 1990 in western Europe (Baudry and Burel, 1984; Vitikainen, 2004), and is still ongoing in some regions. The use of ^{137}Cs -technique can therefore offer a solution to evaluate soil redistribution over these last decades (e.g., Rogowski and Tamura, 1965; Ritchie and McHenry, 1990; Walling and Quine, 1992).

This study aims to quantify and improve our understanding of the effects of land consolidation and field borders on mid-term soil erosion and on agricultural landscape evolution. In order to achieve this objective, we analysed spatial patterns of ^{137}Cs inventories for an agricultural hillslope that was subjected to land consolidation. The study area is representative for field consolidation in intensively cultivated areas of the southwestern Parisian basin (France). We used a spatially-distributed model to convert these observations into soil redistribution rates. Emphasis will be put on the impact of small-scale topographical features, especially lynchets and undulations landforms associated with local soil accumulation along current and former field borders. Finally, we discuss the wider implications of our findings in relation to agricultural landscape evolution and model development.

2. Materials and methods

2.1. Study area

2.1.1. Location and physiographical settings

The study was conducted on a 3.7 ha field located at the downslope part of a south-east facing hillslope in the southwestern Parisian Basin, France (47°08.31'N, 0°10.97'E) (Fig. 1). The area is part of the Quincampoix watershed and is characterised by an undulating topography commonly observed in terrains underlain by Cretaceous chalks in this region. In the study field, the elevation ranges between 43 to 60m with slope between 0 to 8.8%. Two types of linear anthropogenic landforms - lynchets and undulations – were also identified within this field (see further).

Soils are calcareous Cambisols, leptic calcareous Cambisols and colluvic Cambisols with clay and loam textures (Boutin et al., 1990; FAO, 1998). The mean annual precipitation reaches ca. 600 mm and is evenly distributed throughout the year.

We reconstructed the changes in field boundaries using aerial photographs for 1945, 1959 and 2009. The study area is presently a unique field but it was divided into seven individual fields before the last important campaign of land consolidation that occurred in 1967 (Fig. 2). Interviews with local farmers and aerial photographs have demonstrated that this area has been cultivated with cereals and oilseed crops (maize, sunflower, wheat, barley and rape) since at least 1945.

2.1.2. Characteristics of the anthropogenic linear landforms

We observed one lynchets (L1) and two undulations (U1, U2) in the study area. Locations of the field borders associated with these linear landforms are presented in Figure 2. The lynchets L1 and undulations U1 and U2 are oriented perpendicularly to the steepest slope and separated by mostly regular hillslope sections in the study area (Chartin et al., 2011).

Lynchets landforms, also known as terraces, soil banks, or locally as “rideaux” in northern France and Belgium, are predominantly shaped by the progressive accumulation of

soil material upslope of a field border (Bollinne, 1971; Papendick and Miller, 1977; Van Dijk et al., 2005; Salvador-Blanes et al., 2006; Follain et al., 2007). A lynchet is defined by two morphological components separated by a field border, i.e., the axis (Fig. 3a). Uphill, progressive slope gentling from up- to downslope shapes a wide concave area (from 15 to 30m width in this region). Downhill, there is a break-in-slope that in this specific case, is 2 to 5 m-wide and reaches more than 2 m height creating a sharp discontinuity in the landscape. The soil accumulation in a lynchet evolves as a pseudo right-angle triangle (Fig. 3c).

An undulation consists of a wide gentle central convexity gradually connected by slight external concavities to the general hillslope morphology (Fig. 3b). The top of the convexity - considered as its axis - coincides closely with a former field border (Houben, 2008; Chartin et al., 2011). Undulation landforms correspond to a more or less thickened soil with a lenticular convex shape (Fig. 3d).

2.2. Topographical data

A 2 m resolution raster DEM was built from LIDAR data with the following method. Raw, tiled LIDAR datasets for the study area were assembled. Only the datasets identified as ground floor points were kept for this study (vegetation and infrastructure top points were not considered). The 2 m cells covering the area contained from 0 to 247 LIDAR points with different Z values. The median Z value was attributed to each cell centroid for cells with points whereas centroids for cells without points were deleted. Only elevation points of each cell belonging to the same landuse as the centroid were used to calculate this median value. This prevents from an artificial smoothing of sharp topographic discontinuities such as lynchet borders. A triangular irregular network (TIN) was then built from the remaining centroids. The TIN was then interpolated to a 2 m resolution raster. The elevation raster was

176 finally filtered on a plot by plot basis depending on the land use.

178 2.3. ^{137}Cs analysis

179 2.3.1. Field sampling

180 Sixty-eight soil cores were sampled in the field in 2009 for ^{137}Cs activity analysis.
181 Thirteen cores were collected along each of the five transects parallel to the steepest slope and
182 referred to as I to V in Figure 4a to cover the entire range of topographic settings observed in
183 the area. Sample lines 1 and 2 are located in the concavity of lynchets L1; sample lines 5 and 9
184 are located along the axis of undulations U1 and U2, respectively, e.g. in their central
185 convexities; lines 4, 6, 8 and 10 are located in the external concavities of the latter undulations;
186 and, finally, sample lines 3, 7, 11, 12 and 13 are located in areas characterised by more or less
187 regular slopes. Three additional cores (IV-1a, IV-1b and IV-1c on Figure 4b) were sampled
188 along transect IV to detail the effect of the present field border on the recent soil redistribution
189 over a short distance in lynchets L1 (Fig. 4b).

