Combining 137Cs measurements and a spatially distributed erosion model to assess soil redistribution in a hedgerow landscape in northwestern France (1960–2010)

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Abstract

Erosion is one of the main threats to the soils and it is associated with numerous environmental and economic impacts. At the landscape scale, soil redistribution patterns induced by water and tillage erosion are complex, and landscape structures play an important role on their spatial distribution. In this study, soil redistribution patterns were estimated in the vicinity of hedges in an agricultural landscape, as generated by both water and tillage erosion. Two complementary methods were employed to estimated soil redistribution for the time period from 1960 to 2010: ^{137}Cs conversion models and a spatially-distributed soil erosion model (LandSoil model). Both methods established that hedges affected soil redistribution patterns, leading to soil deposition or limiting soil erosion uphill from hedges, even if soil erosion rates were always higher than soil deposition rates. Depending on the method, the mean soil redistribution rates ranged between -15.9 and -4.7 t ha^{-1} yr^{-1} for all the study sites, -4.8 t ha^{-1} yr^{-1} or 2.2 t ha^{-1} yr^{-1} in positions uphill from hedges, while the rates reached -4.8 to -11.2 t ha^{-1} yr^{-1} in positions located downhill from hedges. The impact of tillage on the soil redistribution in the vicinity of hedges was found to be more important than water processes because 87% of the soil net redistribution was linked to tillage. This confirmed the importance to take landscape structures into account and to work at the

landscape scale rather than at the plot scale to better estimate soil redistribution in agricultural areas.

Keywords: soil redistribution; ^{137}Cs; spatial modelling; hedge; agricultural landscape

1. Introduction

In 2006, the European Commission identified soil erosion as one of the major threats on soils. Soil erosion may affect all soil functions (Boardman and Poesen, 2006), also described as soil ecosystem services (Dominati et al., 2010): physical support of life and human activity, food and fibre production, water filter, carbone storage and climate regulation, etc. Soil erosion has been recognised to have direct consequences on these services both on-site (because of the soil loss from fields), and off-site: during the last decades, a significant increase in environmental issues such as eutrophication, pollution of water bodies and reservoir sedimentation has been observed in Europe, as a result of soil erosion on agricultural land (Boardman and Poesen, 2006). In numerous cases soil erosion leads to a significant reduction in soil thickness. If soil thickness decline is not compensated by soil formation, soil erosion may induce the loss of soil nutrients (Bakker et al., 2004) or soil organic carbon (Papiernik et al., 2005; Papiernik et al., 2009), and threaten the sustainability of crop production (Bakker et al., 2004). Methods and models have been developed to estimate soil redistribution by erosion and to understand the effect of several parameters on this redistribution (climate, soil properties, land use and agricultural practices, landscape structure, etc.). Before the 1990s, studies focused mostly on water erosion, because this was the most obvious process contributing to soil exportation out of cultivated fields (Govers et al., 1996). However, it is now recognized that tillage erosion is also an important process to consider, especially when studying soil loss and deposits within individual fields (Govers et al., 1994). Regarding erosion rates, tillage erosion can have an equivalent or even a higher impact than water erosion on soil redistribution (Govers et al., 1999; Lobb et al., 2007; Van Oost et al., 2005; Chartin et al., 2013). Both water and tillage erosion depend on topography, but have distinct impacts on soil redistribution regarding spatial patterns (Li et al., 2007). Water erosion is maximal on steep mid-slopes and where water concentrates, whereas tillage induces maximum erosion at convexities and deposition at concavities (Govers et al., 1996; Li et al., 2007; Thiessen et al., 2009; Van Oost et al., 2005). Moreover, there are linkages and interactions between water and tillage erosion (Li et al. 2007).

Runoff and soil erosion have been studied at different scales, from plots to catchments, and it appears that both landscape management and structure have an impact on soil erosion and sedimentation in agricultural land. Impact of land use on soil redistribution has been investigated in many studies, over a large range of spatial extents. Cerdan et al. (2010) considered European soil erosion studies conducted at plot scale and showed that spring crops and vineyards were the most sensitive land uses to soil erosion. From a long-term survey of soil erosion at the catchment scale, Prasuhn (2012) showed that potatoes were the crop inducing the most serious soil erosion. Consequently, land use change has an impact on soil redistribution dynamics. Vanniere et al. (2003) examined the impact of historical human occupation on soil redistribution at the hillslope scale. They explained some recorded variations in erosion by changes in agricultural activities. Bakker et al. (2008) estimated that past land-use change (de-intensification or intensification) in four European landscapes directly impacted soil erosion and sediment export to rivers. Besides land use, the farming practices, and particularly tillage practices, impact the soil redistribution. Van Muysen et al. (2000) showed that soil distribution depends on tillage speed and depth. Prasuhn (2012) observed that conventional plough tillage induced higher soil erosion rates than reduced tillage practices. However, it has been shown that these factors (land use and farming practices) were not sufficient to understand soil redistribution at landscape and catchment scales. Bakker et al. (2008) underlined that the spatial pattern of land use change strongly impacted soil redistribution and export out of the studied catchments. In this context, the spatial distribution and the connectivity between areas producing soil erosion and the zones where deposition takes place should be taken into account in the framework of studies conducted at the landscape or catchment scale (Cerdan et al., 2012; Delmas et al. 2012). Vegetated filter strips are part of the anthropogenic structures that impact connectivity inside a landscape, with an effect on water and sediments transfer (Bracken and Croke, 2005; Evrard et al., 2008; Gumiere et al., 2011). More particularly, linear structures such as hedges have been recognised as key elements of the landscape to prevent or limit erosion (Baudry et al., 2000; Boardman and Poesen, 2006; Kiepe, 1995b; Skinner and Chambers, 1996). During the last decades, important changes in landscape structure and soil use have been observed in Western Europe: land use homogenisation, removal of linear structures such as hedges and loss of connectivity between landscape elements were outlined to be the main observed changes (Burel and Baudry, 1990; Deckers et al., 2005; Petit et al., 2003). Such changes in landscape modify soil redistribution dynamics (Evrard et al., 2010) and should be taken into account in soil redistribution modelling.

Ability of empirical models (e.g. USLE) to integrate the dominant processes of soil redistribution at the catchment scale is uncertain (Kirkby et al., 1996), whereas process-based models require numerous input data, which are generally not available and difficult to measure (Takken et al., 1999). In such a context, spatially-distributed and expert-based models (e.g. STREAM; Cerdan et al., 2002a) can offer an alternative solution, especially when dealing with connectivity issues in landscapes (Gumiere et al., 2010). Such models focus on the dominant processes to avoid over-parameterisation and the associated uncertainties, and model simulations rely on decision rules derived by expert judgment from databases of field measurements carried out in a specific region. However, validation of such models remains an important issue in areas where experimental data, i.e. runoff and erosion measurements, are missing. This issue can be addressed by using ^{137}Cs. ^{137}Cs is an artificial radionuclide (half-life of 30 years) produced by the thermonuclear bomb tests conducted during the 1960s as well as, in certain regions of the world (i.e., mainly in Europe), by the Chernobyl accident in 1986. ^{137}Cs is now stored in soils, and its stock decreases by radioactive decay and by fine sediment transfer due to water and tillage erosion. ^{137}Cs has been widely used as a tracer of soil redistribution and it proved to be useful in soil erosion studies conducted around the world (Ritchie and McHenry, 1990; Zapata, 2003). Several studies showed a good correlation between soil redistribution obtained from ^{137}Cs inventories and field measurements (Kachanoski, 1987; Mabit et al., 2002; Porto and Walling, 2012; Porto et al., 2001; Porto et al., 2003a; Porto et al., 2003b) and ^{137}Cs has been used previously to validate or calibrate erosion models (Bacchi et al., 2003; Li et al., 2000; Li et al., 2007; Li et al., 2008; Porto et al., 2003b; Quine, 1999; Tiessen et al., 2009; Walling et al., 2003). The use of ^{137}Cs estimates of soil redistribution relies on several hypotheses, especially that the distribution of local fallout was uniform (Walling and Quine, 1992). Such a statement could be uncertain in complex hedgerow landscapes, especially close to hedges (Follain et al., 2009). Moreover, Parsons and Foster (2011) underlined that the conditions necessary to the use of ^{137}Cs as a soil redistribution indicator are most often not verified. Another limitation is that ^{137}Cs is not a spatially integrative measurement and its high cost may limit sampling at landscape scale.

In this study, we aim to combine two methods to estimate the spatial and temporal soil redistribution dynamics near hedges from 1960 to 2009, in a rural hedgerow landscape. A new model simulating soil redistribution at the landscape scale (LandSoil, Ciampalini et al., 2012) and ^{137}Cs measurements have been used to this end. The results of both methods will be compared and discussed.

2. Materials and methods

2.2. Study sites

The study sites were selected within the study area of Pleine-Fougères (NW France, 48° 505’ N, 1° 565’ W), which belongs to the European Long-Term Ecosystem Research Network, and covers an area of 10 km² (Fig. 1). This area is characterised by a high soil spatial heterogeneity. Soil types are mainly Cambisols and Luvisols, but Leptsosols and Fluvisols from alluvial and colluvial deposits are also found (IUSS Working Group WRB, 2007). These soils exhibit features reflecting variable redoximorphic conditions, soil and A-horizon thickness, and soil parent material (granite, hard and soft schist with a heterogeneous cover of superficial deposits such as aeolian loam, alluvium and colluviums). Main topographical features may be associated with the presence of different geological substrates: granite under the plateau (south of the study area), Brioverian schists under plains (north of the area), metamorphic schists at the transition between granite and soft schist under hillslides. In addition, the presence of linear anthropogenic structures at field boundaries (hedges, banks, ditches, lanes and roads) delineates microtopographical changes. Landscape evolution has been driven by former agricultural policies and local farming practices, consisting in the enlargement of fields in order to facilitate the use of large machinery. Numerous hedges have been removed during a land consolidation programme conducted in the early 1990s, and some are still selectively removed by farmers. The result is that hedge density decreased from 250 m ha^{-1} in 1952 to 90 m ha^{-1} in 2000 (unpublished data, derived from analysis of aerial photographs using Geographical Information Systems). The main land uses are annual crops (maize (*Zea mays*), winter wheat (*Triticum aestivum*)) and temporary or permanent grasslands (mostly Italian ryegrass, *Lolium multiflorum*). Hedges mainly consist of grass strips planted with trees such as oaks (*Quercus robur)*, chesnuts (*Castanea sativa)* and blackthorns (*Prunus spinosa*).

