Sediment tracers in water erosion studies: current approaches and challenges

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Abstract

Introduction Interest in the use of sediment tracers as a tool to complement traditional water erosion or deposition measurements has increased due to the additional information they provide, such as sediment source identification, tracking of sediment movement across the landscape at various temporal and spatial scales, and estimation of soil erosion rates. For these reasons, the utility and robustness of sediment tracing approaches using a wide range of substances and soil properties have been evaluated in numerous studies.

Conclusions A review of established tracing approaches identified five distinct groups of tracing approaches: fallout radionuclides, rare earth elements, soil magnetism and magnetic substances, other tracers, and sediment fingerprinting techniques. This paper describes the basic theory of each tracing approach in assessing soil erosion and sediment redistribution, describing their methodology and main applications, and summarizing the commonalities and differences between the approaches. It also identifies research gaps and future trends.

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1 Introduction

Current rates of soil erosion are unsustainable (Pimentel 2006) and rates of erosion by water could increase significantly over the next century due to changes in both land cover (Yang et al. 2003) and precipitation (Nearing et al. 2005). Concerns about the adverse effects associated with accelerated soil losses have led to an urgent need for reliable quantitative data on the extent and rates of soil erosion worldwide (Lal 2001). Such data are required to: (a) obtain a better understanding of the processes and the main controlling factors of water erosion; (b) validate soil erosion/sedimentation prediction models; and (c) provide a basis for developing scientifically sound land use policies and selecting effective soil conservation measures and land management strategies, including assessment of their economic and environmental impacts (Toy et al. 2002; Morgan 2005; Boardman 2006). Traditional techniques, such as erosion plots and surveying methods for monitoring water erosion, are capable of meeting some of these information requirements but they have a number of important limitations in terms of the representativeness of the data obtained, their spatial and temporal resolution, associated spatial patterns over extended areas, and the costs involved (Higgitt 1991). Traditional monitoring and modelling techniques for soil erosion/sedimentation require many parameters and many years of measurement. For example, to obtain long-term erosion data in agro-ecosystems, experimental plots have to be carried out over decades to integrate the inter-annual variability of climate and cropping practices (Mabit et al. 1999).

The quest for alternative methods of soil loss assessment, due to water erosion to complement and enhance existing

methods, has directed attention to the use of tracing approaches for documenting rates and spatial patterns of soil redistribution within the landscape. These approaches are used to determine the rate of soil loss or sediment generation/production and to track soil redistribution through the landscape. One of the main reasons for the number of different tracing techniques available is that no single tracer fulfils all the requirements of a tracer for erosion and sediment dynamics. According to Zhang et al. (2001), the ideal tracer would have the following characteristics: (a) strong binding to soil particles or ready incorporation into soil aggregates, (b) high analytical sensitivity, (c) easy and inexpensive to quantify, (d) low background soil concentration, (e) no interference with sediment transport, (f) low plant uptake, (g) environmentally benign, and (h) available in variants with similar, but distinguishable, physicochemical properties for multiple tracking. These demands have led to increased specialization in the use of different tracer approaches. Most studies to date have compared the estimates of erosion and sedimentation rates obtained from tracer analysis with model predictions (e.g. Busacca et al. 1993) or conventional measurements (e.g. Mabit et al. 2009), but no comprehensive literature review has been published describing the uses and the applications of the various existing sediment tracer approaches.

A bibliographic research was carried out in June 2011. Figure 1 presents the results of a search of the Web of Science database (http://wokinfo.com/webtools/searchbox/)

using the following as search terms in the title or as keywords: "erosion and tracer" or "sediment and tracer" or "sediment and tracking". The search excluded reviews and tillage and/or wind erosion studies. Only studies that used tracers to make a determination of water erosion or sedimentation rates, or in some cases relative erosion contribution, were considered in this study and were further refined by manually checking that the articles corresponded to experiments involving sediment studies using tracers, as defined within the context of this review. The threshold between small catchment and large basin was set at 100 ha. Table 1 lists all the references consulted to establish Fig. 1.

The aim of this review is to produce a synthesis of the commonly used tracer approaches used in water erosion studies, with three specific objectives: (a) to describe the current tracing approaches, with an indication of the scale of observation; (b) to describe the main applications of the most utilized tracer approaches; and (c) to discuss limitations of each approach that should be improved in the future. The review begins with the most commonly used technique, fallout radionuclides, followed by rare earth elements, magnetism, and other forms of tracers. The final section presents an overview of sediment fingerprinting techniques, which generally use multiple soil/sediment characteristics that are usually naturally present, but heterogeneously concentrated, across the landscape.
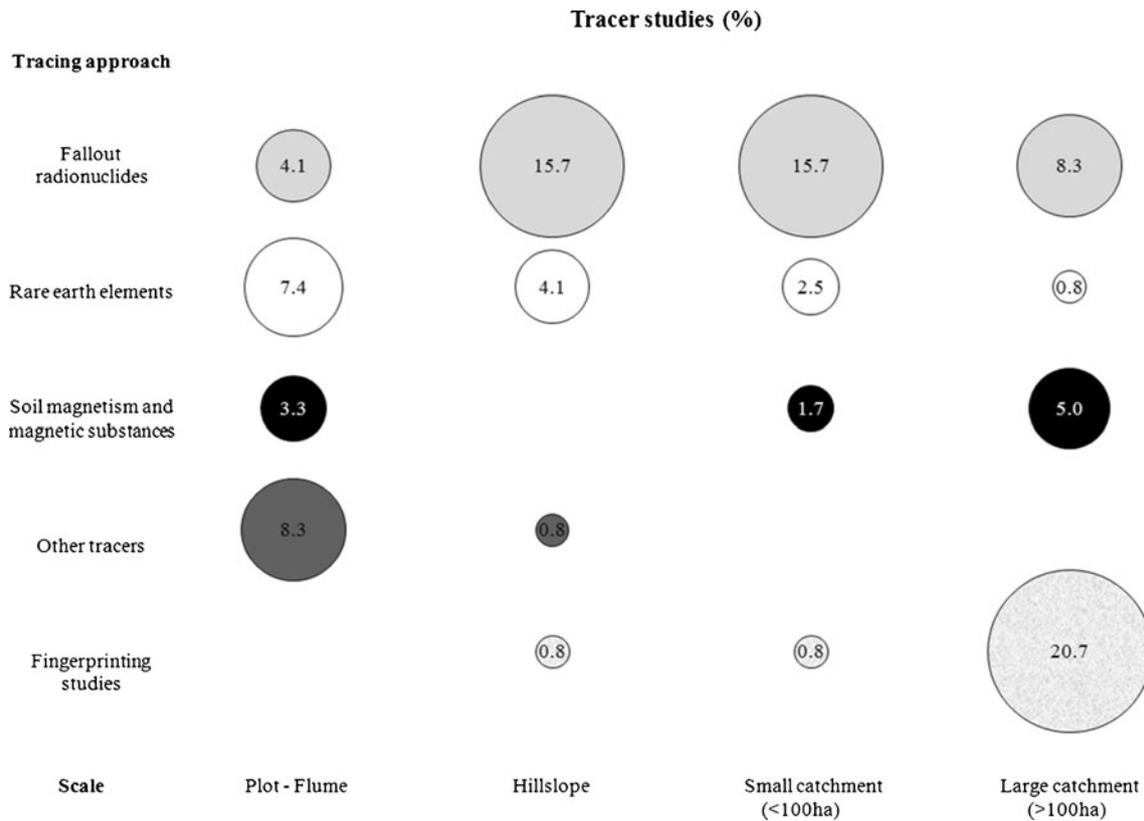