190 Soil cores with 8 cm or 10 cm external diameter were collected with a percussion
191 drilling machine (Eijkelkamp) up to a soil depth of about 70 to 100 cm and immediately cut
192 into sections in the field. Depending on the local soil thickness, the first section corresponded
193 to the uppermost 20 to 40 cm of the core. Then, successive 10-cm sections were cut off until
194 the bedrock was reached. Two of the sixty-eight soil cores (sample points IV-1a and V-8
195 highlighted in Fig. 4) were cut into 5-cm sections to detail the vertical distribution of ^{137}Cs at
196 particular locations. The V-8 soil core was collected on the downslope external concavity of
197 undulation U2, e.g. just downslope of a former field border that was removed in 1967. Core
198 IV-1a was sampled on the gentle slope of L1 lynchets, 3 m upslope of the present lower field-
199 border. Each section of the 68 cores was oven-dried for 48 h at 40 °C, sieved to 8-mm. Soil in
200 the study area contains large quantities of coarser grains and chalk fragments. Attention was

paid to collect the fine particle fraction deposited on the surface of those coarser grains. The <8mm fraction was ground to a fine powder. For each sample, a representative subsample of the <8mm fraction (approx.. 70 g) was then collected to measure ^{137}Cs activities.

2.3.2. *Measuring ^{137}Cs activities*

^{137}Cs in the soil cores was measured at 661 keV using low background Germanium gamma-ray detectors (Germanium hyperpure – GeHP, N-type, coaxial model) for 24 to 48 h. An initial qualitative assessment was performed on successive subsamples of each core to determine the maximum depth of the ^{137}Cs signal. The ^{137}Cs activity (A_i in Bq.kg^{-1}) was then derived for each core section. Finally, the ^{137}Cs total inventory (A_{surf} ; Bq.m^{-2}) of each core was calculated according to Eq. (1) (Sutherland, 1992).

$$A_{\text{surf}} = \sum_{i=1}^n \left(A_i \times \frac{M_i}{S} \right) \quad (1),$$

where A_i is the ^{137}Cs activity in the successive sections of the core (Bq.kg^{-1}); M_i is the mass (kg) of the <8 mm soil fraction of the i^{th} sampled section; S is the surface area (m^2) of the soil core cylinder, n the total number of sampled sections of the core.

To estimate whether soil erosion or deposition occurred in the investigated field since the beginning of ^{137}Cs fallout (1954), ^{137}Cs inventories were compared to the local reference inventory. This latter was the mean inventory obtained in four cores sampled in neighbouring undisturbed sites such as orchards and pastures. Then, ^{137}Cs residuals were calculated according to Eq. (2):

$$Cs_{\text{res},x} = A_{\text{surf},x} - A_{\text{surf,ref}} \quad (2),$$

where $Cs_{\text{res},x}$ are the ^{137}Cs residuals at the x core location (Bq.m^{-2}), $A_{\text{surf},x}$ is the ^{137}Cs total inventory at the the x core location (Bq.m^{-2}), and $A_{\text{surf,ref}}$ is the Cs reference inventory (Bq.m^{-2}).

Negative residuals mean that the concerned location has underwent net soil erosion since 1954, whereas positive residuals indicate the occurrence of net soil deposition.

2.4 Converting ^{137}Cs activities into erosion-deposition rates

2.4.1. Conversion model description

We used a spatially explicit model to convert ^{137}Cs residuals into soil erosion and deposition rates ($\text{Mg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$ or $\text{mm}\cdot\text{yr}^{-1}$). The model consists in a combination of a punctual mass-balance model and a spatially distributed and process-based model of water and erosion tillage processes (Van Oost et al, 2003). The mass-balance model includes both the annual fallouts and susceptible losses of ^{137}Cs due to runoff (before its incorporation within the tilled layer) during the whole period of the radionuclide fallout (Quine, 1995). The conversion model exploits the different patterns of water and tillage erosion in a spatial analysis of ^{137}Cs inventories to assess the relative contribution of water and tillage erosion processes.

The water-induced processes of soil erosion (rill and interrill erosion) are defined as a power function of slope gradient and contributing area, following Eqs. (3) and (4):

$$E_{\text{rill}} = k_1 \rho_b S^a A^b, \quad (3)$$

$$E_{\text{irill}} = c \rho_b S^d, \quad (4)$$

where E_{rill} and E_{irill} are rill and interrill potentials, respectively ($\text{kg}\cdot\text{m}^{-2}$), ρ_b is the dry bulk density of the soil ($\text{kg}\cdot\text{m}^{-3}$), S is the slope ($\text{m}\cdot\text{m}^{-1}$), A is the contributing area per unit contour width ($\text{m}^2\cdot\text{m}^{-1}$), and k_1 , a , b , c and d are coefficients (-).

The local rate of soil erosion is then calculated as the summation of both potentials for rill and interrill erosion unless the local transport capacity is exceeded. The transport capacity on a given slope segment is then considered as being proportional to the potential for rill erosion (e.g., Desmet and Govers, 1995), Eq. (5):

$$T_c = k_2 E_{\text{rill}}, \quad (5)$$

250 where T_c is the transport capacity (kg.m^{-1}) and k_2 is a coefficient (m).

251 When local sediment inflow exceeds the transport capacity T_c , deposition occurs. The amount
252 of soil translocated is then equal to T_c .

253 The net soil flux induced by tillage translocation on a hillslope of infinitesimal length
254 and unit width is considered as proportional to the local slope gradient (Govers et al., 1994;
255 Van Oost et al., 2000), Eq. (6):

256
$$Q_t = k_3 S = -k_3 \frac{dh}{dx}, \quad (6)$$

257 where Q_t represents the net downslope flux due to tillage ($\text{kg.m}^{-1}.\text{a}^{-1}$), k_3 is the tillage transport
258 coefficient ($\text{kg.m}^{-1}.\text{a}^{-1}$), S is the local slope gradient (m.m^{-1}), h is the height at a given point of
259 the hillslope (m), and x is the horizontal distance (m).

