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1 **Combining <sup>137</sup>Cs measurements and a spatially distributed erosion model to assess soil**  
2 **redistribution in a hedgerow landscape of northwestern France (1960 – 2010)**

3  
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17  
18 **Abstract**

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

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

37

38 **Keywords:** soil redistribution; <sup>137</sup>Cs; spatial modelling; hedge; agricultural landscape

39

## 40 **1. Introduction**

41 In 2006, the European Commission identified soil erosion as one of the major threats on soils.  
42 Soil erosion may affect all soil functions (Boardman and Poesen, 2006), also described as soil  
43 ecosystem services (Dominati et al., 2010): physical support of life and human activity, food  
44 and fibre production, water filter, carbone storage and climate regulation, etc. Soil erosion has  
45 been recognised to have direct consequences on these services both on-site (because of the  
46 soil loss from fields), and off-site: during the last decades, a significant increase in  
47 environmental issues such as eutrophication, pollution of water bodies and reservoir  
48 sedimentation has been observed in Europe, as a result of soil erosion on agricultural land  
49 (Boardman and Poesen, 2006). In numerous cases soil erosion leads to a significant reduction  
50 in soil thickness. If soil thickness decline is not compensated by soil formation, soil erosion  
51 may induce the loss of soil nutrients (Bakker et al., 2004) or soil organic carbon (Papiernik et  
52 al., 2005; Papiernik et al., 2009), and threaten the sustainability of crop production (Bakker et  
53 al., 2004). Methods and models have been developed to estimate soil redistribution by  
54 erosion and to understand the effect of several parameters on this redistribution (climate, soil  
55 properties, land use and agricultural practices, landscape structure, etc.). Before the 1990s,  
56 studies focused mostly on water erosion, because this was the most obvious process  
57 contributing to soil exportation out of cultivated fields (Govers et al., 1996). However, it is  
58 now recognized that tillage erosion is also an important process to consider, especially when  
59 studying soil loss and deposits within individual fields (Govers et al., 1994). Regarding  
60 erosion rates, tillage erosion can have an equivalent or even a higher impact than water  
61 erosion on soil redistribution (Govers et al., 1999; Lobb et al., 2007; Van Oost et al., 2005;  
62 Chartin et al., 2013). Both water and tillage erosion depend on topography, but have distinct  
63 impacts on soil redistribution regarding spatial patterns (Li et al., 2007). Water erosion is  
64 maximal on steep mid-slopes and where water concentrates, whereas tillage induces  
65 maximum erosion at convexities and deposition at concavities (Govers et al., 1996; Li et al.,  
66 2007; Thiessen et al., 2009; Van Oost et al., 2005). Moreover, there are linkages and  
67 interactions between water and tillage erosion (Li et al. 2007).

68 Runoff and soil erosion have been studied at different scales, from plots to catchments, and it  
69 appears that both landscape management and structure have an impact on soil erosion and  
70 sedimentation in agricultural land. Impact of land use on soil redistribution has been  
71 investigated in many studies, over a large range of spatial extents. Cerdan et al. (2010)  
72 considered European soil erosion studies conducted at plot scale and showed that spring crops  
73 and vineyards were the most sensitive land uses to soil erosion. From a long-term survey of  
74 soil erosion at the catchment scale, Prasuhn (2012) showed that potatoes were the crop  
75 inducing the most serious soil erosion. Consequently, land use change has an impact on soil  
76 redistribution dynamics. Vanniere et al. (2003) examined the impact of historical human  
77 occupation on soil redistribution at the hillslope scale. They explained some recorded  
78 variations in erosion by changes in agricultural activities. Bakker et al. (2008) estimated that  
79 past land-use change (de-intensification or intensification) in four European landscapes  
80 directly impacted soil erosion and sediment export to rivers. Besides land use, the farming  
81 practices, and particularly tillage practices, impact the soil redistribution. Van Muysen et al.  
82 (2000) showed that soil distribution depends on tillage speed and depth. Prasuhn (2012)  
83 observed that conventional plough tillage induced higher soil erosion rates than reduced  
84 tillage practices. However, it has been shown that these factors (land use and farming  
85 practices) were not sufficient to understand soil redistribution at landscape and catchment  
86 scales. Bakker et al. (2008) underlined that the spatial pattern of land use change strongly  
87 impacted soil redistribution and export out of the studied catchments. In this context, the  
88 spatial distribution and the connectivity between areas producing soil erosion and the zones  
89 where deposition takes place should be taken into account in the framework of studies  
90 conducted at the landscape or catchment scale (Cerdan et al., 2012; Delmas et al. 2012).  
91 Vegetated filter strips are part of the anthropogenic structures that impact connectivity inside  
92 a landscape, with an effect on water and sediments transfer (Bracken and Croke, 2005; Evrard  
93 et al., 2008; Gumiere et al., 2011). More particularly, linear structures such as hedges have  
94 been recognised as key elements of the landscape to prevent or limit erosion (Baudry et al.,  
95 2000; Boardman and Poesen, 2006; Kiepe, 1995b; Skinner and Chambers, 1996). During the  
96 last decades, important changes in landscape structure and soil use have been observed in  
97 Western Europe: land use homogenisation, removal of linear structures such as hedges and  
98 loss of connectivity between landscape elements were outlined to be the main observed  
99 changes (Burel and Baudry, 1990; Deckers et al., 2005; Petit et al., 2003). Such changes in  
100 landscape modify soil redistribution dynamics (Evrard et al., 2010) and should be taken into  
101 account in soil redistribution modelling.

102 Ability of empirical models (e.g. USLE) to integrate the dominant processes of soil  
103 redistribution at the catchment scale is uncertain (Kirkby et al., 1996), whereas process-based  
104 models require numerous input data, which are generally not available and difficult to  
105 measure (Takken et al., 1999). In such a context, spatially-distributed and expert-based  
106 models (e.g. STREAM; Cerdan et al., 2002a) can offer an alternative solution, especially  
107 when dealing with connectivity issues in landscapes (Gumiere et al., 2010). Such models  
108 focus on the dominant processes to avoid over-parameterisation and the associated  
109 uncertainties, and model simulations rely on decision rules derived by expert judgment from  
110 databases of field measurements carried out in a specific region. However, validation of such  
111 models remains an important issue in areas where experimental data, i.e. runoff and erosion  
112 measurements, are missing. This issue can be addressed by using  $^{137}\text{Cs}$ .  $^{137}\text{Cs}$  is an artificial  
113 radionuclide (half-life of 30 years) produced by the thermonuclear bomb tests conducted  
114 during the 1960s as well as, in certain regions of the world (i.e., mainly in Europe), by the  
115 Chernobyl accident in 1986.  $^{137}\text{Cs}$  is now stored in soils, and its stock decreases by  
116 radioactive decay and by fine sediment transfer due to water and tillage erosion.  $^{137}\text{Cs}$  has  
117 been widely used as a tracer of soil redistribution and it proved to be useful in soil erosion  
118 studies conducted around the world (Ritchie and McHenry, 1990; Zapata, 2003). Several  
119 studies showed a good correlation between soil redistribution obtained from  $^{137}\text{Cs}$  inventories  
120 and field measurements (Kachanoski, 1987; Mabit et al., 2002; Porto and Walling, 2012;  
121 Porto et al., 2001; Porto et al., 2003a; Porto et al., 2003b) and  $^{137}\text{Cs}$  has been used previously  
122 to validate or calibrate erosion models (Bacchi et al., 2003; Li et al., 2000; Li et al., 2007; Li  
123 et al., 2008; Porto et al., 2003b; Quine, 1999; Tiessen et al., 2009; Walling et al., 2003). The  
124 use of  $^{137}\text{Cs}$  estimates of soil redistribution relies on several hypotheses, especially that the  
125 distribution of local fallout was uniform (Walling and Quine, 1992). Such a statement could  
126 be uncertain in complex hedgerow landscapes, especially close to hedges (Follain et al.,  
127 2009). Moreover, Parsons and Foster (2011) underlined that the conditions necessary to the  
128 use of  $^{137}\text{Cs}$  as a soil redistribution indicator are most often not verified. Another limitation is  
129 that  $^{137}\text{Cs}$  is not a spatially integrative measurement and its high cost may limit sampling at  
130 landscape scale.

131 In this study, we aim to combine two methods to estimate the spatial and temporal soil  
132 redistribution dynamics near hedges from 1960 to 2009, in a rural hedgerow landscape. A  
133 new model simulating soil redistribution at the landscape scale (LandSoil, Ciampalini et al.,  
134 2012) and  $^{137}\text{Cs}$  measurements have been used to this end. The results of both methods will be  
135 compared and discussed.

