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Upscaling the use of fallout radionuclides in soil erosion and sediment budget investigations: Addressing the challenge

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Abstract

The application of fallout radionuclides in soil erosion investigations and related sediment budget studies has provided a widely used tool for improving understanding of soil erosion and sediment transfer processes. However, most studies using fallout radionuclides undertaken to date have focussed on small areas. This focus on small areas reflects both the issues addressed and practical constraints associated with sample collection and analysis. Increasing acceptance of the important role of fine sediment in degrading aquatic habitats and in the transfer and fate of nutrients and contaminants within terrestrial and fluvial systems has emphasised the need to consider larger areas and the catchment or regional scale. The need to upscale existing approaches to the use of fallout radionuclides to larger areas represents an important challenge. This contribution provides a brief review of existing and potential approaches to upscaling the use of fallout radionuclides and presents two examples where such approaches have been successfully applied. These involve a national scale assessment of soil erosion rates in England and Wales based on ^{137}Cs measurements and an investigation of the sediment budgets of three small/intermediate-size catchments in southern Italy.

Key Words: Fallout radionuclides, Caesium-137, Soil erosion, Soil redistribution, Upscaling, Catchment-scale, National scale, Sediment budget

1 Introduction

The potential for using fallout radionuclides to provide information on rates and patterns of soil loss and soil redistribution has now been clearly demonstrated by a wide range of investigations undertaken in many different areas of the world (e.g. IAEA, 2011, 2014; Zapata and Nguyen, 2010; Matisoff and Whiting, 2012; Walling, 2012). The key principle involved is that the fallout reaching the soil surface is rapidly and strongly adsorbed by the surface soil and its subsequent redistribution by erosion processes directly reflects the intensity and spatial distribution of those processes. Most work has focussed on the use of caesium-137 (^{137}Cs), a man-made fallout radionuclide associated with the testing of atomic weapons in the 1950s and early 1960s (see Walling, 1998; Zapata, 2002). Since most of the ^{137}Cs fallout occurred during the period extending from the late 1950s to the early 1970s, this radionuclide now affords a valuable means of documenting soil redistribution over the past ca. 50 years. Particular advantages of the approach include, firstly, the ability to obtain retrospective data on the basis of a single site visit and without the need to install permanent monitoring equipment and structures; secondly, the ability to integrate the impact of all processes resulting in soil redistribution; thirdly, the spatially distributed nature of the data; and fourthly, provision of time-integrated average rates of soil redistribution (see Walling and Quine, 1995; Mabit et al., 2008). The ability to generate spatially distributed data has coincided with a need for such data for validating physically-based distributed soil erosion models (e.g. de Roo and Walling, 1994; He and Walling, 2003; Norouzi Banis et al., 2004). An additional input of Chernobyl-derived ^{137}Cs fallout

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in some areas of Europe and adjacent regions provided further opportunities to exploit the approach for the period since 1986 (e.g. Golosov, 2002). Ongoing developments in using and interpreting ^{137}Cs measurements have also made it possible to document the impact of changing landuse, such as a shift from conventional to minimum-till management practices, on soil redistribution rates (see Schuller et al., 2007). Porto et al. (2014) have also demonstrated the potential for using repeat ^{137}Cs measurements to obtain information on two or more periods of time within the overall time window covered by ^{137}Cs measurements.

Attention has also been directed to the use of other fallout radionuclides. Some of these, including Plutonium-239 and 240 ($^{239-240}Pu$) (e.g. Tims et al., 2010; Alewell et al., 2014), are also the product of the same bomb tests that caused the fallout of ^{137}Cs, but others are natural geogenic or cosmogenic fallout radionuclies, including excess lead-210 ($^{210}Pb_{ex}$) and beryllium-7 (^{7}Be) (see Mabit et al., 2008). Plutonium-239 and 240 have much longer half-lives than ^{137}Cs, which means that it will be possible to use them further into the future, particularly in regions such as Australia where the global pattern of bomb fallout resulted in low fallout inputs. Use of the natural fallout radionuclides $^{210}Pb_{ex}$ and ^{7}Be makes it possible to derive information relating to time windows different from than that associated with ^{137}Cs. Excess lead-210 provides the potential to extend the length of the period considered to ca. 100 years (Walling and He, 1999a; Mabit et al., 2014), whereas ^{7}Be can be used to document soil redistribution associated with individual events or short periods of heavy rainfall and therefore timescales of days to weeks (see Mabit et al., 2008; Walling et al., 2009; Walling, 2013). Conjunctive use of several fallout radionuclides in a single study can potentially provide information on the erosional history of a site, by providing information for different time windows (e.g. Benmansour et al., 2012).

There has recently been some questioning of the reliability of soil redistribution rates derived from fallout radionuclide measurements (Parsons and Foster, 2011), but this would appear to ignore the rapidly growing number of studies that have provided empirical validation of the approach (e.g. Porto et al., 2003a; Belyaev et al., 2008). Furthermore, Mabit et al. (2013) have provided a response to many of the specific criticisms. It is also important to recognise that all other techniques employed to document soil redistribution possess important inherent limitations (Loughran, 1990) and that fallout radionuclides can offer important and essentially unique advantages over other techniques.

To date, most work in applying fallout radionuclides in soil erosion investigations has focussed on relatively small areas, such as individual fields, although there have been some attempts to investigate larger areas (e.g. Loughran et al., 1996; Mabit et al., 2007). In these small-scale studies, relatively large numbers of samples are collected from a field, commonly using a grid or transect sampling strategy, and the resulting point estimates of soil redistribution rate are used to establish the gross and net erosion rate for the field or study area. This focus on small areas has been largely a response to practical constraints associated with collecting and analysing large numbers of samples. Sampling commonly involves collection of one or more bulk cores from individual sampling points and this can prove labour intensive. However, the processing and radiometric analysis of the resulting cores for radionuclide activity by gamma spectrometry can represent an even greater constraint in terms of both the cost and time involved when dealing with large numbers of samples. Count times for an individual sample are generally of the order of 12-24 hours and this will be directly reflected by sample throughput and the cost of analyses. The resulting emphasis on individual fields or small areas can be seen as appropriate for addressing concerns related to on-site impacts of soil loss and soil redistribution and associated issue of soil degradation and reduced productivity, since it provides site-specific data. However, this small scale focus can be seen as less appropriate for addressing other issues.

The need to also consider the offsite impacts of soil erosion linked to the important role of the mobilised sediment in degrading aquatic ecosystems and in the transfer and fate of nutrients and contaminants in terrestrial and fluvial systems is being increasingly recognised (Waters, 1995; Wood and Armitage, 1997; Warren et al., 2003; Owens et al., 2005; Kemp et al., 2011). This necessarily directs attention to net, rather than gross soil loss, to the transfer of sediment from individual fields to streams, and to the wider landscape rather than the field. As emphasised by Walling and Collins (2008) and Gellis and Walling (2001), the catchment sediment budget is increasingly seen as a key tool for providing the understanding of the behaviour and fate of sediment across larger areas required for the design of effective sediment control strategies. It directs attention to the mobilisation transfer and storage of sediment within the catchment and the relationship between the sediment output and internal sources and sinks. Fallout radionuclides can provide an invaluable and essentially unique source of information on rates of soil loss associated with sheet and rill erosion and the subsequent redistribution and transfer of that sediment within the landscape.

The use of fallout radionuclides to provide information on mobilisation and redistribution of sediment within the wider landscape and on the sediment budget of a catchment necessitates an up-scaling of the approach. It is essentially impossible to extend the detailed sampling commonly applied to an individual field across an entire landscape or anything but a very small area or catchment, due to the cost and other practical constraints mentioned above. As a result, there is a need to develop new approaches that permit such up-scaling without introducing insuperable obstacles. Such approaches could involve use of different sampling or measurement techniques in order to facilitate data collection from larger areas, optimising the information content of a small number of measurements, extrapolating or interpolating the data obtained from a small number of measurements over a much larger area, or the use of reconnaissance type sampling programmes aimed at sampling the total population, These potential approaches are discussed further below.

2 Approaches to upscaling: A brief overview

2.1 *Modifying sampling techniques and laboratory measurements*

Most existing studies that have used fallout radionuclides to document soil erosion or soil redistribution rates are based on the collection of bulk cores from individual sampling points and analysis of the radionuclide inventories of those cores by gamma spectrometry. Field sampling can prove laborious and time-consuming if a large number of cores is to be collected from a wide area. Use of motorised, percussion-coring, equipment can facilitate core collection, but the size and weight of such equipment can introduce problems for its use in remote or difficult terrain. There is limited scope to modify sampling techniques, if these continue to be based on the collection of bulk cores from individual sampling points, although carefully designed sampling strategies that involve combining several cores to permit measurement of the mean inventory for those cores could reduce the number of laboratory measurements required (see below). As indicated above, the long count times (e.g. 12-24 hours) commonly associated with laboratory gamma spectrometry measurements necessarily introduces an important constraint on the number of samples that can be analysed. Improvements in detector design and the development of alternative measuring techniques could potentially increase the number of samples that can be analysed, but no major change would seem to be on the horizon. Recent interest in the use of $^{239-240}$Pu as an alternative to ^{137}Cs has provided an opportunity to use alternative analytical techniques. In this case both ICPMS (e.g. Alewell et al., 2014) and AMS (e.g. Lal et al., 2013) have been successfully used to measure the radionuclide activity. However, sample preparation requirements and analytical costs mean that the potential for analysing large number of samples remains limited. Similar issues exist with alternative methods for analysing ^{210}Pb$_{ex}$ (see Mabit et al., 2014).