260 The local intensity of tillage-induced erosion (E_t in kg.m^{-2}) is then modelled as a diffusive
261 process controlled by slope gradient changes, according to Eq. (7):

262
$$E_t = \rho_b \frac{dh}{dt} = -\frac{dQ_t}{dx} = \frac{d^2 h}{dx^2}, \quad (7)$$

263 where t is the time (s).

264 Finally, field borders were considered as lines of zero flux. This means that no soil material is
265 translocated from one field to another during tillage.

266

267 2.4.2. Conversion model application

268 The simulation process is iterative, and each iteration corresponds to one year. Annual
269 ^{137}Cs fallouts used in the model were based on the measured annual fallout for the northern
270 hemisphere (Cambrai et al., 1989). This value was rescaled to the local reference inventory
271 using the factor α (-), Eq. (8):

272
$$\alpha = \frac{A_{surf,ref}}{A_{surf,NH}}, \quad (8)$$

where $A_{\text{surf,ref}}$ is the Cs reference inventory (Bq.m^{-2}) for the study area and $A_{\text{surf,NH}}$ is the mean Cs reference inventory (Bq.m^{-2}) for the northern hemisphere. ^{137}Cs fallouts associated with the Chernobyl accident were reported to be negligible in the study area and were therefore not considered in Cs reference inventories of Eq. (8) (Fourmont, 2001).

The parameter values proposed by Quine (1997) were used for the punctual mass-balance model. No a-priori assumptions were made about rates of water and tillage erosion. A specific procedure developed by Van Oost et al. (2003) and based on the Generalized Likelihood Uncertainty Estimation (Beven and Binley, 1992) was implemented to explore the parameter space for which the model was in close agreement with the observed ^{137}Cs inventories. Value ranges for parameters k_1 , k_2 and k_3 (Eqs. 2, 4 and 5) used in this procedure are presented in Table 1. Coefficients a, b and d (Eqs. 2 and 3) were based on studies carried out by Desmet and Govers (1995, 1997), Van Oost et al (2000) and Van Rompaey et al. (2001).

The conversion model was applied to compare simulated and observed ^{137}Cs patterns for 3000 randomly chosen parameter sets k_1 , k_2 and k_3 over the whole study area. The observational dataset used in the evaluation procedure, called “total set”, contains 67 sampling points: the point IV-1b was excluded as it is located just 1 m from point IV-1a. Likelihood values for each set of parameters were determined and associated with the corresponding model output values (Van Oost et al., 2003). The cumulative likelihood distribution of water and tillage erosion rates was then defined separately for the whole study area. From these cumulative distributions, median (M), 5th (P5) and 95th (P95) percentile values were calculated to provide a quantification of model uncertainty.

3. Results and discussion

3.1. Recent soil redistribution patterns and relations with topographical settings

The observed ^{137}Cs reference inventory was $1367 \pm 30 \text{ Bq.m}^{-2}$ in 2009 in Seuilly (Fig. 5a). About 40% and 65% of the total ^{137}Cs inventory was concentrated in the uppermost 5 cm and 10 cm, respectively. As observed at many other undisturbed locations, the ^{137}Cs content declined almost exponentially with soil depth in these profiles (Walling and Quine, 1992).

The ^{137}Cs residuals (Eq. 2) ranged from -1030 Bq.m^{-2} to 980 Bq.m^{-2} in the 68 analysed cores. The largest variations were observed along the steepest slope direction (NNW-SSE) and were strongly affected by anthropogenic undulations U1, U2, and lynchet L1 (Fig. 6). About 50% of the ^{137}Cs contained in the soil at the time of sampling (2009) originates from the period between 1962 and 1964. Consequently, currently observed patterns mostly reflected the soil redistribution that occurred after the 1967 land consolidation, when field borders associated with undulations U1 and U2 were removed (Fig. 2).

The ^{137}Cs residuals measured on undulation central convexities (sample lines 5 and 9) ranged from -320 Bq.m^{-2} to 20 Bq.m^{-2} and were mostly negative, whereas residuals measured on undulation external concavities (sample lines 4, 6, 8 and 10) ranged from 30 Bq.m^{-2} to more than 730 Bq.m^{-2} . These results reflected the progressive levelling of the undulation landforms since the last land consolidation. These patterns of ^{137}Cs residuals were then most likely induced by redistribution through tillage practice such as it leads to soil erosion in convexities and soil accumulation in concavities (Govers et al., 1994; 1996; De Alba et al., 2004). The profile V-8 provided an example of ^{137}Cs vertical distribution in an undulation external concavity, i.e. in a depositional context ($A_{\text{surf}} = 1786 \text{ Bq.m}^{-2}$; Fig. 5b). The ^{137}Cs vertical distribution was homogeneous in the uppermost 30 cm and then declined rapidly in the next five centimetres, reflecting the mixing induced by tillage operations (Kachanovski and de Jong, 1984; Ritchie and McCarty, 2003). As undulation landforms initially correspond to local soil accumulation induced by former field borders, their morphology in 2009

potentially derived from the levelling of two other types of anthropogenic landforms consecutive to the field border removal: i) the undulations are former lynchets as suggested by Bollinne (1971) and Houben (2008); ii) undulations are headlands created by an asymmetric accumulation of soil due to tillage translocation on both sides of the borders (Callot, 1980; Leturcq, 2008). In both cases, the morphology corresponding to current undulations in the vicinity of the former field borders shows a strong evolution.