Fig. 1 *Bubble plot* indicating the distribution of erosion studies with tracers found in the review by scale and kind of tracer

Table 1 References of the erosion studies with tracers summarized in Fig. 1 by scale and kind of tracer

Tracer type	References
Fallout radionuclides	[a]Benninger et al. (1998), Dai et al. (2011), de Roo (1991), Kronvang et al. (1997), Li et al. (2010), Martz and de Jong (1987), Olley et al. (1993), Porto et al. (2011), Walling et al. (2000), Zhang and Walling (2005),
	[b]Bacchi et al. (2000), Busacca et al. (1993), Brown et al. (1981), Chiu et al. (2008), di Stefano et al. (1999), Estrany et al. (2010), Lance et al. (1986), Loughran et al. (1988), Lu and Higgitt (2000), Mabit et al. (2009), Ming-Yi et al. (2006), Martz and de Jong (1991), Mizugaki et al. (2008), Montgomery et al. (1997), Spomer et al. (1988), Walling and He (1999b), Walling et al. (1999), Wilson et al. (2008), Yin and Li (2008)
	[c]Bajracharya et al. (1998), Belyaev et al. (2010), Cuesta and Delgado (1997), Fifield et al. (2010), Hassouni and Bouhlassa (2006), Higgitt et al. (2000), Kachanoski (1988), Kachanoski and de Jong (1984), Li et al. (2009), Mabit et al. (2008b), Olson et al. (2008), Quine et al. (1999a), Schuller et al. (2000), Sutherland (1992), Wallbrink et al. (2002), 1999: Walling et al. (2009), Xinbao et al. (1990), Zhang et al. (1998)
	[d]Bernard et al. (1992), Quine et al. (1999b), Syversen et al. (2001), Wallbrink and Murray (1993), Woolridge (1965)
Rare earth elements	[a]Mahler et al. (1998)
	[b]Kimoto et al. (2006a), Polyakov et al. (2009, 2004)
	[c]Deasy and Quinton (2010), Matisoff et al. (2001), Stevens and Quinton (2008), Wude et al. (2008), Yang et al. (2008)
	[d]Kimoto et al. (2006b), Li et al. (2006), Michaelides et al. (2010), Polyakov and Nearing (2004), Pu-Ling et al. (2004), Tian et al. (1994), Wei et al. (2003), Xue et al. (2004), Zhang et al. (2003)
Soil magnetism and magnetic substances	[a]Caitcheon (1993), Dearing et al. (2001), Maher et al. (2009), Slattery et al. (2000), Walling et al. (1979), Yu and Oldfield (1993)
	[b]Hardy et al. (2000), Royall (2001)
	[c]
	[d]Armstrong et al. (2010), Guzmán et al. (2010), Parsons et al. (1993), Ventura et al. (2002)
Other tracers	[a]
	[b]
	[c]Schwertmann and Schmidt (1980)
	[d]Bennett et al. (2010), Mentler et al. (2009), Olmez and Pink (1994), Plante et al. (1999), Riebe (1995), Sharma et al. (2009), Spencer et al. (2011), Wheatcroft et al. (1994), Young and Holt (1968), Yu et al. (2011)
Fingerprinting studies	[a]Barcellos et al. (1997), Collins and Walling (2002), Collins et al. (1998), Cunha et al. (2006), de Junet et al. (2009), Devereux et al. (2010), Fox and Papanicolaou (2008a, b), Juracek and Ziegler (2009), Kouhpeima et al. (2011), Martínez-Carreras et al. (2010a, c), Martinotti et al. (1997), Miller et al. (2005), Minella et al. (2008), Motha et al. (2003), Nosrati et al. (2011), Poulenard et al. (2009), Rhoton et al. (2008), Rowan et al. (2000), Russell et al. (2001), Rustomji et al. (2008), Sawhney and Frink (1978), Schoonover et al. (2007), Walling et al. (2007)
	[b]Fox and Papanicolaou (2007)
	[c]Bellanger et al. (2004)
	[d]

[a] Studies made at large catchments (>100 ha)

[b] Studies made at small catchments (<100 ha)

[c] Studies made at hillslope scale

[d] Studies made at small plot or laboratory scale

2 Tracing approaches

2.1 Fallout radionuclides

Fallout radionuclides (FRN), such as ^{137}Cs (half-life, $t_{1/2}=$ 30.17 years), ^{7}Be ($t_{1/2}=53.12$ days), and ^{210}Pb ($t_{1/2}=$ 22.26 years), are the most extensively used soil redistribution tracers reported in the scientific literature (Walling 2003; Mabit et al. 2008a), representing approximately half of all the studies included in Fig. 1. Of the various FRN used, ^{137}Cs is by far the dominant one. ^{137}Cs is an anthropogenically derived radioisotope produced in great quantities and released into the

stratosphere and globally distributed during the atmospheric testing of thermonuclear weapons from the mid-1950s to the 1960s. Fallout of ^{137}Cs began in 1954, peaked in the 1960s after moratoriums on testing and the Test Ban Treaty signed in 1963, and fell to almost zero levels by the 1980s. Locally, additional ^{137}Cs was released by the Chernobyl nuclear power plant accident in 1986, adding a second spike. This radioisotope has been used in a wide variety of depositional environments to determine erosion and sedimentation rates for medium time scales (tens of years) across a broad range of spatial scales: from hillslope (e.g. Wallbrink and Murray 1993) and small catchment (e.g. Higgitt et al. 2000) to large basins (e.g. de Roo 1991). By combining field measurements with model analysis, ^{137}Cs has also been used to determine rates of water and tillage erosion (e.g. Quine et al. 1999a) or to differentiate between surface, subsurface and stream bank sources (Li et al. 2003; Zhang and Walling 2005). Two other radionuclides have been tested and validated: ^{210}Pb to provide longer-term estimates (approx. 100 years) of soil redistribution magnitudes (e.g. Walling and He 1999b), and more recently, ^{7}Be to estimate short-term erosion and sedimentation rates (e.g. Walling et al. 1999). ^{210}Pb is a naturally occurring product of the ^{238}U decay series derived from the decay of gaseous ^{222}Rn, the daughter of ^{226}Ra. ^{226}Ra exists naturally in soils and rocks and decays *in situ* to generate ^{210}Pb. This is termed supported ^{210}Pb and is in equilibrium with the amount of ^{226}Ra in soils. The ^{210}Pb used as a soil erosion tracer is unsupported ^{210}Pb (^{210}Pb$_{ex}$) and reaches the soil as fallout, as a result of the diffusion of small amounts of ^{222}Rn to the atmosphere. ^{7}Be is a naturally occurring cosmogenic radionuclide produced in the stratosphere and troposphere as a result of nitrogen and oxygen spallation and subsequently deposited as fallout.

These three FRN have a strong affinity to fine soil particles and are mostly mobilized and transported by physical processes such as water, tillage, and wind erosion (Mabit et al. 2008a). In fact, for using an FRN as a soil erosion tracer it is important that it has a specific environmental behaviour including chemical stability and a limited mobility in soil environments. For example, ceasium is the most electropositive and the most alkaline element among alkali metals. Therefore, after its deposition at the soil surface it is rapidly and strongly adsorbed on the cation exchange sites of fine soil particles (clay and organic particles) and can thus be considered as being essentially non-exchangeable (Davis 1963; Lomenick and Tamura 1965; Ritchie and McHenry 1990). The ^{137}Cs intercepted by the plant canopy can be transferred to the soil via wash-off and the biological uptake from soils by vegetation can be considered negligible (Dahlman et al. 1975). If adsorbed by the vegetation, the ^{137}Cs is released to soils when the vegetation dies and decays (Davis 1963; Rogowski and Tamura 1970a; b; Dahlman et al. 1975). Therefore, lateral redistribution of ^{137}Cs in soils by biological and chemical processes is insignificant in comparison with the movement

of ^{137}Cs by physical processes, i.e. erosion and transport by water and wind (Ritchie and McHenry 1990). For these reasons, and since ^{137}Cs fallout was relatively uniformly distributed across the landscape and strongly adsorbed by soil particles, it has been used worldwide under various agri-environmental conditions as a soil tracer for studying the physical processes of erosion and sedimentation and to provide quantitative information on their rates and spatial patterns (e.g. Mabit et al. 2008a; Ritchie and Ritchie 2008).