Six 30-m long transects, perpendicular to six hedges (one transect per hedge) and parallel to the slope have been determined (Table 1). They have been chosen to document the variability observed across the study area in terms of soil depth, local slope and drainage area. Five transects were intersecting currently existing hedges, whereas one transect was selected along a hillslope where the hedge was removed in 1970. We ensured that each transect was representative of soil redistribution along the hedge by conducting a slingram electromagnetic survey (EM31). Along each transect, eight locations were sampled at increasing distances from the hedge: one point in the hedge bank, four points uphill from the hedge at 20, 10, 5 and 2 m and three points downhill of the hedge (at 2, 5 and 10 m). A field survey was carried out in 2009. Soils were described at every sampling point using auger (soil type, soil thickness, A-horizon thickness, colour, texture, gravel content, soil organic carbon content) and a precise toposequence of soil horizons was drawn. Soil redistribution along transects was inferred from comparison between the observed A-horizon thickness and the tilled layer thickness. The size of the fields where the transects were located ranged from 0.3 to 3 ha (mean size = 1 ha).

< Figure 1 near here >

< Table 1 near here >

2.3. Estimating soil redistribution rates based on ^{137}Cs measurements

For each hedge, two soil profiles were sampled for ^{137}Cs measurements in March 2010: samples were systematically collected at 5 m uphill and downhill from the hedge, respectively. Undisturbed soil cores were sampled with a 7.5 cm diameter hydraulic core sampler (SEDIDRILL 80) to document the total A-horizon thickness (up to 90 cm). All cores were cut into sections of variable length: the first section corresponded to the uppermost 30 cm (i.e. the ploughed mixed layer), whereas the deeper sections were cut into 5 cm sections. All samples were dried at 105°C, weighed and sieved to 2 mm. ^{137}Cs activity was then measured for 102 samples, at 661 keV using Germanium gamma-ray detectors (Germanium hyperpure – GeHP, N-type, coaxial model) for 80,000 to 300,000 s.

Total ^{137}Cs inventory (^{137}Cs surface activity; A_S in Bq m^{-2}) of each core was then calculated according to Eq. (1):

$$A_S = \sum_{i=1}^{n} A_i \times \frac{M_i}{S} \qquad (1)$$

where A_i is the ^{137}Cs concentration in each sub-sample i of the core containing ^{137}Cs (Bq kg^{-1}); M_i is the mass (kg) of the soil fine fraction of each sub-sample i; S is the surface area (m^2) of the soil core; n is the number of sections in the soil core.

In order to estimate whether soil deposition or erosion occurred in the investigated area, ^{137}Cs inventories were compared to an additional soil core, used as reference and sampled at an undisturbed site, i.e. in a pasture located to the south of the study area (a flat part of the plateau), where we considered that soil redistribution was very unlikely to occur during the past 60 years. Soil redistribution rates (t ha^{-1} yr^{-1}) were calculated using the ^{137}Cs conversion models developed by Walling and He (2001). The use of these models relies on four hypotheses (Walling and Quine, 1992): (i) the ^{137}Cs fallout is locally and spatially uniform, (ii) the fallout is rapidly and irreversibly fixed onto soil particles; (iii) the subsequent redistribution of fallout is due to the movement of soil particles; and (iv) estimates of soil erosion can be derived from measurements of ^{137}Cs inventories. Regarding the irreversible binding issue, Kato et al. (2012a) observed after the Fukushima Dai-ichi nuclear accident that the bulk of radiocaesium was absorbed in the upper 2.0 cm in the soil profile.

In this study we applied four ^{137}Cs conversion models: two proportional models (PM) and two mass-balance models (MBM). PM are based on the premise that ^{137}Cs fallout inputs are completely mixed within the plough layer, and that the soil loss is directly proportional to the amount of ^{137}Cs removed from the soil profile since the beginning of the ^{137}Cs accumulation. PM-1954 considers ^{137}Cs fallout since 1954, whereas PM-1963 takes ^{137}Cs into account since 1963 (when the major fallout occurred). MBM are more complex. The simplified mass balance model (MBM1) takes into account the progressive decrease of ^{137}Cs content in the plough layer because of the mixing with soil which lowers concentration of ^{137}Cs. Additionally, MBM2 takes into account the variation in ^{137}Cs fallout over time (based on the annual ^{137}Cs fallout measured in the northern hemisphere) and the ^{137}Cs redistribution before its incorporation in soil by ploughing. Both MBM1 and MBM2 consider the major ^{137}Cs fallout since 1963. These ^{137}Cs conversion models have been implemented in software developed by Walling and He (2001).

2.4. Modelling soil redistribution over a 50-year period (1960-2010)

The LandSoil model (Ciampalini et al, 2012) was used to simulate soil redistribution between 1960 and 2009. This model is based on a combination of the STREAM runoff and soil erosion model (Cerdan et al., 2002a; Cerdan et al., 2002b; Souchere et al., 2003; Souchere et al., 1998) and the WaTEM/SEDEM tillage erosion model (Van Oost et al., 2000, Van Rompaey et al., 2001, Verstraeten et al., 2002). LandSoil is an expert-based model designed to simulate soil redistribution at a fine spatial resolution scale (1–10 m), a medium-term temporal scale (10–100 yrs), and for study areas ranging from the field to the catchment. LandSoil is spatially-distributed, event-based, and aims to simulate soil erosion (interrill, concentrated and tillage erosion) and deposition. The model assumes that surface characteristics (soil surface crusting, surface roughness, vegetation cover (crops and residues) and residual water storage after the previous event) are the main drivers of runoff and infiltration at the field scale. A soil infiltration rate value (IR), i.e. a constant soil infiltration rate reached during prolonged rainfall which determines the production of runoff during a rainfall event, is assigned to each combination of these soil surface characteristics (Le Bisonnais et al., 1998;2005). For instance, soils with thick depositional crusts and low vegetation cover were assigned with low values of IR, whereas soils with high roughness and fragmentary structure have higher IR values. In the same way, the model assumes that surface characteristics and the maximum intensity of rainfall are the main drivers of sediment concentration in runoff. This concentration is also determined for each combination of the soil surface characteristics and rainfall maximum intensity (Cerdan et al. 2002c). To estimate the soil water erosion, the IR parameter is combined with the potential sediment concentration in runoff and with landscape properties (mainly the slope intensity). An adaptation of the rules to the local context (types of crops planted and climate) is required before running the model (Evrard et al., 2009; Evrard et al., 2010).

Several studies have estimated IR on loamy soils from Western France using plot measurements under rainfall simulation or field scale monitoring networks under natural rainfalls. Soils considered in these studies differed slightly by texture and carbon content, but showed large variations of IR associated with the variations of soil surface characteristics throughout time: Cros-Cayot (1996) and Lacoste (2012) found IR ranging from 1 to 25 mm h^{-1} depending on surface characteristics. In different sites characterised by similar soil types, Le Bissonnais et al. (1992) and Evrard et al. (2008) estimated variations between 2 to 50 mm h^{-1}. To take into account this variability, three ranges of IR variation were considered (Table 2): 1 to 25 mm h^{-1} (IR_1), 1.5 to 37.5 mm h^{-1} (IR_2) and 2 to 50 mm h^{-1} (IR_3). Concerning hedges, previous studies found a wide range of IR under hedges, according to hedge composition, thickness and climate of the study area (Table 3). In this study, IR under hedges was fixed at 150 mm h^{-1}. Regarding the soil concentration in runoff, the default values were determined after Cerdan et al. (2002c) and ranged from 0 to 25 g L^{-1}. These default values have been estimated for loamy soils with similar characteristics as those of the soils in our study area, and local studies confirmed the relevance of their use in our study (Gascuel-Odoux et al, 1996; Lacoste, 2012).

LandSoil simulates soil deposition in two cases: for a given pixel, (i) soil deposit is modelled when water infiltration is higher than runoff, and (ii) when the sediment transport by water erosion is limited. For this second case, the maximum sediment concentration is controlled by several threshold functions based on the local topography and soil cover (Cerdan et al, 2002c). These functions include profile curvature (concavity > 0.055 m^{-1}), slope gradient (< 5 %), land use (permanent grassland and wood) and vegetation cover (> 60 %).

< Table 2 near here >

< Table 3 near here >

The LandSoil model was run for all rainfall/tillage events that occurred from 1960 to 2009 in the areas where ^{137}Cs inventories were available (Table 1). The modelling areas included one field uphill and one downhill from the hedges where transects were investigated, and the field boundaries (hedges, banks and ditches).

The following inputs were provided to LandSoil to model soil redistribution:

(i) Initial elevation: a 2-m LiDAR DEM (light detection and ranging digital elevation model), produced in 2009, was used. It allows taking into account fine topographic variations close to the hedges.