136

## 137 **2. Materials and methods**

138

### 139 *2.2. Study sites*

140 The study sites were selected within the study area of Pleine-Fougères (NW France, 48° 505'  
141 N, 1° 565' W), which belongs to the European Long-Term Ecosystem Research Network, and  
142 covers an area of 10 km<sup>2</sup> (Fig. 1). This area is characterised by a high soil spatial  
143 heterogeneity. Soil types are mainly Cambisols and Luvisols, but Leptosols and Fluvisols  
144 from alluvial and colluvial deposits are also found (IUSS Working Group WRB, 2007). These  
145 soils exhibit features reflecting variable redoximorphic conditions, soil and A-horizon  
146 thickness, and soil parent material (granite, hard and soft schist with a heterogeneous cover of  
147 superficial deposits such as aeolian loam, alluvium and colluviums). Main topographical  
148 features may be associated with the presence of different geological substrates: granite under  
149 the plateau (south of the study area), Brioverian schists under plains (north of the area),  
150 metamorphic schists at the transition between granite and soft schist under hillslides. In  
151 addition, the presence of linear anthropogenic structures at field boundaries (hedges, banks,  
152 ditches, lanes and roads) delineates microtopographical changes. Landscape evolution has  
153 been driven by former agricultural policies and local farming practices, consisting in the  
154 enlargement of fields in order to facilitate the use of large machinery. Numerous hedges have  
155 been removed during a land consolidation programme conducted in the early 1990s, and some  
156 are still selectively removed by farmers. The result is that hedge density decreased from 250  
157 m ha<sup>-1</sup> in 1952 to 90 m ha<sup>-1</sup> in 2000 (unpublished data, derived from analysis of aerial  
158 photographs using Geographical Information Systems). The main land uses are annual crops  
159 (maize (*Zea mays*), winter wheat (*Triticum aestivum*)) and temporary or permanent grasslands  
160 (mostly Italian ryegrass, *Lolium multiflorum*). Hedges mainly consist of grass strips planted  
161 with trees such as oaks (*Quercus robur*), chesnuts (*Castanea sativa*) and blackthorns (*Prunus*  
162 *spinosa*).

163 Six 30-m long transects, perpendicular to six hedges (one transect per hedge) and parallel to  
164 the slope have been determined (Table 1). They have been chosen to document the variability  
165 observed across the study area in terms of soil depth, local slope and drainage area. Five  
166 transects were intersecting currently existing hedges, whereas one transect was selected along  
167 a hillslope where the hedge was removed in 1970. We ensured that each transect was  
168 representative of soil redistribution along the hedge by conducting a slingram electromagnetic  
169 survey (EM31). Along each transect, eight locations were sampled at increasing distances

170 from the hedge: one point in the hedge bank, four points uphill from the hedge at 20, 10, 5  
171 and 2 m and three points downhill of the hedge (at 2, 5 and 10 m). A field survey was carried  
172 out in 2009. Soils were described at every sampling point using auger (soil type, soil  
173 thickness, A-horizon thickness, colour, texture, gravel content, soil organic carbon content)  
174 and a precise toposequence of soil horizons was drawn. Soil redistribution along transects was  
175 inferred from comparison between the observed A-horizon thickness and the tilled layer  
176 thickness. The size of the fields where the transects were located ranged from 0.3 to 3 ha  
177 (mean size = 1 ha).

178

179 < Figure 1 near here >

180 < Table 1 near here >

181

### 182 2.3. Estimating soil redistribution rates based on $^{137}\text{Cs}$ measurements

183 For each hedge, two soil profiles were sampled for  $^{137}\text{Cs}$  measurements in March 2010:  
184 samples were systematically collected at 5 m uphill and downhill from the hedge,  
185 respectively. Undisturbed soil cores were sampled with a 7.5 cm diameter hydraulic core  
186 sampler (SEDIDRILL 80) to document the total A-horizon thickness (up to 90 cm). All cores  
187 were cut into sections of variable length: the first section corresponded to the uppermost 30  
188 cm (i.e. the ploughed mixed layer), whereas the deeper sections were cut into 5 cm sections.  
189 All samples were dried at 105°C, weighed and sieved to 2 mm.  $^{137}\text{Cs}$  activity was then  
190 measured for 102 samples, at 661 keV using Germanium gamma-ray detectors (Germanium  
191 hyperpure – GeHP, N-type, coaxial model) for 80,000 to 300,000 s.  
192 Total  $^{137}\text{Cs}$  inventory ( $^{137}\text{Cs}$  surface activity;  $A_S$  in  $\text{Bq m}^{-2}$ ) of each core was then calculated  
193 according to Eq. (1):

194

$$195 A_S = \sum_{i=1}^n A_i \times \frac{M_i}{S} \quad (1)$$

196

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

200

201 In order to estimate whether soil deposition or erosion occurred in the investigated area,  $^{137}\text{Cs}$   
202 inventories were compared to an additional soil core, used as reference and sampled at an

203 undisturbed site, i.e. in a pasture located to the south of the study area (a flat part of the  
204 plateau), where we considered that soil redistribution was very unlikely to occur during the  
205 past 60 years. Soil redistribution rates ( $\text{t ha}^{-1} \text{ yr}^{-1}$ ) were calculated using the  $^{137}\text{Cs}$  conversion  
206 models developed by Walling and He (2001). The use of these models relies on four  
207 hypotheses (Walling and Quine, 1992): (i) the  $^{137}\text{Cs}$  fallout is locally and spatially uniform,  
208 (ii) the fallout is rapidly and irreversibly fixed onto soil particles; (iii) the subsequent  
209 redistribution of fallout is due to the movement of soil particles; and (iv) estimates of soil  
210 erosion can be derived from measurements of  $^{137}\text{Cs}$  inventories. Regarding the irreversible  
211 binding issue, Kato et al. (2012a) observed after the Fukushima Dai-ichi nuclear accident that  
212 the bulk of radiocaesium was absorbed in the upper 2.0 cm in the soil profile.  
213 In this study we applied four  $^{137}\text{Cs}$  conversion models: two proportional models (PM) and two  
214 mass-balance models (MBM). PM are based on the premise that  $^{137}\text{Cs}$  fallout inputs are  
215 completely mixed within the plough layer, and that the soil loss is directly proportional to the  
216 amount of  $^{137}\text{Cs}$  removed from the soil profile since the beginning of the  $^{137}\text{Cs}$  accumulation.  
217 PM-1954 considers  $^{137}\text{Cs}$  fallout since 1954, whereas PM-1963 takes  $^{137}\text{Cs}$  into account since  
218 1963 (when the major fallout occurred). MBM are more complex. The simplified mass  
219 balance model (MBM1) takes into account the progressive decrease of  $^{137}\text{Cs}$  content in the  
220 plough layer because of the mixing with soil which lowers concentration of  $^{137}\text{Cs}$ .  
221 Additionally, MBM2 takes into account the variation in  $^{137}\text{Cs}$  fallout over time (based on the  
222 annual  $^{137}\text{Cs}$  fallout measured in the northern hemisphere) and the  $^{137}\text{Cs}$  redistribution before  
223 its incorporation in soil by ploughing. Both MBM1 and MBM2 consider the major  $^{137}\text{Cs}$   
224 fallout since 1963. These  $^{137}\text{Cs}$  conversion models have been implemented in software  
225 developed by Walling and He (2001).

226

#### 227 *2.4. Modelling soil redistribution over a 50-year period (1960-2010)*

228 The LandSoil model (Ciampalini et al, 2012) was used to simulate soil redistribution between  
229 1960 and 2009. This model is based on a combination of the STREAM runoff and soil  
230 erosion model (Cerdan et al., 2002a; Cerdan et al., 2002b; Souchere et al., 2003; Souchere et  
231 al., 1998) and the WaTEM/SEDEM tillage erosion model (Van Oost et al., 2000, Van  
232 Rompaey et al., 2001, Verstraeten et al., 2002). LandSoil is an expert-based model designed  
233 to simulate soil redistribution at a fine spatial resolution scale (1–10 m), a medium-term  
234 temporal scale (10–100 yrs), and for study areas ranging from the field to the catchment.  
235 LandSoil is spatially-distributed, event-based, and aims to simulate soil erosion (interrill,  
236 concentrated and tillage erosion) and deposition. The model assumes that surface

237 characteristics (soil surface crusting, surface roughness, vegetation cover (crops and residues)  
238 and residual water storage after the previous event) are the main drivers of runoff and  
239 infiltration at the field scale. A soil infiltration rate value (IR), i.e. a constant soil infiltration  
240 rate reached during prolonged rainfall which determines the production of runoff during a  
241 rainfall event, is assigned to each combination of these soil surface characteristics (Le  
242 Bissonnais et al., 1998;2005). For instance, soils with thick depositional crusts and low  
243 vegetation cover were assigned with low values of IR, whereas soils with high roughness and  
244 fragmentary structure have higher IR values. In the same way, the model assumes that surface  
245 characteristics and the maximum intensity of rainfall are the main drivers of sediment  
246 concentration in runoff. This concentration is also determined for each combination of the soil  
247 surface characteristics and rainfall maximum intensity (Cerdan et al. 2002c). To estimate the  
248 soil water erosion, the IR parameter is combined with the potential sediment concentration in  
249 runoff and with landscape properties (mainly the slope intensity). An adaptation of the rules to  
250 the local context (types of crops planted and climate) is required before running the model  
251 (Evrard et al., 2009; Evrard et al., 2010).