One approach which might be seen as offering potential to 'revolutionise' sample collection and measurement is the use of field portable in-situ gamma detectors. In this case the detector is taken into the field and in-situ measurements are made at individual sampling points, thereby avoiding the need to collect samples or return samples to the laboratory (e.g. He and Walling, 2000). In addition, the field of view of an in-situ detector commonly means that the data collected will relate to a much larger surface area (e.g. a circle of 1 m diameter) than that associated with conventional manual samples. This can reduce problems associated with obtaining a representative core sample due to the small-scale spatial variability of soil inventories. However, although potentially promising, this approach has to date fallen short of providing a clear proven alternative to traditional field sampling and laboratory analysis. Virtually all studies using this approach have focussed on ^{137}Cs, since, despite suggestions to the contrary by Li et al. (2010), self absorption effects introduce important problems for ^{210}Pb measurements and ^{7}Be inventories are often too low. Early work by He and Walling (2000) in the UK indicated that the required count times at individual measuring points were still significant (e.g. 5000 s), meaning that the total time required to make measurements at a large number of points could be substantial. Furthermore, the requirement for liquid nitrogen for the germanium detector cryostat could represent a major constraint in remote areas. More success has been achieved in areas that received high amounts of Chernobyl fallout, since activities and inventories can be several orders of magnitude higher than those associated with bomb fallout and the high activity associated with ^{137}Cs means that count times are greatly reduced (e.g. 100 s). Furthermore, potential measurement problems introduced by interference from other radionuclides with similar energy levels are greatly reduced in such areas, due to the dominance of the ^{137}Cs peak. It is therefore possible to use a simpler and cheaper scintillation (NaI) detector, that does not require liquid nitrogen for cooling, as an alternative to a conventional higher resolution Ge detector (See Golosov et al., 2000). Recent field trials of an in-situ lanthanum bromide ($LaBr_3$) scintillation detector undertaken by the IAEA in Austria reported by Gonsalves et al. (2014) would appear to offer considerable promise for use in areas where the ^{137}Cs signal reflects bomb fallout. Such detectors are relatively inexpensive, they are lightweight and highly portable, they do not require cooling and have sufficient resolution to resolve the ^{137}Cs signal. However, a count time of 900 s was used in the study reported and this could still represent a significant limitation in attempting to collect data from a large number of measurement points.

A further limitation of the measurements obtained from in-situ detector is that they reflect the gamma decay signal received from the soil within the field of view of the detector and this will in turn be influenced by the depth distribution of the radionuclide in the soil and attenuation of the gamma decay signal by the soil. Calibration of a detector to provide estimates of the total radionuclide inventory in the soil requires information on the depth distribution, which may not be known in detail, and must also take account of soil properties, including moisture content. Calibration problems are likely to introduce considerable uncertainty into the resulting estimates of radionuclide inventory and thus restrict their potential use.

2.2 *Optimising information return from a reduced number of samples or laboratory measurements*

An alternative approach to modifying the techniques used for sample collection and analysis to facilitate dealing with a larger area could involve modifying the sampling strategy, in order to generate the required information from a reduced number of samples or measurements. Chappell & Warren (2003) indicate how geostatistical analysis can be used to inform the design of the network of sampling points, particularly the spacing of sampling points, to maximise the information provided by a limited number of samples. Alternatively, the number of samples requiring analysis can be considerably reduced by compositing (bulking) individual samples and analysing only the composite samples. The authors have employed this strategy when sampling fields characterised by planar slopes (i.e. no curvature along the contour). A sampling grid orientated parallel to the slope is established and the samples collected along individual grid lines orthogonal to the slope are composited. The resulting composite samples effectively reflect a single downslope transect, which is representative of the sampled field. Use of a similar procedure is reported by Schuller et al. (2003). Using this approach, the number of samples requiring analysis could be reduced by ca. 90%, although the sampling effort is not reduced and the need to homogenise the composite sample is likely to add significantly to the task of sample processing. A more radical alternative could conceivably involve compositing all the samples from a given field and analysing a single sample. However, this approach is likely to be of limited value, due to the problems of interpreting the final single measured inventory, if the field is characterised by areas of both erosion and deposition, and the non-linear nature of the relationship between soil redistribution rate and the radionuclide inventory.

Walling et al. (2010) describe the use of another strategy which falls within this general approach. This reduces both the number of samples collected and the number of samples requiring analysis. This is applicable to fields in areas of simple topography (i.e. planar slopes). The authors demonstrated that a single carefully positioned downslope transect comprising ca 8-12 samples can provide a reliable estimate of the gross and net erosion rate within a given field and therefore remove the need for detailed transect or grid-based sampling. Loughran et al. (2004) used a similar strategy as the basis for a national soil erosion survey undertaken in Australia. In this case the sampling strategy was underpinned by a generalised model of the variation of soil redistribution rates along a typical slope (Loughran et al., 1989). By collecting samples from 20 pre-selected points along a representative downslope transect it was possible to characterize the gross and net erosion rate for the site. A total of 206 sites representative of different regions of Australia was sampled to provide information on typical soil redistribution rates across the country. These sites comprised 100 on cultivated land, 52 on uncultivated pasture and forest and 54 on rangeland.

2.3 *Extrapolation and interpolation techniques*

Extrapolation techniques have been used by a number of researchers to permit evaluation of soil redistribution rates within large areas, based on a limited number of samples/laboratory measurements. The basis of this approach is commonly to subdivide the study area into a number of homogeneous units and to undertake ^{137}Cs measurements at representative locations (e.g. fields) within each unit. If the gross and/or net erosion rate is obtained for each representative location (field) within a given unit, using grid or transect sampling, and these values are averaged, the result can be extrapolated across the entire unit. A good example of the use of this approach is provided by the work of Mabit et al. (2007) in the 217 km^2 Boyer River watershed in Quebec, Canada. In this study, the homogeneous units were termed isosectors and the study watershed was subdivided into 6 isosectors. Five of the isosectors were characterized by different combinations of soil texture and slope steepness and forest areas were identified as an additional isosector. Within each of these five isosectors, at least three representative fields were sampled and the mean erosion rate for each isosector was established and assumed to be representative of the unit. In total 24 fields were sampled. Fourteen sites were sampled within the forested area where erosion was assumed to be minimal, to establish the reference inventory. A GIS was used to facilitate the extrapolation of the results for each isosector across the watershed. Other studies that have used this general approach have involved study areas of different sizes, have employed different criteria to define the homogeneous units and have varied the total number of units used (see Table 1). Land use and geology have frequently been included in the criteria. It is clearly important to ensure that the units are relatively homogeneous in terms of soil redistribution rates and that the values of soil redistribution rates obtained for a given unit are representative of that unit. GIS tools clearly offer considerable potential when applying extrapolation procedures.

A variant of this approach which recognises that homogeneous units may still incorporate considerable variability is reported by Walling et al. (2002). In this study, attention focussed on two small catchments in the UK, the Rosemaund catchment (1.5 km^2) in Herefordshire and the Smisby catchment (3.6 km^2) in Derbyshire. Several fields within each catchment (8 in the Rosemaund catchment and 10 in the Smisby catchment) were selected to be representative of different land use and the range of slope gradients and lengths. These fields were sampled using multiple transects and estimates of gross and net erosion rates were derived for each field from the associated ^{137}Cs measurements. These results were extrapolated to the remaining unsampled fields using a topographically driven soil erosion model. This grid-based GIS based model was applied to the DEM of each catchment to generate relative values of gross and net erosion for each field. These relative values were calibrated against the estimates of gross and net erosion rate obtained for the sampled fields, enabling estimates of gross and net erosion to be obtained for each field within the two catchments. This calibration was only applicable to fields under arable or temporary grass and erosion rate estimates were obtained for fields under permanent grassland by directly extrapolating the values provided by the ^{137}Cs measurements for this land use. This approach could clearly be developed further by using a relatively small number of ^{137}Cs measurements undertaken at carefully selected representative locations within a study area to calibrate a soil loss model and employing that model to establish the spatial distribution of soil loss across that area. An example of this approach, involving the national-scale application of a typology model, is provided in section 3.1.

Table 1 Examples of the criteria used for defining homogeneous units or isosectors when extrapolating a limited number of ^{137}Cs measurements to a larger area

Source	Study location	Size of study Area (km^2)	Key criteria for defining homogeneous units	Number of Units
Mabit et al. (2007)	Boyer Watershed, Quebec, Canada	217	Soil texture, slope, forest cover	6
Walling et al. (2006)	Pang and Lambourn Catchments, Berkshire, UK	166 and 234	Land use – cultivated and pasture	2
Walling et al. (2001)	Kaleya Catchment, Zambia	63	Land use – communal cultivation, commercial cultivation, bush grazing	3
Blake et al. (2009)	Blue Gum Creek, subcatchment, NSW, Australia	0.89	Landscape (topographic) units	5
Estrany et al. (2012)	Can Revull catchment, Majorca, Spain	1.03	Land use, topography	1
Estrany et al. (2010)	Na Borges catchment, Majorca, Spain	319	Slope gradient, land use, soil conservation practices	2

The use of geostatistical techniques or spatial statistics can be viewed as another approach for extrapolating a limited number of measurements across a larger area. Here the emphasis is on the spatial structure of the data rather than the influence of key physical controls such as soil type or land use. The sampling undertaken within the study area aims to define the variogram, which characterizes the spatial variability of ^{137}Cs across the area. This variogram is used to weight the influence of surrounding values when estimating values for unsampled locations (see Chappell et al., 1998). The approach can be combined with kriging and could be seen as involving interpolation rather than extrapolation *sensu strictu*. Additional information involving physical controls, such as that provided by remote sensing could be used to improve the accuracy and precison of the estimates (Chappell, 1998). A useful example of the approach is provided by the work of Chappell and Warren (2003) within an area of arable cultivation in Suffolk, England. Here the emphasis was on wind erosion, but ^{137}Cs was used to quantify medium-term rates of net soil redistribution. A total of 181 soil samples was collected from the 19 km^2 study area for ^{137}Cs analysis and geostatistical techniques were used to generate a map of ^{137}Cs inventories across the study area, which provided the basis for mapping soil redistribution rates. Chappell et al (2011) applied similar geostatistical techniques to produce maps of soil redistribution rates for Australia, based on a limited number of samples. The study was based on the ^{137}Cs data assembled by Loughran et al. (2003) from about 200 sampling sites located across the country. Because the estimates of soil redistribution rates obtained for the individual sites based on the sample transects showed no spatial structure, data analysis focussed on the ^{137}Cs inventories documented at the sites. These were used in combination with soil classification data and an existing national scale geostatistical analysis of ^{137}Cs reference inventories to produce national maps of soil redistribution rates.