(ii) Crop rotations and associated soil surface characteristics (soil surface crusting, surface roughness, and vegetation cover). We used aerial photographs, taken during summer in 1966, 1968 and from 1993 to 2009, to create crop transition matrices (based on Markov chain) and allocated a main crop per field for each year from 1960 to 2009. Crop rotations consisted of a succession of maize, winter cereals and temporary grassland. Monthly soil surface characteristics were attributed to each crop from expert knowledge and field survey data.

(iii) Soil tillage operation data (direction of tillage and coefficient of tillage erosion). Two tillage transport coefficients (KTILL) are used by LandSoil to model soil redistribution by tillage: KTILLmax for describing soil redistribution parallel to the tillage direction and KTILLmin for characterising soil redistribution perpendicular to the tillage direction. For years with maize or winter cereals sowing, the sequence of tillage operations consisted in the use of mouldboard plough (25 to 30 cm depth), chisel, rotary harrow and air seeder. For years where grassland was established, the sequence of tillage operations consisted in the use of chisel and air seeder. Van Muysen et al. (2006) showed that the KTILL of a sequence of tillage operations can be predicted by summing the KTILL of all individual tillage operations. They also calculated that the mean annual $KTILL_{max}$, associated with mechanized agriculture, is in the order of 781 kg m^{-1} yr^{-1}. Mean annual KTILL coefficients were defined based on the results of previous studies (Table 4). Values for $KTILL_{max}$ and $KTILL_{min}$ reached respectively 631 and 376 g m^{-1} for the years with maize or winter wheat sowing, and 291 and 139 g m^{-1} for the years with grassland sowing.

(iv) Rainfall event characteristics: rainfall amount (mm), rainfall maximum intensity (mm h^{-1}), effective rainfall duration (h) and rainfall amount during the 48-h before the event (mm). These parameters were defined from predictive 6-minute meteorological data from 1960 to 2009, estimated from hourly meteorological data from Lacoste (2012). Hourly data were provided by the INRA unity Agroclim for the Rennes Station.

< Table 4 near here >

From LandSoil outputs, we computed for each 2-m pixel of the modelled areas the soil redistribution rates and the proportion of soil redistribution by water erosion processes compared to the absolute net soil redistribution.

2.5. Comparing soil redistribution rates derived from ^{137}Cs and from modelling

For the two methods, negative values of soil redistribution rate correspond to soil erosion rates, whereas positive values are soil deposition rates. The soil redistribution rates estimated by ^{137}Cs were located and valid for the sampling points alone, whereas those estimated from LandSoil modelling were grid maps, spatially explicit across the fields located uphill and downhill from the studied hedges. To facilitate comparison between both estimation methods, soil redistribution rates derived from LandSoil modelling were averaged over 10*10 meters windows centred on the ^{137}Cs sampling points. The redistribution rates showed in the results refer to these two spatial extents: points for the ^{137}Cs-derived estimations and 10*10 meters windows for the LandSoil modelling. These results were compared to the soil redistribution rates obtained from the ^{137}Cs survey using the correlation coefficient R^2 (Eq. 2) and Lin's concordance coefficient (Eq. 3).

$$R^2 = \frac{\sum_{i=1}^{n}(x_i - \hat{x}_i)^2}{\sum_{i=1}^{n}(x_i - \bar{x})^2} \quad (2)$$

$$ccc = \frac{2S_{xy}}{S_x^2 + S_y^2 + (\bar{x} - \bar{y})^2} \quad (3)$$

Where $S_x^2 = \frac{1}{n}\sum_{i=1}^{n}(x_i - \bar{x})^2$, $S_y^2 = \frac{1}{n}\sum_{i=1}^{n}(y_i - \bar{y})^2$, $S_{xy} = \frac{1}{n}\sum_{i=1}^{n}(x_i - \bar{x})(y_i - \bar{y})$

In both equations *x* and *y* are estimates of soil redistribution rate by ^{137}Cs and LandSoil, respectively, y_i and x_i are estimate of soil redistribution rate for the sampling site i, $\bar{x}$ is the mean value x and n is the number of sampling sites.

R^2 indicates how well two datasets are correlated (linear correlation), whereas CCC measures the agreement between two variables. CCC combines measurements of both precision and accuracy to determine how far the observed data deviate from the line of perfect concordance (i.e. the 45°-line on a square scatter plot).

3. Results

3.1. Soil redistribution rates based on ^{137}Cs measurements

Fig. 2 gives an example of the soil redistribution patterns observed along the transects crossing the hedges H4 and H5, and the associated ^{137}Cs concentration distributions with depth. These two hedges were located in the centre of the study area, on the hillside, on deep soils developed in aeolian loams. For both H4 and H5, we observed soil deposition uphill from the hedge (characterised by an A-horizon thicker than the ploughed horizon), and soil erosion downhill from the hedge (characterised by a thinner A-horizon).

The reference ^{137}Cs inventory was 1590 Bq m^{-2}. The results of ^{137}Cs inventories calculated uphill and downhill from each hedge were compared in Table 5. Except for H4 and H5, ^{137}Cs inventories measured uphill and downhill from the hedges were lower than the reference ^{137}Cs inventory. For all the hedges, H3 exepted, ^{137}Cs inventories uphill from all the other hedges differed from the ones calculated for downhill positions and ranged from 123 Bq m^{-2} (H6) to 1010 Bq m^{-2} (H2). For three hedges (H1, H4 and H5), ^{137}Cs inventories were higher in uphill positions. For three hedges (H2 and H6) the opposite result was observed. For the hedge H3, ^{137}Cs inventories were similar in both positions (difference of 8 Bq m^{-2} between both positions).

The soil redistribution rates estimated from the four ^{137}Cs conversion models were well correlated, with an R^2 ranging from 0.94 to 1 and CCC ranging from 0.80 to 0.97 (Table 5). The largest difference between model estimates was obtained for $H2_{up}$ (where the hedge was removed in 1970), with a difference of 41 t ha^{-1} yr^{-1}. According to ^{137}Cs inventories, the mean soil redistribution rate was estimated at -12.60 t ha^{-1} yr^{-1}, the mean soil erosion rate was estimated at -15.89 t ha^{-1} yr^{-1}, and the mean soil deposition rate at 3.82 t ha^{-1} yr^{-1}. Among the six sampling sites, four were estimated to be erosion sites, both for positions uphill and downhill from the hedge (H1, H2, H3 and H6). For transects across hedges H4 and H5, positions uphill from the hedge were estimated to be deposition sites, whereas positions downhill from the hedge were estimated to be erosion sites. Focusing of the 5 currently existing hedges, the mean soil redistribution rate was estimated at -4.77 t ha^{-1} yr^{-1} for the positions uphill from the hedges, and at -11.15 t ha^{-1} yr^{-1} for the positions downhill from the hedges.

< Figure 2 near here >
< Table 5 near here >

3.2. Soil redistribution rates over 50 years based on soil redistribution modelling with the LandSoil model

Soil redistribution rates obtained with LandSoil are summarized in Table 6. The soil redistribution rates estimated using the three sets of soil infiltration rates were strongly correlated, with an R^2 ranging from 0.96 to 1 and CCC ranging from 0.91 to 0.99. The largest differences between estimates were obtained for $H4_{down}$ and $H5_{down}$, with a difference of 3.3 t ha^{-1} yr^{-1}. Overall mean soil redistribution rates showed that the model usually predicted erosion along the simulated transects. The mean soil redistribution rate was estimated at -1.20t ha^{-1} yr^{-1}, the mean soil erosion rate at -4.73 t ha^{-1} yr^{-1}, and the mean soil deposition rate at 2.34 t ha^{-1} yr^{-1}. Among the six sampling sites, LandSoil modelled soil deposition uphill from hedges and soil erosion downhill from hedges for five hedges (H1, H3, H4, H5 and H6). The opposite pattern was modelled for the hedge H2 (i.e. the area uphill from hedge experienced soil deposition and area downhill from hedge experienced soil erosion). H2 corresponds to the situation where the hedge was removed in 1970. Focusing of the 5 currently existing hedges, the mean soil redistribution rate was estimated at 2.18 t ha^{-1} yr^{-1} for the positions uphill from the hedges, and at -4.84 t ha^{-1} yr^{-1} for the positions downhill from the hedges.

Relative contribution of water and tillage erosion processes to soil redistribution is given in Table 6. Contribution of water redistribution to absolute net soil redistribution ranged from 7 to 64% (mean value: 13%). Considering soil redistribution due to the single tillage operations only, the mean soil redistribution rate was estimated at -0.84 t ha^{-1} yr^{-1}, the mean soil erosion rate at -4.30 t ha^{-1} yr^{-1}, and the mean soil deposition rate at 2.61 t ha^{-1} yr^{-1}. LandSoil only modelled water erosion in the vicinity of the hedges, and the mean soil erosion rate was estimated at -0.36 t ha^{-1} yr^{-1}.

< Table 6 near here >

Fig. 3 shows the annual variations of the contribution of water erosion processes to absolute soil redistribution between 1960–2009, and the cumulative soil redistribution. Depending on the year, the transect and the position from the hedge, the contribution of water erosion processes to absolute soil redistribution ranged from 0 to 76%. The contribution of water redistribution to absolute soil redistribution remained lower than 50% for five hedges (H1, H2, H4 and H5), and reached 50% and more for two hedges (H3 and H6). For all hedges except H3, the contribution of water erosion processes to absolute soil redistribution was similar or higher for positions located uphill of hedges than for positions situated downhill from hedges. For H3 the opposite pattern was obtained from the model. For all hedges except H2, results showed no change in soil redistribution dynamics: positions uphill from hedges always proved to be deposition sites, whereas positions downhill from hedges were erosion sites. For hedge H2, both positions uphill and downhill from the hedge were deposition sites before the hedge removal in 1970, and then the position uphill from hedges became an erosion site, whereas the position downhill from hedges was a deposition site.The intra and inter-annual variations of the modelled soil redistribution were due to the combination of meterological conditions (extreme rainfall events) and soil cover linked to land use.