252 Several studies have estimated IR on loamy soils from Western France using plot  
253 measurements under rainfall simulation or field scale monitoring networks under natural  
254 rainfalls. Soils considered in these studies differed slightly by texture and carbon content, but  
255 showed large variations of IR associated with the variations of soil surface characteristics  
256 throughout time: Cros-Cayot (1996) and Lacoste (2012) found IR ranging from 1 to 25 mm h<sup>-1</sup>  
257 <sup>1</sup> depending on surface characteristics. In different sites characterised by similar soil types, Le  
258 Bissonnais et al. (1992) and Evrard et al. (2008) estimated variations between 2 to 50 mm h<sup>-1</sup>.  
259 To take into account this variability, three ranges of IR variation were considered (Table 2): 1  
260 to 25 mm h<sup>-1</sup> (IR<sub>1</sub>), 1.5 to 37.5 mm h<sup>-1</sup> (IR<sub>2</sub>) and 2 to 50 mm h<sup>-1</sup> (IR<sub>3</sub>). Concerning hedges,  
261 previous studies found a wide range of IR under hedges, according to hedge composition,  
262 thickness and climate of the study area (Table 3). In this study, IR under hedges was fixed at  
263 150 mm h<sup>-1</sup>. Regarding the soil concentration in runoff, the default values were determined  
264 after Cerdan et al. (2002c) and ranged from 0 to 25 g L<sup>-1</sup>. These default values have been  
265 estimated for loamy soils with similar characteristics as those of the soils in our study area,  
266 and local studies confirmed the relevance of their use in our study (Gascuel-Oudoux et al,  
267 1996; Lacoste, 2012).

268 LandSoil simulates soil deposition in two cases: for a given pixel, (i) soil deposit is modelled  
269 when water infiltration is higher than runoff, and (ii) when the sediment transport by water  
270 erosion is limited. For this second case, the maximum sediment concentration is controlled by

271 several threshold functions based on the local topography and soil cover (Cerdan et al,  
272 2002c). These functions include profile curvature (concavity  $> 0.055 \text{ m}^{-1}$ ), slope gradient ( $< 5$   
273 %), land use (permanent grassland and wood) and vegetation cover ( $> 60 \%$ ).

274 < Table 2 near here >

275 < Table 3 near here >

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

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

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

284 (ii) Crop rotations and associated soil surface characteristics (soil surface crusting, surface  
285 roughness, and vegetation cover). We used aerial photographs, taken during summer in 1966,  
286 1968 and from 1993 to 2009, to create crop transition matrices (based on Markov chain) and  
287 allocated a main crop per field for each year from 1960 to 2009. Crop rotations consisted of a  
288 succession of maize, winter cereals and temporary grassland. Monthly soil surface  
289 characteristics were attributed to each crop from expert knowledge and field survey data.

290 (iii) Soil tillage operation data (direction of tillage and coefficient of tillage erosion). Two  
291 tillage transport coefficients (KTILL) are used by LandSoil to model soil redistribution by  
292 tillage: KTILL<sub>max</sub> for describing soil redistribution parallel to the tillage direction and  
293 KTILL<sub>min</sub> for characterising soil redistribution perpendicular to the tillage direction. For  
294 years with maize or winter cereals sowing, the sequence of tillage operations consisted in the  
295 use of mouldboard plough (25 to 30 cm depth), chisel, rotary harrow and air seeder. For years  
296 where grassland was established, the sequence of tillage operations consisted in the use of  
297 chisel and air seeder. Van Muysen et al. (2006) showed that the KTILL of a sequence of  
298 tillage operations can be predicted by summing the KTILL of all individual tillage operations.  
299 They also calculated that the mean annual KTILL<sub>max</sub>, associated with mechanized agriculture,  
300 is in the order of  $781 \text{ kg m}^{-1} \text{ yr}^{-1}$ . Mean annual KTILL coefficients were defined based on the  
301 results of previous studies (Table 4). Values for KTILL<sub>max</sub> and KTILL<sub>min</sub> reached respectively  
302 631 and  $376 \text{ g m}^{-1}$  for the years with maize or winter wheat sowing, and 291 and  $139 \text{ g m}^{-1}$  for  
303 the years with grassland sowing.

304 (iv) Rainfall event characteristics: rainfall amount (mm), rainfall maximum intensity (mm h<sup>-1</sup>)  
 305 <sup>1</sup>), effective rainfall duration (h) and rainfall amount during the 48-h before the event (mm).  
 306 These parameters were defined from predictive 6-minute meteorological data from 1960 to  
 307 2009, estimated from hourly meteorological data from Lacoste (2012). Hourly data were  
 308 provided by the INRA unity Agroclim for the Rennes Station.

309

310 < Table 4 near here >

311

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

315

### 316 2.5. Comparing soil redistribution rates derived from <sup>137</sup>Cs and from modelling

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

328

$$329 \quad R^2 = \frac{\sum_{i=1}^n (x_i - \hat{x}_i)^2}{\sum_{i=1}^n (x_i - \bar{x})^2} \quad (2)$$

$$330 \quad ccc = \frac{2S_{xy}}{S_x^2 + S_y^2 + (\bar{x} - \bar{y})^2} \quad (3)$$

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

332

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

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

340

### 341 **3. Results**

342

#### 343 *3.1. Soil redistribution rates based on $^{137}\text{Cs}$ measurements*

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

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

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

370 positions uphill from the hedges, and at  $-11.15 \text{ t ha}^{-1} \text{ yr}^{-1}$  for the positions downhill from the  
371 hedges.

372

373 < Figure 2 near here >

374 < Table 5 near here >

375

### 376 *3.2. Soil redistribution rates over 50 years based on soil redistribution modelling with the* 377 *LandSoil model*

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

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

399

400 < Table 6 near here >

401

402 Fig. 3 shows the annual variations of the contribution of water erosion processes to absolute  
403 soil redistribution between 1960–2009, and the cumulative soil redistribution. Depending on

404 the year, the transect and the position from the hedge, the contribution of water erosion  
405 processes to absolute soil redistribution ranged from 0 to 76%. The contribution of water  
406 redistribution to absolute soil redistribution remained lower than 50% for five hedges (H1,  
407 H2, H4 and H5), and reached 50% and more for two hedges (H3 and H6). For all hedges  
408 except H3, the contribution of water erosion processes to absolute soil redistribution was  
409 similar or higher for positions located uphill of hedges than for positions situated downhill  
410 from hedges. For H3 the opposite pattern was obtained from the model. For all hedges except  
411 H2, results showed no change in soil redistribution dynamics: positions uphill from hedges  
412 always proved to be deposition sites, whereas positions downhill from hedges were erosion  
413 sites. For hedge H2, both positions uphill and downhill from the hedge were deposition sites  
414 before the hedge removal in 1970, and then the position uphill from hedges became an  
415 erosion site, whereas the position downhill from hedges was a deposition site. The intra and  
416 inter-annual variations of the modelled soil redistribution were due to the combination of  
417 meteorological conditions (extreme rainfall events) and soil cover linked to land use.  
418 Fig. 4 and Fig. 5 show the spatial distribution of the soil redistribution simulated by LandSoil  
419 for the hedges H1, H3 and H4. For the three hedges shown in Fig. 5, LandSoil modelling  
420 results showed (i) soil deposition uphill from the hedge; (ii) soil erosion downhill from the  
421 hedge, and (iii) higher soil redistribution at the field boundaries rather than within the fields.  
422 For the modelling areas shown in Fig. 4, the mean soil redistributions across the entire fields  
423 ranged from -0.07 to -0.42 t ha<sup>-1</sup> yr<sup>-1</sup>, whereas it ranged from -8.26 to 3.95 t ha<sup>-1</sup> yr<sup>-1</sup> in the  
424 vicinity of the hedges. Fig. 6 shows the relative contribution of water erosion processes to the  
425 net soil redistribution. For the three hedges H1, H4 and H5, tillage redistribution tended to be  
426 dominant at the vicinity of the hedge. Water redistribution was nevertheless dominant within  
427 the fields.

428

429 < Figure 3 near here >

430 < Figure 4 near here >

431 < Figure 5 near here >

432

### 433 *3.3. Comparison of soil redistribution rates from <sup>137</sup>Cs measurements and LandSoil model*

434 The correlation coefficient R<sup>2</sup> and Lin's concordance coefficient (CCC) were calculated to  
435 compare the soil redistribution rates estimated from <sup>137</sup>Cs inventories and from LandSoil.

436 Considering all the study sites, R<sup>2</sup> and CCC reached 0.17 and 0.12, respectively. Considering  
437 only the positions uphill from the hedges, R<sup>2</sup> and CCC amounted to 0.99 and 0.15,

438 respectively. Finally, considering only the positions downhill from the hedges,  $R^2$  and CCC  
439 was equal to 0.29 and 0.17, respectively.