The ^{137}Cs residuals ranged from 420 to 560 Bq.m^{-2} 16 m upslope of the lynchet L1 field border (sample line 2). This area is characterised by a particularly gentle slope that is clearly concave. In contrast, four of the five cores collected 2m upslope of same border (sample line 1) showed negative residuals ranging from -1000 Bq.m^{-2} to -250 Bq.m^{-2} (Fig.6). Fig. 7 shows that lynchet L1 (sample lines 1 and 2) was characterised by a large variability of ^{137}Cs residuals revealing the occurrence of both erosion and deposition processes. It has nevertheless mostly undergone soil accumulation since 1967. Indeed, ^{137}Cs residuals ranged from -426 Bq.m^{-2} to 946 Bq.m^{-2} , with negative values exclusively concentrated within the first 4m upslope the field border. Even though lynchets are typically developed by the progressive accumulation of soil upslope of a field border, our results reflected that soil erosion occurred along this downfield border during the period 1954-2009. This complex behaviour could be due to specific tillage practices performed along the field border. Farmers systematically end up their tillage operations by contouring the field, i.e. perpendicularly to the main up- and downslope tillage direction in the case of the upslope and downslope field borders. This operation with the mouldboard plough results in the translocation of soil to the inside of the field. This process could explain the slight local convexity observed in the first meters of the field along the lower field border (Fig. 7). Considering this short-distance variability, sampling schemes have to be carefully designed to describe in an appropriate way

the major soil redistribution processes (erosion or deposition) occurring in landforms associated with field borders.

A depositional context was also observed 10 m downslope of the current upslope field border (^{137}Cs residuals varied from 475 Bq.m^{-2} to 800 Bq.m^{-2} along the sampling line 13; Fig. 5). Indeed, soil is thicker at this location than in surrounding areas with a depth locally up to 80 cm (Chartin, 2011). Moreover, the local slope is convex in the direction of the steepest slope. Here again, tillage practices performed along this field border could explain the local soil accumulation and the topographical convex settings. Unfortunately, sampling density was not sufficient to check the relevance of this hypothesis in this specific study case.

Finally, spatial patterns of ^{137}Cs residuals in undifferentiated areas are rather complex and do not seem to be directly influenced by slope gradient nor curvature (Fig. 6).

3.2. Modelling erosion and deposition rates from 1954 to 2009

3.2.1. Conversion results

The factor α used to determine mean annual ^{137}Cs fallouts in this area was 0.786 (Eq. 7). Fig. 8 presents the simulated optimal ^{137}Cs inventories, tillage and water erosion simulated by the model over the whole study area.

The major observed patterns of ^{137}Cs inventories related to lynchets and undulations were reproduced by the simulations. Convex landforms were characterised by soils with low ^{137}Cs inventories when compared to the higher inventories simulated on concave landforms (Fig. 8a). Tillage was found to be the dominant process controlling ^{137}Cs patterns since the 1967 land consolidation. Tillage erosion rates were often higher than $6 \text{ Mg.ha}^{-1}.\text{yr}^{-1}$ and reached locally up to $15 \text{ Mg.ha}^{-1}.\text{yr}^{-1}$ in central convex parts of U1 and U2 (Fig. 8b). Soil accumulation by tillage observed in external concave areas of undulations and over the whole area of lynchet L1 seemed to be correctly reproduced with a maximum deposition rate

reaching 22 Mg.ha⁻¹.yr⁻¹. Water erosion output patterns were also greatly influenced by undulation and lynchets landforms (Fig. 8c). U1 and U2 subdivided the study field in three areas characterised by similar patterns of simulated water erosion. In each area, erosion of soil material was simulated over most of the upslope part whereas deposition was simulated in the concave depositional settings located downslope of the area. Maximum soil erosion and deposition rates by water were simulated in the lower area, i.e. between U1 and L1, where the general slope is the steepest. Here, erosion rates reached 0.6 Mg.ha⁻¹.yr⁻¹ within the U1 external concavity and net deposition reached 5 Mg.ha⁻¹.yr⁻¹ within L1 concavity.

Simulated tillage erosion was an order of magnitude higher than water erosion in both erosional and depositional areas (e.g., the median water erosion rates was -0.20 Mg.ha⁻¹.yr⁻¹ whereas the median tillage erosion rate was -4.50 Mg.ha⁻¹.yr⁻¹; Tab. 2). The mean tillage erosion rate over the whole study area was three to four times lower than the maximum erosion rates simulated in undulations (Tab. 2; Fig. 8b). Considering the mean tillage deposition rate over the whole study area, it was more than four times lower than maximum deposition rates simulated within the lynchets L1 (Tab. 2; Fig. 8b). The uncertainty associated with simulated water erosion and deposition was much higher than the one associated with tillage simulation. This could be explained by the dominance of tillage erosion process (Tab. 2). Moreover, deposition rates and patterns of water erosion are largely controlled by k_2 whereas tillage process simulations are characterised by soil redistribution patterns that remained stable for the entire range of the tested k_3 values (Van Oost et al., 2003).

The model runs confirmed our observations: soil redistribution by tillage is dominant in the area since 1967. Tillage redistributes sediment in fields whereas water exports it from fields. In this context, we estimated here that 98.5 % of the total eroded soil was deposited in the study field, whereas only 1.5% was exported by water erosion. This result was in agreement with previous findings obtained in the Parisian basin showing that land

consolidation generates a dramatic increase of soil redistribution within cultivated catchments, and to a slight (Evrard et al., 2010; Delmas et al., 2012) or a large increase (Boardman et al., 1994; Evrard et al., 2008) of sediment exports from the catchment, depending on the catchment topography, land use characteristics and the implementation of soil conservation measures. As soon as mechanization appeared in the 1950s, land consolidation campaigns favoured its introduction across the world and, inversely, mechanization motivated the implementation of land consolidation operations (Van Huylenbroeck et al., 1996; Crecente et al., 2002; Niroula and Thapa, 2005). However, mechanized agriculture involved higher tillage erosion rates when compared to previous non-mechanized tillage (Govers et al., 1996; Van Oost et al., 2000, 2006). Land consolidation enhances hydrological and sedimentological connectivity and leads to important increases of sediment exports in regions particularly sensitive to water erosion (Evrard et al., 2007b). However, this increase is not observed when tillage largely dominates soil redistribution. Moreover, soils developed on Cretaceous chinks appear not to be particularly sensitive to water erosion, thereby limiting the hydrological and sedimentological connectivity (Boutin et al., 1990).

