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Combining ^{137}Cs 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 soils and 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 in their spatial distribution. In this study, soil redistribution patterns, generated by both water and tillage erosion, were estimated in the vicinity of hedges in an agricultural landscape. Two complementary methods were employed to estimate soil redistribution from 1960 to 2010: ^{137}Cs conversion models and a spatially distributed soil erosion model (LandSoil model). Both methods determined that hedges affected soil redistribution patterns, which led to soil deposition or limited soil erosion uphill from hedges, even though soil erosion rates were consistently higher than soil deposition rates. Depending on the method, mean soil redistribution rates ranged from -15.9 to -4.7 t ha^{-1} yr^{-1} for all sampling points, from -4.8 to 2.2 t ha^{-1} yr^{-1} in positions uphill from hedges and from -4.8 to -11.2 t ha^{-1} yr^{-1} in positions located downhill from hedges. The impact of tillage on soil redistribution in the vicinity of hedges was found to be higher than that of water processes because 87% of net soil redistribution was linked to tillage. This confirmed the importance of including landscape structure and working at the landscape scale rather than at the plot scale to better estimate soil redistribution in agricultural areas.

1. Introduction

In 2006, the European Commission identified soil erosion as one of the major threats to 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 filtration, carbon storage and climate regulation, among others. Soil erosion has been recognised as having direct consequences on these services both on-site (because of soil loss from fields) and off-site: in recent 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 the decrease in soil thickness 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, 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 of this redistribution (e.g. climate, soil properties, land use and agricultural practices, landscape structure). Before the 1990s, studies focused mostly on water erosion because it was the most obvious process exporting soil out of cultivated fields (Govers et al., 1996). However, it is now recognised that tillage erosion is also an important process to consider, especially when studying soil loss and deposits within individual fields (Govers et al., 1994). Tillage erosion can have an equivalent or even a higher influence on soil redistribution than water erosion (Chartin et al., 2013; Govers et al., 1999; Lobb et al., 2007; Van Oost et al., 2005). Both water and tillage erosion depend on topography, but have distinct impacts on soil redistribution regarding spatial patterns (Li et al., 2007). Water erosion peaks on steep mid-slopes and areas where water concentrates, whereas tillage induces maximum erosion at convexities and deposition at concavities (Govers et al., 1996; Li et al., 2007; Tiessen et al., 2009; Van Oost et al., 2005). Moreover, linkages and interactions exist between water and tillage erosion (Li et al., 2007).

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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 on agricultural land. Impacts of land use on soil redistribution have been investigated in many studies and over a large range of spatial extents. Cerdan et al. (2010) considered European soil erosion studies performed at the plot scale and showed that spring crops and vineyards were the most sensitive to soil erosion. From a long-term survey of soil erosion at the catchment scale, Prasuhn (2012) showed that potatoes induced the highest 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 and explained recorded variations in erosion caused 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, farming practices, particularly tillage practices, impact 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) highlighted that the spatial pattern of land-use change strongly impacted soil redistribution and export out of the catchments studied. In this context, the spatial distribution and connectivity of areas that produce soil erosion and zones where deposition occurs should be included in studies performed at the landscape or catchment scale (Cerdan et al., 2012; Delmas et al., 2012). Vegetated filter strips are some of the anthropogenic structures that impact connectivity within a landscape and affect water and sediment transfer (Bracken and Croke, 2007; Evrard et al., 2008; Gumiere et al., 2011). More particularly, linear structures such as hedges have been recognised as key elements of the landscape that prevent or limit erosion (Baudry et al., 2000; Boardman and Poesen, 2006; Kiepe, 1995b; Skinner and Chambers, 1996). In recent decades, important changes in landscape structure and soil use have been observed in Western Europe, the main ones being land-use homogenisation, removal of linear structures such as hedges and loss of connectivity between landscape elements (Burel and Baudry, 1990; Deckers et al., 2005; Petit et al., 2003). Such changes in the landscape modify soil redistribution dynamics (Evrard et al., 2010) and should be included in soil redistribution modelling.