2.4 *Sampling the total population*

As indicated above, extrapolation procedures based on the ability to define homogeneous units or isosectors within a study area may be of limited value in larger area characterized by considerable spatial variability of terrain characteristics and land use. A somewhat different approach to upscaling the use of fallout radionuclide measurements to investigate soil redistribution rates across larger areas that aims to address this problem has been proposed by Porto et al. (2009). This approach is based on simple sampling theory and assumes that the slopes of a catchment or larger area can be represented by an infinite number of points each characterized by a soil redistribution rate. The gross or net rate of soil loss from those slopes could in theory be calculated by integrating the soil redistribution rates for the total population of points. If a representative sample of those points is obtained, this can generate a frequency distribution of soil redistribution rates which will provide information on the magnitude of erosion and deposition rates and the frequency of occurrence of erosion and deposition rates of different magnitude, as well as the relative incidence of erosion and deposition within the area under investigation. Porto et al. (2009) successfully tested this approach in the 31.6 km^2 Trionto catchment in Central Calabria, Italy. Replicate samples, which were bulked to provide a single sample from each sampling point, were collected from 128 locations within the catchment. Unlike the extrapolation techniques outlined above, the primary product of this approach is a frequency distribution of soil redistribution rates for the study area, rather than information on the spatial distribution of soil redistribution rates and controlling factors. However, such information represents a key requirement for establishing a catchment sediment budget. To be representative of the overall catchment, the sampling points would need to be distributed fairly uniformly across the catchment, although the precise location of the sampling points would need to be random to avoid bias towards particular slope positions etc. Key issues to be addressed in designing such a sampling network clearly include the number sampling points required to provide a meaningful estimate of the mean rate of gross or net soil loss. This could be informed by calculation of standard error statistics.

The authors have recently been exploring further the potential for up-scaling the use of fallout radionuclides, and particularly ^{137}Cs measurements, to larger areas, in terms of both region-wide soil erosion assessments/ inventories and establishing catchment sediment budgets. Details of two such studies are provided here to demonstrate different approaches and objectives. The first focuses on the use of ^{137}Cs measurements to provide the data required for a national-scale soil erosion inventory and involves the use of extrapolation procedures. The second involves the use of reconnaissance sampling of ^{137}Cs inventories to assemble information on the sediment budgets of several small to medium scale catchments in Southern Italy.

3 Some examples

3.1 *A national-scale soil erosion inventory*

The national soil erosion inventory study briefly reported here was funded by the UK Department for Environment, Food and Rural Affairs (DEFRA). It focussed on agricultural land in England and Wales and aimed to use ^{137}Cs measurements to provide national-scale information on the magnitude of soil erosion rates and the sustainability of the soil resource, as well as the importance of soil erosion from agricultural land as a sediment source and a cause of off-site problems such as degradation of aquatic ecosystems. In the absence of a national network of erosion plots or other ongoing soil erosion monitoring, ^{137}Cs measurements were seen as a valuable means of assembling infornation on mean annual rates of soil loss over the past ~50 years and the influence of topography, soil type, land use and other controlling factors on those rates. The approach involved using a limited number of ^{137}Cs measurements to assemble representative information on rates of soil loss in different parts of England and Wales in response to the influence of the key controlling factors and extrapolating those data across England and Wales to provide a national soil erosion inventory. The approach taken and the results are described in more detail by Walling and Zhang (2010). The study involved three key elements. Firstly, preliminary fieldwork was undertaken to confirm that the collection of a relatively small number of bulk cores (e.g. 10 — 12) from a field with a simple topography, using a single representative transect, could provide a reliable estimate of the longer-term (50 year) gross and net soil loss from that field using standard conversion models for cultivated and uncultivated soils (e.g. permanent pasture) (Walling et al., 2002, 2011). This was confirmed. Secondly, by sampling a substantial number of such fields, representative of a wide range of slope steepness and form, soil texture, land use, and hydrometeorological conditions (e.g. mean annual rainfall and hydrologically effective rainfall), and obtaining reliable estimates of the reference inventory for these fields, it was possible to assemble information on the range of soil erosion rates found on agricultural land in England and Wales and their key controls. Thirdly, these data were used to populate a typology model which was employed to extrapolate the available estimates of erosion rates over the entire area of agricultural land in England and Wales.

Within the national sampling programme, attention was restricted to areas of agricultural land below an altitude of 300 m. National Parks, urban areas and areas of minimal slope (<1º), where water erosion was likely to be negligible, were excluded. In total, 248 fields were sampled, of which 133 were arable fields and 115 were under permanent pasture. Fig.1 indicates the locations where fields were sampled. At each location, several

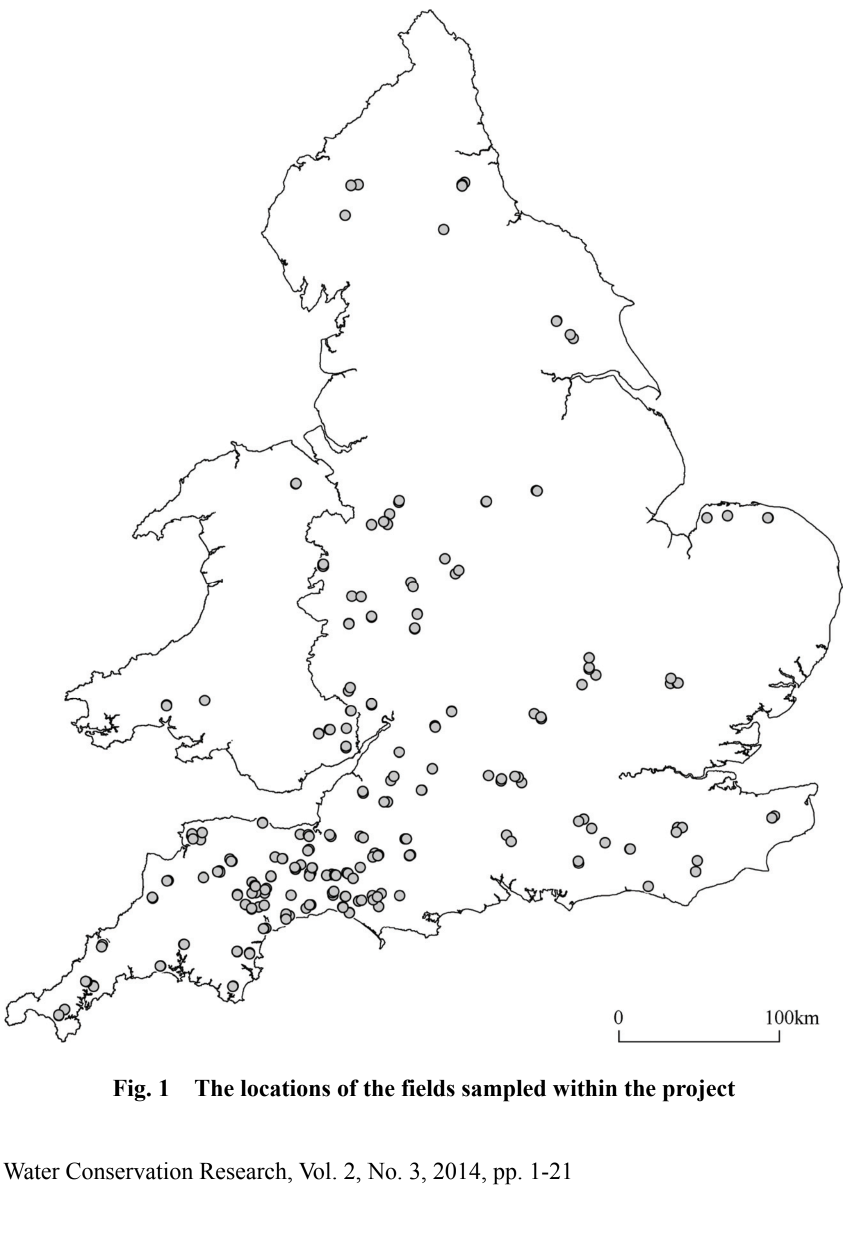


Fig. 1 The locations of the fields sampled within the project

individual fields with different features (e.g. land use) were sampled. The range of terrain characteristics, including relief, slope steepness and convexity, length, soil texture and organic matter content and hydrological effective rainfall (HER), covered by the sampled fields is summarised in Table 2. The ^{137}Cs measurements undertaken on the cores collected from the 248 fields were used to derive estimates of several measures of the longer-term (ca. 50 year) mean annual soil erosion rate, using the conversion models referred to above. These measures included the proportion of the field that was eroding, the erosion rate within the eroding area and the gross and net erosion rates for the field. A summary of the information on soil erosion rates provided by the 248 fields is presented in Table 3. Erosion rates associated with areas of arable cultivation are considerably greater than those for pasture areas. The ratio of net to gross erosion, which provides a measure of the efficiency of sediment transfer out of the field also differed between these two categories of land use and was typically >0.6 for arable fields and <0.6 for pasture.