In addition to these three main FRN (^{137}Cs, ^{7}Be, and ^{210}Pb$_{ex}$) already present into the soil, other radionuclides that have been used as erosion tracers include ^{59}Fe ($t_{1/2}=$ 44.51 days; e.g. Woolridge 1965) and ^{134}Cs ($t_{1/2}=2.06$ years; e.g. Syversen et al. 2001). These two radionuclides were incorporated into small controlled experimental areas (points and lines of 1.5 m in a bare area and a tilled plot of 225 m^2, respectively) as a liquid solution, providing a usable method for tracing surface soil movement at a small scale for local investigations.

The determination of erosion and/or sedimentation rates in the landscape is based on: (a) the determination of the reference inventory of the radionuclide in stable and undisturbed locations (sites such as flat permanent pastures or forests reflecting the original fallout input, without erosion and/or deposition processes) and (b) the comparison of this undisturbed tracer inventory and its vertical distribution to the investigated disturbed soil profile. This inventory is periodically replenished in the case of ^{7}Be and ^{210}Pb$_{ex}$. In the case of ^{137}Cs fallout, the inventory decreases according to its radioactive decay. Figure 2 illustrates the typical distribution of the areal activity of the various FRN in reference sites (undisturbed soil) and cultivated sites exhibiting deposition or erosion. In undisturbed areas, ^{137}Cs and ^{210}Pb$_{ex}$ are distributed mostly in the first 10 cm of the soil, with the highest concentration close to the surface and quickly diminishing with depth. This reflects the existence of downward migration due to soil processes, such as bioturbation or preferential water flow. Because of its short half-life, ^{7}Be is only found in the first few centimetres of the topsoil. When applied to the soil surface as a solution in short-term experiments, the radionuclides are mostly concentrated in the top 2–4 cm of the soil (e.g. Syversen et al. 2001).

The first step in using the FRN approach is to select a representative reference site as close as possible to the study area and to determine a sufficient number of samples to include the uncertainty linked to the spatial variability of the initial fallout. To avoid bias of the FRN reference level, a repeated-sampling concept can be applied. It requires two sets of samples and use of point-specific values, the FRN inventories of the first set of samples being considered as the reference levels (e.g. for ^{137}Cs see Tiessen et al. 2009; Li et al. 2011). Because of the mechanisms regulating fallout
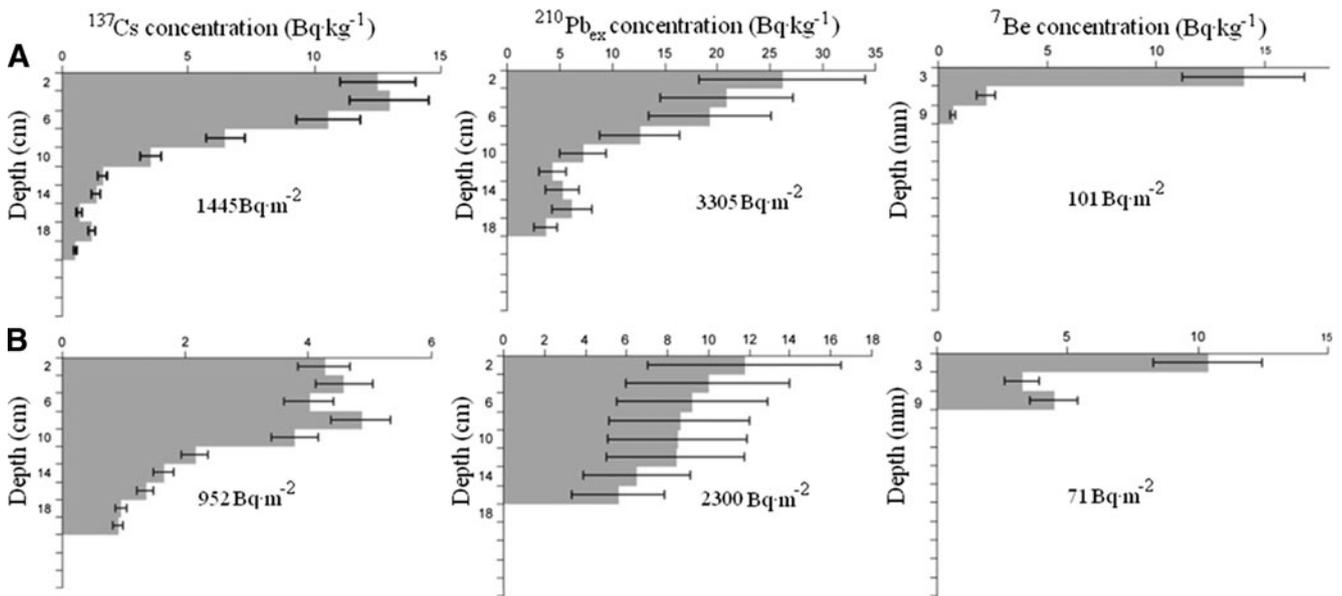
Fig. 2 Typical depth distributions of Fallout radionuclides (from *left* to *right*: ^{137}Cs, ^{210}Pb$_{ex}$, and ^{7}Be) in an undisturbed (**a**) and a cultivated (**b**) eroded soil in Morocco. NB: The uncertainty of the measured mass activity of each radioisotope is expressed at 2σ (adapted from Mabit et al. 2008a, published with the permission of Elsevier 2012)

distribution, the concentration of these tracers in reference areas is subjected to spatial variability at global, regional, field and plot scales. Sutherland (1991, 1996) suggested that in undisturbed areas, approximately 10 sampling points will usually be necessary to provide an accurate estimate for the reference inventory. In investigated cultivated areas, the number of samples taken to determine inventory values varied widely from a few cores per hillslope transect (e.g. Cuesta and Delgado 1997) to up to 60 on agricultural fields (Owens and Walling 1996), and between 68 and 80 in small catchments (Ritchie et al. 2009 and Wallbrink et al. 2002, respectively). Soil samples are usually collected to a depth (usually 50–60 cm) sufficient to measure the total content of radionuclides present in the profile and are often subdivided into regular depth increments (usually 5–10 cm in cultivated area; 2.5 or 5 cm in the case of the reference site) to obtain the radionuclide's vertical distribution.

Typically the amount of a specific radionuclide within the soil profile is quantified in mass activity (Bqkg^{-1}) or areal activity (Bqm^{-2}) using gamma spectroscopy.

The next step is to translate the radiotracer areal activities into erosion or sedimentation rates (tha^{-1}year^{-1}) using appropriate theoretical conversion models. These models are formulated for both cultivated and uncultivated soils, for each FRN (^{137}Cs, ^{210}Pb$_{ex}$, and ^{7}Be), and are associated with a user-friendly software for their implementation (Walling et al. 2002). The election of the model and the assumptions made in its calibration have a direct impact on the soil redistribution magnitude obtained (Walling and He 1999a). As an example, the ^{137}Cs conversion models are summarized in Table 2. One

of the most widely used models is the Mass Balance Model (MBM 1).

$$\frac{dA(t)}{dt} = I(t) - \left(\lambda + \frac{R}{d}\right)A(t) \qquad (1)$$

Where t is time since start of ^{137}Cs accumulation, $A(t)$ is cumulative activity per unit area, R is erosion rate, d is cumulative mass representing average plough depth, λ is a decay constant for ^{137}Cs, and $I(t)$ is an annual deposition flux at time t.