The 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., 2011). Such models focus on the dominant processes to avoid over-parameterisation and the associated uncertainties, and model simulations rely on decision rules derived from expert judgment using databases of field measurements performed 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) that was produced and deposited globally by atmospheric nuclear-weapon tests (1945–1980) and, in Europe, by the Chernobyl nuclear accident in 1986. ^{137}Cs is now stored in soils, and its stock decreases due to radioactive decay and fine sediment transfer caused by water and tillage erosion. ^{137}Cs has been widely used as a tracer of soil redistribution and has proved useful in soil erosion studies performed around the world (Ritchie and McHenry, 1990; Zapata, 2003). Several studies demonstrated a strong 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, 2003a, 2003b), and ^{137}Cs has been used to calibrate or validate erosion models (Bacchi et al., 2003; Li et al., 2000, 2007, 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). Uniform distribution could be uncertain, however, in complex hedgerow landscapes, especially near hedges (Follain et al., 2009). Moreover, Parsons and Foster (2011) emphasised that the conditions necessary for the use of ^{137}Cs as a soil redistribution indicator are usually not verified. Another limitation is that ^{137}Cs is a point measurement done on a specific date, which doesn't allow assessing the spatiotemporal variation of soil redistribution, and its high cost may limit sampling at the landscape scale.

In this study, we aim to compare two methods to estimate spatial and temporal soil redistribution dynamics near hedges in a rural hedgerow landscape from 1960 to 2010. A new model simulating soil redistribution at the landscape scale (LandSoil; Ciampalini et al., 2012) and ^{137}Cs measurements were used to this end.

2. Materials and methods

2.1. Study sites

The study sites were selected within the study area of Pleine-Fougères (NW France, 48° 32.2′ N, 1° 33.5′ W), which belongs to the European Long-Term Ecosystem Research Network and covers an area of 10 km^2 (Fig. 1). This area is characterised by a high soil spatial

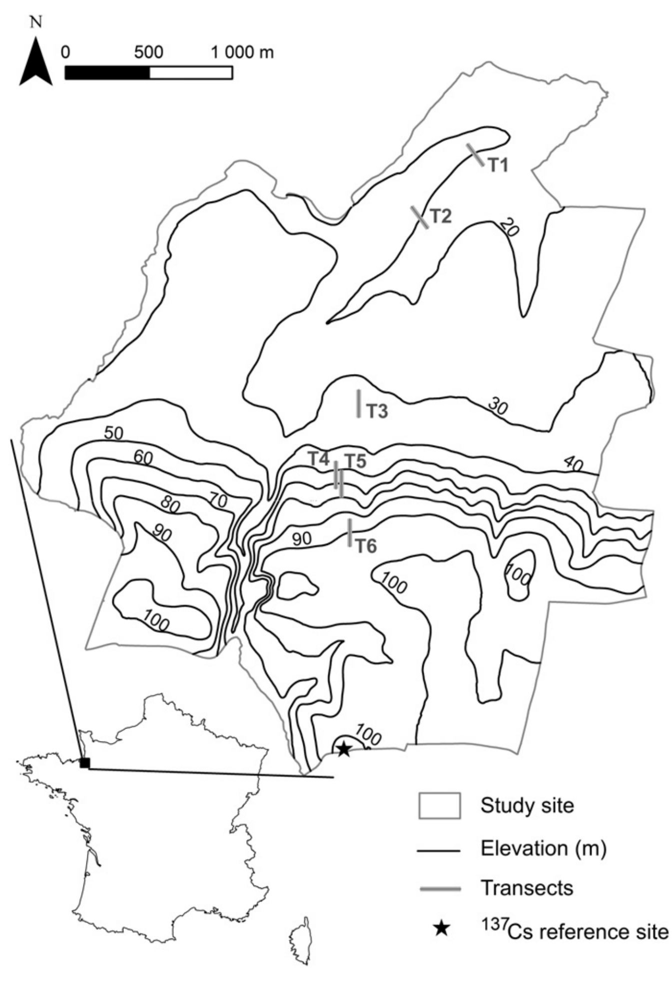

Fig. 1. Location of the transects within the study area of Pleine-Fougères, France.