Table 2 **A summary of the characteristics of the sampled fields**

Arable Fields	Field relief (m)	Maximum slope (degrees)	Convex slope* (degrees)	Average slope (degrees)	Field length (m)	Convexity ratio**	Clay content (%)	Silt content (%)	Sand content (%)	Organic C content*** (%)	HER*** (mm)
Minimum	7.9	4.5	3.0	2.8	73	0.3	4.7	7.4	4.7	0.8	137
Maximum	57.8	18.9	14.9	13.7	524	1.0	50.6	80.7	83.2	31.3	778
Standard Dev.	10.6	3.0	2.5	2.2	83	0.2	10.6	15.5	19.2	5.2	134
Average	24.5	10.1	7.7	7.2	207	0.6	23.9	39.9	36.2	4.3	371
Median	22.3	10.1	7.6	7.0	198	0.5	23.3	38.1	30.8	3.0	355
C.V. (%)	43.3	29.8	32.2	30.5	40	32.5	44.4	38.8	52.9	121.4	36
Pasture fields	**Field relief (m)**	**Maximum slope (degrees)**	**Convex slope* (degrees)**	**Average slope (degrees)**	**Field lengh (m)**	**Convexity ratio****	**Clay content (%)**	**Silt content (%)**	**Sand content (%)**	**Organic C content*** (%)**	**HER*** (mm)**
Minimum	5.7	6.6	5.2	4.8	68	0.4	4.3	6.8	3.8	1.3	160
Maximum	48.0	23.4	20.7	19.8	399	1.0	66.5	80.7	88.9	19.8	880
Standard Dev.	9.6	4.2	3.7	3.3	58	0.2	10.7	15.6	18.9	2.9	149
Average	25.1	13.3	10.7	10.0	151	0.6	25.2	43.1	31.8	4.2	427
Median	24.3	13.0	10.2	9.3	143	0.6	24.3	44.3	26.8	3.6	392
C.V. (%)	38.5	31.3	34.3	32.6	38	30.0	42.7	36.3	59.5	70.1	35

* slope gradients from convex sections of a transect were averaged

** the lengths of the convex sections of a transect were summed and divided by the total length of a transect

*** interpolated values from existing national datasets

Table 3 **Summary statistics for the erosion rates for the sample of 248 fields (t ha^{-1} year^{-1})**

	Arable			Pasture		
	Eroding area rate	Gross erosion rate	Net erosion rate	Eroding area rate	Gross erosion rate	Net erosion rate
Minimum	1.2	1.2	0.0	0.7	0.5	0.0
Maximum	30.3	29.3	27.3	16.0	13.0	11.7
Average	10.6	8.4	6.7	3.2	2.5	1.8
Median	8.8	6.6	5.2	2.6	1.8	1.2
1st quartile	6.5	4.7	2.0	1.7	0.9	0.2
3rd quartile	13.3	11.2	9.1	3.8	3.0	2.6
CV * (%)	80	99	140	82	118	200

* CV was calculated as: (Q1-Q3) / median × 100

A schematic representation of the typology model used to extrapolate the assembled data across the entire study area in England and Wales area is shown in Fig. 2. This was based on a detailed analysis of the key variables controlling the spatial variability of soil erosion rates at the national scale. The final model involved three primary variables, i.e. slope steepness, soil texture (% clay, silt and sand) and land use, and provided estimates of both gross and net erosion for individual grid cells. This model was coupled with a GIS, comprising 1 km×1 km grid cells and incorporating a 50 m DEM (Digital Elevation Model) for use in deriving information on slope steepness, as well as information on soil texture and land use for each grid cell. The extrapolation procedure permitted estimation of the likely proportion of cultivated and pasture land in each grid cell and the mean gross erosion rate within a grid square reflecting these proportions. An estimate of the net erosion rate within each grid square was derived from the values of gross erosion using the empirical relationships between net and gross erosion rates established for arable and cultivated fields using the data obtained from the 248 sampled fields (see Fig. 3).

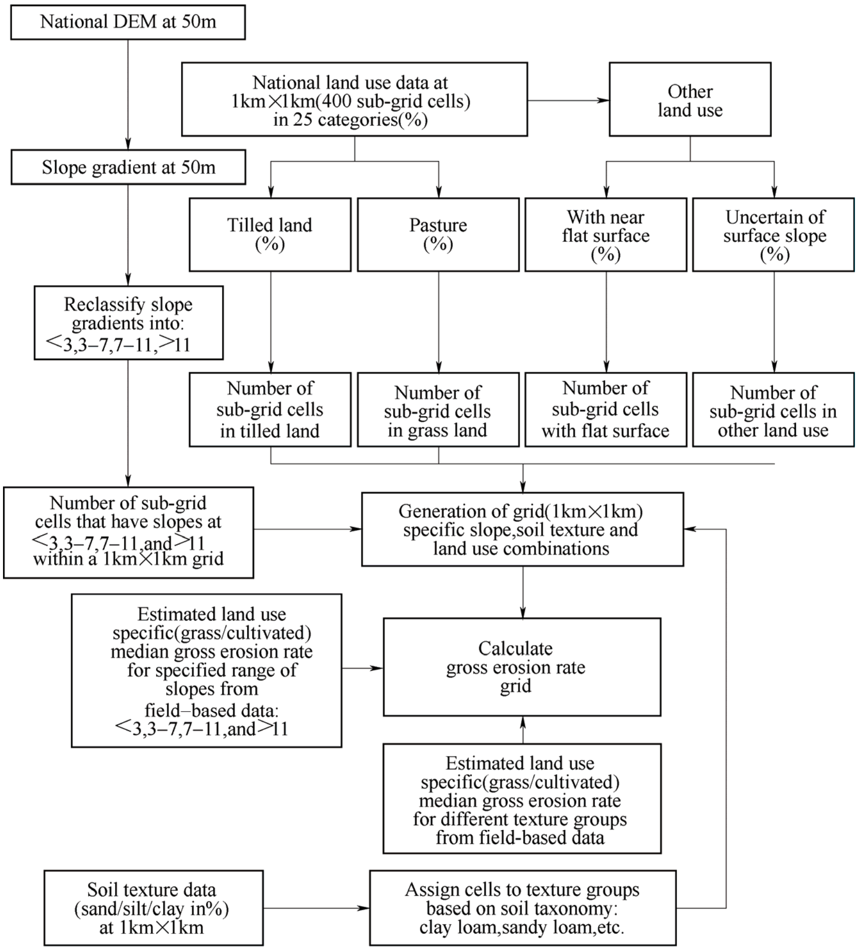


Fig. 2 The typology model used to extrapolate the data obtained from the sampled fields across the agricultural land of England and Wales

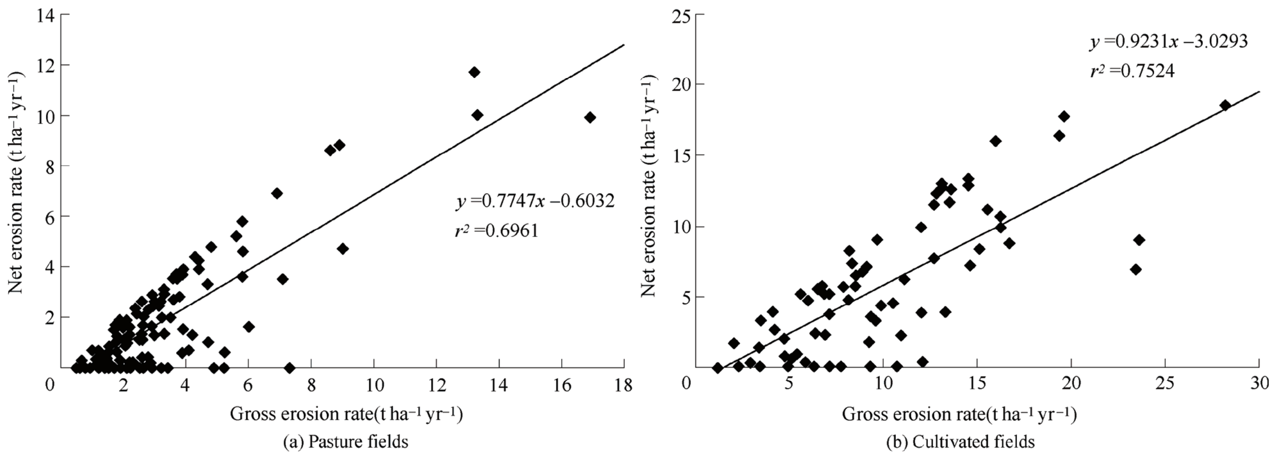


Fig. 3 The emprical relationships between net and gross erosion rates for pasture and culrivated fields used to estimate the net erosion rates for individual grid squares

National maps of gross and net erosion rates associated with arable, pasture and the local mixture of land use within individual grid squares were produced. As an example, Fig. 4 presents the national map of gross erosion rates associated with the likely combination of arable and pasture land use in each grid cell. In the absence of other maps or detailed information on the spatial distribution of soil erosion rates within England and Wales, it is difficult to validate Fig. 4 and other maps produced by the study. Furthermore, it is questionable whether erosion rate estimates derived using different procedures can be directly compared (see Brazier, 2004), due to both fundamental differences in what is being measured or reported, as well as contrasts in the time period or spatial scale involved. The results generated by this study arguably represent the first attempt to generate a comprehensive set of estimates of soil erosion rates for England and Wales, that include the effects of sheet erosion as well as rill erosion. However, some comparisons can usefully be introduced to place the results presented into context. The estimates of erosion rates obtained by this project must be seen as consistent with the annual erosion rates based on field surveys, reported by the Soil Survey of England and Wales (0.5 – 4.8 m^3 ha^{-1} $year^{-1}$), the Agricultural Development and Advisory Service (0.8 – 11 m^3 ha^{-1} $year^{-1}$), and the Soil Survey and Land Research Centre (0.1 – 1.5 m^3 ha^{-1} $year^{-1}$).