Fig. 4 and Fig. 5 show the spatial distribution of the soil redistribution simulated by LandSoil for the hedges H1, H3 and H4. For the three hedges shown in Fig. 5, LandSoil modelling results showed (i) soil deposition uphill from the hedge; (ii) soil erosion downhill from the hedge, and (iii) higher soil redistribution at the field boundaries rather than within the fields. For the modelling areas shown in Fig. 4, the mean soil redistributions across the entire fields ranged from -0.07 to -0.42 t ha^{-1} yr^{-1}, whereas it ranged from -8.26 to 3.95 t ha^{-1} yr^{-1} in the vicinity of the hedges. Fig. 6 shows the relative contribution of water erosion processes to the net soil redistribution. For the three hedges H1, H4 and H5, tillage redistribution tended to be dominant at the vicinity of the hedge. Water redistribution was nevertheless dominant within the fields.

< Figure 3 near here >

< Figure 4 near here >

< Figure 5 near here >

3.3. Comparison of soil redistribution rates from ^{137}Cs measurements and LandSoil model

The correlation coefficient R^2 and Lin's concordance coefficient (CCC) were calculated to compare the soil redistribution rates estimated from ^{137}Cs inventories and from LandSoil. Considering all the study sites, R^2 and CCC reached 0.17 and 0.12, respectively. Considering only the positions uphill from the hedges, R^2 and CCC amounted to 0.99 and 0.15, respectively. Finally, considering only the positions downhill from the hedges, R^2 and CCC was equal to 0.29 and 0.17, respectively.

Fig. 6 compares the soil redistribution rates modelled from ^{137}Cs inventories and LandSoil. The rates overlap only for two transects in uphill hedge positions (H4 and H5), and two transects in downhill hedge positions (H4 and H6). The two methods predicted a different dominant redistribution process for three hedges. For the hedges H3, H6 and H1, positions uphill from the hedges were estimated to be erosion sites by ^{137}Cs inventories conversion models, whereas they were estimated to be deposition sites by LandSoil model. The same pattern was observed for the hedge H2 (for the position downhill from the hedge).

Soil redistribution rates estimated from ^{137}Cs inventories conversion models were higher for all study sites, except for hedge H5 (uphill from the hedge) and the hedges H6 and H4 (downhill from the hedge). Hedge H2, removed in 1970, showed the largest estimate differences: for the position uphill from the hedge both methods estimated soil erosion, but the mean soil erosion was estimated at -4.2 t ha^{-1} yr^{-1} by LandSoil and -66 t ha^{-1} yr^{-1} from the ^{137}Cs inventories.

< Figure 6 near here >

4. Discussion

4.1. Methods to estimate soil redistribution patterns and rates

4.1.2. Soil redistribution estimated from ^{137}Cs inventories

Among the five transects on a still existing hedge, three were estimated to be erosion sites, both for positions uphill and downhill from the hedge (H1, H3 and H6). In these cases, soil erosion rates were estimated lower on uphill positions than on downhill positions. For the two others (H4 and H5), positions uphill from the hedge were estimated to be deposition sites, whereas positions downhill from the hedge were estimated to be erosion sites. This last result was more consistent with previous studies (Follain, 2006; Follain et al., 2006; Walter et al., 2003).

The use of ^{137}C inventories to estimate soil redistribution raise several issues for discussion, especially in the context of a heterogeneous hedgerow landscape. The first point to consider is the choice of the ^{137}C conversion model, used to convert ^{137}C inventories to soil redistribution rates. In this study, two PM models (PM-1954 and PM-1963) and two mass balance models (MBM1 and MBM2) have been applied. The soil redistribution rates estimated from these four models were very similar and very well correlated (Table 5).

The second important point is the initial deposit of ^{137}C. In a hedgerow landscape, hedges obstruct winds, so the diffusion of ^{137}Cs in the landscape by the wind may not be uniform. Moreover, the canopy of trees constituting the hedge is prone to intercept ^{137}Cs before soil deposition (Kato et al., 2012b), which could be another explanation to the non-uniform ^{137}Cs fallout. The ^{137}Cs reference inventory that we used was located in the south of the study area (Fig. 1), inside a flat cultivated field without hedges at its boundaries. It could mean that this reference inventory that we compared to ^{137}Cs inventories on transects was not representative of the initial ^{137}Cs fallout for the whole study area, and specifically for the study sites. If the reference inventory was not representative of the initial ^{137}Cs fallout, soil redistribution patterns and modelled rates using ^{137}Cs conversion models could be biased. This could explain why the patterns of soil redistribution modelled by ^{137}Cs inventory conversion models were not always consistent with known soil redistribution at hedge proximity in such a landscape. For example, soil erosion was estimated uphill from the hedge H1, whereas the survey of A-Horizon thickness (thicker in positions uphill from the hedges than in positions downhill from the hedge) suggested the occurrence of soil deposition. Moreover, previous studies showed that positions uphill from hedges were more prone to soil deposition (Follain, 2006; Follain et al., 2006; Walter et al., 2003). However, such soil deposition could have occurred prior to ^{137}Cs fallout, and positions uphill from the hedges could have become, during the last decades, areas where soil erosion was prevailing over soil deposition. A better estimation of soil redistribution could have been achieved by the use of a more representative reference ^{137}Cs inventory, e.g. a local reference profile for each hedge, sampled in the proximity of each of those landscape features. However, it seems difficult to find such a location, without soil redistribution, in an agricultural landscape. An estimate of the soil redistribution dynamics in the vicinity of hedges could be done without using a ^{137}Cs reference inventory, by comparing for each transect the uphill/downhill ^{137}Cs inventory. For the hedge H1 for example, the ^{137}Cs inventory uphill from the hedge was higher than the ^{137}Cs inventory downhill from the hedge. This is insufficient to conclude that soil deposition occurred uphill from H1, but we may assume that soil erosion was higher downhill from H1. Regarding the hedge H2, which was removed in 1970, the soil redistribution estimate using the ^{137}Cs inventories can be biased because of the possible soil redistribution in the fields by farmers during the hedge removal.

4.1.2. Soil redistribution estimated by LandSoil model

LandSoil is an expert-based model, which allows minimizing the number of parameters to calibrate. However, parameters requiring calibration remain, and these parameters are prone to uncertainties:

(i) Parameters linked to soil redistribution by water processes

Three main parameters have to be defined to simulate soil redistribution by water: infiltration rate, potential sediment concentration in runoff, and factors of soil deposition (or factors controlling the maximum sediment concentration in runoff). In this study, we used three ranges of soil infiltration rates to take soil variability into account, but the limited contribution of water erosion processes to soil redistribution made the results very similar. The default values used in LandSoil for potential sediment concentration in runoff (Cerdan et al, 2002b and 2002c) were consistent with the available data in our study area (Gascuel-Odoux et al., 1996; Lacoste, 2012). Finally, the maximum sediment concentration in runoff was controlled by threshold functions based on four factors: profile curvature, slope gradient, land use and vegetation cover. Threshold values for these factors have been determined by Cerdan et al (2002b and 2002c) using data on runoff and soil redistribution under natural rainfall on loamy soils from Northern France. No data were available in our study area to calibrate these thresholds but, given the observed similarities between the study sites, we assumed that these values were valid in our study site. The threshold values for topographic parameters (curvature and slope), estimated for a 5x5 m DEM, have only been adapted for a 2x2 m DEM.

(ii) Parameters linked to soil redistribution by tillage processes

Regarding soil redistribution by tillage, the most important parameters to calibrate are the tillage erosion coefficients. In this study, sequences of tillage operations have been derived from surveys among farmers. The resulting tillage erosion coefficients have been estimated from previous studies (Table 4).

(ii) Other source of uncertainties

All the input data could be source of uncertainties. In this study, the most sensitive data was the DEM. In fact, a 2-m resolution DEM was used, to take into account the fine topography at hedge vicinity. However, such a DEM may be noisy and needs to be pre-processed. A non-controlled DEM could lead to mis-modelling the drainage network, and consequently the soil erosion and deposition areas.

The soil redistribution patterns modelled by LandSoil were consistent with previous knowledge of soil redistribution in such a landscape, i.e. soil deposition uphill from hedges and soil erosion downhill from hedges (Follain et al., 2006). H2 was the only hedge where an opposite pattern was modelled. H2 was also the only hedge removed during the simulation process, and in such conditions this result is consistent: soil previously deposited uphill from the hedge could be eroded and redistributed downhill from the hedge. LandSoil provides quantitative estimations of the spatial variability of soil redistribution and its variability over time by dynamic modelling. LandSoil also allows distinguishing and quantifying the contribution of the different processes taking part in soil redistribution (in this study water and tillage redistribution). These types of models are particularly interesting to better understand the impacts of landscape structure on soil redistribution processes or to estimate soil variation in time according to various global change scenarios.