The results obtained with the MBM 2 model (see Table 2) are likely to be more realistic than those provided by the simplified MBM 1. The results obtained with the MBM 2 involve additional parameters that take into account several specificities of ^{137}Cs that improve the precision of the soil loss assessment as compared to real measured erosion rates (e.g. Fulajtar 2000). However, in order to use this model, some parameters (e.g. relaxation mass depth and the proportion of the annual ^{137}Cs input susceptible to removal by erosion) are more difficult to specify. The MBM 3 model (see Table 2) represents an important improvement over the other existing models because it takes into account the effects of tillage-induced soil movement processes which could lead to soil loss rates as high as those by water erosion (see Li et al. 2007). However, the advanced models (MBM 2 and 3) can only be used for individual downslope transects. By using one of the conversion models in Table 2 it is possible to obtain point values of erosion or sedimentation rates from punctual ^{137}Cs activity levels, interpolate the data and create a soil redistribution map (Ritchie et al. 2009)

Table 2 Review of models for estimating erosion rates from ^{137}Cs measurements from models requiring a lower number of information (less parameters) to be used (upper part of the Table) to those requiring more information (more parameters), to be used (lower part of the Table)

Model	Approach
Empirical relationship[a, b]	Empirical measurements from plots measurements.
Proportional model[a]	Assume complete mixing of ^{137}Cs in the plow layer. Soil loss is proportional to ^{137}Cs loss since the beginning of accumulation.
Mass balance model (MBM 1)[a]	It models the change of ^{137}Cs in soil profile due to time changes in ^{137}Cs inputs, losses due to erosion and incorporation of soils without ^{137}Cs from below the plow layer.
Profile distribution model[b]	Assuming a given shape of the ^{137}Cs profile distribution with depth in undisturbed soil.
Refined mass balance model (MBM 2)[a]	Including also the removal of freshly deposited ^{137}Cs before cultivation and grain size selectivity associated with sediment mobilization and transport.
Mass balance model including tillage (MBM 3)[a]	It is a mass balance model including soil redistribution caused by tillage.
Diffusion and migration model[b]	It includes the redistribution of ^{137}Cs into the soil trough diffusion and migration processes.

[a] Applicable to cultivated soils

[b] Applicable to undisturbed soils

and/or average the values in homogeneous areas and establish a full sediment budget (Wallbrink et al. 2002). Complementary and in-depth information about the different FRN conversion models and their parameters have been recently provided by Walling et al. (2011).

Recently, other radioisotopic soil tracers have attracted the attention of the research community, for example plutonium isotopes (Everett et al. 2008; Tims et al. 2010; Hoo et al. 2011; Ketterer et al. 2011), such as ^{239}Pu ($t_{1/2}=$ 24110 years) and ^{240}Pu ($t_{1/2}=$6561 years), which are alpha emitters and originated from nuclear weapon test fallout like ^{137}Cs. Their main advantage over other FRN is the long half-life of ^{239}Pu and ^{240}Pu, which ensures their long-term availability to be used as tracers for environmental purposes. However, they do have limitations, such as the need for acquisition of accurate detection.

2.2 Rare earth elements

Rare earth elements (REE) represent approximately 13 % of tracer studies found in Fig. 1 and provide a good example of the issues that can arise when developing the use of a specific tracer in water erosion studies for different soils and different scales. This approach relies on REE being incorporated into the soil prior to the experiment, and the determination of the concentration in the soil and sediment after a period, from weeks (e.g. Zhang et al. 2003) to years (e.g. Kimoto et al. 2006a).

There are many small-scale studies using REE as tracers, including: at the flume scale to study detachment and deposition processes (e.g. Michaelides et al. 2010) and rill erosion processes (e.g. Lei et al. 2006); at the plot scale (e.g. Tian et al. 1994) to study the impact of different topographical positions on soil erosion under simulated rainfall; and at the hillslope scale (e.g. Deasy and Quinton 2010) or the small catchment scale (e.g. Polyakov et al. 2004, 2009). Deasy and Quinton (2010) tagged only the upper millimetres of undisturbed soil in short-term experiments. They used three REEs to tag upper, mid and downslope hillslope areas with a length varying from 66 to 99 m. Within these hillslopes a fourth tracer was used to tag the soil in the tractor wheel marks. They showed after three rainfall events that the upslope area was the most eroding area of the hillslope, while the wheel tracks acted mainly as conduits for sediment transport. Polyakov et al. (2004) divided a 0.68-ha catchment into geomorphological homogeneous areas and tagged each one using a different REE. The sampling density used in this study was 94 sampling points over 6,800 m^2, each of which comprised 30 sub-samples taken in a 2-m radius around the sample point. Using sediment samples from different rainfall events and soil samples taken from the upper 30 mm of soil, the authors studied sediment redistribution and sediment budgets within the catchment for a period of approximately 5 months. Kimoto et al. (2006a) followed a similar approach in the same agricultural catchment to obtain sediment redistribution and sediment budgets for the different geomorphological units of the catchment. However, they point out that just the upper millimetre of the soil was tagged and therefore was suitable only for short-term studies as the tagged soil layer may be rapidly eroded. REE can also be used at larger spatial erosion scales, such as studying the sediment transport in a karst area in which a lanthanide-labelled clay was deployed (Mahler et al. 1998).

REE can be detected at very low concentrations (up to parts per billion) using neutron activation analysis (INAA, e.g. Orvini et al. 2000) or by inductively coupled plasma-mass spectrometry (ICP-MS) after acid extraction (Zhang et al.

2001). The majority of the REE studies use ICP-MS with a similar acid extraction procedure, most of them following the method of Zhang et al. (2001), although some use a slightly modified procedure (e.g. Deasy and Quinton 2010).

To convert REE concentrations into erosion and deposition rates, conversion models (equations) are necessary. Polyakov and Nearing (2004) confirmed the validity of the formulas through an indoor-flume experiment using simulated rainfall over a uniform slope of 4 m×4 m divided in segments and tagged with different REE. By sampling the soil surface and the sediment measured at the outlet of the flume and determining the REE concentration before and after each rainfall simulation, they were able to determine erosion and sedimentation rates along the flume slope as well as sediment source during the simulations. The amount of sediment delivered from segment i to the flume outlet in the time-step j (L_i^j), can be calculated as:

$$L_i^j = T^j \frac{CC_i^j}{O_i} \qquad (2)$$

Where i is the tracer or segment index, j is the time-step index, T^j is the total mass of the sediment delivered at the flume outlet during time-step j, and O_i is the original average concentration of the tracer for segment i. REE concentration CC_i^j is obtained as the result of correcting the direct ICP-MS measurements for soil background and the extraction efficiency of each element. Eq. (2) can be rearranged to determine a ratio of sediment discharged to that originated from each segment i during time j, R_i^j:

$$R_i^j = \frac{CC_i^j}{O_i} = \frac{L_i^j}{T^j} \qquad (3)$$

The sum of the sediment ratio for each segment gives the overall sediment ratio, defined as the ratio between the amount of sediment delivered to the outlet and the amount of sediment generated on hillslopes, for the whole flume at each time step. This should be equal to 1 if the tracer is completely recovered. Weighting each R_i^j for the sediment mass delivered at each time step j, it is possible to determine the sediment ratio for the whole rainfall simulation. Zhang et al. (2003) also determined erosion and sedimentation rates at a given flume position comparing erosion rates calculated from REE concentrations in surface soil samples taken before and after six rainfall simulations with determinations of sediment loss based on high resolution scanning of the soil surface at the same time intervals. The close agreement between the approaches and R values close to 1 demonstrated the applicability of the REE method in laboratory experiments, which then led to its successful use at the field scale (e.g., Polyakov et al. 2004). In addition, by analysing sediment collected at the end of flume during the

simulations for REE, it was possible to provide information on changing sediment sources during the simulations.