440 Fig. 6 compares the soil redistribution rates modelled from  $^{137}\text{Cs}$  inventories and LandSoil.  
441 The rates overlap only for two transects in uphill hedge positions (H4 and H5), and two  
442 transects in downhill hedge positions (H4 and H6). The two methods predicted a different  
443 dominant redistribution process for three hedges. For the hedges H3, H6 and H1, positions  
444 uphill from the hedges were estimated to be erosion sites by  $^{137}\text{Cs}$  inventories conversion  
445 models, whereas they were estimated to be deposition sites by LandSoil model. The same  
446 pattern was observed for the hedge H2 (for the position downhill from the hedge).  
447 Soil redistribution rates estimated from  $^{137}\text{Cs}$  inventories conversion models were higher for  
448 all study sites, except for hedge H5 (uphill from the hedge) and the hedges H6 and H4  
449 (downhill from the hedge). Hedge H2, removed in 1970, showed the largest estimate  
450 differences: for the position uphill from the hedge both methods estimated soil erosion, but  
451 the mean soil erosion was estimated at  $-4.2 \text{ t ha}^{-1} \text{ yr}^{-1}$  by LandSoil and  $-66 \text{ t ha}^{-1} \text{ yr}^{-1}$  from the  
452  $^{137}\text{Cs}$  inventories.

453 < Figure 6 near here >

454

## 455 **4. Discussion**

456

### 457 *4.1. Methods to estimate soil redistribution patterns and rates*

#### 458 *4.1.2. Soil redistribution estimated from $^{137}\text{Cs}$ inventories*

459 Among the five transects on a still existing hedge, three were estimated to be erosion sites,  
460 both for positions uphill and downhill from the hedge (H1, H3 and H6). In these cases, soil  
461 erosion rates were estimated lower on uphill positions than on downhill positions. For the two  
462 others (H4 and H5), positions uphill from the hedge were estimated to be deposition sites,  
463 whereas positions downhill from the hedge were estimated to be erosion sites. This last result  
464 was more consistent with previous studies (Follain, 2006; Follain et al., 2006; Walter et al.,  
465 2003).

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

472 The second important point is the initial deposit of  $^{137}\text{C}$ . In a hedgerow landscape, hedges  
473 obstruct winds, so the diffusion of  $^{137}\text{Cs}$  in the landscape by the wind may not be uniform.  
474 Moreover, the canopy of trees constituting the hedge is prone to intercept  $^{137}\text{Cs}$  before soil  
475 deposition (Kato et al., 2012b), which could be another explanation to the non-uniform  $^{137}\text{Cs}$   
476 fallout. The  $^{137}\text{Cs}$  reference inventory that we used was located in the south of the study area  
477 (Fig. 1), inside a flat cultivated field without hedges at its boundaries. It could mean that this  
478 reference inventory that we compared to  $^{137}\text{Cs}$  inventories on transects was not representative  
479 of the initial  $^{137}\text{Cs}$  fallout for the whole study area, and specifically for the study sites. If the  
480 reference inventory was not representative of the initial  $^{137}\text{Cs}$  fallout, soil redistribution  
481 patterns and modelled rates using  $^{137}\text{Cs}$  conversion models could be biased. This could  
482 explain why the patterns of soil redistribution modelled by  $^{137}\text{Cs}$  inventory conversion models  
483 were not always consistent with known soil redistribution at hedge proximity in such a  
484 landscape. For example, soil erosion was estimated uphill from the hedge H1, whereas the  
485 survey of A-Horizon thickness (thicker in positions uphill from the hedges than in positions  
486 downhill from the hedge) suggested the occurrence of soil deposition. Moreover, previous  
487 studies showed that positions uphill from hedges were more prone to soil deposition (Follain,  
488 2006; Follain et al., 2006; Walter et al., 2003). However, such soil deposition could have  
489 occurred prior to  $^{137}\text{Cs}$  fallout, and positions uphill from the hedges could have become,  
490 during the last decades, areas where soil erosion was prevailing over soil deposition. A better  
491 estimation of soil redistribution could have been achieved by the use of a more representative  
492 reference  $^{137}\text{Cs}$  inventory, e.g. a local reference profile for each hedge, sampled in the  
493 proximity of each of those landscape features. However, it seems difficult to find such a  
494 location, without soil redistribution, in an agricultural landscape. An estimate of the soil  
495 redistribution dynamics in the vicinity of hedges could be done without using a  $^{137}\text{Cs}$   
496 reference inventory, by comparing for each transect the uphill/downhill  $^{137}\text{Cs}$  inventory. For  
497 the hedge H1 for example, the  $^{137}\text{Cs}$  inventory uphill from the hedge was higher than the  $^{137}\text{Cs}$   
498 inventory downhill from the hedge. This is insufficient to conclude that soil deposition  
499 occurred uphill from H1, but we may assume that soil erosion was higher downhill from H1.  
500 Regarding the hedge H2, which was removed in 1970, the soil redistribution estimate using  
501 the  $^{137}\text{Cs}$  inventories can be biased because of the possible soil redistribution in the fields by  
502 farmers during the hedge removal.

503

504 *4.1.2. Soil redistribution estimated by LandSoil model*

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

508 *(i) Parameters linked to soil redistribution by water processes*

509 Three main parameters have to be defined to simulate soil redistribution by water: infiltration  
510 rate, potential sediment concentration in runoff, and factors of soil deposition (or factors  
511 controlling the maximum sediment concentration in runoff). In this study, we used three  
512 ranges of soil infiltration rates to take soil variability into account, but the limited contribution  
513 of water erosion processes to soil redistribution made the results very similar. The default  
514 values used in LandSoil for potential sediment concentration in runoff (Cerdan et al, 2002b  
515 and 2002c) were consistent with the available data in our study area (Gascuel-Oudoux et al.,  
516 1996; Lacoste, 2012). Finally, the maximum sediment concentration in runoff was controlled  
517 by threshold functions based on four factors: profile curvature, slope gradient, land use and  
518 vegetation cover. Threshold values for these factors have been determined by Cerdan et al  
519 (2002b and 2002c) using data on runoff and soil redistribution under natural rainfall on loamy  
520 soils from Northern France. No data were available in our study area to calibrate these  
521 thresholds but, given the observed similarities between the study sites, we assumed that these  
522 values were valid in our study site. The threshold values for topographic parameters  
523 (curvature and slope), estimated for a 5x5 m DEM, have only been adapted for a 2x2 m DEM.

524 *(ii) Parameters linked to soil redistribution by tillage processes*

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

529 *(ii) Other source of uncertainties*

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

535 The soil redistribution patterns modelled by LandSoil were consistent with previous  
536 knowledge of soil redistribution in such a landscape, i.e. soil deposition uphill from hedges  
537 and soil erosion downhill from hedges (Follain et al., 2006). H2 was the only hedge where an  
538 opposite pattern was modelled. H2 was also the only hedge removed during the simulation

539 process, and in such conditions this result is consistent: soil previously deposited uphill from  
540 the hedge could be eroded and redistributed downhill from the hedge. LandSoil provides  
541 quantitative estimations of the spatial variability of soil redistribution and its variability over  
542 time by dynamic modelling. LandSoil also allows distinguishing and quantifying the  
543 contribution of the different processes taking part in soil redistribution (in this study water and  
544 tillage redistribution). These types of models are particularly interesting to better understand  
545 the impacts of landscape structure on soil redistribution processes or to estimate soil variation  
546 in time according to various global change scenarios.

547

#### 548 *4.2. Soil redistribution dynamics in agricultural landscapes*

##### 549 *Scale issues*

550 Soil erosion has been studied on a range of temporal and spatial scales. Results showed that  
551 there is no simple relationship between erosion rates when up- or downscaling (Chaplot and  
552 Poesen, 2012; Delmas et al., 2012; Le Bissonnais et al., 1998). At the scale of Europe, Delmas  
553 et al (2012) showed that soil erosion rates decrease from the field to the catchment scale.  
554 Moreover, Le Bissonnais et al. (1998) showed that the size of the plot used for soil erosion  
555 measurement have an impact on the results. Therefore, comparisons of studies should be  
556 conducted carefully. In this study, soil redistribution rates are given at the plot scale (400 m<sup>2</sup>),  
557 located in the vicinity of hedges. However, both methods used to estimate these rates integrate  
558 soil redistribution on larger spatial extents: (i) <sup>137</sup>Cs inventories take into account all soil  
559 particle movements at a given point without scale restriction. In our hedgerow landscape, we  
560 assume that the spatial extent of soil redistribution was limited by the hedges located at the  
561 field boundaries (field size ranging from 0.3 to 3 ha, with a mean size of 1 ha). Therefore,  
562 <sup>137</sup>Cs inventories provided a way to estimate soil redistribution rates from point-to-field  
563 scales; (ii) the LandSoil model was run at field scale, so it also allowed assessing soil  
564 redistribution rates from point-to-field scales. According to <sup>137</sup>Cs inventories, the mean soil  
565 redistribution rate was estimated at -12.6 t ha<sup>-1</sup> yr<sup>-1</sup>, the mean soil erosion rate was estimated  
566 at -15.9 t ha<sup>-1</sup> yr<sup>-1</sup>, and the mean soil deposition rate at 3.8 t ha<sup>-1</sup> yr<sup>-1</sup>. Considering LandSoil  
567 estimates, the mean soil redistribution rate was estimated at -1.2 t ha<sup>-1</sup> yr<sup>-1</sup>, the mean soil  
568 erosion rate was estimated at -4.7 t ha<sup>-1</sup> yr<sup>-1</sup>, and the mean soil deposition rate at 2.3 t ha<sup>-1</sup> yr<sup>-1</sup>.  
569 These rates are close to those estimated in previous studies. Verheijen et al. (2009)  
570 synthesized studies at field scales and estimated that soil erosion rates in Europe ranged from  
571 -3.2 to -19.8 t ha<sup>-1</sup> yr<sup>-1</sup> (considering water, wind and tillage erosion). Both methods used in  
572 this study estimated that erosion processes were more pronounced than deposition processes.

