Using environmental radionuclides as tracers in sediment budget investigations

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Abstract The many problems associated with the presence of fine sediment in river basins has generated an increasing need for an improved understanding of the mobilization, transfer and fate of such sediment, in order to plan appropriate sediment management and control strategies. The sediment budget concept affords a valuable framework for assembling information on the sediment response of a catchment, but it is difficult to assemble the data required to construct a reliable sediment budget. The use of environmental or fallout radionuclides offers considerable potential for developing improved understanding and characterization of catchment sediment budgets. The basis for using caesium-137, unsupported lead-210 and beryllium-7 as tracers in sediment budget investigations is outlined, and examples of the use of these radionuclides in studies of soil erosion and sediment delivery from agricultural land, flood plain sedimentation and associated sediment conveyance losses, and sediment source fingerprinting are provided.

Key words environmental radionuclides; flood plain sedimentation; sediment budget; sediment delivery; sediment tracing; soil erosion

INTRODUCTION

Fine sediment is the cause of a wide range of environmental problems in river basins, in both the developed and the developing world. Such problems range from the siltation of reservoirs, navigable waterways and irrigation canals, through the transfer and storage of sediment-associated nutrients and contaminants, including pesticides and heavy metals, to the siltation of fish spawning gravels and the more general degradation of aquatic habitats. As long ago as the 1970s, Eckholm (1976) identified "excess sediment" as "the major form of human-induced water pollution in the world today" and the significance of sediment-related problems has greatly increased, as their wider ecological implications have been increasingly recognized. Such problems must now be seen as having both an economic dimension, through their impact on water resource development and related infrastructure, as well as a broader ecological/environmental dimension, related to the longer-term sustainability and conservation of aquatic ecosystems (cf. Mahmood, 1987; Clark *et al.*, 1985). Against this background, there is a growing need for the implementation of effective sediment management and control strategies in river basins. Since much of the fine sediment transported by a river will have been mobilized from the catchment surface by soil erosion, such strategies must consider the entire catchment or river basin. The design and implementation of sediment management and control strategies is, however, frequently hampered by a lack of data on erosion rates and sediment yields, as well as understanding of the delivery and storage of fine sediment within catchment systems. The linkages between erosion, transport, deposition, storage and sediment yield at the catchment outlet can be highly complex (cf. Phillips, 1992; Walling, 2000), especially in situations where sediment storage equals or exceeds sediment export (cf. Trimble, 1983).

The sediment budget concept affords a valuable framework for assembling the detailed information required to elucidate and interpret drainage basin sediment delivery (Swanson *et al.*, 1982; Golosov *et al.*, 1992; Reid & Dunne, 1996; Walling, 1998b, 2000). By quantifying the sources, transfer pathways, sinks and output of sediment for a drainage basin, it is possible to identify and quantify the key areas of sediment mobilization and storage and to assess the efficiency of the sediment delivery system and its sensitivity to both extrinsic and intrinsic controls. However, despite its utility, the sediment budget concept has proved difficult to apply, due to the problems of assembling the necessary data for anything except a very small drainage basin. There is a need for new approaches to obtaining the data required to establish a catchment sediment budget (cf. Walling *et al.*, 2001) and the potential for using environmental radionuclides as tracers in sediment budget investigations has been increasingly recognized and exploited. Key advantages of the use of environmental radionuclides include the potential for assembling retrospective (medium-term) information on the basis of a single site visit, the possibility of using effectively the same measurements within different components of the sediment budget and thus tracing the movement of sediment through the delivery system, the provision of point estimates of sediment mobilization and deposition that are directly compatible with the current generation of spatially distributed numerical models, as well as the ability to apply the measurements at a range of spatial scales. This contribution aims to demonstrate further the potential for using environmental radionuclides as tracers in sediment budget investigations.