There are some other concerns that need to be addressed regarding the use of REE as tracers. Several studies have explored methods of incorporating these tracers into the soil and the selectivity of their binding to different aggregate sizes, all of them using REE in their oxide forms. Zhang et al. (2001) found that the best way to incorporate these tracers into the soil was by serial dilutions with dried soil sieved to 6 mm and pre-wetted with deionised water to approximately 15 % water content. When applied in larger field experiments, the application methods varied: spreading incubated tagged soil on the surface followed by a light tillage, as done by Polyakov et al. (2004) in a small agricultural catchment; spreading the tagged soil on the surface followed by water spray to improve binding to soil aggregates (e.g. Polyakov et al. 2009); excavating soil pits and refilling with tagged soil in selected areas of a hillslope (e.g. Yang et al. 2008); spraying as water solution in wheel track areas along a hillslope (e.g. Deasy and Quinton 2010); or by mixing REE with sand before spreading (e.g. Stevens and Quinton 2008). In all the cases, the interpretation and analysis of the experimental measurements were conditioned by the initial distribution of the tracer along the soil profile, which was restricted to the upper few millimetres in the case of surface applications of Polyakov et al. (2009) and Deasy and Quinton (2010), and deeper in the case of Polyakov et al. (2004).

The second consideration is the selectivity in REE binding to soil aggregates of different sizes. Zhang et al. (2001) determined the concentration of five REE in tagged soil aggregates separated by wet sieving. The soil used was well-structured silt-loam soil and the results in Fig. 3 suggest a more or less homogeneous distribution along different aggregate sizes. These results support the assumption of non-selective binding among the soil aggregates used in the calculations of tracer erosion rates made in experiments using REE on these kinds of soils (e.g. Zhang et al. 2003; Polyakov and Nearing 2004). However, this assumption of non-selective binding to different soil aggregates is not always achieved, as in Kimoto et al. (2006b), who made a similar analysis using the same five REE in a coarse-textured soil in a semi-arid area. Table 3 shows the trend towards an increased concentration in the finer aggregate sizes. Kimoto et al. (2006b) explained this trend as a function of the higher clay content of the finer aggregate sizes, which provide the chemically active clay surfaces to which the REE bind. They suggest that in situations where selective binding takes place, the REE analysis should be made by separating samples into sub-samples of aggregate sizes that present homogeneous binding of REE—which increases the time, cost and complexity of the analysis—and including those differences in the calculations of soil erosion

Fig. 3 Total extractable rare earth element (REE) concentrations of the tagged REE-tagged soil (whole) and individual aggregate size groups after sieving (adapted from Zhang et al. 2001, published with the permission of Soil Science Society of America 2012)
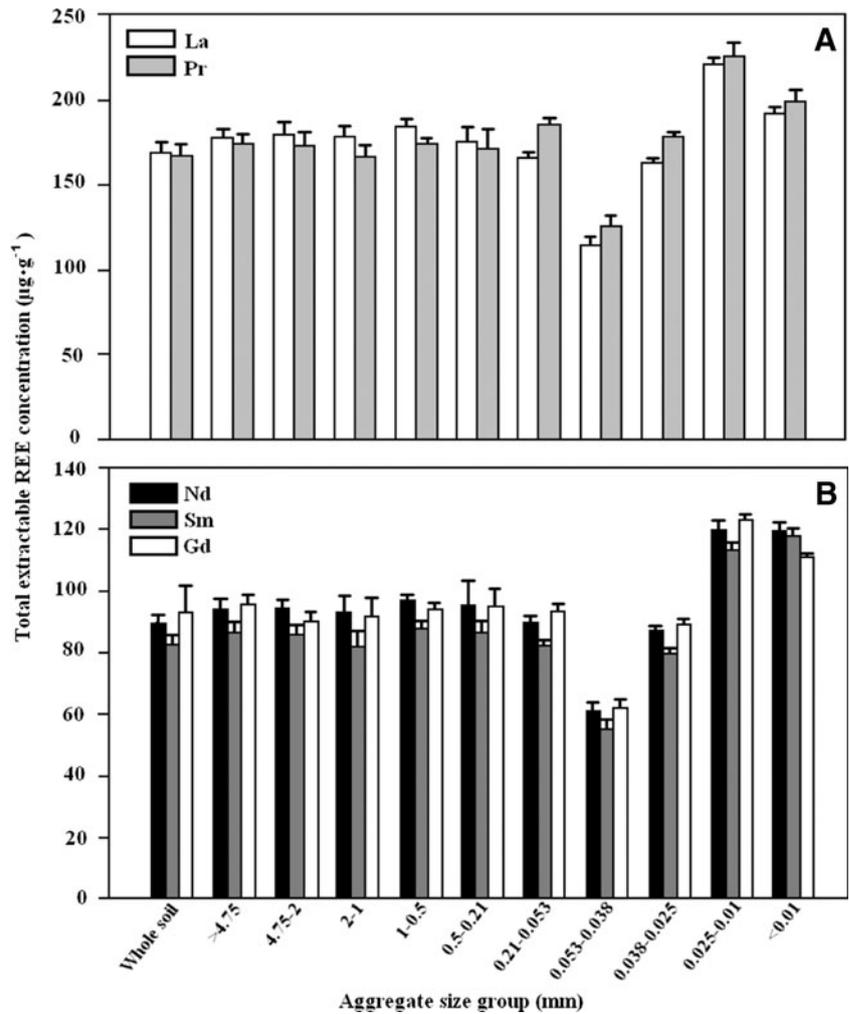
rates. Alternatively, the tagged soil that is applied to the soil surface may be separated by size fractions prior to field application so that each size fraction may be incubated with the same, desired concentration of REE (Polyakov et al.

Table 3 Total extractable REE concentrations of the REE-tagged soil and individual aggregate size groups after sieving (adapted from Kimoto et al. 2006b, published with the permission of Elsevier 2012)

Particle size group	Particle size (mm)	Ratio									
		Kendall area					Lucky Hills area				
		La	Pr	Nd	Sm	Gd	La	Pr	Nd	Sm	Gd
Group D	4.7–8.0	2.5	1.9	7.1	2.1	5.8	2.1	1.4	4.3	3.1	5.5
Group C	2.0–4.7	4.2	4.0	8.6	5.0	11.9	4.0	2.8	7.6	6.6	10.3
	.7–2.0	5.8	6.7	11.5	8.6	23.1	4.4	3.0	9.7	9.0	14.2
	.3–.7	7.4	9.2	15.3	12.4	31.5	4.4	3.3	10.0	10.6	16.4
Group B	.18–.3	5.7	7.2	11.7	10.1	23.3	7.0	5.2	15.4	17.0	24.9
	.09–.18	6.7	9.0	14.0	12.4	31.3	8.5	7.2	15.3	19.7	30.2
Group A	.04–.09	33.0	51.6	43.9	47.5	126.3	19.3	18.1	27.7	34.8	60.9
	.02–.04	96.9	110.3	77.3	103.2	234.4	55.1	55.8	53.1	50.6	131.5
	.01–.02	95.7	103.1	76.0	77.9	215.8	49.3	104.9	50.4	39.3	173.2
	<.01	41.0	47.3	35.3	24.1	66.5	50.3	54.0	40.8	26.3	96.5

2009). Soil losses would be overestimated in these situations of preferential binding and selective transport of different aggregate sizes. Therefore, it is important to determine this selectiveness by comparing aggregate distributions of tagged and untagged soils and textural class of sediment. The studies of Zhang et al. (2001) and Kimoto et al. (2006b) also demonstrated that, as expected, due to their strong binding to soil particles, REE was not leached in soil column experiments.