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

575

#### 576 *Tillage vs. water erosion*

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

594

#### 595 *Impacts of linear anthropogenic structures*

596 Both methods were associated with uncertainties and it turns out to be difficult to directly  
597 compare their respective results. Soil redistribution rates were not well correlated (overall  $R^2$   
598 =  $0.17$ ), estimates of soil redistribution patterns were not always consistent, and for most of  
599 the sampling sites soil redistribution rates estimated from  $^{137}\text{Cs}$  inventories were higher than  
600 those estimated from LandSoil.

601 However, both methods showed that hedges had an impact on soil redistribution. For  
602 positions uphill from the hedges, we modelled either soil accumulation or soil erosion with a  
603 lower rate than the one modelled for positions downhill from hedges. Focusing of the 5 still  
604 existing hedges, the mean soil redistribution rates estimated by LandSoil and  $^{137}\text{Cs}$  inventories  
605 were  $2.2$  and  $-4.5 \text{ t ha}^{-1} \text{ yr}^{-1}$  respectively for the positions uphill from the hedges, and  $-4.8$  and  
606  $-11.2 \text{ t ha}^{-1} \text{ yr}^{-1}$  respectively for the positions downhill from the hedges. This result was

607 consistent with those of previous studies: hedges act as a trap for soil particles in runoff and  
608 can enhance runoff infiltration (Baudry et al., 2000); moreover, hedges act as a zero flux line  
609 regarding tillage erosion processes (Govers et al., 1996). Both processes result in the  
610 differentiation of soil redistribution on both sides of hedges. Soil redistribution at the  
611 landscape scale considering hedge networks has been modeled by Follain et al. (2006). They  
612 concluded that hedges have an impact on soil redistribution and landscape topography  
613 evolution. In hedge-less landscape, soil can be redistributed over longer distances, which  
614 induces landscape leveling. By contrast, in a hedged landscape, hedges contribute to bank  
615 creation in soil deposit areas (uphill from hedges). It has also been shown that hedges have an  
616 impact on soil horizon succession, particularly regarding the A-horizon thickness (Follain et  
617 al., 2009). It comes out that, at the field-to-landscape scale, hedges are a factor of  
618 heterogeneity when dealing with soil redistribution (Fig. 4 and 5; Follain et al., 2006; Lacoste,  
619 2012): they impact soil redistribution within fields but they also impact soil export from the  
620 fields and from landscapes or catchments. Hedges should therefore be taken into account in  
621 studies dealing with soil redistribution. Nevertheless, it could be difficult to take them  
622 explicitly into account in studies conducted over large areas (landscape-to-region or country).  
623 One solution could be to use a connectivity index as described by Cerdan et al. (2012). This  
624 index combines information on slope, lithology and rainfall and has been used to estimate the  
625 connectivity and the soil deposit processes in the major French river basins.

626

627

## 628 **5. Conclusions**

629 This study aimed to estimate soil redistribution patterns and rates in a hedgerow landscape for  
630 areas close to hedges. We compared two methods, one derived from  $^{137}\text{Cs}$  survey and the  
631 other based on a spatially distributed soil redistribution model (LandSoil). Soil redistribution  
632 rate estimates obtained with  $^{137}\text{Cs}$  survey were higher than those obtained with LandSoil, but  
633 both were consistent with other values found in previous studies. Estimates from both  
634 methods showed that soil erosion processes were dominant in the vicinity of the hedge.  
635 Depending on the method, mean soil redistribution rate varied between  $-4.8 \text{ t ha}^{-1} \text{ yr}^{-1}$  and  $2.2$   
636  $\text{t ha}^{-1} \text{ yr}^{-1}$  in positions uphill from hedges, whereas they reached  $-4.8$  to  $-11.2 \text{ t ha}^{-1} \text{ yr}^{-1}$  in  
637 positions downhill from hedges. Both methods modelled hedges as anti-erosive landscape  
638 elements: estimates of soil redistribution uphill from the hedges showed either soil  
639 redistribution or lower soil erosion than positions downhill from hedges. Estimates from  
640 LandSoil were consistent with previous studies on soil redistribution in relation to landscape

641 structure, and showed its ability to model soil redistribution in complex hedgerow landscapes.  
642 At the field scale, the estimates of soil redistribution by LandSoil showed that soil  
643 redistribution was more important at the vicinity of field boundaries. Moreover, the impact of  
644 tillage on the soil redistribution in the vicinity of the hedges was found more important than  
645 water processes (an average of 87% of the soil net redistribution was due to tillage). However,  
646 soil redistribution processes varied in space and in time, and water erosion processes were  
647 dominant within the fields. Further work will focus on the impacts of landscape structure on  
648 soil redistribution in the context of global change.

649

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656

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**Table 1.**  
**Characteristics of sampling locations selected for calculating <sup>137</sup>Cs inventories and modelling.**

Study site	Date of hedge removal (where applicable)	Orientation	Hedges				<sup>137</sup> Cs sampling sites			
			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 <sub>up</sub>	40	40
								H1 <sub>down</sub>	21	21
H2	1970	NE-SW	Brioverian schist	plain	3.0	10.3	1.68	H2 <sub>up</sub>	55	40
								H2 <sub>down</sub>	70	36
H3	-	E-W	Brioverian schist	plain	1.3	5.6	2.47	H3 <sub>up</sub>	70	64
								H3 <sub>down</sub>	60	53
H4	-	E-W	Aeolian loam	hillside	6.9	7.0	1.64	H4 <sub>up</sub>	500	90
								H4 <sub>down</sub>	490	30
H5	-	E-W	Aeolian loam	hillside	6.5	24.0	0.91	H5 <sub>up</sub>	550	70
								H5 <sub>down</sub>	500	30
H6	-	E-W	Hornfels	upper hillside	4.0	30.1	1.21	H6 <sub>up</sub>	50	38
								H6 <sub>down</sub>	30	30

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

**Table 2**[Click here to download Table: Table2.docx](#)**Table 2.**

Sets of soil infiltration rates ( $\text{mm h}^{-1}$ ) used to model soil redistribution. IR depend on combinations of soil surface characteristics driving runoff (values for  $\text{IR}_1 - \text{IR}_2 - \text{IR}_3$ ).

Soil roughness <sup>a</sup>	Vegetation cover <sup>b</sup>	Crusting stage <sup>c</sup>			
		F0	F11	F12	F2
0 - 1 cm	< 20 %	5 - 7.5 - 10	5 - 7.5 - 10	2.5 - 3.75 - 5	1 - 1.5 - 2
	21-60 %	10 - 15 - 20			
	> 61 %	25 - 37.5 - 50	10 - 15 - 20	5 - 7.5 - 10	2.5 - 3.75 - 5
1 - 2 cm	< 20 %	10 - 15 - 20	5 - 7.5 - 10	2.5 - 3.75 - 5	1 - 1.5 - 2
	21-60 %	25 - 37.5 - 50	10 - 15 - 20	5 - 7.5 - 10	
	> 61 %				2.5 - 3.75 - 5
2 - 5 cm	< 20 %	25 - 37.5 - 50	10 - 15 - 20	5 - 7.5 - 10	2.5 - 3.75 - 5
	21-60 %			5 - 7.5 - 10	2.5 - 3.75 - 5
	> 61 %		25 - 37.5 - 50	10 - 15 - 20	5 - 7.5 - 10
5 - 10 cm	< 20 %	25 - 37.5 - 50	10 - 15 - 20	5 - 7.5 - 10	2.5 - 3.75 - 5
	21-60 %		25 - 37.5 - 50	10 - 15 - 20	5 - 7.5 - 10
	> 61 %			25 - 37.5 - 50	
> 10 cm	< 20 %	25 - 37.5 - 50	10 - 15 - 20	10 - 15 - 20	5 - 7.5 - 10
	21-60 %		25 - 37.5 - 50	25 - 37.5 - 50	
	> 61 %				

<sup>a</sup> 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).

<sup>b</sup> Vegetation cover classes are defined as the percentage of soil surface covered by canopy or litter

<sup>c</sup> 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.