Our results explicitly identified hotspot areas of soil erosion induced by land consolidation, i.e., in the vicinity of removed field borders, and hotspot areas of soil deposition, i.e. in the vicinity of present field borders and external concavities of undulations. We thereby show that land consolidation led progressively to the general levelling of the hillslope morphology. This was particularly true when considering the convex undulations. From sinks for soil material before land consolidation, the corresponding areas progressively became sources for soil material redistribution.

3.2.2. Analysis of the discrepancies

Even though the conversion model reproduced the major trends of soil erosion and deposition recorded in the study field, observed and simulated ^{137}Cs inventories fitted rather poorly ($r^2 = 0.20$, $p < 0.05$; Fig. 9). Indeed, the model tended to systematically underestimate inventories in depositional settings and to overestimate inventories in highly erosive areas.

Simulated ^{137}Cs inventories along the linear landforms were more or less underestimated (from -150 to -750 Bq.m^{-2}) on observed depositional settings associated with lynchet L1 and undulations U1 and U2, i.e. on concave parts (lines 2, 4, 6, 8 and 10; Fig. 10a). However, most of these locations were simulated as being in a depositional context (Fig. 10b). Similarly, central convexities located on undulations were simulated as eroded areas. The largest under-estimations ($< -600 \text{ Bq.m}^{-2}$) were concentrated along the upslope field border (Fig. 10a). In this case, observed depositional locations were simulated as eroded sites (Fig. 10b). Although deposition is simulated within a few-meter width area along this upslope border (Fig. 8a), actual deposition occurring ten meters downslope of this border (along the sample line 13) was not well reproduced by the model. The largest over-estimations were concentrated along the downslope field-border (sample line 1 on lynchet L1) where simulation errors reached up to 1000 Bq.m^{-2} . As mentioned before, we infer that specific tillage operations performed along the upper and lower field borders could have induced processes differing from the classical diffusive process associated with “infield” operations and considered as the only tillage-induced process in the conversion model. The mechanisms of those specific processes could therefore be implemented to improve the model efficiency. Furthermore, sampling could be intensified in the vicinity of the current field borders in order to describe more accurately their impact on local variations of soil redistribution.

We therefore ran the model with a dataset (subsequently referred to as the “infield set”) excluding points sampled at the vicinity of upslope and downslope field borders, i.e. sampling points IV-1a, IV-1b and those located along lines 1 and 13 ($n = 55$). Observed and simulated

inventories fitted much better when using the “infield set” ($r^2 = 0.42$; $p < 0.05$; Fig. 11) than when using the “total set” ($r^2 = 0.20$), but discrepancies remained.

Both simulations outlined that tillage-induced processes were the driving factor explaining more than 95 % of the total soil redistribution that occurred after the 1967 land consolidation (Table 3). We demonstrated that tillage practices levelled the field morphology, and we can then hypothesise that local curvatures derived from the DEM based on the 2009 LIDAR data were smoothened compared to those existing in 1967. Consequently, tillage erosion and deposition rates simulated for the period 1954-2009 remained necessarily underestimated when considering curvatures as the morphological parameter controlling the intensity of soil translocation by this diffusive process. The more the landscape morphology would have evolved since ^{137}Cs fallouts, the less the conversion of ^{137}Cs inventories into soil redistribution rates would be accurate. This should be taken into account when using the ^{137}Cs method to validate the parameterization of erosion models dedicated to the prediction of medium-term erosion (from decades to centuries).

4. Conclusions

This study focused on the impact of land consolidation and field borders on medium term soil erosion (1954-2009) and landscape morphology within a 3.7-ha field consolidated in 1967. The spatial patterns of ^{137}Cs inventories were analyzed and a spatially-distributed conversion model simulating the respective implication of water and tillage erosion was applied. The model simulated tillage erosion as the most dominant process across the study area, by generating 95% of soil redistribution movements (i.e., about $4.50 \text{ Mg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$) and deeply modifying the landscape morphology. Hence, soil redistribution was greatly affected by the presence of current but also former field borders, where hotspots areas of erosion (on undulations) and deposition (on lynchets; i.e., about $20 \text{ Mg}\cdot\text{ha}^{-1}\cdot\text{yr}^{-1}$) are identified. Land

consolidation contributed to the acceleration of soil redistribution within the field through the conversion of depositional areas into sediment delivering areas. The conversion model reproduced the main patterns of ^{137}Cs inventories, but simulated inventories fitted rather poorly with these latter ($r^2 = 0.20$). Discrepancies were identified and associated with sampling points located along the current field borders where tillage is performed perpendicularly to the main tillage direction in the field. This tillage operation is observed in most cultivated plots and corresponds to a contouring of the field when finishing a tillage operation. This particular tillage operation could be taken into account in a future version of the model to accurately simulate rates and patterns of past soil redistribution in fragmented cultivated hillslopes. An other outcome of this specific process is that ^{137}Cs sampling schemes have to be carefully designed to describe in an appropriate way the major soil redistribution processes occurring in landforms associated with field borders. We also suggest that the use of an accurate DEM representing the actual topography in the model leads to the underestimation of soil redistribution rates, especially in this landscape where morphology was submitted to recent and significant changes. These results have important implications for the modelling of tillage erosion and to better quantify and determine the spatial patterns of soil redistribution processes across cultivated areas in a changing environment. This is of particular interest to improve our knowledge and prediction of patterns of soil physical parameters, as carbon storage or water content, particularly sensitive to surface erosion and landscape structuration.