4.2. Soil redistribution dynamics in agricultural landscapes

Scale issues

Soil erosion has been studied on a range of temporal and spatial scales. Results showed that there is no simple relationship between erosion rates when up- or downscaling (Chaplot and Poesen, 2012; Delmas et al., 2012; Le Bissonnais et al., 1998). At the scale of Europe, Delmas et al (2012) showed that soil erosion rates decrease from the field to the catchment scale. Moreover, Le Bissonnais et al. (1998) showed that the size of the plot used for soil erosion measurement have an impact on the results. Therefore, comparisons of studies should be conducted carefully. In this study, soil redistribution rates are given at the plot scale (400 m²), located in the vicinity of hedges. However, both methods used to estimate these rates integrate soil redistribution on larger spatial extents: (i) ^{137}Cs inventories take into account all soil particle movements at a given point without scale restriction. In our hedgerow landscape, we assume that the spatial extent of soil redistribution was limited by the hedges located at the field boundaries (field size ranging from 0.3 to 3 ha, with a mean size of 1 ha). Therefore, ^{137}Cs inventories provided a way to estimate soil redistribution rates from point-to-field scales; (ii) the LandSoil model was run at field scale, so it also allowed assessing soil redistribution rates from point-to-field scales. According to ^{137}Cs inventories, the mean soil redistribution rate was estimated at -12.6 t ha^{-1} yr^{-1}, the mean soil erosion rate was estimated at -15.9 t ha^{-1} yr^{-1}, and the mean soil deposition rate at 3.8 t ha^{-1} yr^{-1}. Considering LandSoil estimates, the mean soil redistribution rate was estimated at -1.2 t ha^{-1} yr^{-1}, the mean soil erosion rate was estimated at -4.7 t ha^{-1} yr^{-1}, and the mean soil deposition rate at 2.3 t ha^{-1} yr^{-1}. These rates are close to those estimated in previous studies. Verheijen et al. (2009) synthesized studies at field scales and estimated that soil erosion rates in Europe ranged from -3.2 to -19.8 t ha^{-1} yr^{-1} (considering water, wind and tillage erosion). Both methods used in this study estimated that erosion processes were more pronounced than deposition processes.

This result was also described by Van Oost et al. (2005), who studied soil redistribution at field scale.

Tillage vs. water erosion

In this study, the mean soil redistribution rate due to tillage operation was estimated by the LandSoil model -0.84 t ha^{-1} yr^{-1} considering all situations, -4.30 t ha^{-1} yr^{-1} considering erosion sites and 2.61 t ha^{-1} yr^{-1} considering deposition sites. Considering soil redistribution by water processes, Landsoil estimated that soil erosion (and not soil deposition) was prevailing at the observed locations (erosion rates ranged from -0.09 to -0.52 t ha^{-1} yr^{-1}, mean value = -0.36 t ha^{-1} yr^{-1}). Therefore, these results show that (i) tillage operations were the main processes inducing soil redistribution at the observed locations, and (ii) water processes mainly induce soil erosion at the vicinity of hedges. This first result was shared by Quine et al. (1994) and Van Oost et al. (2005), who showed that field scale soil erosion was mainly due to tillage operations during the last decades (i.e. contemporary erosion). According to various studies (Govers et al.,1996; Van Oost et al., 2005), the mean tillage erosion rates in Europe ranged from 3.0 to 9.0 t ha^{-1} yr^{-1} (studies at plot-to-field scale) and, locally, often exceed 10 t ha^{-1} yr^{-1}, which is consistent with our results. At the field scale, Cros-Cayot (1996) estimated water soil erosion at 1.55 t ha^{-1} yr^{-1} for loamy soils in Brittany. Cerdan et al. (2010) estimated from a synthesis of existing field measurements that the mean soil erosion rate in Europe (rill and interill erosion) was -1.2 t ha^{-1} yr^{-1} considering all land uses, and -3.6 t ha^{-1} yr^{-1} considering only arable land.

Impacts of linear anthropogenic structures

Both methods were associated with uncertainties and it turns out to be difficult to directly compare their respective results. Soil redistribution rates were not well correlated (overall R^2 = 0.17), estimates of soil redistribution patterns were not always consistent, and for most of the sampling sites soil redistribution rates estimated from ^{137}Cs inventories were higher than those estimated from LandSoil.

However, both methods showed that hedges had an impact on soil redistribution. For positions uphill from the hedges, we modelled either soil accumulation or soil erosion with a lower rate than the one modelled for positions downhill from hedges. Focusing of the 5 still existing hedges, the mean soil redistribution rates estimated by LandSoil and ^{137}Cs inventories were 2.2 and -4.5 t ha^{-1} yr^{-1} respectively for the positions uphill from the hedges, and -4.8 and -11.2 t ha^{-1} yr^{-1} respectively for the positions downhill from the hedges. This result was consistent with those of previous studies: hedges act as a trap for soil particles in runoff and can enhance runoff infiltration (Baudry et al., 2000); moreover, hedges act as a zero flux line regarding tillage erosion processes (Govers et al.,1996). Both processes result in the differentiation of soil redistribution on both sides of hedges. Soil redistribution at the landscape scale considering hedge networks has been modeled by Follain et al. (2006). They concluded that hedges have an impact on soil redistribution and landscape topography evolution. In hedge-less landscape, soil can be redistributed over longer distances, which induces landscape leveling. By contrast, in a hedged landscape, hedges contribute to bank creation in soil deposit areas (uphill from hedges). It has also been shown that hedges have an impact on soil horizon succession, particularly regarding the A-horizon thickness (Follain et al., 2009). It comes out that, at the field-to-landscape scale, hedges are a factor of heterogeneity when dealing with soil redistribution (Fig. 4 and 5; Follain et al., 2006; Lacoste, 2012): they impact soil redistribution within fields but they also impact soil export from the fields and from landscapes or catchments. Hedges should therefore be taken into account in studies dealing with soil redistribution. Nevertheless, it could be difficult to take them explicitly into account in studies conducted over large areas (landscape-to-region or country). One solution could be to use a connectivity index as described by Cerdan et al. (2012). This index combines information on slope, lithology and rainfall and has been used to estimate the connectivity and the soil deposit processes in the major French river basins.

5. Conclusions

This study aimed to estimate soil redistribution patterns and rates in a hedgerow landscape for areas close to hedges. We compared two methods, one derived from ^{137}Cs survey and the other based on a spatially distributed soil redistribution model (LandSoil). Soil redistribution rate estimates obtained with ^{137}Cs survey were higher than those obtained with LandSoil, but both were consistent with other values found in previous studies. Estimates from both methods showed that soil erosion processes were dominant in the vicinity of the hedge. Depending on the method, mean soil redistribution rate varied between -4.8 t ha^{-1} yr^{-1} and 2.2 t ha^{-1} yr^{-1} in positions uphill from hedges, whereas they reached -4.8 to -11.2 t ha^{-1} yr^{-1} in positions downhill from hedges. Both methods modelled hedges as anti-erosive landscape elements: estimates of soil redistribution uphill from the hedges showed either soil redistribution or lower soil erosion than positions downhill from hedges. Estimates from LandSoil were consistent with previous studies on soil redistribution in relation to landscape structure, and showed its ability to model soil redistribution in complex hedgerow landscapes. At the field scale, the estimates of soil redistribution by LandSoil showed that soil redistribution was more important at the vicinity of field boundaries. Moreover, the impact of tillage on the soil redistribution in the vicinity of the hedges was found more important than water processes (an average of 87% of the soil net redistribution was due to tillage). However, soil redistribution processes varied in space and in time, and water erosion processes were dominant within the fields. Further work will focus on the impacts of landscape structure on soil redistribution in the context of global change.

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Table 1.
Characteristics of sampling locations selected for calculating ^{137}Cs inventories and modelling.

Hedges								^{137}Cs sampling sites		
Study site	Date of hedge removal (where applicable)	Orientation	Soil parent material	Topographic location	Local slope (%)	Uphill slope length (m)	Modeling area extent (ha)	Position relative to the hedge*	Soil thickness (cm)	A-hz thickness (cm)
H1	-	NE-SW	Brioverian schist	plain	3.3	10.9	1.68	$H1_{up}$	40	40
								$H1_{down}$	21	21
H2	1970	NE-SW	Brioverian schist	plain	3.0	10.3	1.68	$H2_{up}$	55	40
								$H2_{down}$	70	36
H3	-	E-W	Brioverian schist	plain	1.3	5.6	2.47	$H3_{up}$	70	64
								$H3_{down}$	60	53
H4	-	E-W	Aeolian loam	hillside	6.9	7.0	1.64	$H4_{up}$	500	90
								$H4_{down}$	490	30
H5	-	E-W	Aeolian loam	hillside	6.5	24.0	0.91	$H5_{up}$	550	70
								$H5_{down}$	500	30
H6	-	E-W	Hornfels	upper hillside	4.0	30.1	1.21	$H6_{up}$	50	38
								$H6_{down}$	30	30

* up: sampling site uphill from the hedge, down: sampling site downhill from the hedge

Table 2.

Sets of soil infiltration rates (mm h^{-1}) used to model soil redistribution. IR depend on combinations of soil surface characteristics driving runoff (values for $IR_1 - IR_2 - IR_3$).