ENVIRONMENTAL RADIONUCLIDES

The term "environmental radionuclide" is commonly used to refer to those radionuclides which are commonly occurring and widely distributed in the environment or landscape and, whilst occurring at relatively low levels, are readily measurable. In most cases they are of natural origin, but in some cases they are man-made. For applications relating to sediment budget investigations, most work to date has focused on a particular group of environmental radionuclides, namely, fallout radionuclides, or radionuclides which reach the land surface by fallout from the atmosphere. In this case, the fallout input can generally be assumed to be spatially uniform, at least over a relatively small area. Because the radionuclides employed are rapidly and strongly adsorbed by the soil, they accumulate at or near the surface, affording a means of tracing sediment mobilization and deposition by documenting the subsequent redistribution of the radionuclide tracer, which moves in association with soil or sediment particles. In essence, it is therefore possible to view the fallout as equivalent to the artificial application of a tracer to the surface of a study area. Observation of the subsequent redistribution of the radionuclide provides a basis for establishing rates and patterns of sediment transfer within the landscape.

The radionuclide that has been most widely used as a sediment tracer is caesium-137(^{137}Cs) (cf. Ritchie & McHenry, 1990; Zapata, 2002). Caesium-137 is a man-made radionuclide, with a half-life of 30.2 years, that was produced during the atmospheric testing of thermonuclear weapons during the period extending from the mid 1950s to the 1960s. The radiocaesium was released into the stratosphere and globally distributed. Global fallout of ^{137}Cs began in 1954, peaked in the early 1960s and subsequently decreased, reaching near zero levels in the mid 1980s. Fallout levels were globally variable, reflecting both annual precipitation amount and location relative to the main weapons tests (cf. Walling, 2002). Smaller amounts of ^{137}Cs have also been released into the atmosphere by accidents at nuclear power stations, notably the Chernobyl disaster in 1986, which resulted in additional inputs of ^{137}Cs fallout over large areas of Europe.

Use of other fallout radionuclides in sediment budget investigations has focused on unsupported lead-210 (^{210}Pb) and beryllium-7 (^{7}Be). These differ from ^{137}Cs in two important respects. In the first place, they are both of natural origin, and, secondly, their fallout input can be treated as essentially constant over time. Lead-210 is a naturally occurring product of the ^{238}U decay series, with a half-life of 22.2 years, that is derived from the decay of gaseous ^{222}Rn, the daughter of ^{226}Ra. Radium-226 exists naturally in soils and rocks and the ^{210}Pb in soils generated *in situ* by the decay of ^{226}Ra is termed supported ^{210}Pb and is in equilibrium with ^{226}Ra. However, upward diffusion of a small portion of the ^{222}Rn produced in the soil and rock introduces ^{210}Pb into the atmosphere and its subsequent fallout provides an input of this radionuclide to surface soils and sediments that will not be in equilibrium with its parent ^{226}Ra (Robbins, 1978). Fallout ^{210}Pb is commonly termed unsupported or excess ^{210}Pb, when incorporated into soils and sediments, to distinguish it from the ^{210}Pb produced *in situ* by the decay of ^{226}Ra. The amount of unsupported ^{210}Pb in a soil or sediment sample can be calculated by measuring both the ^{210}Pb and ^{226}Ra activities and subtracting the ^{226}Ra-supported ^{210}Pb component from the total ^{210}Pb in the sample. In contrast to ^{137}Cs and ^{210}Pb, ^{7}Be has a very short half-life (53 days). It is produced by the bombardment of the Earth's atmosphere by cosmic rays and is subsequently deposited as fallout.

The different half-lives of the three fallout radionuclides considered above and the different temporal distributions of their fallout mean that their inventories (i.e. the total amount of radionuclide contained within a soil or sediment profile (Bq m^{-2})) will reflect different temporal behaviour. In the case of unsupported ^{210}Pb, the essentially constant fallout input means that the inventory of a stable soil unaffected by erosion or deposition will also remain essentially constant and in steady state, with loss by decay being balanced by new fallout input. In contrast the ^{137}Cs inventory of a stable soil will have been zero prior to the onset of fallout in the mid 1950s, it will then have increased through to the mid or late 1960s in response to the main period of fallout input, and subsequently it will have decreased as the rate of decay exceeded the rate of replenishment by new fallout. However, because of its relatively long half-life (30.2 years), significant amounts of ^{137}Cs will still remain some 40 years after the main period of fallout input. As a result of its short half-life, the ^{7}Be inventory of a stable soil will evidence considerable short-term variability. During periods of dry weather the inventory will rapidly decline due to decay, only to increase again as a result of rainfall and associated fallout.