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References

- Andraski, B.J. and Lowery, B., 1992. Erosion Effects on Soil Water Storage, Plant Water Uptake, and Corn Growth. *Soil Science Society of America Journal* 56(6), 1911-1919.
- Baudry, J., Burel, F., 1984. "Remembrement": Landscape consolidation in France. *Landscape Planning* 11, 235-241.
- Boardman, J., Poesen, J., 2006. Soil erosion in Europe: major processes, causes and consequences. In: J. Boardman and J. Poesen (Editors), *Soil erosion in Europe*. Wiley, Chichester, pp. 479-489.
- Berger, G., Kaechele, H. and Pfeffer, H., 2006. The greening of the European common agricultural policy by linking the European-wide obligation of set-aside with voluntary agri-environmental measures on a regional scale. *Environmental Science & Policy* 9(6), 509-524.
- Beven, K. and Binley, A., 1992. The Future Of Distributed Models – Model Calibration And Uncertainty Prediction. *Hydrological Processes* 6(3), 279-298.
- Boardman, J., Ligneau, L., de Roo, A., Vandaele, K., 1994. Flooding of property by runoff from agricultural land in northwestern Europe. *Geomorphology* 10, 183-196.
- Bollinne, A., 1971. Les rideaux en Hesbaye gembloutoise - Etude morphologique et sédimentologique. *Bulletin de la Société géographique de Liège* 7, 61-67.
- Boutin, D., Froger, D., Rassineux, J., 1990. Feuille Loudun (1724-1624), Carte des sols du Département de la Vienne et de la région Centre au 1:50000, Chambre d'Agriculture de la Vienne - IGN - INRA.
- Callot, H.J., 1980. La plaine d'Alsace. Modelé agraire et parcellaire. Université de Nancy II, France.
- Cambrai, R.S., Playford, K., Carpenter, R.C., 1989. Radioactive fallout in air and rain: results to the end of 1988. UK Atomic Energy Authority Report AERE-R 10155, HMSO.
- Caubel, V., Grimaldi, C., Merot, P., Grimaldi, M., 2003. Influence of a hedge surrounding bottomland on seasonal soil-water movement. *Hydrological Processes* 17, 1811-1821.
- Chaplot, V., Le Bissonnais, Y., 2003. Runoff features for interrill erosion at different rainfall intensities, slope lengths, and gradients in agricultural loessial hillslope. *Soil Science Society of America Journal* 67(3), 844-851.

- Chartin, C., 2011. Effet de l'évolution du parcellaire agricole sur la redistribution des sols et la morphologie des versants cultivés – exemple du Bassin parisien. Université de Tours, France, 322 pp.
- Crecente, R., Alvarez, C., Fra, U. 2002. Economic, social and environmental impact of land consolidation in Galicia. *Land Use Policy* (19), 135-147.
- Chartin, C., Bourennane H., Salvador-Blanes, S., Hinschberger, F., Macaire, J.-J., 2011. Classification and mapping of anthropogenic landforms on cultivated hillslopes using DEMs and soil thickness data — Example from the SW Parisian Basin, France. *Geomorphology* 135, 8-20.
- Dabney, S.M., Liu, Z., Lane, M., Douglas, J., Zhu, J., Flanagan, D.C., 1999. Landscape benching from tillage erosion between grass hedges. *Soil & Tillage Research* 51, 219-231.
- De Alba, S., 2003. Simulating long-term soil redistribution generated by different patterns of mouldboard ploughing in landscapes of complex topography. *Soil and Tillage Research* 71, 71–86.
- De Alba, S., Lindstrom, M.J., Schumacher, T.E., Malo, D.D., 2004. Soil landscape evolution due to soil redistribution by tillage: new model of soil catena evolution in agricultural landscapes. *Catena* 58, 77-100.
- Delmas, M., Pak, L.T., Cerdan, O., Souchère, V., Le Bissonnais, Y., Couturier, A, Sore, L., 2012. Erosion and sediment budget across scale: A case study in a catchment of the European loess belt. *Journal of Hydrology* 420, 255-263.
- Desmet, P.J.J., Govers, G., 1995. GIS-based simulation of erosion and deposition patterns in an agricultural landscape: a comparison of model results with soil map information. *Catena* 25, 389-401.
- Desmet, P.J.J., Govers, G., 1997. Two-dimensional modelling of the within-field variation in rill and gully geometry and location related to topography. *Catena* 29, 283-306.
- Evrard, O., Bielders, C.L., Vandaele, K., Van Wesemael, B., 2007a. Spatial and temporal variation of muddy floods in central Belgium, off-site impacts and potential control measures. *Catena* 70, 443-454.
- Evrard, O., Persoons, E., Vandaele, K., van Wesemael, B., 2007b. Effectiveness of erosion mitigation measures to prevent muddy floods: A case study in the Belgian loam belt. *Agriculture, Ecosystems & Environment* 118, 149-158.
- Evrard, O., Vandaele, K., van Wesemael, B., Bielders, C.L, 2008. A grassed waterway and earthen dams to control muddy floods from a cultivated catchment of the Belgian loess belt. *Geomorphology* 100, 419-428.
- Evrard, O., Nord, G., Cerdan, O., Souchère, V., Le Bissonnais, Y., Bonté, P., 2010. Modelling the impact of land use change and rainfall seasonality on sediment export from an agricultural catchment of the northwestern European loess belt. *Agriculture, ecosystems and environment* 138, 83-94.