2.3 Soil magnetism and magnetic substances

Magnetic tracing studies refer to approaches using magnetic properties in two different ways. One discriminates sediment sources through the natural magnetic properties of soil constituents (e.g. Dearing et al. 2001), and the other approach uses magnetic tracing substances, incorporating them into the soil and measuring their concentration and distribution in soil and sediment before and after an experiment. Table 4 summarizes some of these magnetic properties measured in both kinds of studies, which are based on measuring the magnetization of the sample under different magnetic fields. It is beyond the scope of this review to provide a thorough discussion of the magnetic properties of minerals, but a good introduction can be found in several texts (e.g. Maher 2007).

For source discrimination, soil and sediment processing is similar to that described in the fingerprinting approach (see below), as it is based on the use of magnetic properties naturally present in soils materials. Magnetic properties of soil mineral constituents, especially iron oxides (magnetite, maghemite, hematite, goethite and pyrrhite), allow for the discrimination of different types of soil. Bias introduced by selective transport is a major concern and has been addressed by processing the samples and analysing only the material most likely to be transported. For example, Slattery et al. (2000) analysed material screened to <63 μm, because in their

6.2 km^2 English study catchment there was no evidence of suspended sediment of a larger size. Maher et al. (2009) analysed the medium sand fraction between 250 and 355 μm, because this was best suited for the transport processes in their study of tropical sediment provenance in NE Australia. On other occasions, the sample has been divided into sand, clay and silt fractions, as in Yu and Oldfield's (1993) sediment source study for a reservoir in Spain. It is also possible to correct the bias by specific surface or empirical functions, as in the case of the fingerprinting studies described in the next section. The interpretation of magnetic fingerprinting has relied on a variety of statistical approaches, from full un-mixing models (e.g. Yu and Oldfield 1993; Slattery et al. 2000) and to alternative approaches such as cluster analysis (e.g. Maher et al. 2009).

The second main approach is the incorporation of a magnetic tracer to the soil, whose concentration can be determined from measurements of its magnetic properties. In the case of incorporated magnetic tracers, these studies have been made at small scales (from 1 to 522 m^2). Parsons et al. (1993) applied crushed magnetite in selected transects perpendicular to the longitudinal slope on a 18 m wide× 29 m long plot during rainfall simulations to study interrill sediment transport, noticing potential problems due to the different density of the tracer compared to soil aggregates. Ventura et al. (2002) applied magnetic beads which were encased in resin over the soil surface to study detachment and deposition on a 4-m^2 plot under simulated rainfall. The magnetic tracer allowed the identification of net detachment and deposition areas. However, the quantification of erosion rates would need a wider range of sizes and densities of the tracer as a preferential sediment sorting was observed, which enriched the tracer concentration in the sediment. Hu et al. (2011) identified areas of deposition or detachment using simulated rainfall and inflow experiments to simulate the interrill and rill components of soil erosion. For that purpose, they developed five combinations of fine soil,

Table 4 Summary of magnetic soil properties measured in erosion tracers in soils

Property	Brief description
Low frequency magnetic susceptibility χ_{LF}	Magnetic susceptibility is the ratio between the magnetization of the material and the magnetic field strength under which is measured. In this case at low frequency (0.47 kHz)
High frequency magnetic susceptibility χ_{LF}	Magnetic susceptibility at high frequency (4.7 kHz)
Isothermal remanent magnetization IRM	Isothermal remanent magnetization is the remanence left in the sample after a steady field (1–1000 mT) has been applied for a short time (e.g. 100 s) and then switched off at a constant (isothermal) temperature, often room temperature. It is classified depending on the strength of the field, e.g. soft (0–20 mT), mid (20–300 mT), hard (300–1000 mT)
Saturated isothermal remanent magnetization $SIRM$	It is the maximum IRM
Anhysteretic remanent magnetization ARM	Magnetization acquired by the combined effects of a large alternating field and a small DC field

magnetic powder and fly ash with cement or bentonite, three of which could not move in phase with soil. Most of these studies measured the magnetic susceptibility of soils ($m^3 kg^{-1}$), which can be determined in the laboratory or directly in the field.

Guzmán et al. (2010) used silt-sized magnetite mixed by serial dilutions with four different soils. They carried out rainfall simulations in 1 m^2 boxes to study the behaviour of the magnetite as an estimator of soil losses, measuring the variations of magnetic susceptibility before and after the simulations. This approach identified the need to correct for differences in soil bulk density after the simulations and the selective transport of the finer soil fraction that tended to be enriched in tracer using:

$$\Delta\chi = \left[\left(\frac{\chi_0 - \chi_f}{\chi_0} \right) - \left(\frac{\chi_0 - \chi_0 f_c}{\chi_0} \right) \right] \frac{1}{f_s} \tag{4}$$

Where χ_o and χ_f are the net magnetic susceptibility of the soil before and after the rainfall simulation and f_s represents an increase of the magnetic susceptibility of the sediment compared to the original tagged soil due to the possibility of selectivity in the transport of the tracer with sediment. f_c is the ratio of the initial bulk density divided by the bulk density at the sampling time. One limitation of this model is the upper limit of f_c, is equal to one, which considers the decrease of bulk density of expansive soils after the rainfall. As with previous tracing approaches, magnetic tracers also require the use of a conversion model to calculate total soil losses, based on the assumption that losses of net magnetic susceptibility, and therefore, losses of the tracer are proportional to soil losses S:

$$S = \frac{w}{a} \Delta\chi \tag{5}$$

Where w is the weight of the tagged layer before the simulations and a is the tagged area

A recent approach has been proposed by Armstrong et al. (2010), who enhanced soil magnetic properties through intense and localized heating. This could be useful at small scales where areas do not present significant differences in natural magnetic properties. Magnetism is also commonly used as a tracer in the study of sediment dynamics following wildfire, as fire has been shown to enhance the magnetic signature of soil allowing for the discrimination between burnt and unburnt areas (e.g. Blake et al. 2006).

2.4 Other tracers

A final set of erosion tracer studies are those identified as "others", grouping alternative approaches that have been, or are being, developed as potential soil tracers. Most of them are preliminary studies at a small scale and found only in a few instances in the literature review.

Young and Holt (1968) used fluorescent glass particles. Their bulk density (2.6 gcm^{-3}) and size (44–125 μm) were similar to that of the soil mineral particles. They were observed with UV light after rainfall simulation experiments at plot scale (4 m×10 m). Plante et al. (1999) developed laboratory scale ceramic tracer spheres with a 13 % Dysprosium concentration that could be quantified using instrumental neutron activation analysis. Schwertmann and Schmidt (1980) described the use of Cu applied to the soil as fungicide as a tracer to determine long-term soil erosion in a way that resembles some of the early applications of ^{137}Cs. Another alternative has been the use of sediment material segregated by grain size and labelled with gold and silver after immersion in a solution with these elements; their concentration is then measured using instrumental neutron activation analysis (Olmez and Pink 1994). Wheatcroft et al. (1994) used this approach in a preliminary study of sediment movement in Massachusetts Bay, USA. Sharma et al. (2009) presented preliminary results of a polymer microsphere tracer. Each microsphere was coded with a unique DNA sequence, of which there are essentially limitless combinations, and can be measured using real-time polymerase chain reactions. These polymers are also relatively short-lived, which can be an advantage in some experimental conditions. At the laboratory scale, Mentler et al. (2009) presented the first results of the potential use of organophilic clays as sediment tracers in erosion studies, obtaining extraction efficiencies of about 85 %, and Spencer et al. (2011) presented the results of a field experiment using holmium-labelled montmorillonite to track fine sediment in urban water management systems.