APPLICATION OF FALLOUT RADIONUCLIDES AS TRACERS

Figure 1 illustrates typical distributions of ^{137}Cs, unsupported ^{210}Pb and ^{7}Be in adjacent permanent pasture and cultivated soils at a site near Crediton in Devon, UK. The depth scale has been expressed in terms of cumulative mass depth (kg m^{-2}), rather than linear depth (cm), in order to remove the effects of down-core changes in bulk density. At pasture sites, the radionuclides are typically concentrated at the surface, with, in the case of ^{137}Cs and unsupported ^{210}Pb, ~90% of the total inventory occurring in the upper 15 cm of the soil and concentrations exhibiting an exponential decline below the surface. Although the ^{137}Cs and ^{210}Pb fallout is delivered to the surface, the existence of the radionuclides to depths of ~30 cm reflects downward diffusion and migration, particularly in association with biological activity. The minor differences between the vertical distributions of ^{137}Cs and unsupported ^{210}Pb primarily reflect the different temporal patterns of fallout input associated with the two fallout radionuclides. More particularly, ^{137}Cs fallout effectively ceased in the 1980s and there has been no surface replenishment since that time. In the case of unsupported ^{210}Pb, however, fallout has been effectively continuous and maximum concentrations are, expectedly, found at the surface. The vertical distribution of ^{7}Be differs significantly from that of the other two radionuclides and this difference directly reflects its very much shorter half-life. Beryllium-7 is only found at or very near the surface where it is replenished by fallout, since any of the radionuclide moving down into the profile will soon disappear due to decay. If an undisturbed site, which has been influenced by neither erosion nor deposition, can be identified, measurement of the total inventory of the individual radionuclides at that site (Bq m^{-2}) will provide an estimate of the local fallout input. Such sites are usually referred to as reference sites (cf. Loughran *et al.*, 2002) and are normally located in areas with limited relief, and particularly on interfluves.

The ^{137}Cs and unsupported ^{210}Pb depth profiles from cultivated areas adjacent to the equivalent pasture areas, which are also shown in Fig. 1, clearly demonstrate the effects of cultivation or tillage in mixing the soil contained within the plough layer. In both cases the radionuclide concentrations are effectively constant throughout this layer, which extends to a depth of ~25 cm. The total ^{137}Cs and unsupported ^{210}Pb contents of the soil profiles from the cultivated sites are less than those for the adjacent pasture sites, and this reflects removal of soil containing both radionuclides from the profiles by soil erosion. As soil is progressively eroded from the surface, soil from below the original plough depth is mixed into the plough layer and the radionuclide concentration in this layer will progressively decline. The contrasting behaviour of ^{7}Be again reflects the short half-life of this radionuclide. Due to its short half-life, the total inventory of ^{7}Be always remains relatively low and when the soil is cultivated mixing of the available ^{7}Be inventory into the plough layer will produce very low concentrations, which, in most cases, will be below the level of detection. Furthermore, the rapid decay of any ^{7}Be mixed into the plough layer will cause the ^{7}Be concentration in this layer to decline rapidly. As a result, the presence of ^{7}Be is limited to a thin surface layer, which reflects the recent fallout to the surface. As with ^{137}Cs and unsupported ^{210}Pb, the total inventory associated with the cultivated soil reflects loss of ^{7}Be in association with eroded soil. This is further indicated by the reduced depth to which ^{7}Be is found in the cultivated soil, relative to the pasture soil.

At sites in the landscape where deposition occurs, both the depth distribution and total inventories of ^{137}Cs, unsupported ^{210}Pb and ^{7}Be will differ from those shown in Fig. 1. Deposition of soil or sediment containing these radionuclides will cause the depth to which the radionuclide is found to increase and the total inventory will also