- FAO, 1998. World reference base for soil resources. Food and Agriculture Organization of the United Nations. World Soil Resources Report No 84, FAO, Rome.
- Follain, S., Minasny, B., McBratney, A.B., Walter, C., 2006. Simulation of soil thickness evolution in a complex agricultural landscape at fine spatial and temporal scales. *Geoderma* 133, 71-86.
- Forman, R.T.T., Alexander, L.E., 1998. Roads and major ecological effects. *Annual Review of Ecology and Systematics* 29, 207-231.
- Foster, G.R., 1986. Understanding ephemeral gully erosion. In: B.i.A. National Research Council (Editor), *Soil conservation: Assessing the national research inventory*. National Academy Press, Washington, pp. 90-118.
- Fourmont, A., 2001. Analyse de l'évolution subactuelle des banquettes colluviales anthropiques par la méthode du Césium-137. Influence sur le bilan sédimentaire du bassin du Quincampoix (Sud-ouest du Bassin Parisien, France), Rapport de stage de DEA, Université de Tours, France, pp. 40.
- Govers, G., Lobb, D.A., Quine, T.A., 1999. Tillage erosion and translocation: emergence of a new paradigm in soil erosion research. *Soil & Tillage Research* 51, 167-174.
- Govers, G., Quine, T.A., Desmet, P.J.J., Walling, D.E., 1996. The relative contribution of soil tillage and overland flow erosion to soil redistribution on agricultural land. *Earth Surface Processes and Landforms* 21, 929-946.
- Govers, G., Vandaele, K., Desmet, P.J.J., Poesen, J., Bunte, K., 1994. The role of tillage in soil redistribution on hillslopes. *European Journal of Soil Science* 45, 469-478.
- Hooke, D., 1988. Cosmogenic nuclides, topography, and the spatial variation of soil depth. *Geografiska Annaler* 70 B(1), 123-131.
- Houben, P., 2008. Scale linkage and contingency effects of field-scale and hillslope-scale controls of long-term soil erosion: anthropogeomorphic sediment flux in agricultural loess watersheds of Southern Germany. *Geomorphology* 101, 172-191.
- Kachanovski, R. G., de Jong, E., 1984. Predicting the temporal relationship between soil Cesium-137 and erosion rate. *Journal of Environmental Quality* 13(2), 301-304.
- Knapen, A., Poesen, J., Govers, G., De Baets, S., 2008. The effect of conservation tillage on runoff erosivity and soil erodibility during concentrated flow. *Hydrological Processes* 22, 1497-1508.
- Leturcq, S., 2008. Fonction et devenir d'un réseau invisible: les crêtes de labour dans les terroirs beaucerons (XIVe-XXe siècles), COST du Mans. Marqueurs des paysages et systèmes socio-économiques. R.Compatangelo-Soussignan, J.R.Bertrand, J.Chapman, P.Y. Laffont, Rennes (France), pp. 163-174.

- Lindstrom, M.J., Nelson, W.W., Schumacher, T.E., 1992. Quantifying tillage erosion rates due to moldboard plowing. *Soil & Tillage Research* 24, 243-255.
- Lindstrom, M.J., Nelson, W.W., Schumacher, T.E., Lemme, G.D., 1990. Soil movement by tillage as affected by slope. *Soil & Tillage Research* 17, 255-264.
- Montgomery, J.A., McCool, D.K., Busacca, A.J., Frazier, B.E., 1999. Quantifying tillage translocation and deposition rates due to moldboard plowing in the Palouse region of the Pacific Northwest, USA. *Soil & Tillage Research* 51, 175-187.
- Niroula, G.S., Thapa, G.B. 2005. Impacts and causes of land fragmentation, and lessons learned from land consolidation in South Asia. *Land use Policy* 22, 358-372.
- Papendick, R.I., Miller, D.E., 1977. Conservation tillage in Pacific Northwest. *Journal of Soil and Water Conservation* 32, 49-56.
- Papiernick, S.K., Schumacher, T.E., Lobb, D.A., Lindstrom, M.J., Lieser, M.L., Eynard, A., Schumacher, J.A., 2009. Soil properties and productivity as affected by topsoil movement within an eroded landform. *Soil & Tillage Research* 102, 67-77.
- Pimentel, D., Harvey, C., Resosudarmo, P., Sinclair, K., Kurz, D., McNair, M., Crist, S., Shpritz, L., Fitton, L., Saffouri, R., Blair, R. 1995. Environmental and economic costs of soil erosion and conservation benefits. *Science* 267(5201), 1117-1123.
- Poesen, J., 1984. The influence of slope gradient on infiltration rate and Hortonian overland flow volume. *Z. Geomorphol., N.F. Suppl.-Bd.* 49, 117-131.
- Quine, T.A., 1995. Estimation of erosion rates from caesium-137 data: the calibration question. In: I.D.L. Webster, A.M. Gurnell and B.W. Webb (Editors), *Sediment and Water Quality in River Catchments*. John Wiley & Sons, Chicester, pp. 307-329.
- Quine, T.A., Govers, G., Walling, D.E., Zhang, X., Desmet, P., Zhang, Y., Vandaele, K., 1997. Erosion processes and landform evolution on agricultural land - new perspectives from caesium-137 measurements and topographic-based erosion modelling. *Earth Surface Processes and Landforms* 22, 799-816.
- Ritchie, J.C., McCarty, G.W., 2003. Using ¹³⁷Cesium to understand soil carbon redistribution on agricultural watersheds. *Soil and tillage Research* 69, 45-51.
- Ritchie, J.C., McHenry, J.R., 1990. Application of radioactive fallout Cesium-137 for measuring soil erosion and sediment accumulation rates and patterns: a review. *Journal of Environmental Quality* 19, 215-233.
- Rogowski, A.S., Tamura, T., 1965. Movement of ¹³⁷Cs by runoff, erosion and infiltration on the alluvial Captina silt loam. *Health Physics* 11(12), 1333-1340.
- Salvador-Blanes, S., Cornu, S., Couturier, A., King, D., Macaire, J.-J., 2006. Morphological and geochemical properties of soil accumulated in hedge-induced terraces in the Massif Central, France. *Soil and Tillage Research* 85, 62-77.