<table>
<tr><th rowspan="2">Soil roughness[a]</th><th rowspan="2">Vegetation cover[b]</th><th colspan="4">Crusting stage[c]</th></tr>
<tr><th>F0</th><th>F11</th><th>F12</th><th>F2</th></tr>
<tr><td rowspan="3">0 - 1 cm</td><td>< 20 %</td><td>5 - 7.5 - 10</td><td rowspan="2">5 - 7.5 - 10</td><td rowspan="2">2.5 - 3.75 - 5</td><td rowspan="2">1 - 1.5 - 2</td></tr>
<tr><td>21-60 %</td><td>10 - 15 - 20</td></tr>
<tr><td>> 61 %</td><td>25 - 37.5 - 50</td><td>10 - 15 - 20</td><td>5 - 7.5 - 10</td><td>2.5 - 3.75 - 5</td></tr>
<tr><td rowspan="3">1 - 2 cm</td><td>< 20 %</td><td>10 - 15 - 20</td><td>5 - 7.5 - 10</td><td>2.5 - 3.75 - 5</td><td rowspan="2">1 - 1.5 - 2</td></tr>
<tr><td>21-60 %</td><td rowspan="2">25 - 37.5 - 50</td><td rowspan="2">10 - 15 - 20</td><td rowspan="2">5 - 7.5 - 10</td></tr>
<tr><td>> 61 %</td><td>2.5 - 3.75 - 5</td></tr>
<tr><td rowspan="3">2 - 5 cm</td><td>< 20 %</td><td rowspan="3">25 - 37.5 - 50</td><td rowspan="2">10 - 15 - 20</td><td>5 - 7.5 - 10</td><td>2.5 - 3.75 - 5</td></tr>
<tr><td>21-60 %</td><td>5 - 7.5 - 10</td><td>2.5 - 3.75 - 5</td></tr>
<tr><td>> 61 %</td><td>25 - 37.5 - 50</td><td>10 - 15 - 20</td><td>5 - 7.5 - 10</td></tr>
<tr><td rowspan="3">5 - 10 cm</td><td>< 20 %</td><td rowspan="3">25 - 37.5 - 50</td><td>10 - 15 - 20</td><td>5 - 7.5 - 10</td><td>2.5 - 3.75 - 5</td></tr>
<tr><td>21-60 %</td><td rowspan="2">25 - 37.5 - 50</td><td>10 - 15 - 20</td><td rowspan="2">5 - 7.5 - 10</td></tr>
<tr><td>> 61 %</td><td>25 - 37.5 - 50</td></tr>
<tr><td rowspan="3">> 10 cm</td><td>< 20 %</td><td rowspan="3">25 - 37.5 - 50</td><td>10 - 15 - 20</td><td>10 - 15 - 20</td><td rowspan="3">5 - 7.5 - 10</td></tr>
<tr><td>21-60 %</td><td rowspan="2">25 - 37.5 - 50</td><td rowspan="2">25 - 37.5 - 50</td></tr>
<tr><td>> 61 %</td></tr>
</table>

[a] Soil surface roughness state is defined by the elevation difference between the deepest part of micro depressions and the lowest point of their divide (2006).

[b] Vegetation cover classes are defined as the percentage of soil surface covered by canopy or litter

[c] Soil surface crusting stages from Bresson and Boiffin (1990). F0 = initial fragmentary structure; F11 = altered fragmentary state with structural crusts; F12 = local appearance of depositional crusts; F2 = continuous state with depositional crusts.

Table 3 revised

Table 3.
Literature review of infiltration rates under hedges.

Reference	Study site	Hedge type, wide	Infiltration rate (mm h^{-1})
Alegre and Rao, 1996	Loreto, Peru	Contour hedgerow (*Inga edulis*), 0.5 m	500
Anderson et al , 2009	Missouri, USA	Contour strip (*Agrostis gigantea*, *Bromus* spp., *Lotus Corniculatus* L., *Quercus palustris* Muench, *Quercus bicolor* Willd., *Quercus macrocarp* Michx), 4.5 m	17
Bharati et al., 2002	Iowa, USA	Grass strip (*Panicum virgatum* L.), 7.1 m ; Shrub strip (*Cornus stolonifera* Michx., *Physocarpus opulifolius* L.), 2 rows ; Tree strip (*Populus X euramericana* 'Eugenei', *Fraxinus pennsylvanica* Marsh. , *Acer saccharinum* L., *Juglans nigra* L.), 5 rows	90-450
Kiepe et al, 1995a	Kenya	Contour hedgerows of *Cassi siamea* Lam.	69-135
Perret et al., 1996	Reunion Island	*Caliendra* hedges	225
Rachman et al., 2004	Iowa, USA	Grass strip, 0.75-1 m	144-208
Richet et al. 2006	Normandy, France	Grass strip, 0.75-1 m	120-200
Souiller et al., 2002	Loire-Atlantique, I	Grass strip, 3 m^2	83-123 (mean value: 106)

Table 4.
Tillage erosion coefficients available from the literature and chosen in this study

Reference	Implement	Soil bulk density (g cm^{-3})	Tillage depth (cm)	Tillage speed (m s^{-1})	KTILL$_{max}$ (kg m^{-1})	KTILL$_{min}$ (kg m^{-1})
Values for ligth cultivators plus seeders					*60*	*n.a.*
Li et al., 2007	Air seeder	1.27	3	2.23	18	n.a.
Li et al., 2007	Light cultivator	1.25	8	2.23	42	n.a.
Mean values for chisel ploughs					*274*	*139*
Govers et al. , 1994	Chisel plough	1.35	12	1.25	111	n.a.
Poesen et al., 1997	Duckfoot chisel	1.58	15	0.65	282	139
Quine et al, 1999	Duckfoot chisel	1.38	19	2.3	657	n.a.
Tiessen et al. 2007	Chisel plough	1.37	16.2	1.92	64.4	n.a.
Van Muysen et al., 2001	Chisel plough	1.25	20	2.02	258	n.a.
Mean values for mouldboards					*297*	*237*
Lindstrom et al. , 1992	Mouldboard	1.35	24	2.1	330	363
Montgomery et al. 1999	Mouldboard	1.31	23	1	n.a.	110
Revel et al. 1993	Mouldboard	1.35	27	1.8	263	n.a.
Values used for a full tillage sequence for maize or wheat sowing (mouldboard, chisel, light cultivator and air seeder)					*631**	*376**
Values used for a full tillage sequence for grassland sowing (chisel, light cultivator and air seeder)					*292**	*139**

*Values used in this study for soil redistribution modelling
Not available data: n.a.

Table 5
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Table 5.

Statistics of soil redistribution rates derived from the four ^{137}Cs conversion models (reference value = 1590 Bq m^{-2}).

Location	^{137}Cs inventory (Bq m^{-2})	Soil redistribution rate estimation (t ha^{-1} yr^{-1})			
		minimum	mean	median	maximum
$H1_{up}$	1412	-8.71	-7.61	-7.93	-5.87
$H1_{down}$	1274	-15.95	-13.88	-14.24	-11.10
$H2_{up}$	438	-88.48	-65.70	-63.49	-47.35
$H2_{down}$	1448	-6.96	-5.94	-6.18	-4.45
$H3_{up}$	1285	-16.41	-13.83	-13.73	-11.43
$H3_{down}$	1293	-15.93	-13.44	-13.37	-11.09
$H4_{up}$	1741	5.19	6.64	6.82	7.73
$H4_{down}$	1579	-5.29	-2.47	-2.08	-0.46
$H5_{up}$	1611	0.85	1.00	0.95	1.23
$H5_{down}$	1150	-25.32	-20.70	-19.80	-17.86
$H6_{up}$	1326	-12.92	-10.05	-10.28	-6.73
$H6_{down}$	1449	-6.88	-5.25	-5.37	-3.40

Negative values: soil erosion, positive values: soil deposition.

Table 6.

Statistics of soil redistribution rates from 1960 to 2009 derived from LandSoil model, considering 3 infiltration rates (IR_1, IR_2 and IR_3).

Location	Soil redistribution rate (1960-2009, t ha^{-1} yr^{-1}) minimum	mean	median	maximum	Mean tillage	Mean water	Relative contribution of water erosion processes to total soil redistribution (%)
$H1_{up}$	1.42	1.54	1.46	1.73	1.78	-0.24	10
$H1_{down}$	-6.53	-5.61	-5.89	-4.41	-5.28	-0.33	5
$H2_{up}$	-5.16	-4.20	-4.17	-3.26	-3.72	-0.48	7
$H2_{down}$	2.64	3.12	3.24	3.49	3.35	-0.23	7
$H3_{up}$	0.78	0.96	0.89	1.22	1.34	-0.38	18
$H3_{down}$	-1.01	-0.52	-0.39	-0.17	-0.13	-0.40	64
$H4_{up}$	2.80	3.47	3.66	3.95	3.76	-0.40	8
$H4_{down}$	-8.24	-6.75	-7.06	-4.96	-6.25	-0.51	7
$H5_{up}$	2.42	3.15	3.41	3.63	3.58	-0.42	9
$H5_{down}$	-8.26	-6.78	-7.08	-4.99	-6.26	-0.52	7
$H6_{up}$	1.62	1.77	1.76	1.94	1.86	-0.09	10
$H6_{down}$	-5.24	-4.53	-4.67	-3.67	-4.18	-0.35	7

Negative values: soil erosion; positive values: soil deposition.

Figure captions

Fig. 1. Location of the transects within the study area of Pleine-Fougères.

Fig. 2. Soil distribution patterns in the vicinity of hedges H4 and H5. a) Location of transects in the hillside, b) Vertical distribution of 137Cs activities and 137Cs inventories for reference, H4 and H5 sites. Hz A: organo-mineral horizon, Hz B: structural horizon, Hz M: aeolian loam, Hz C: eroded bedrock.

Fig. 3. Soil redistribution modelled between 1960 and 2009: cumulative soil redistribution (lower half of graphs) and contribution of water erosion processes to annual and absolute net soil redistribution (upper half of graphs). Both were calculated by averaging soil redistribution on 10*10 m windows centred on ^{137}Cs sampling locations.

Fig. 4. Maps of mean soil redistribution rate simulated by LandSoil in the vicinity of hedges H1, H4 and H5 from 1960 to 2009.

Fig. 5. Spatial distribution of the proportion of water redistribution in absolute net soil redistribution estimated by the LandSoil model in the vicinity of hedges H1, H4 and H5.

Fig. 6. Comparison of soil redistribution rates obtained from 137Cs measurements (blue) and predicted by LandSoil (red). Black cross and black points figure mean soil redistribution rates predicted by LandSoil model and ^{137}Cs inventories, respectively. Error bars stand for minimum and maximum values of soil redistribution rates.