## Literature review of infiltration rates under hedges.

Reference	Study site	Hedge type, wide	Infiltration rate (mm h <sup>-1</sup> )
Alegre and Rao, 1996	Loreto, Peru	Contour hedgerow ( <i>Inga edulis</i> ), 0.5 m	500
Anderson et al, 2009	Missouri, USA	Contour strip ( <i>Agrostis gigantea</i> , <i>Bromus</i> spp., <i>Lotus Corniculatus</i> L., <i>Quercus palustris</i> Muench, <i>Quercus bicolor</i> Willd., <i>Quercus macrocarp</i> Michx), 4.5 m	17
Bharati et al., 2002	Iowa, USA	Grass strip ( <i>Panicum virgatum</i> L.), 7.1 m ; Shrub strip ( <i>Cornus stolonifera</i> Michx., <i>Physocarpus opulifolius</i> L.), 2 rows ; Tree strip ( <i>Populus X euramericana</i> 'Eugenei', <i>Fraxinus pennsylvanica</i> Marsh., <i>Acer saccharinum</i> L., <i>Juglans nigra</i> L.), 5 rows	90-450
Kiepe et al, 1995a	Kenya	Contour hedgerows of <i>Cassia siamea</i> Lam.	69-135
Perret et al., 1996	Reunion Island	<i>Caliendra</i> 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 <sup>2</sup>	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 <sup>-3</sup> )	Tillage depth (cm)	Tillage speed (m s <sup>-1</sup> )	KTILL <sub>max</sub> (kg m <sup>-1</sup> )	KTILL <sub>min</sub> (kg m <sup>-1</sup> )
<i>Values for light cultivators plus seeders</i>					60	<i>n.a.</i>
Li et al., 2007	Air seeder	1.27	3	2.23	18	<i>n.a.</i>
Li et al., 2007	Light cultivator	1.25	8	2.23	42	<i>n.a.</i>
<i>Mean values for chisel ploughs</i>					274	139
Govers et al., 1994	Chisel plough	1.35	12	1.25	111	<i>n.a.</i>
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	<i>n.a.</i>
Tiessen et al., 2007	Chisel plough	1.37	16.2	1.92	64.4	<i>n.a.</i>
Van Muysen et al., 2001	Chisel plough	1.25	20	2.02	258	<i>n.a.</i>
<i>Mean values for mouldboards</i>					297	237
Lindstrom et al., 1992	Mouldboard	1.35	24	2.1	330	363
Montgomery et al., 1999	Mouldboard	1.31	23	1	<i>n.a.</i>	110
Revel et al., 1993	Mouldboard	1.35	27	1.8	263	<i>n.a.</i>
<i>Values used for a full tillage sequence for maize or wheat sowing (mouldboard, chisel, light cultivator and air seeder)</i>					631*	376*
<i>Values used for a full tillage sequence for grassland sowing (chisel, light cultivator and air seeder)</i>					292*	139*

\*Values used in this study for soil redistribution modelling

Not available data: *n.a.*

Table 5.

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

Location	$^{137}\text{Cs}$ inventory ( $\text{Bq m}^{-2}$ )	Soil redistribution rate estimation ( $\text{t ha}^{-1} \text{yr}^{-1}$ )			
		minimum	mean	median	maximum
H1 <sub>up</sub>	1412	-8.71	-7.61	-7.93	-5.87
H1 <sub>down</sub>	1274	-15.95	-13.88	-14.24	-11.10
H2 <sub>up</sub>	438	-88.48	-65.70	-63.49	-47.35
H2 <sub>down</sub>	1448	-6.96	-5.94	-6.18	-4.45
H3 <sub>up</sub>	1285	-16.41	-13.83	-13.73	-11.43
H3 <sub>down</sub>	1293	-15.93	-13.44	-13.37	-11.09
H4 <sub>up</sub>	1741	5.19	6.64	6.82	7.73
H4 <sub>down</sub>	1579	-5.29	-2.47	-2.08	-0.46
H5 <sub>up</sub>	1611	0.85	1.00	0.95	1.23
H5 <sub>down</sub>	1150	-25.32	-20.70	-19.80	-17.86
H6 <sub>up</sub>	1326	-12.92	-10.05	-10.28	-6.73
H6 <sub>down</sub>	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<sub>1</sub>, IR<sub>2</sub> and IR<sub>3</sub>).**

Location	Soil redistribution rate (1960-2009, t ha <sup>-1</sup> yr <sup>-1</sup> )						Relative contribution of water erosion processes to total soil redistribution (%)
	minimum	mean	median	maximum	Mean tillage	Mean water	
H1 <sub>up</sub>	1.42	1.54	1.46	1.73	1.78	-0.24	10
H1 <sub>down</sub>	-6.53	-5.61	-5.89	-4.41	-5.28	-0.33	5
H2 <sub>up</sub>	-5.16	-4.20	-4.17	-3.26	-3.72	-0.48	7
H2 <sub>down</sub>	2.64	3.12	3.24	3.49	3.35	-0.23	7
H3 <sub>up</sub>	0.78	0.96	0.89	1.22	1.34	-0.38	18
H3 <sub>down</sub>	-1.01	-0.52	-0.39	-0.17	-0.13	-0.40	64
H4 <sub>up</sub>	2.80	3.47	3.66	3.95	3.76	-0.40	8
H4 <sub>down</sub>	-8.24	-6.75	-7.06	-4.96	-6.25	-0.51	7
H5 <sub>up</sub>	2.42	3.15	3.41	3.63	3.58	-0.42	9
H5 <sub>down</sub>	-8.26	-6.78	-7.08	-4.99	-6.26	-0.52	7
H6 <sub>up</sub>	1.62	1.77	1.76	1.94	1.86	-0.09	10
H6 <sub>down</sub>	-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  $^{137}\text{Cs}$  activities and  $^{137}\text{Cs}$  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  $10 \times 10$  m windows centred on  $^{137}\text{Cs}$  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  $^{137}\text{Cs}$  measurements (blue) and predicted by LandSoil (red). Black cross and black points figure mean soil redistribution rates predicted by LandSoil model and  $^{137}\text{Cs}$  inventories, respectively. Error bars stand for minimum and maximum values of soil redistribution rates.

Figure 1

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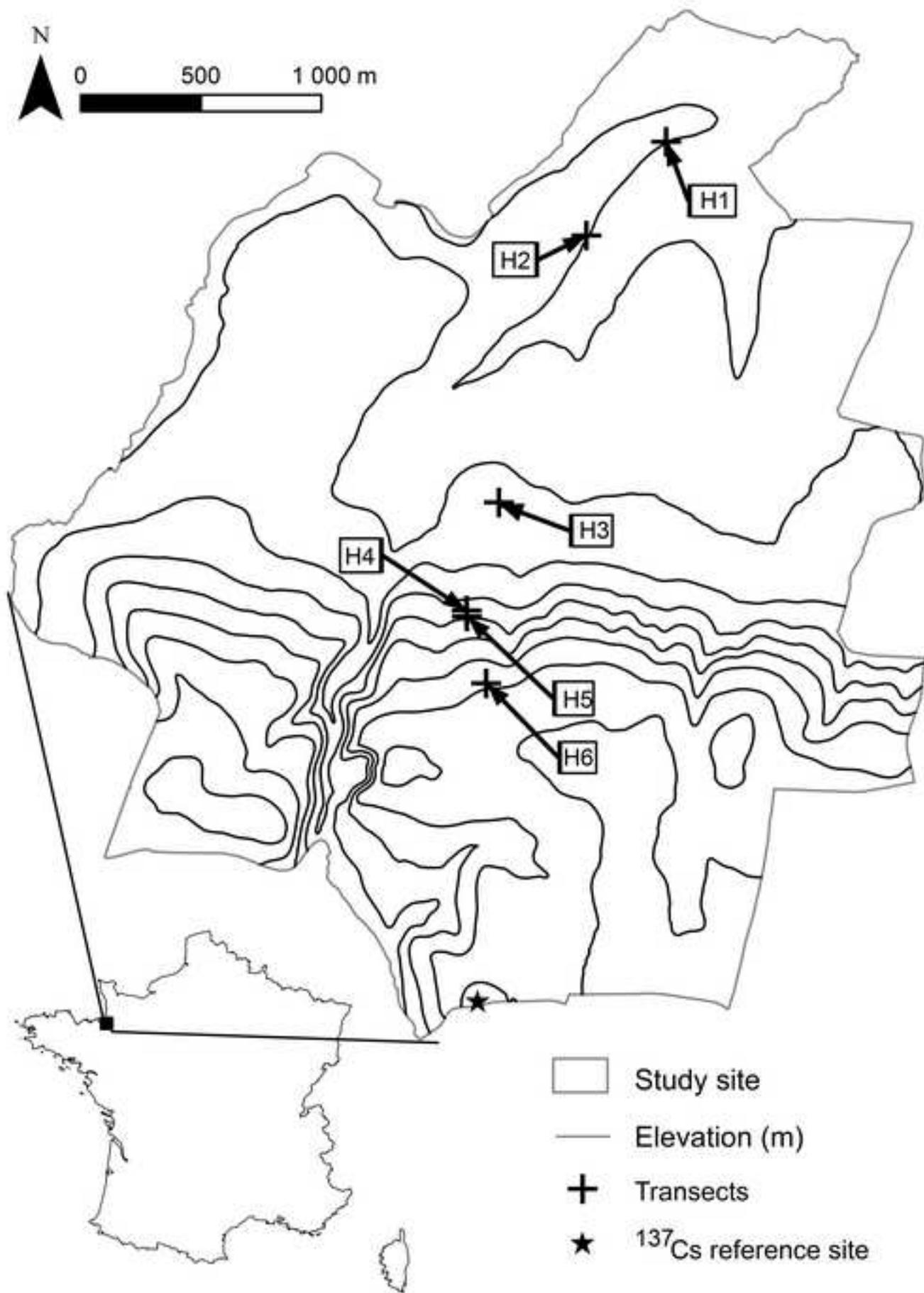