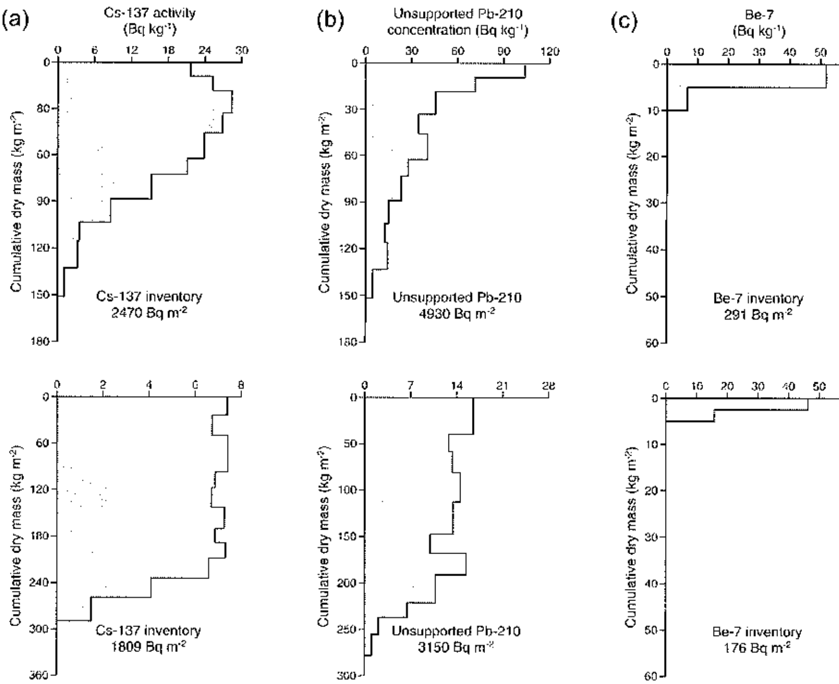

Fig. 1 Typical depth distributions of ^{137}Cs, unsupported ^{210}Pb and ^{7}Be concentrations in undisturbed pasture (upper) and cultivated soils (lower) in Devon, UK.

increase. This situation is illustrated in Fig. 2, which compares the depth profiles of the three radionuclides in sediment cores collected from river flood plains in Devon, UK with those in adjacent pasture soils above the level of inundating floodwater. In all cases, the soils are uncultivated and the profiles are therefore not significantly disturbed by tillage mixing. In the case of ^{137}Cs, the depth profile provides clear evidence of the progressive accretion of the flood plain above the level marked by the maximum ^{137}Cs activity, which represents the level of the surface of the flood plain in the mid 1960s. This accretion is caused by overbank deposition of fine sediment containing ^{137}Cs that has been mobilized by erosion from the upstream catchment and which results in a total ^{137}Cs inventory that is considerably in excess of that associated with the core collected from the site above the level of inundation, which will have received only fallout inputs. The decline in radiocaesium activity towards the surface reflects the reduction and subsequent cessation of fallout after the mid 1960s, which would result in both reducing fallout inputs to the flood plain surface, and a gradual reduction in the ^{137}Cs activity associated with sediment mobilized from the upstream catchment, as erosion proceeds. The response of the unsupported ^{210}Pb profile to progressive accretion shown in Fig. 2 differs from that shown by the ^{137}Cs depth profile, due to the continuous fallout input. In this case, progressive accretion is marked by a more gradual exponential decline in unsupported ^{210}Pb activity with depth, with the base of the unsupported ^{210}Pb profile being found at a much greater depth than in the core collected from the site above the level of flood plain inundation. In the case of ^{7}Be, the short half-life of this radionuclide means that contrasts between