Figure 1
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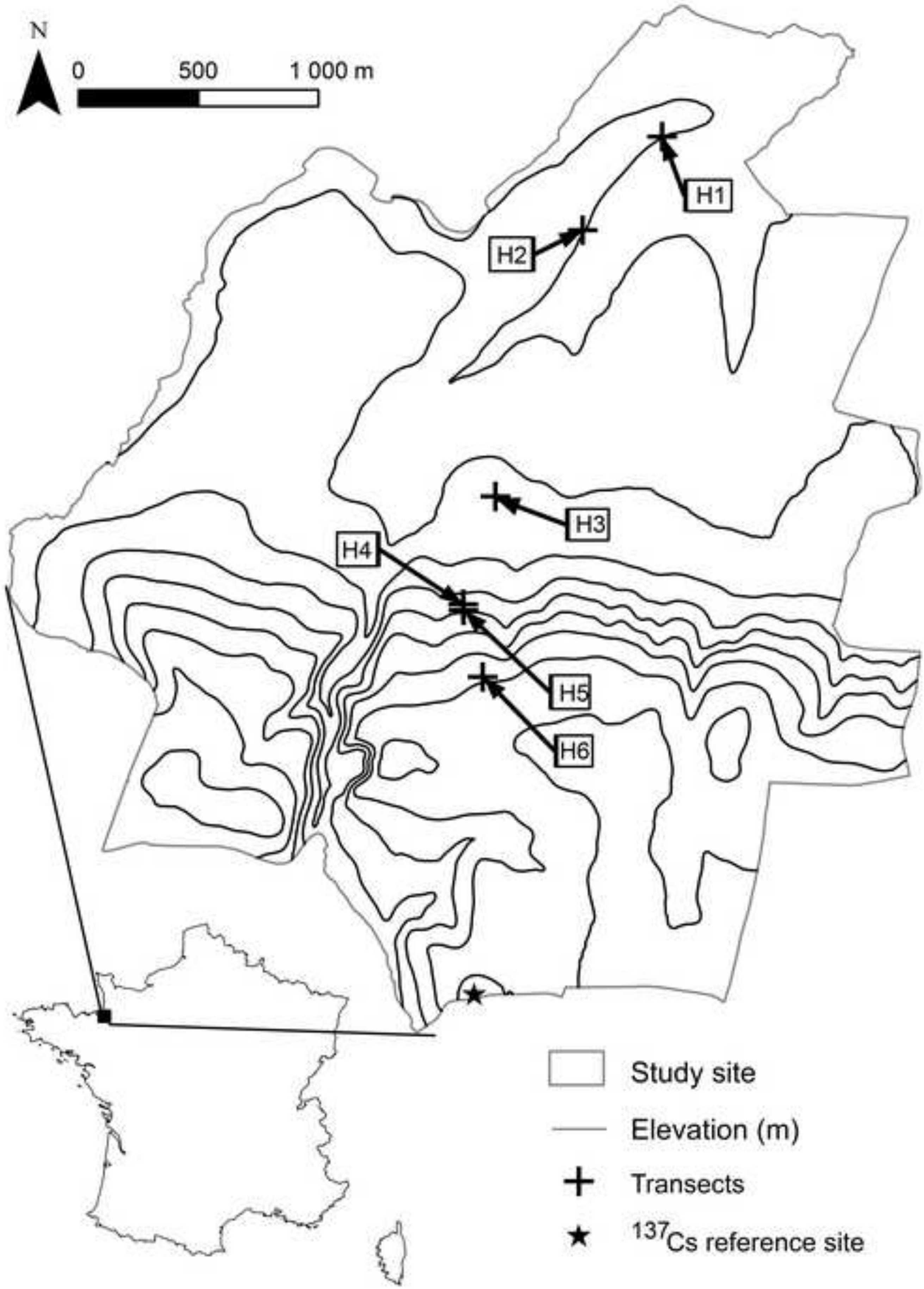

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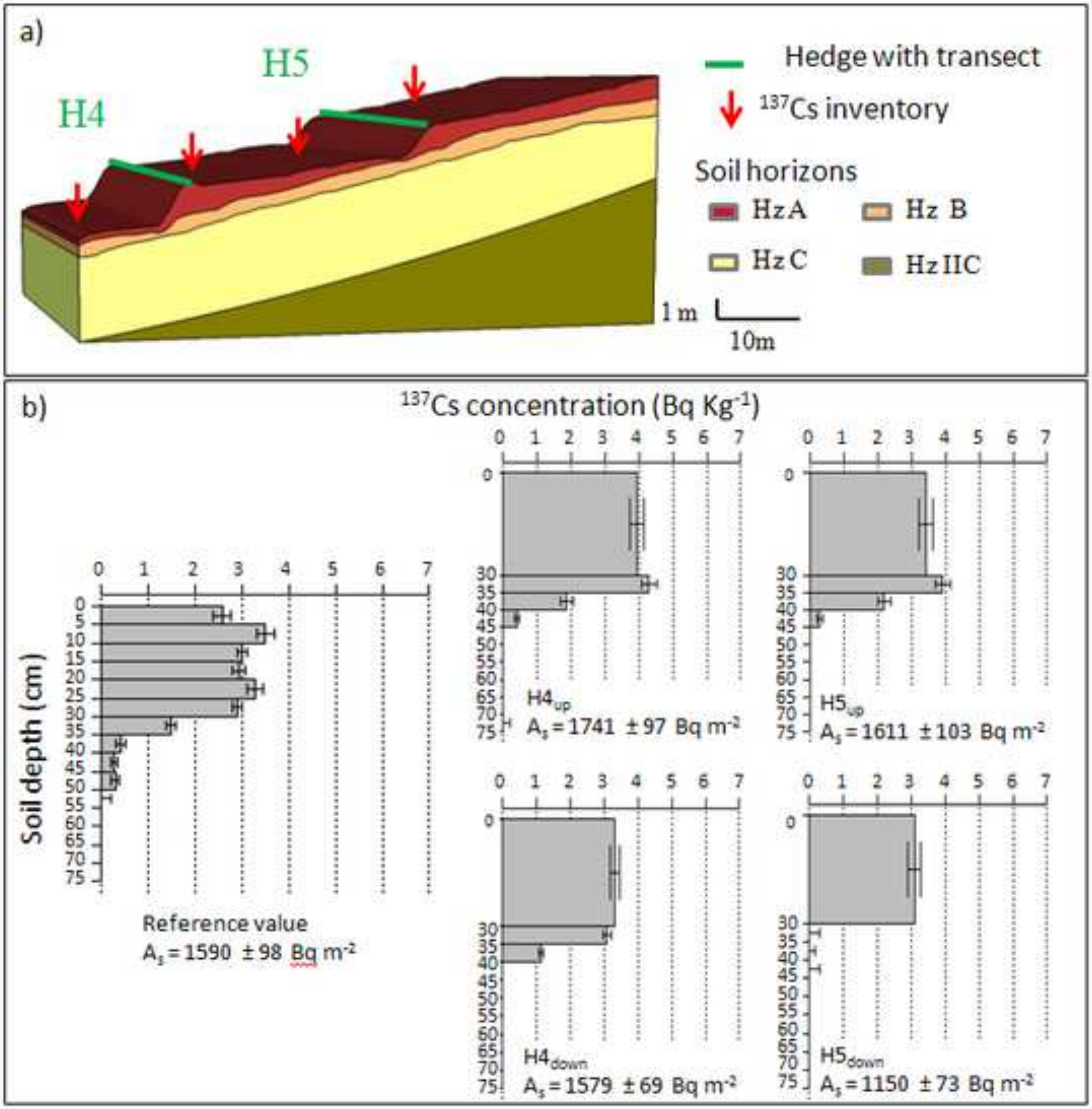

Figure 3

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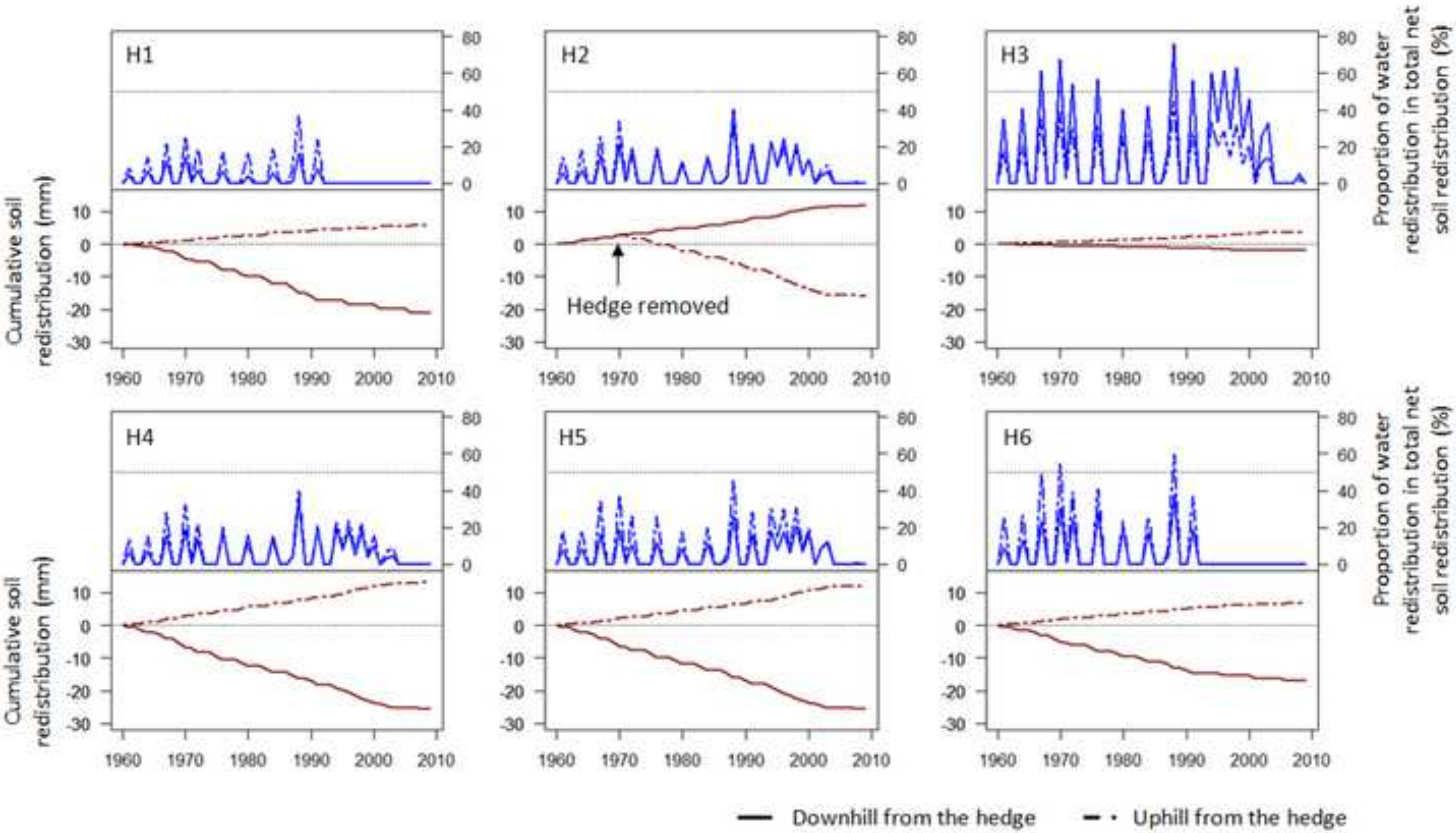