Figure 2

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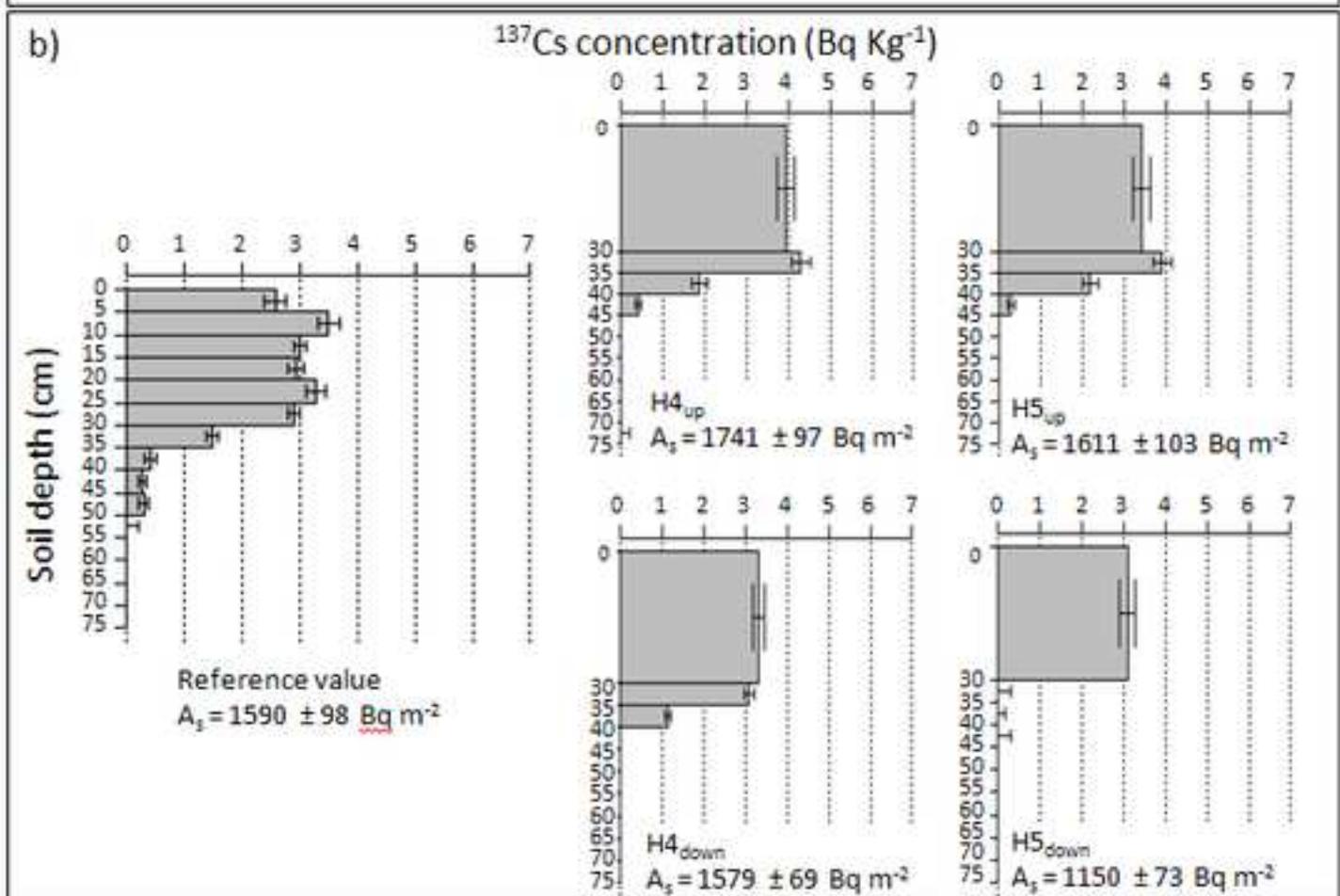
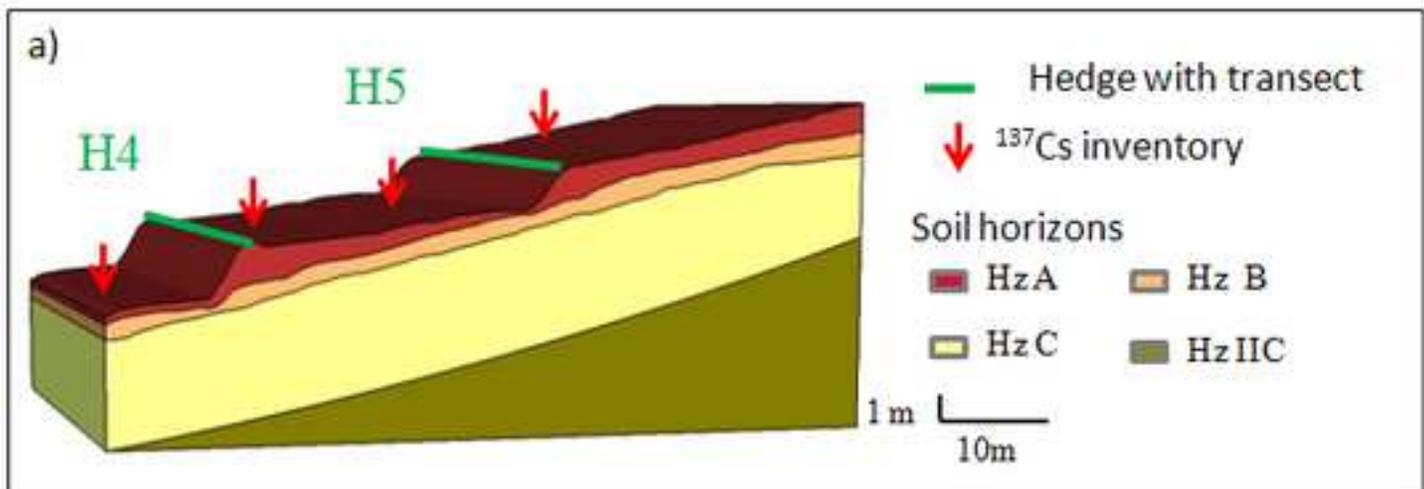


Figure 3  
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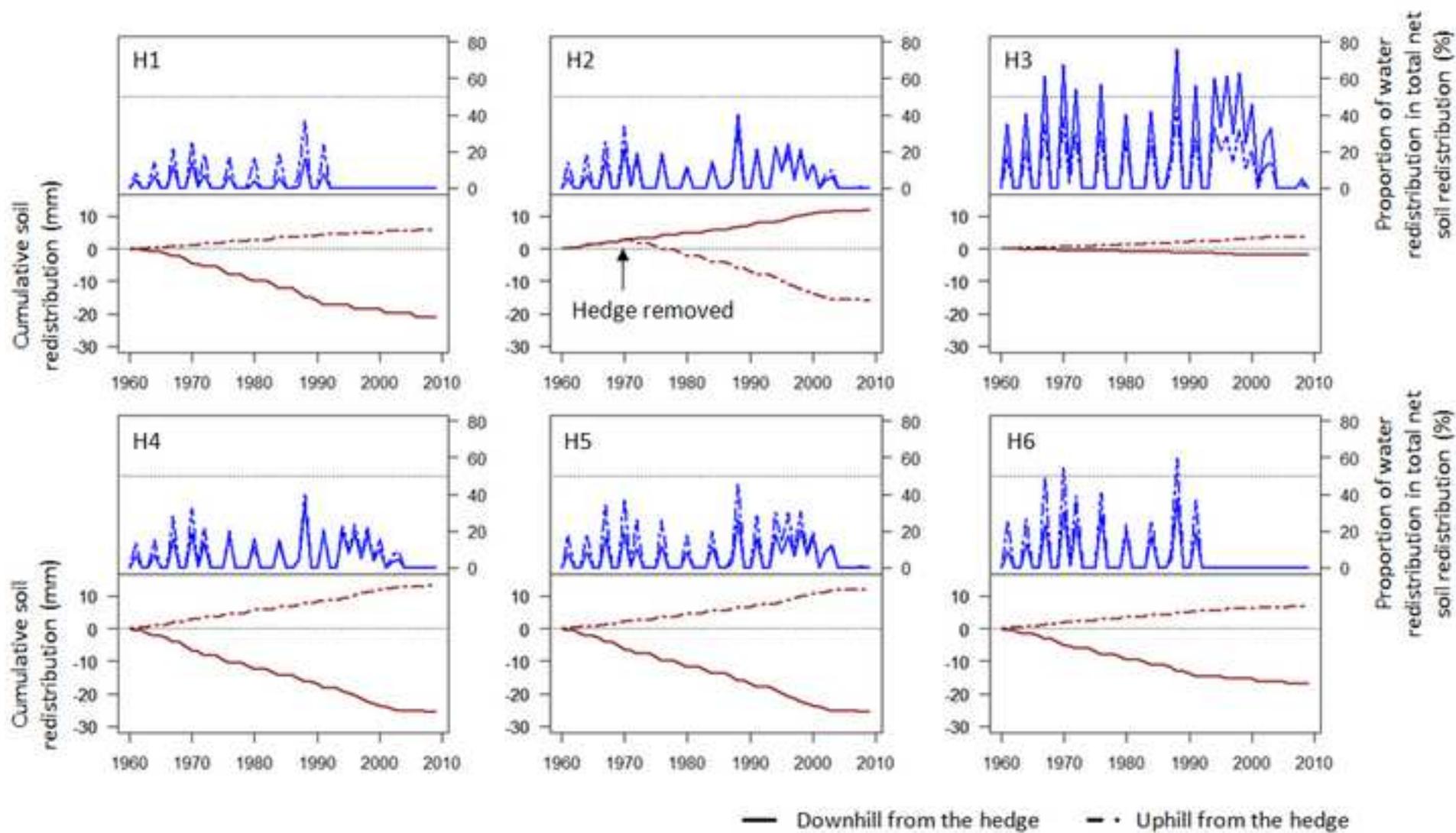