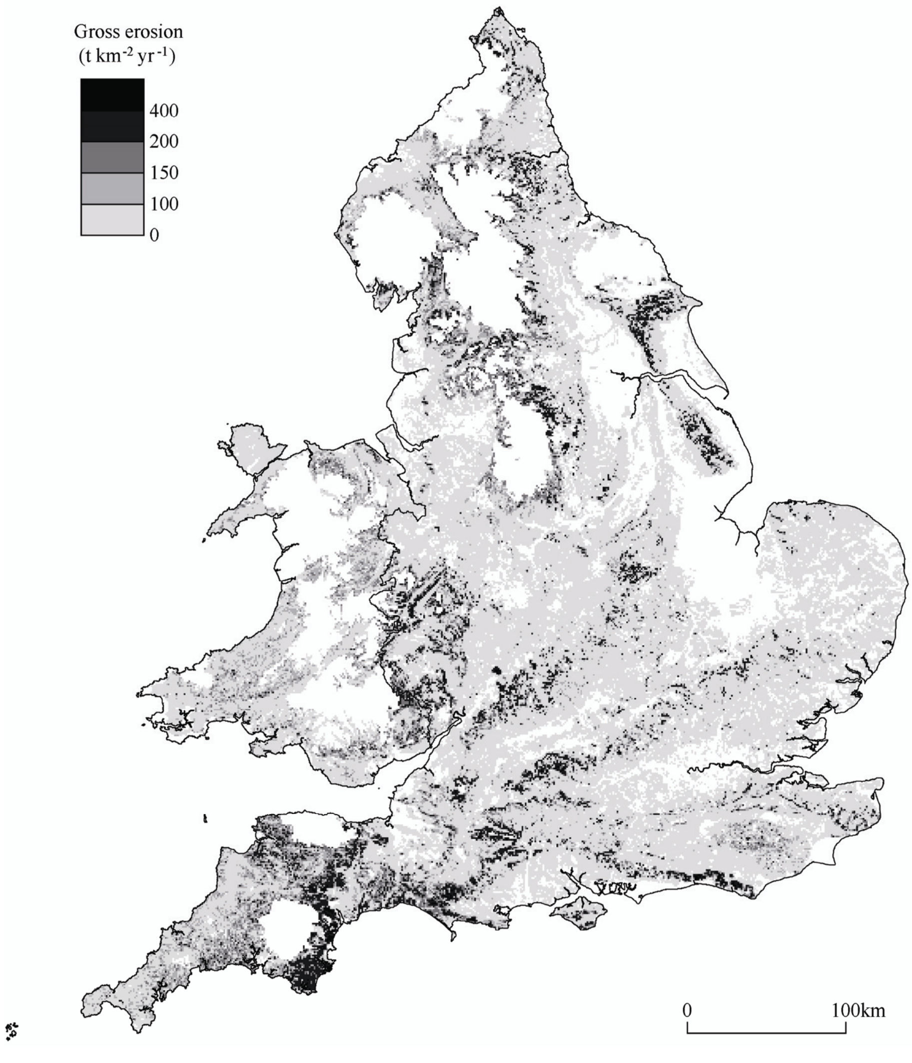
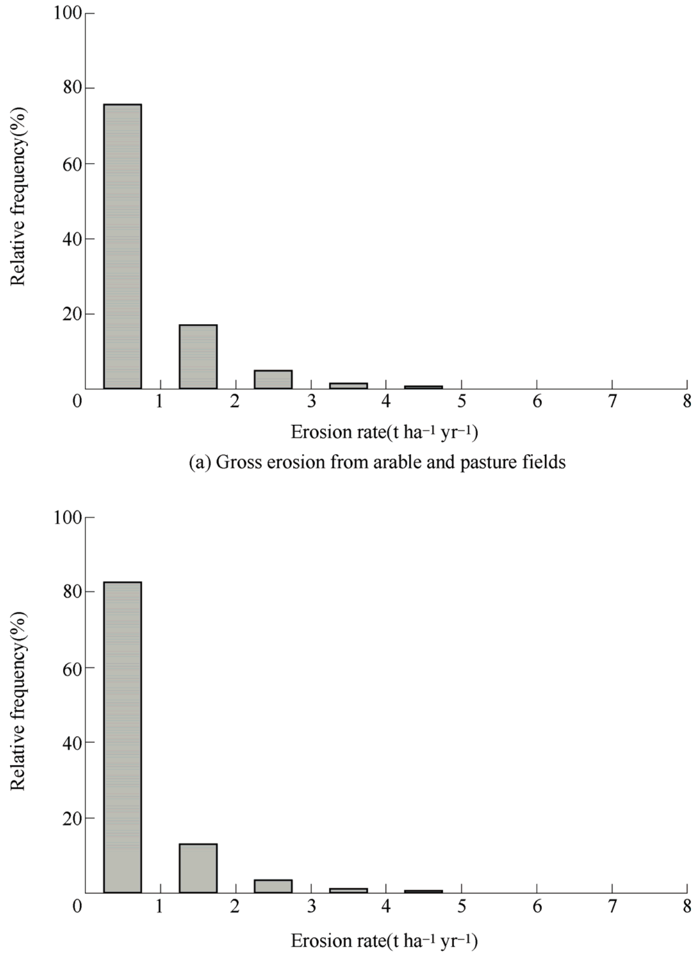


Fig. 4 A national map of gross erosion rates for the likely local combination of cultivated and pasture land use, based on spatial extrapolation of the field-scale erosion rates documented by ^{137}Cs measurements. The areas shown in white are areas excluded from the study as indicated in the text.

Spatial integration of the data presented on the maps produced by the study provided a basis for generating information on the frequency distributions of gross and net erosion rates from agricultural land in England and Wales. These frequency distributions shown in Fig. 5 indicate that, based on the time window of ca. 50 years associated with the use of ^{137}Cs measurements to estimate soil redistribution rates, a soil loss tolerance of 2 t ha^{-1} $year^{-1}$ is exceeded on ~7 percent of the agricultural land in England and Wales for gross erosion and on ~5 percent of the agricultural land for net erosion.



(a) Gross erosion from arable and pasture fields

(b) Net erosion from arable and pasture fields

Fig. 5 A national scale overview of soil erosion rates on agricultural land in England and Wales for local combinations of arable and pasture land use

The maps of gross and net soil loss produced by this study represent soil loss from individual fields and cannot be used directly to represent sediment inputs to river channels, which may be of primary interest when constructing a sediment budget and developing a sediment management strategy for a catchment. A significant proportion of the soil exported from a field as a result of erosion may be deposited between the field and the stream channel to which it is linked. The magnitude of this conveyance loss will be influenced by a range of factors including the drainage density, slope gradient, slope form, surface roughness and sediment texture. Walling and Zhang (2004) developed an index of slope-channel connectivity that reflected these controls and produced a national map of slope-channel connectivity for England Wales, that like the maps of soil loss presented above was based on 1 km grid cells. This map was combined with the map of net soil loss to produce a national map of sediment input to the local channel system. This map is presented as Fig. 6. It indicates that contributions of surface erosion from agricultural land to the sediment input to the channel system typically range between 5 and 50 t km^{-2} $year^{-1}$. These values are again consistent with available data on suspended sediment yields for catchments in England and Wales (see Walling et al., 2007; Cooper et al., 2008). Such yields will include contributions from sources other than the surface of agricultural land, such as channel or bank erosion (Walling and Collins, 2005) and can thus be significantly greater than those associated solely with mobilisation of sediment from agricultural land. In some areas of England and Wales, the surface contribution may account for only ca. 50% of the sediment transported through the channel system and thus the sediment yield. The map presented in Fig. 6 could provide a valuable basis for identifying areas where sediment inputs to the channel system from agricultural land on the catchment slopes are high and where there could therefore be scope to reduce such inputs through implementation of improved land management and sediment control measures.