- Sutherland, R.A., 1992. Caesium-137 estimates of erosion in agricultural areas. *Hydrological Processes* 6, 215-225.
- Szilassi, P., Jordan, G., Van Rompaey, A., Csillag, G., 2006. Impacts of historical land use changes on erosion and agricultural soil properties in the Kali Basin at the Lake Balaton, Hungary. *Catena* 68, 96-108.
- Van Dijk, P.M., Auzet, A.-V., Lemmel, M., 2005. Rapid assessment of field erosion and sediment transport pathways in cultivated catchments after heavy rainfall events. *Earth Surface Processes and Landforms* 30(2), 169-182.
- Van Dijk, P.M., Kwaad, F.J.P.M., Klapwijk, M., 1996. Retention of water and sediment by grass strips. *Hydrological Processes* 10, 1069-1080.
- Van Huylenbroeck, G., Castro Coelho, J., Pinto, P.A. 1996. Evaluation of Land Consolidation Projects (LCPs): A Multidisciplinary Approach. *Journal of Rural Studies* 12(3), 297-310.
- Van Muysen, W., Govers, G., Bergkamp, G., Roxo, M., Poesen, J., 1999. Measurement and modelling of the effects of initial soil conditions and slope gradient on soil translocation by tillage. *Soil & Tillage Research* 51, 303-316.
- Van Oost, K., Govers, G., De Alba, S., Quine, T.A. 2006. Tillage erosion: a review of controlling factors and implications for soil quality. *Progress in Physical geography* 30(4), 443-466.
- Van Oost, K., Govers, G., Desmet, P.J.J., 2000. Evaluating the effects of changes in landscape structure on soil erosion by water and tillage. *Landscape Ecology* 15, 577-589.
- Van Oost, K., Govers, G., Van Muysen, W., 2003. A process-based model for Caesium-137 derived erosion rates on agricultural land: an integrated spatial approach. *Earth Surface Processes and Landforms* 28, 187-207.
- Van Rompaey, A.J.J., Verstraeten, G., Van Oost, K., Govers, G., Poesen, J., 2001. Modelling mean annual sediment yield using a distributed approach. *Earth Surface Processes and Landforms* 26(11), 1221-1236.
- Vitikainen, A., 2004. An overview of land consolidation in Europe. *Nordic Journal of Surveying and Real Estate Research* 1, 25-44.
- Walling, D.E., Quine, T.A., 1992. The use of Caesium-137 measurements in soil erosion surveys, Erosion and sediment transport monitoring programmes in river basins. IAHS, Oslo, pp. 143-152.
- Wemple, B.C., Jones, J.A., Grant, G.E., 1996. Channel network extension by logging roads in two basins, Western Cascades, Oregon. *Water Resources Bulletin* 32, 1195-1207.
- Zadora-Rio, E., 1991. Les terroirs médiévaux dans le Nord-Ouest de l'Europe, Pour une Archéologie agraire. J. Guilaine, Paris, pp. 165-192.

Table 1. Values of parameters and coefficients input in the model application.

Parameter / coefficient	Related to Equation	Values	
a (-)	(3)	1.45	
b (-)	(3)	0.75	
d (-)	(4)	0.8	
		Minimum	Maximum
k ₁ (-)	(3)	0	0.40
k ₂ (m)	(5)	20	220
k ₃ (Mg.m ⁻¹ .yr ⁻¹)	(6)	300	1000

Table 2. Simulated soil redistribution rates for the whole study area for the period 1954-2009. P5 and P95 are the 5th and 95th percentile simulation limits, respectively, and M the median.

	Mean erosion (Mg.ha ⁻¹ .yr ⁻¹)			Mean deposition (Mg.ha ⁻¹ .yr ⁻¹)		
	P5	M	P95	P5	M	P95
Water	-0.69	-0.20	-0.13	0.07	0.13	0.24
Tillage	-4.55	-4.50	-4.46	4.47	4.50	4.55

Table 3. Results of the conversion model applications to “Total” and “Infield” datasets.

Dataset	Elevation inputs	R ²	Mean erosion (Mg.ha ⁻¹ .yr ⁻¹)	Mean deposition (Mg.ha ⁻¹ .yr ⁻¹)	Implication of Tillage (%)
Total	LIDAR	0.20	-4.70	4.63	~ 98
Infield		0.42	-5.82	5.77	~ 98

Fig. 1. Location of the study area.

Fig. 2. Reconstruction of the field borders in 1945, 1959 and 2009.

Fig. 3. Geometrical characteristics of the two types of anthropogenic landforms, and the associated soils: (a) and (c) provide a picture and a cross-section of a lynchet; (b) and (d) provide a picture and a cross-section of an undulation.

Fig. 4. ^{137}Cs sampling schemes. (a) Sampling in the whole study area. (b) Zoom on samples to give details on lynchet L1.

Fig. 5. Vertical distribution of ^{137}Cs in: (a) mean reference profile of Seuilly, (b) profile V-8 and (c) profile IV-1a.

Fig. 6. ^{137}Cs residuals measured in the study area with slope gradient contours.

Fig. 7. ^{137}Cs residuals measured along a topographic cross-section within lynchet L1 (transect IV).

Fig. 8. Model outputs overlayed on an isometric projection of the study area topography: (a) simulated ^{137}Cs inventories, (b) simulated soil redistribution rates by tillage erosion, and (c) simulated soil redistribution rates by water erosion.

Fig. 9. Median simulated ^{137}Cs inventories plotted against observed ^{137}Cs inventories. Black line is the 1:1 line and grey lines correspond to the ^{137}Cs reference inventory (i.e., 1367 Bq.m⁻²).

Fig. 10. Spatial distribution of a) simulation errors and b) simulated ^{137}Cs residuals in the study field.

Fig. 11. Median simulated ^{137}Cs inventories, based on the “infield dataset”, plotted against observed ^{137}Cs inventories. Black line is the 1:1 line and grey lines correspond to the ^{137}Cs reference inventory (i.e., 1367 Bq.m⁻²).

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