Figure 4 revised
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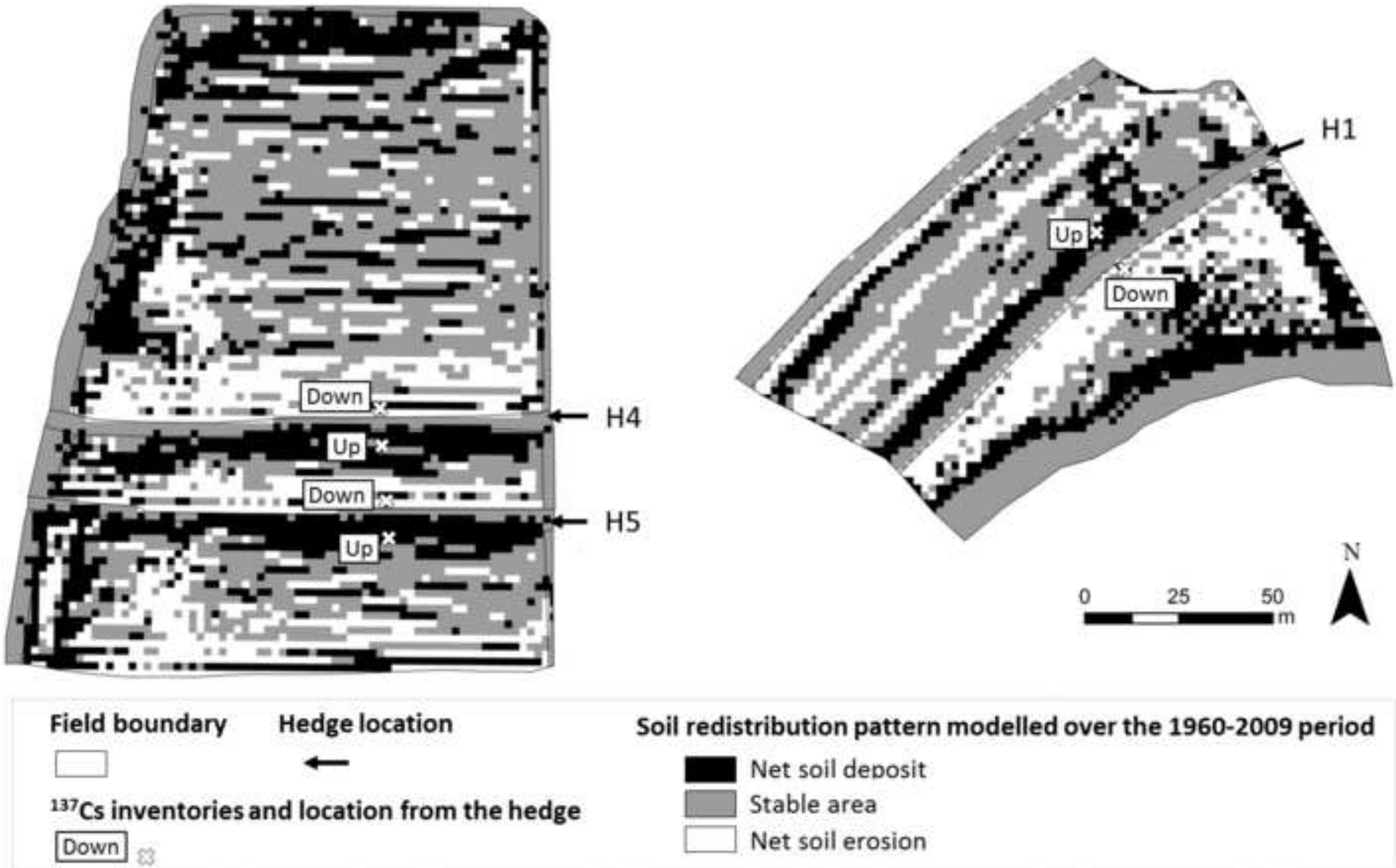

Figure 5 revised

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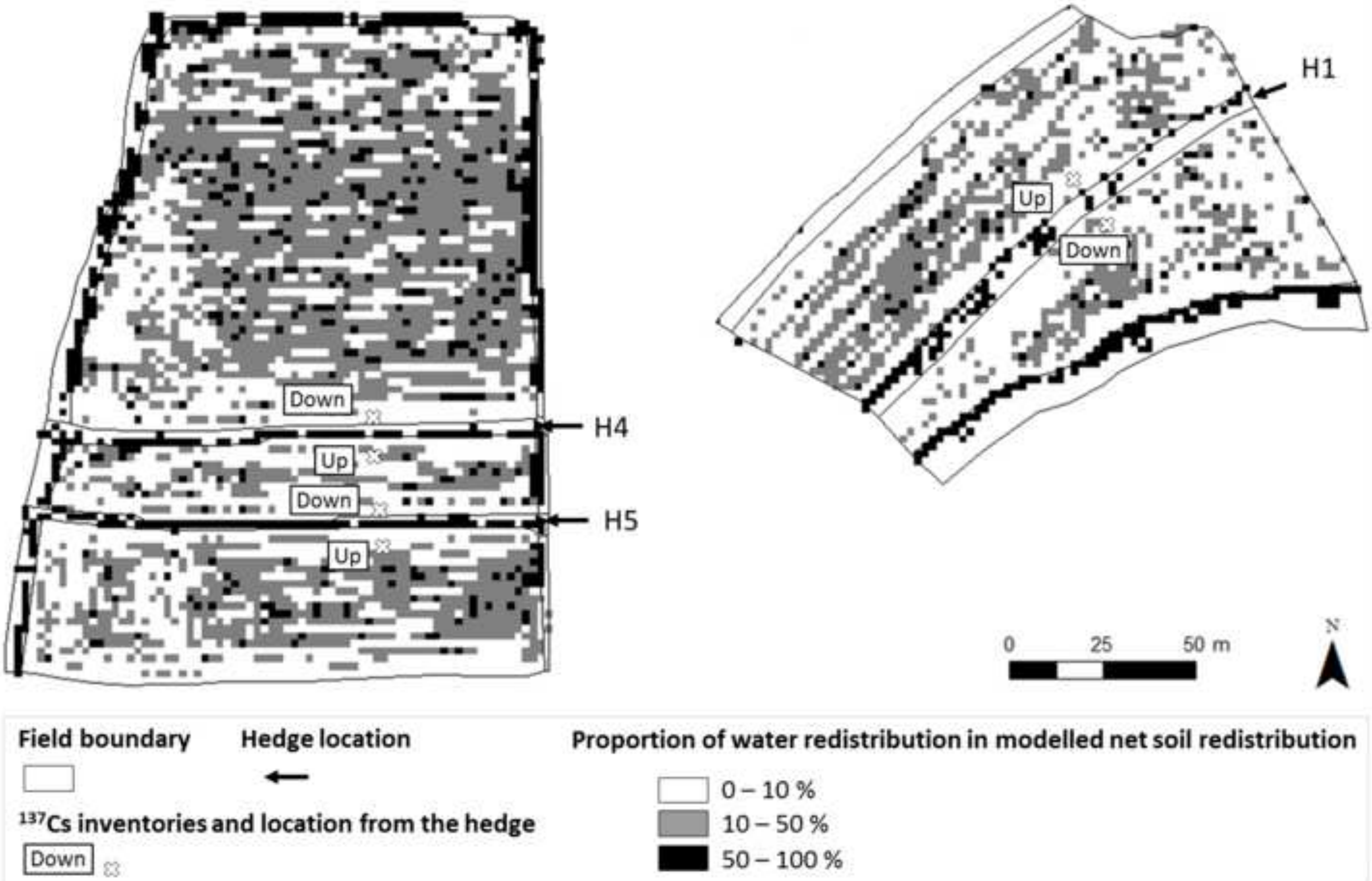

Figure 6 revised
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