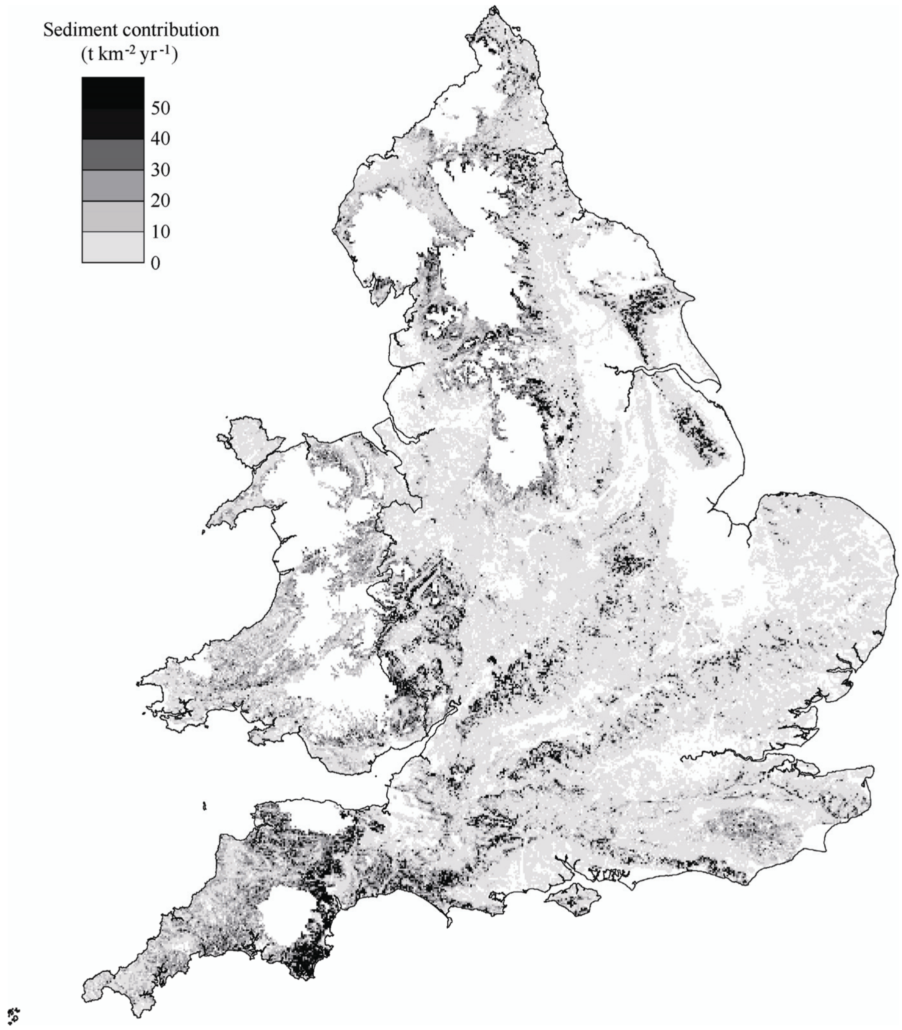


Fig. 6 Estimates of fine sediment inputs to watercourses draining agricultural land in England and Wales from surface sources. The estimates are based on combining data on net erosion rates assembled by the current study with a national map of slope-channel connectivity produced by Walling and Zhang (2004). The areas shown in white are areas excluded from the study as indicated in the text.

3.2 *Use of wide-scale reconnaissance sampling to establish catchment sediment budgets*

The information on soil redistribution rates provided by FRN measurements can potentially represent a valuable data source when establishing a catchment sediment budget, since it can provide information on both gross and net erosion rates and therefore the mobilisation of sediment from the slopes of a catchment and its downslope transfer towards the channel network (e.g. Walling et al., 2002, 2006). The wide-scale sampling approach favoured by Porto et al. (2009) provides an alternative to other upscaling procedures for assembling the required data. Here the emphasis is placed on obtaining a representative frequency distribution of soil redistribution rates for the study catchment. This approach was applied successfully in three small to medium scale catchments in southern Italy (see Porto et al., 2011). The key elements of the study are reported below but further details can be found in Porto et al. (2011).

The three catchments are illustrated in Fig. 7 and further information on their characteristics is provided in Table 4. Their catchment areas ranged from 0.015 km^2 to 31.61 km^2 and this substantial range provided a basis for assessing the impact of scale (i.e. increasing catchment area) on their sediment budgets. Catchment W1 is uncultivated and has a rangeland cover, whereas the Bonis catchment is largely under forest and the Trionto catchment is largely cultivated. Information on longer-term sediment suspended sediment yields was available for the three catchments (see Table 4), based on long-term sediment sampling programmes (W1 and Trionto) and periodic emptying of sediment traps (Bonis).

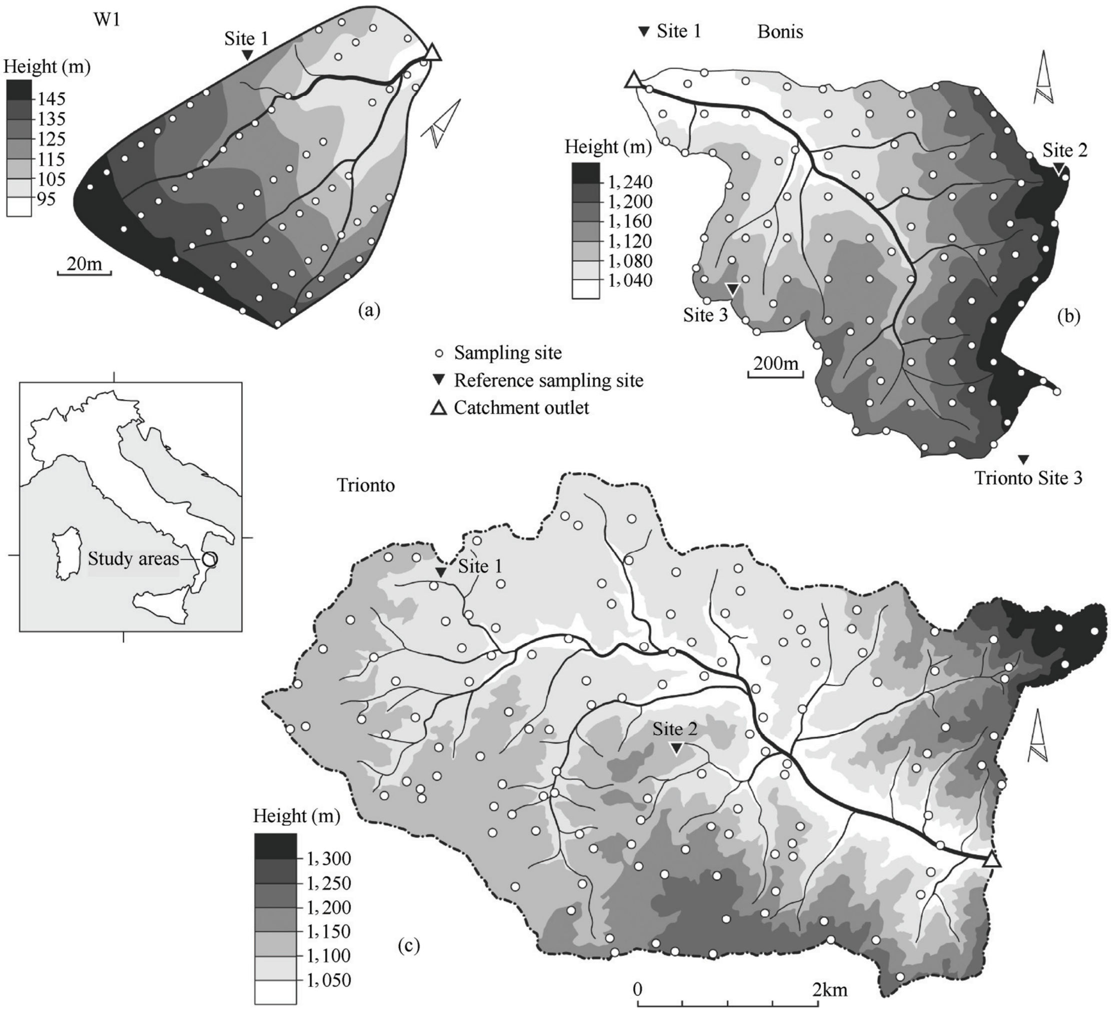


Fig. 7 The study catchments: (a) W1, (b) Bonis and (c) Trionto

Table 4 Characteristics of the study catchments

Catchment	Area (km^2)	Mean altitude (m a.s.l.)	Mean slope (%)	Relief ratio	Primary land use	Sediment yield (t km^{-2} year^{-1})
W1	0.015	122	53	0.17	Rangeland	1,160
Bonis	1.39	1,131	40	0.14	Forest	25
Trionto	31.61	1,100	8.4	0.05	Cultivated	21

Bulk cores were collected from each catchment for establishing the ^{137}Cs inventory at individual sampling points. The cores were collected using a percussion corer fitted with a 6 – 9 cm diameter core tube. The cores extended to depth that ensured inclusion of the full ^{137}Cs depth profile. Different sampling strategies were used for each catchment. In the case of catchment W1, this involved the collection of 68 bulk cores along five essentially parallel linear transects (see Fig. 7a). For the Bonis catchment, 55 bulked replicate cores were collected from the intersections of a 150 m × 150 m grid and these samples were supplemented by bulked cores collected from a further 55 locations selected to extend the coverage of the topographic variability. The collection of replicate cores that were bulked to provide a single sample from each sampling point provided a means of taking account of small-scale spatial variability in ^{137}Cs inventories associated with soil and vegetation heterogeneity. In both the above cases, use of a systematic sampling framework was seen as providing an essentially random set of sampling points within the study catchments, that provided representative coverage of the terrain. The larger size of the Trionto catchment and constraints on access precluded the use of a similar systematic sampling network, and in this case the sampling was based on a network of 128 essentially random sampling points, which aimed to provide representative coverage of the terrain and a relatively uniform spatial distribution of sampling points across the catchment (see Fig. 7c). Again, replicate cores were collected from each sampling point and bulked to provide a single sample from that point. Additional sampling was carried out in each of the three catchments to establish the local reference inventory and to provide catchment-specific information on the depth distributions of ^{137}Cs for use with the conversion models used to estimate soil redistribution rates from the measurements of ^{137}Cs inventory. The considerable range of altitude associated with the larger Trionto catchment meant that three reference inventories, related to three altitudinal zones were established. Table 5 provides information on the reference inventories documented for the three study catchments and Table 6 details the range and standard deviation of the inventories associated with the cores collected from each of the catchments.