3 Fingerprinting studies

The second largest group of tracing studies in Fig. 1 is grouped under the generic name of fingerprinting studies. These are discussed as a separate group because this approach is focused on identifying the source of sediments. These studies are based on comparing the composition, for a given set of soil properties (considered as natural tracers), of the sediment collected at the end of a catchment with the soil properties from different areas within the catchment. Using this approach, it is possible to identify what fraction of the lost sediment at the catchment outlet or deposition reservoir comes from each area of the catchment. A large number of different properties (physical, chemical, or biological) have been used in fingerprinting studies, and these are summarized in Table 5. The number and type of properties used in specific studies varies, depending on the catchment characteristics, objectives of the study, and availability of analytical capability.

Some studies use this approach at a small scale with a small number of properties. Bellanger et al. (2004) identified and monitored sources of soil organic matter in runoff water in

three agricultural plots (30 m²) using the ratio of $\delta^{13}C$ and $\delta^{15}N$ isotopes as discriminating properties. Fox and Papanicolaou (2007) used the same properties to distinguish between sediment coming from upland rill∕interrill erosion and floodplain headcut erosion in a small agricultural catchment (0.71 km²). For larger areas, as in the study of Russell et al. (2001), who established the origin of suspend sediment within two agricultural catchments (<4 km²), a larger number of determinants were selected—between 8 and 12 depending on the study catchment and the objective. Most of these studies have been developed at the large basin scale where differences in soil properties are easier to establish. The study of Rhoton et al. (2008) discriminated sediment sources of a 150-km² catchment from six different sub-catchments using nine physical properties, 15 chemical properties, a stable carbon isotope ($\delta^{13}C$) and radionuclides (^{137}Cs, ^{40}K, ^{226}Ra) measured in soil samples and sediment.

Soil and sediment sampling design depends on the study catchment, the nature of the properties selected and particular conditions of each study and requires the use of geostatistical techniques and variograms to optimize sample numbers (McBratney and Webster 1983). Small et al. (2002) described the principle sources of uncertainty within fingerprinting studies and developed a Bayesian Monte Carlo-based methodology to assess the importance of sampling design.

Viscarra Rossel et al. (2006) provide a review of studies using NIR-Vis and other spectroscopic techniques that demonstrate their potential for soil and sediment characterization. For instance, Cañasveras et al. (2010) used diffuse reflectance spectroscopy for categorising soil zones through soil aggregate stability estimations. Similarly, Martínez-Carreras et al. (2010a, b) used a spectra-reflectance-based fingerprinting approach for documenting suspended sediment sources during storm runoff events in an agricultural catchment in Luxembourg, although these authors found that the confidence associated with the source tracing results decreased with increasing spatial scale (Martínez-Carreras et al. 2010c).

Potential fingerprinting properties should be measurable in both the source area and in sediment and should be conservative between sediment generation and delivery (Mukundan et al. 2010). In some studies, the properties used to discriminate the sediment source are decided *a priori*, and subsequently reduced following a statistical analysis (Fox and Papanicolaou 2007). This process is described well by Collins and Walling (2002) and Walling (2005) and has been used with minor modifications by several researchers (e.g. Minella et al. 2008). In the first stage of this approach, the null hypothesis that the source material samples are drawn from the same population is tested. To maximize the discrimination between the sources, while minimizing the size of the property subset, a second stage of statistical analysis, based on a stepwise multivariate discriminating function, is used to select the optimum set of fingerprint properties from those identified in the first stage. From this analysis it is also possible to establish the associated uncertainties with the fingerprint property values used to characterize each source. The final step is to estimate the relative contribution of each source to the sediment samples collected at the catchment outlet. This is usually done using a multivariate mixing model:

$$y_i = \sum a_{is} P_s \qquad (6)$$

Where y_i is the concentration of the element i in the suspended sediment sample, a_{is} is the concentration of the element i in source s and P_s is the relative contribution of source s. This model assumes that the suspended sediment retains the characteristics of its source and that the suspended sediment comprises material only from the identified sources, so $\Sigma P_i = 1$, and $0 < P_s < 1$. Examples of applying such mixing models are described by Yu and Oldfield (1989) and Walling and Collins (2000). The model in Eq. (6) is over-determined and it must be fitted iteratively by minimizing an objective function. These approaches have been applied to different samples, both intra-event and multi-event. Solution of Eq. (6) is not unique and its equifinality has been explored to provide an estimation of the uncertainty of sediment source predictions. For instance,

Table 5 Summary of soil properties measured in some of the fingerprinting studies of Fig. 1

Class	Soil and/or sediment properties
Chemical[a] (Concentration)	Organic C, Inorganic C, Total C, C/N, pH, Extractable Ca, Extractable Mg, Extractable K, Extractable Na, Al_p, Al_o, Al_d, Fe_p, Fe_o, Fe_d, Mn_p, Mn_o, Mn_d, $\delta^{13}C$, $\delta^{15}N$, Total Si, Al, Ag, Bi, Cd, Cr, Hg, Fe, Ca, Mg, Mn, Na, K, Ti, P, Zn, Sr, Pb, Ni, Cu, As, Mo, Sn, U, Pb, Sb, Sn, Inorganic P, Organic P, Total P, ^{204}Pb, ^{206}Pb, ^{207}Pb, ^{208}Pb
Physical	Sand, clay and silt fraction, Water dispersible clay, Aggregation index, Frequency dependent magnetic susceptibility (0.47 y 4.7 KHz), ^{137}Cs, ^{40}K, ^{226}Ra, Unsupported ^{210}Pb, Anhysteretic remanent magnetization (ARM), Isothermal remanent magnetization (SIRM and IRM at −0.1 T), Infrared spectroscopy
Biological	Sterol rations, *E. coli*, *Enterococci* bacterial signatures

[a] Subscripts $_p$, $_o$, and $_d$ denote pyrophosphate, oxalate, and dithionate extractable, respectively

Rowan et al. (2000) provided an analysis using the GLUE procedure to explore all the possible solutions of the multivariate mixing model. The results of this study provided a means to distinguish between the different contributions of each sediment source in probabilistic terms.