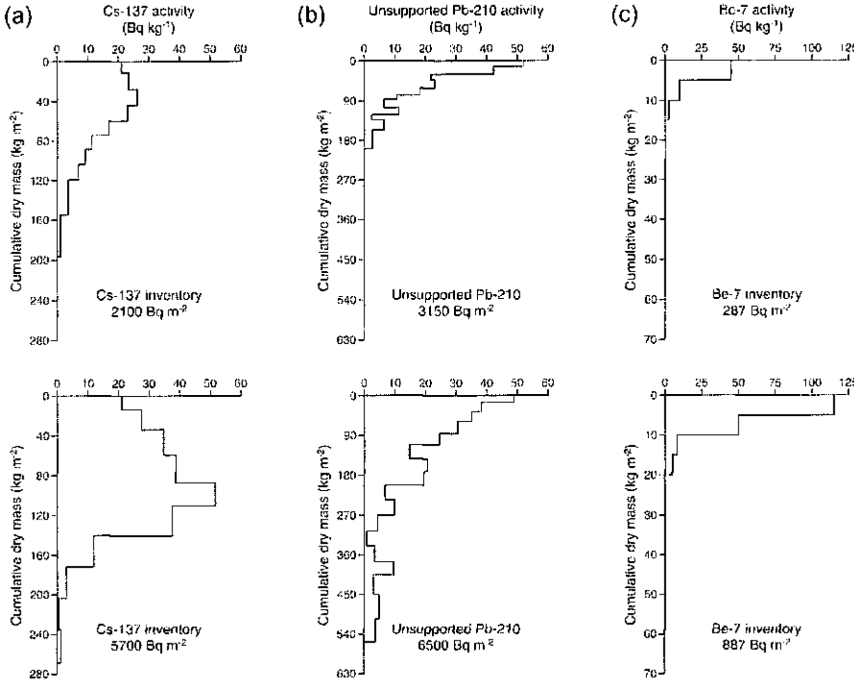

Fig. 2 Typical depth distributions of ^{137}Cs, unsupported ^{210}Pb and ^{7}Be concentrations in overbank sediments from river flood plains in Devon, UK (lower), and in adjacent pasture soils above the level of inundation (upper).

the profiles from the flood plain area and the adjacent area above the level of inundation will only reflect very recent flood plain accretion. If no overbank sedimentation has occurred within the past 6 months or so, there is unlikely to be any difference between the profiles. The influence of any sediment deposited before this date on the total ^{7}Be inventory will be negligible, due to the influence of decay in reducing its activity, and both profiles will therefore simply reflect the fallout input to the surface. The ^{7}Be profile for the flood plain surface depicted in Fig. 2 was measured shortly after a sizeable flood had inundated the flood plain, causing significant deposition. The influence of this accretion is evident in both the increased inventory of the flood plain core and the greater depth to which ^{7}Be is found in this core.

The distinctive behaviour of ^{137}Cs, unsupported ^{210}Pb and ^{7}Be at erosional and depositional sites illustrated in Figs 1 and 2 provides the basis for their use as tracers in sediment budget investigations. Thus, for example, by collecting soil cores from a field, measuring their ^{137}Cs, unsupported ^{210}Pb or ^{7}Be inventories and comparing these with the local reference inventory, it is possible to identify sites where erosion (reduced inventories) and deposition (increased inventories) have occurred. A variety of conversion models are available to convert the measurements of inventory loss or gain to estimates of erosion or deposition rate (cf. Walling & He, 1999a,b; Walling *et al.*, 1999). For ^{137}Cs measurements, the resulting estimates of erosion and deposition rates will reflect erosion and deposition occurring over the last ~45 years (i.e. since the beginning of significant ^{137}Cs fallout), whereas for unsupported ^{210}Pb and ^{7}Be they will relate to longer and much shorter periods, respectively. With its half-life of 22.2 years and essentially continuous input, unsupported ^{210}Pb will provide estimates of erosion and deposition rates extending back over ~100 years (i.e. 4–5 half-lives), whereas for ^{7}Be the estimates could relate to a single event, when there has been little or no erosion in the preceding ~6 months.

Similarly, the ^{137}Cs, unsupported ^{210}Pb and ^{7}Be depth distributions found in river flood plain environments illustrated in Fig. 2 afford a basis for estimating rates of overbank deposition on river flood plains. By collecting cores from an area of flood plain and determining the radionuclide profiles within those cores or, in simpler applications, comparing their total inventories with the local reference inventory, it is possible to establish both rates and patterns of overbank flood plain sedimentation (cf. He & Walling, 1996; Walling & He, 1997; Blake *et al.*, 2002). Again, the time base of the estimates will vary according to the radionuclide involved. With ^{7}Be it is possible to obtain estimates of sedimentation rates associated with an individual flood, whereas with ^{137}Cs the estimates will relate to a period of ~40–45 years, and with unsupported ^{210}Pb the period involved will be still longer, although some workers have succeeded in breaking this down into shorter constituent periods for which the deposition rate can be estimated.