Table 5 **The ^{137}Cs reference inventories for the study catchments**

Catchment	Range of altitude (m a.s.l.)	Mean (Bq m^{-2})
W1	90 – 122	2,456
Bonis	975 – 1,300	3,112
Trionto	996 – 1,080	2,803
	1,080 – 1,140	3,883
	1,140 – 1,418	4,682

Table 6 **The range and standard deviation of the inventories associated with the sampling points in the study catchments**

Catchment	Range of ^{137}Cs inventories (Bq m^{-2})	Standard deviation (Bq m^{-2})
W1	4.1 – 4,053	761.7
Bonis	126 – 6,241	1,294.6
Trionto	0.5 – 8,770	1,650.0

The values of ^{137}Cs inventory obtained for the sampling points in the individual catchments were used to derive estimates of soil redistribution rates using appropriate conversion models (see Walling et al., 2011). In the case of catchment W1 and the Bonis catchment, where the soils were uncultivated, a diffusion and migration model was employed. A mass balance model was employed for the cultivated areas in the Trionto catchment. Where the measured inventory was not significantly different from the reference inventory, taking account of the uncertainty associated with the latter, the sampling point was designated as stable. The resulting estimates of soil redistribution rate are presented in Fig. 8, which depicts frequency distributions of erosion and deposition rates and stable sites for the three catchments. In the case of catchment W1, these data provide little evidence of deposition and suggest that most of the mobilised sediment is transported directly to the catchment outlet. This conclusion was confirmed by visual surveys of the catchment, which indicated that areas of deposition were very limited. In the larger Bonis and Trionto catchments, deposition becomes progressively more important. This is a direct reflection of increasing catchment area and the associated greater opportunities for deposition. There was clear evidence of areas of deposition in these two catchments. Important contrasts in the magnitude of the soil redistribution rates are also apparent between the three catchments. As might be expected, soil redistribution rates, and particularly erosion rates, are generally lowest in the forested Bonis catchment and highest in the cultivated Trionto catchment. Those in catchment W1, which is characterized by a rangeland cover, occupy an intermediate position.

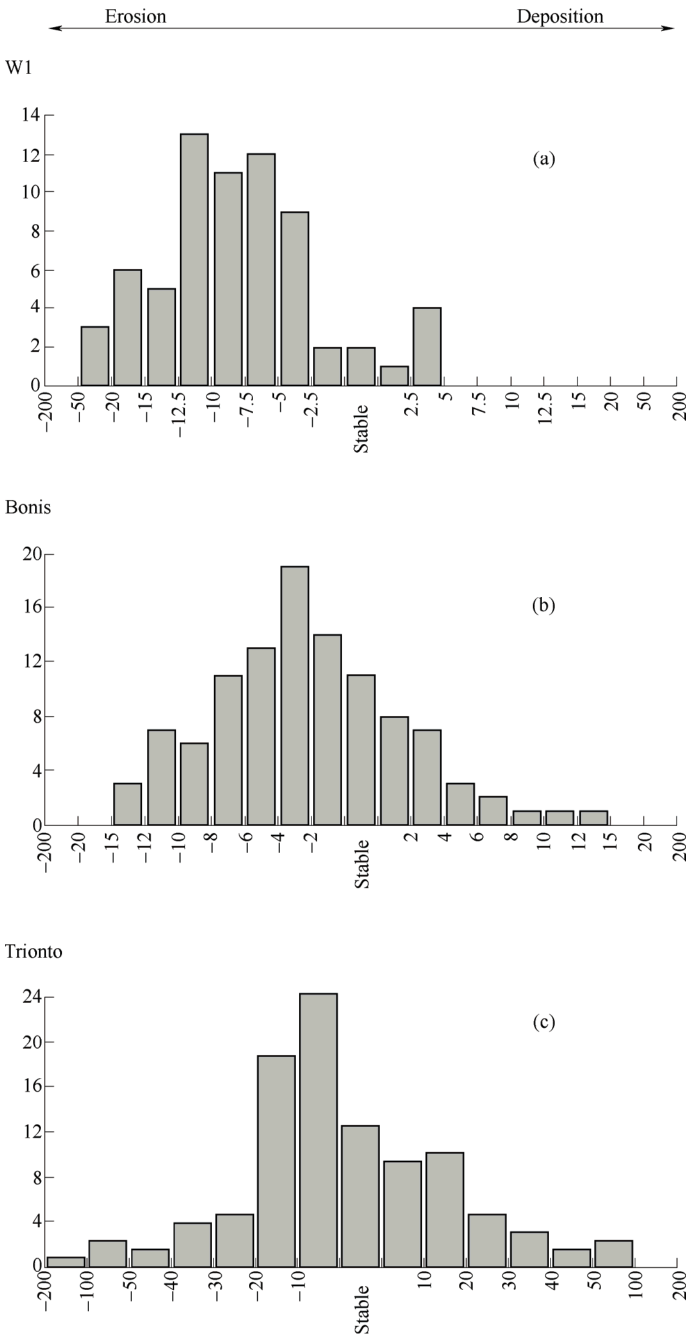


Fig. 8 Frequency distributions of erosion and deposition rates (t ha^{-1} $year^{-1}$) associated with the sampling points within each study catchment: (a) W1, (b) Bonis, (c) Trionto (Based on Porto et al., 2011)

Fig.9 adds a spatial dimension to the results presented in Fig.8, by providing information on both the magnitude and spatial distribution of erosion and deposition rates within the three study catchments. The spatial patterns presented in Fig.8 demonstrate that appreciable rates of sediment mobilisation are widely distributed across the three catchments, although the highest values are frequently located within the higher parts of the catchments or along catchment divides, where the terrain is steepest. In contrast, the sampling points characterized by deposition are preferentially located in the lower parts of the catchments and along sediment delivery pathways, where the potential for deposition is greatest.

If the frequency distributions of soil redistribution rates presented in Fig.8 are assumed to provide representative information on medium-term average rates of erosion and deposition on the slopes of the individual study catchments, they can be used to estimate the gross erosion from the slopes, the on-slope deposition and the net soil loss. These data can be used to characterize the "slope" component of the sediment budgets for the three catchments. The relative incidence of sampling locations providing evidence of deposition and the magnitude of the deposition rates at those locations directly reflects the importance of the storage component of the sediment budgets for the slopes of the individual catchments and thus their sediment delivery ratios. In this context, the storage component of the budget can be seen to be of greatest importance in the Trionto catchment.