Figure 4 revised  
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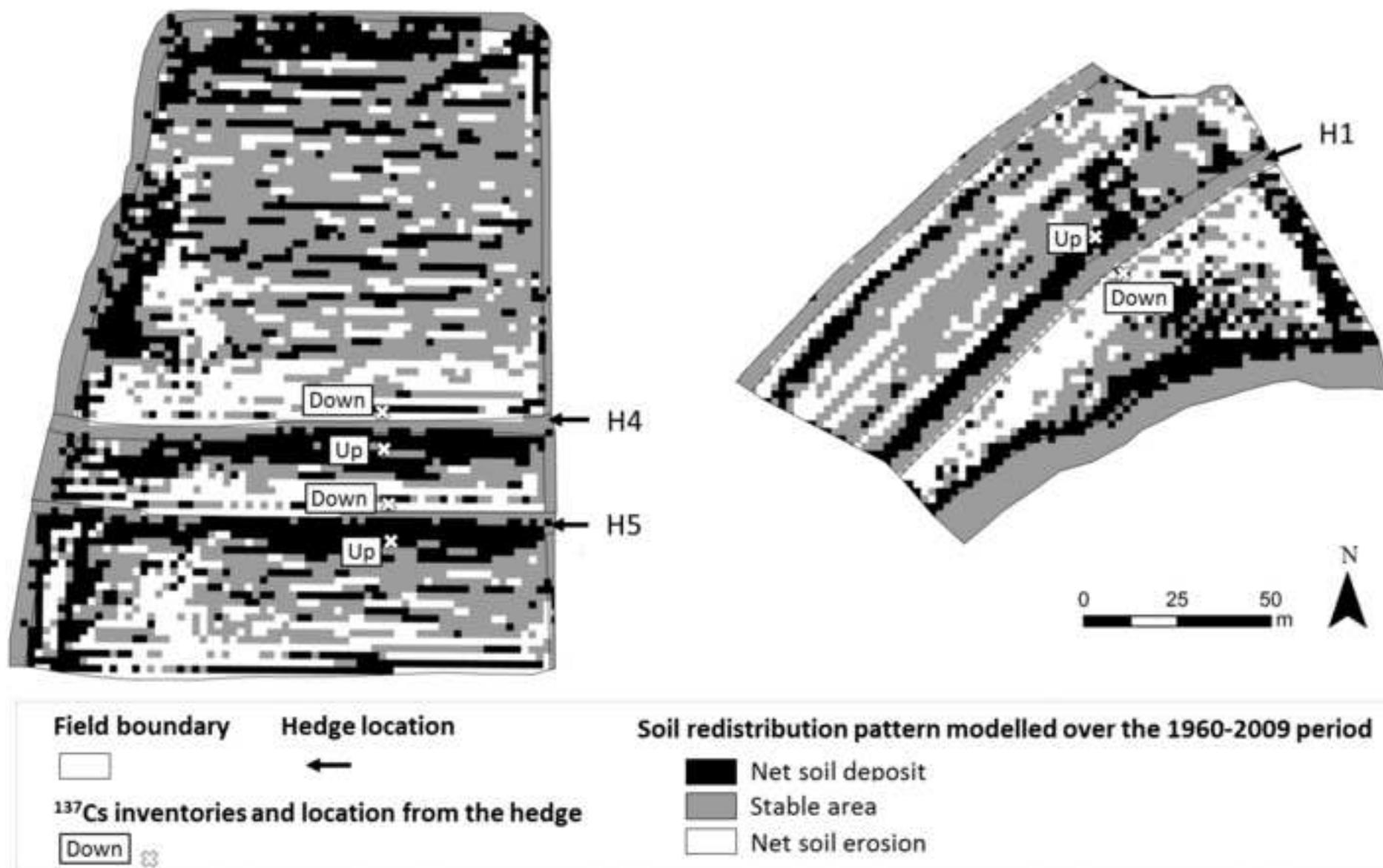


Figure 5 revised

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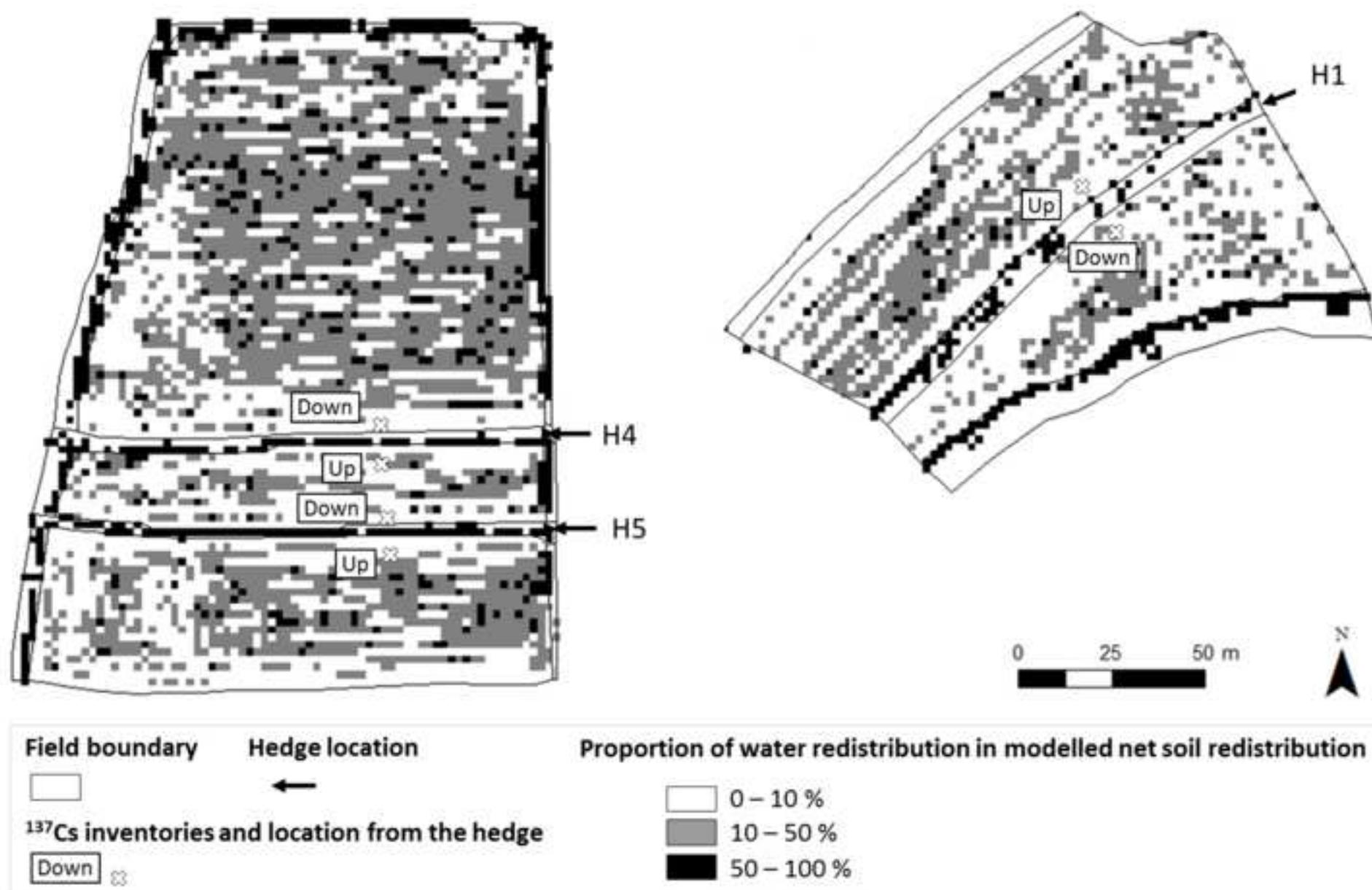


Figure 6 revised  
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