A further example of the potential for exploiting the behaviour of environmental radionuclides illustrated in Figs 1 and 2 in sediment budget investigations relates to their potential use as fingerprints in suspended sediment source tracing investigations (cf. Walling & Woodward, 1992). The fingerprinting approach (cf. Walling *et al.*, 1993; Walling & Woodward, 1995; Collins *et al.*, 1997) is based on the ability to discriminate between potential source materials, based on their physical and chemical properties, and to estimate the relative contribution of a number of potential sources to the river load by comparing the properties of the suspended sediment transported by a river with those of the potential sources, whilst taking account of differences in grain size composition between the sediment and the sources. A key requirement of the approach is the need to identify a number of properties that will clearly discriminate between several potential sources. Fallout radionuclide activities or concentrations are particularly useful in this regard, since they generally provide a means of discriminating between surface and subsurface (e.g. channel bank) source materials within a catchment, and between surface materials from areas under different land use. Thus, considering Fig. 1 for example, the ^{137}Cs and unsupported ^{210}Pb content of sediment eroded from the surface of pasture areas will be several times greater than that mobilized from cultivated areas. Equally, sediment eroded from river banks is likely to be characterized by much lower ^{137}Cs and unsupported ^{210}Pb concentrations than those associated with sediment originating from surface sources, since the bank face will receive little or no direct fallout and significant ^{137}Cs and unsupported ^{210}Pb activities will be found only in the upper 10 or 15 cm of the bank profile. In the case of ^{7}Be, significant concentrations of this radionuclide will only be found where the soil or sediment surface has been recently exposed to rainfall, and thus ^{7}Be fallout, and the radionuclide is likely to be absent from channel banks and other subsurface sources.

More detailed examination of the potential for using environmental radionuclides in sediment budget investigations can usefully be achieved by briefly considering examples drawn from a number of studies undertaken by the author and his co-workers in recent years. These include studies of soil erosion and sediment delivery from agricultural land, flood plain sedimentation and associated sediment conveyance losses and sediment source fingerprinting.

SOIL EROSION AND SEDIMENT DELIVERY FROM AGRICULTURAL LAND

Although most work involving the use of environmental radionuclides in studies of erosion and sediment delivery from agricultural land has been based on measurements of ^{137}Cs (cf. Ritchie & McHenry, 1990; Walling, 1998a), both unsupported ^{210}Pb and ^{7}Be have also been used in similar applications (cf. Walling *et al.*, 1995, 1999; Walling & He, 1999). By collecting cores from a study site, measuring the ^{137}Cs, unsupported ^{210}Pb or ^{7}Be inventories, and applying a conversion model, it is possible to derive point estimates of the erosion and deposition rates associated with the cores, and by integrating these values across the study site it is possible to establish the magnitude and relative importance of erosion and deposition and thus the gross and net erosion and the sediment delivery ratio. Figure 3 presents the results of an investigation of rates and patterns of soil redistribution within a 6.7 ha field at Higher Walton Farm near Crediton in Devon, UK (cf. Walling *et al.*, 1999). The area covered by the field represents a valley head depression with slopes of up to 15° converging towards its outlet. In this case measurements of both ^{137}Cs and ^{7}Be activities were undertaken, with the former providing estimates of average rates of soil redistribution over the past ~40 years, and the latter estimates of the erosion rates associated with a particular period of heavy rainfall (69 mm in 7 days) that occurred in early January 1998.

In this study, the soil cores used for the ^{137}Cs and ^{7}Be measurements were collected during two separate campaigns, although they could have been collected together. In both cases the cores were collected at the intersections of a 20 m × 20 m grid, resulting in a suite of ~140 cores from each sampling campaign. Cores were also collected from adjacent areas of undisturbed land, which were not expected to experience either erosion or deposition, in order to establish the reference inventory. For the ^{137}Cs measurements, the cores were collected in August 1996, using a motorized percussion corer equipped with a 6.9 cm internal diameter steel core tube, which was propelled into the soil to a depth of ~60 cm. The cores used for the ^{7}Be measurements were, in contrast, much shallower and were collected manually to depths of 3–5 cm using a 15 cm diameter plastic core tube, in January 1998. During the preceding spring/summer of 1997, the field had been cultivated and sown to maize and the crop was harvested in early November 1997, when the soil was compacted by the harvesting equipment. After harvesting, the field was left bare and uncultivated over the winter and the period of heavy rainfall in early January 1998 resulted in substantial surface runoff and soil erosion.