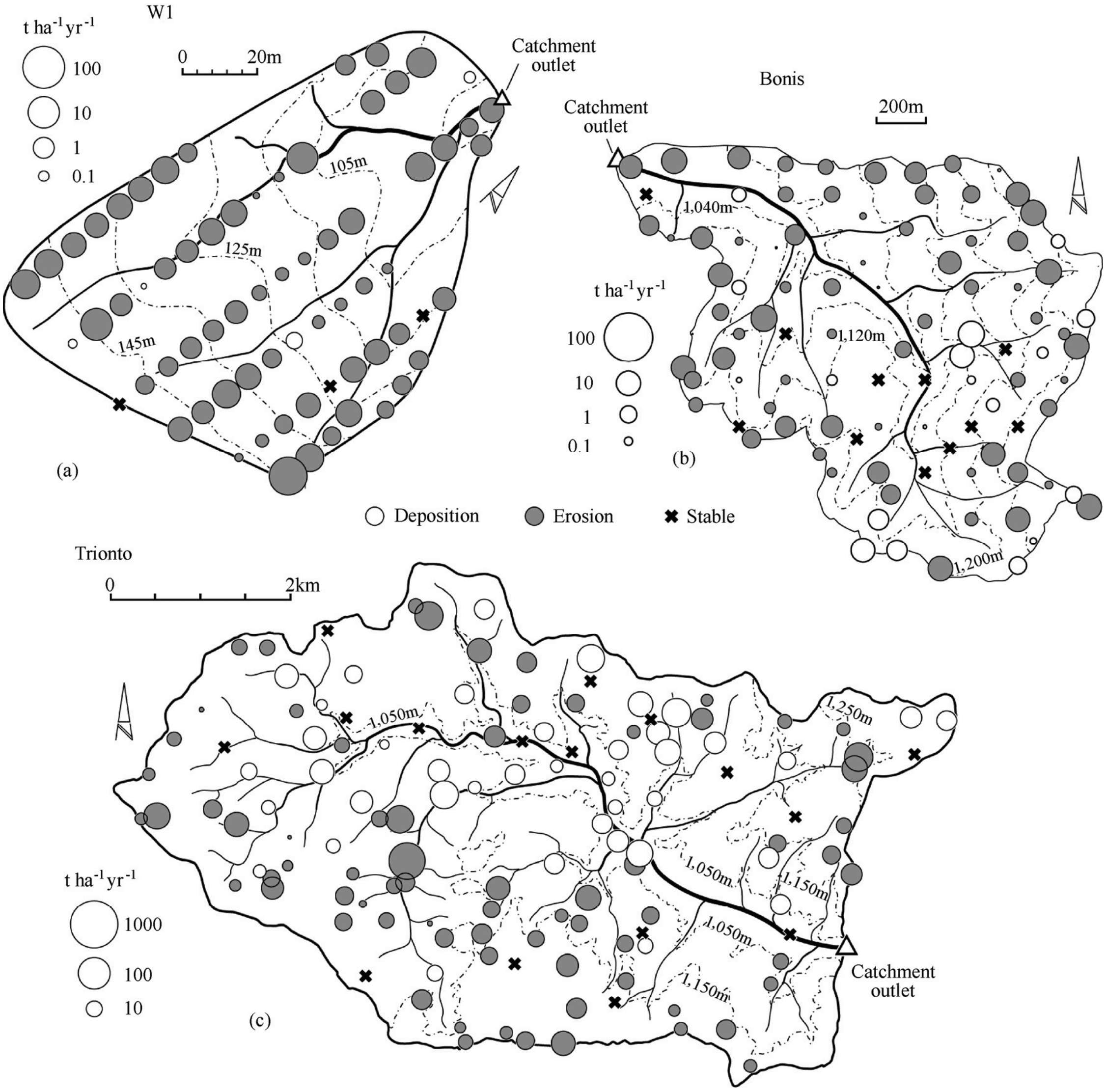


Fig. 9 The spatial distribution of the soil redistribution rates within (a) catchment W1, (b) the Bonis catchment and (c) the Trionto catchment estimated from the ^{137}Cs measurements (Based on Porto et al., 2011)

The estimates of mean annual sediment yield available for the study catchments provide another key component of their sediment budgets, namely the sediment output. If the slope component is assumed to provide a meaningful estimate of the input to the valley and channel system from the slopes, the difference between this and the sediment yield at the catchment outlet can be attributed to deposition during conveyance through the valley channel systems. However, this will provide a minimum estimate of such conveyance losses, since the amount of sediment transported through the channel system could be greater than the input from the slopes, due to additional sediment supply from eroding channel banks. However, this sediment source was judged to be of limited importance in the study catchments. The sediment budgets for the three study catchments established using the above information and assumptions are presented in Fig.10.

These budgets emphasise the increasing importance of sediment deposition, both on the slopes and within the channel system, as catchment scale increases. The catchment sediment delivery ratio (SDR) provides a useful means of assessing the efficiency of sediment delivery from a catchment (Walling, 1983) and Fig.10 indicates that the SDR declines from 98% for catchment C1 to 7% for the Bonis catchment and 2% for the Trionto catchment. This is consistent with existing representations of the inverse relationship between SDR and catchment area (e.g. Roehl, 1962, Walling, 1983).

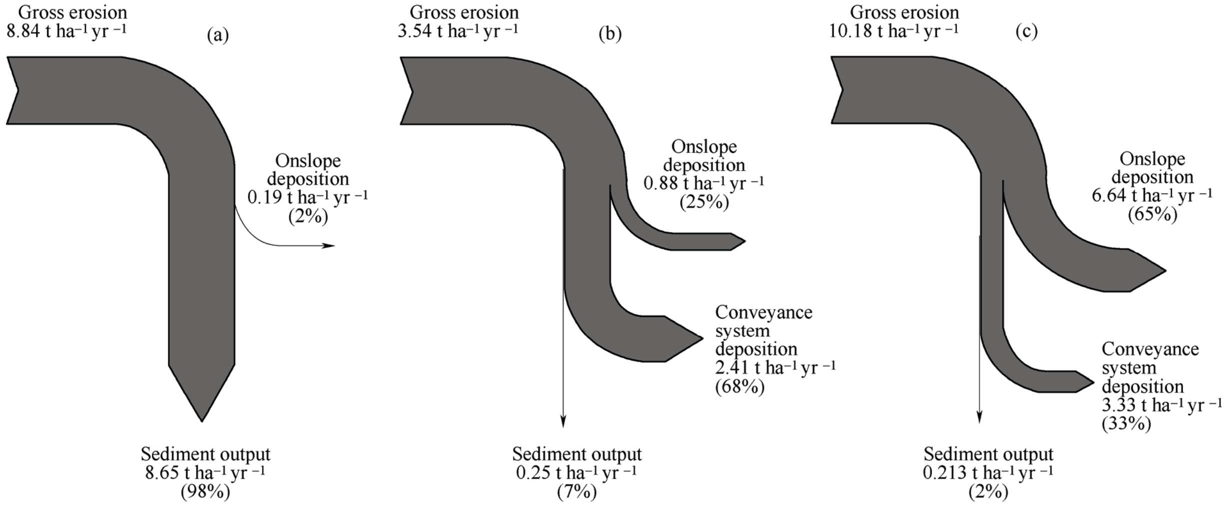


Fig. 10 Schematic sediment budgets for (a) catchment W1, (b) the Bonis catchment and (c) the Trionto catchment. The SDR associated with the sediment transfers and the output from each catchment is indicated as a percentage. (Based on Porto et al., 2011)

Recent work within the Bonis catchment by the same authors has successfully used both ^{137}Cs and $^{210}Pb_{ex}$ measurements to construct catchment sediment budgets (Porto et al., 2013). The sediment budget constructed using the information provided by the $^{210}Pb_{ex}$ measurements is presented in Fig.11. Although $^{210}Pb_{ex}$ measurements respond to a longer time window than ^{137}Cs measurements, by virtue of the essentially continuous nature of the $^{210}Pb_{ex}$ fallout, they will be more sensitive to the recent erosional response. Analysis of the rainfall records for the catchment provided evidence of a progressive increase of the annual erosivity in recent years, which can be linked to the higher gross erosion rates indicated in Fig.11.

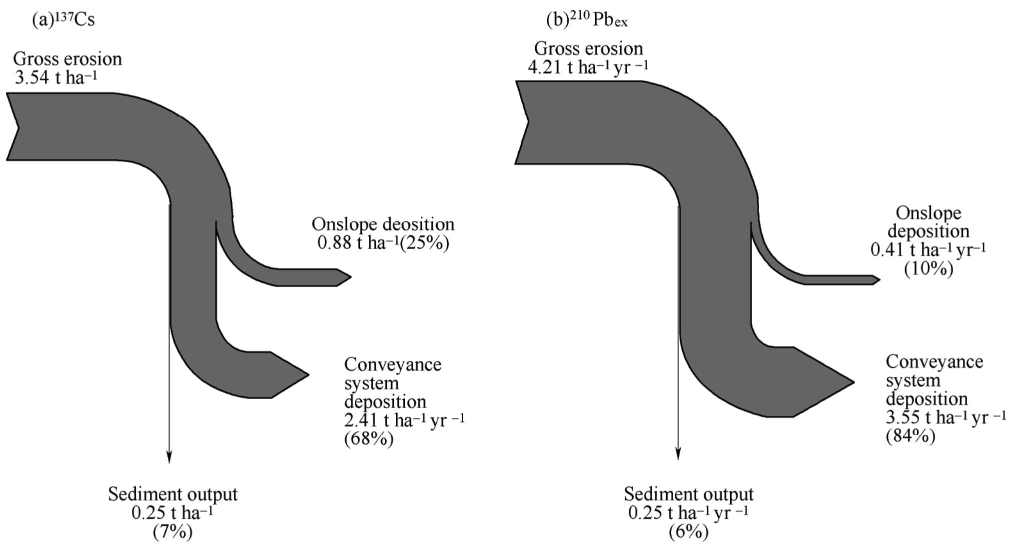


Fig. 11 Schematic sediment budgets for the Bonis catchment derived from the ^{137}Cs (a) and $^{210}Pb_{ex}$ (b) measurements (Based on Porto et al., 2013)

Further work is clearly required to assess the uncertainties associated with various components of the sediment budgets presented in Figs. 10 and 11, but these are seen as representing a meaningful reflection of the basic form of the sediment budgets of the three study catchments. The use of fallout radionuclides and the upscaling approach employed must be seen as providing the primary basis for deriving the information required to construct the sediment budgets.

4 Perspective

The brief overview of potential approaches to upscaling the use of fallout radionuclides in soil erosion and sediment budget investigations and the two more detailed examples described subsequently serve to illustrate both the challenge posed by such upscaling and some ways in which this challenge can be met. Additional work is required to explore further the different approaches available and more particularly the possibilities for integrating individual approaches. Geostatistical techniques clearly offer important opportunities, but are frequently seen as being difficult to combine with more deterministic or physically-based approaches. As the scale of attention in sediment budget investigations changes from a small area to the wider landscape and a large catchment or region, there is a need to recognise that upscaling also introduced the need to direct more attention to conveyance losses and sediment storage. This was demonstrated by the results from the study of the three catchments in southern Italy. However, as catchment size increases further, sediment storage is likely to assume even greater importance. River floodplains can, for example, represent very important sediment sinks. Fallout radionuclides again offer considerable potential for quantifying sedimentation rates on river floodplains and for establishing their importance as a sink within the overall sediment budget (see He and Walling, 1996; Walling and He, 1997; Du and Walling, 2012). However, as with soil erosion assessment, much of this work has to date focussed on small areas of floodplain or individual floodplain reaches and there is again a need to consider how best to upscale such work to larger river systems. Some progress is evident in the studies reported by Walling et al. (1998); Sweet et al. (2003) and Nicholas et al. (2006), but further work is again required.

Acknowledgements

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