Mukundan et al. (2010) followed a multi-step process minimizing Wilks' lambda to separate the best fingerprints to differentiate the sediment sources. The multivariate mixing model of Collins et al. (1998) was subsequently used to assess the source contribution to the suspended sediment. Mukundan et al. (2010) checked the results of the multivariate mixing model with the help of the end-member mixing analysis (EMMA, e.g. Christophersen et al. 1990), generally developed through principal component analysis with conservative tracers. Both models gave similar results in the Mukundan et al. (2010) study. The introduction of a Bayesian Monte Carlo Markov Chain (MCMC) solution scheme in the un-mixing model of Fox and Papanicolaou (2008b) adds a multivariate normal parameterization to the sampling process to account for the episodic character of erosion, which occurs at different rates in different parts of the catchment. The MCMC method allows the posterior distribution of the parameters to be determined (e.g. Press et al. 2007). As Fox and Papanicolaou (2008b) remarked, the introduction of the un-mixing model offers a promising way to analyse non-equilibrium erosion processes. One of the challenges in these kinds of studies is to account for the selectivity in the sediment transport process and its impact on the concentration of the measured soil property in the sediment. A common strategy is to analyse the selected properties in the fine fraction of the soil and sediment, using the <53 to <63 μm fraction (Rowan et al. 2000; Fox and Papanicolaou 2007). Sometimes the screening is limited to a coarser fraction, as in the case of Rhoton et al. (2008), in which the fraction <2 mm was used. An additional step to correct for the selectivity in sediment transport, and the bias introduced into the comparison of soil and sediment tracer concentrations, is correcting by the ratio of the specific surface area and organic matter content of the suspended sediment and that of the potential source material (e.g. Collins et al. 1997, 1998). However, some authors have noted that the relationship between geochemical and radionuclide concentrations and specific surface area is non-linear for specific surface areas >1 $m^2 g^{-1}$, instead proposing relationships that need to be experimentally determined from fractionation of the source material (e.g. Russell et al. 2001).

Biomarkers have also been used as sediment fingerprints. The compound specific stable isotope (CSSI) approach is based on the concept of differences in "land-use", which is typically defined by the plant communities growing on the land. Different plants produce different CSSI signatures (Chikaraishi and Naraoka 2003), such as $\delta^{13}C$ or $\delta^{15}N$. These plant communities label the soil where they grow by exuding organic biomarkers. Using the CSSI approach, the isotopic signatures of soil biomarkers from each sedimentation zone can be used to determine the proportional contribution of each soil source using multi-source mixing models (Phillips and Gregg 2001, 2003; Gibbs 2008). Recently, Nosrati et al. (2011) has proposed enzyme activity as a potential sediment tracer for discriminating sediment sources, although they may show important temporal dynamics, and it seems that new addition tracing methods will probably arise from this rapidly advancing field.

4 Final considerations and challenges for the future

A comprehensive literature review on tracing approaches used in water erosion studies was carried out, describing established tracing approaches and their application to soil erosion research. In addition to these approaches, new and emerging tracing approaches are also mentioned. The FRN approach offers advantages when documenting erosion and deposition rates because: (a) FRN are present worldwide and can provide information on soil redistribution rates and patterns at various temporal scales (there are no scale constraints, apart from the number of samples to be analysed), (b) depending on the FRN approach used, they integrate all processes involving soil particle movements—from a single event up to a period of one century, and (c) erosion/sedimentation estimates are retrospective and can be obtained through a single soil sampling "campaign" and long-term monitoring is not required. Among the major constraints, the following should be considered: (a) accurate selection of the reference site and the determination of its inventory value is vital, (b) the proper application of this approach requires a multi-disciplinary team involving environmental, soil and gamma spectroscopy expertise, and (c) FRN is an indirect approach, therefore a sound understanding of the conversion models and their parameters is needed.

REE tracers have attained significant use in small- and medium-scale erosion studies in the last two decades, providing spatial and temporal information on soil erosion and sediment redistribution. REE have many of the desirable characteristics of an ideal sediment tracer, such as: (a) the possibility of multiple tracers, (b) the strong-binding ability to soil aggregates, and (c) the possibility of being detected at very low concentrations. They thus provide a promising tool to track sediment movement through the landscape. However, their use entails some notable limitations such as: (a) the preferential binding to finer soil aggregates, (b) the incorporation in to the soil profile without disturbing it, and (c) the complexity of the analytical approaches (e.g. ICP-MS after acid extraction), which can present some uncertainty. Nonetheless, even in soils with strong preferential binding across a range of aggregate sizes, the problem can be minimized by either separating the soil used to tag the soil surface into size fractions prior to incubating them with REE (Polyakov et al. 2009), or

conducting the REE analysis after separating the sampled material into sub-samples of aggregate sizes that present relatively homogeneous binding of REE, and including those differences in the calculations of tracer erosion rates (Kimoto et al. 2006b).

Research studies using soil magnetic properties as natural tracers have common issues with fingerprinting studies but have the advantage of simple analytical tools to determine soil and sediment properties. Until now, most of these studies using magnetic properties as tracers have been carried out at large scales (in the case of soil mineral magnetic properties) or at a laboratory/plot scale (for incorporated magnetic tracers). Depending on the magnetic tracer employed: (a) this approach would be useful at a broad range of spatial and temporal scales under different soil management regimes to identify source of sediments, monitor soil movement and estimate soil erosion rates, (b) the use of physical technology to measure the magnetic properties does not require a complex preparation of soil samples and so direct measurements in the field are possible, and (c) this technology allows relatively inexpensive processing of a large number of samples, which facilitates an more accurate mapping of soil magnetic properties. Despite the relative simplicity of using and measuring magnetic properties compared to other tracer measurements, this approach has some drawbacks: (a) some of the incorporated magnetic tracers used show a higher density compared to soil minerals, thus interfering with soil movement; (b) the approach shares some shortcomings with REE, for instance the preferential binding to finer soil particles; and (c) it is impossible to incorporate these tracers into the soil profile in undisturbed soils.

A significant number of new tracers have been developed, or are in development, mostly for use at hillslope or small catchment scales. Their potential as sediment tracers have been evaluated under controlled conditions and most of them are in a preliminary stage of evaluation and currently it is hard to draw conclusions as to their advantages and disadvantages.

Fingerprinting studies combine soil properties with mathematical and statistical methods, allowing the identification and quantification of sediment sources depending on differences in soil properties, especially at large scales where these differences are more apparent. This approach presents several advantages compared to other tracer approaches: (a) it does not present temporal limitations, (b) it avoids many of the operational problems that appear with other sediment tracers, such as the incorporation in to undisturbed soils, and (c) the possibility of using multiple natural sediment tracers to identify sediment sources at large scales. Some limitations of the fingerprinting are that: (a) this approach is well established but requires sophisticated approaches to evaluate the uncertainty of the predictions and complementary methods, such as air photographs, to interpret the results; (b) the variability of bottom sediment properties and possible particle size effects can complicate the differentiation of sources of sediment; and

(c) to obtain consistent results it is important to employ a variety of sediment properties, which could require large amounts of sediment and which is not always easy to obtain. In the future, spectroscopic techniques may also have the ability to provide low cost analytical procedures and thus the ability to process large amounts of soil and sediment samples of small mass, thereby facilitating tracer studies.

Work on radionuclides and fingerprinting studies are well advanced over the other tracing technologies, but still have associated with them significant uncertainties in the calculated erosion and deposition rates, and the attribution of sediment sources. Future research should focus, amongst other issues, on reducing the uncertainty associated with the conversion of tracer concentrations into erosion rates, for which a good understanding of the tracer distribution along the tagged soil profile and its affinity to particular soil aggregate sizes will be key. For new tracers in development, studies already published in fingerprinting and radionuclides might provide significant help. A key line of research for the future is the development of tracers requiring inexpensive and rapid analysis approaches that are able to process quickly a large number of samples. This seems to be feasible using multiple magnetic properties of soils, whether induced or natural, or spectroscopic techniques. Both techniques, allow rapid, non-destructive and quantitative measurements of soil and sediment properties, even in the field. These techniques would allow us to understand the behaviour of the tracer under many different conditions and the important role that spatial variability might play in understanding water erosion. Also important is the development of "ephemeral" tracers that disappear in months or years, as in the case of DNA microsphere spheres, allowing experimental areas to be used for further experiments without the problems caused by contamination with tracers used in previous experiments.

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