The pattern of ^{137}Cs inventories documented for the study field is presented in Fig. 3(a). The cores from the adjacent undisturbed sites provided an estimate of the local reference inventory of ~2500 Bq m^{-2} and the pattern of ^{137}Cs inventories shows clear evidence of both erosion (reduced inventories) and deposition (increased inventories). Use of a conversion model enables estimates of mean annual soil redistribution rates over the past ~40 years to be derived from the measured inventories. The

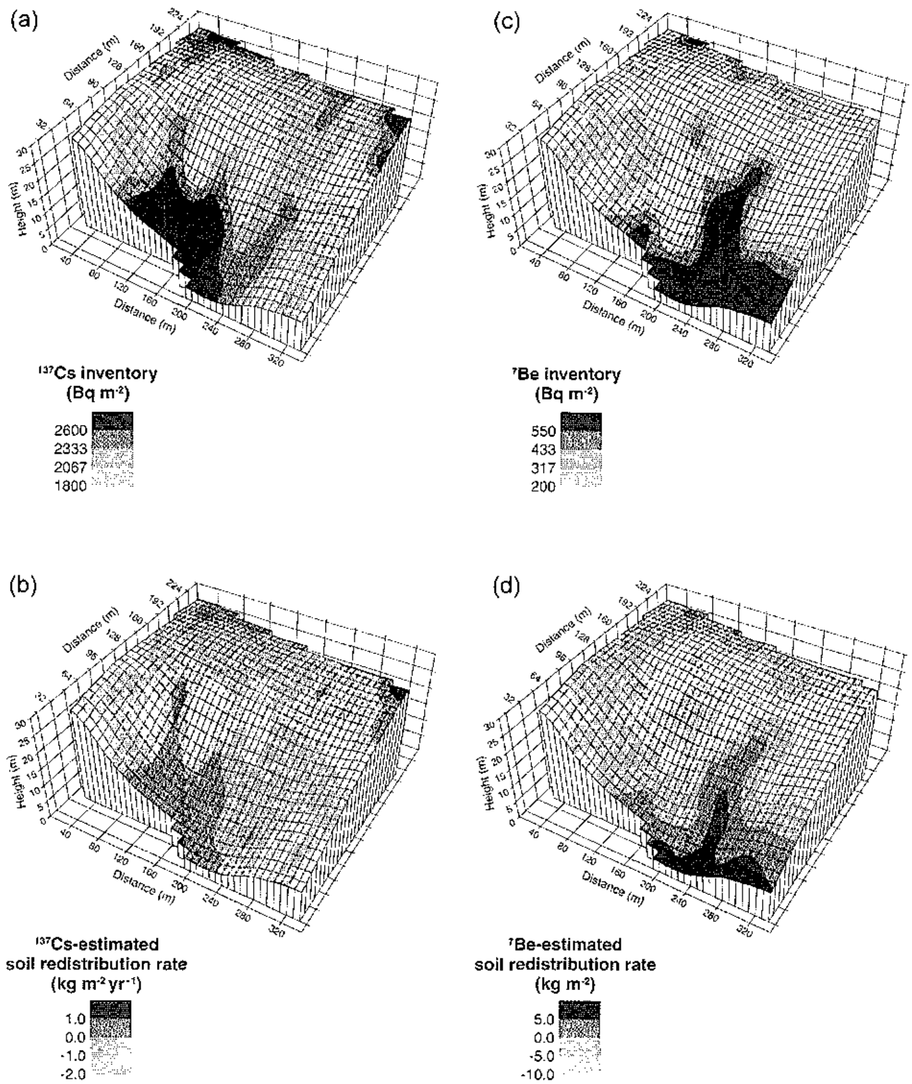

Fig. 3 The spatial distribution of ^{137}Cs and ^{7}Be inventories within a field at Higher Walton Farm, near Crediton Devon, and of the estimates of soil redistribution rates derived from these measurements.

Table 1 A comparison of rates of soil redistribution within the study field at Higher Walton Farm estimated from ^{137}Cs and ^{7}Be measurements on the soil cores collected from the field.

Measure	^{137}Cs (kg m^{-2} year^{-1})	^{7}Be (kg m^{-2})
Range	–4.5 to +2	–11.9 to +9.8
Mean erosion rate for eroding area	–1.1	–5.3
Mean deposition rate for depositional areas	0.69	4.0
Net soil loss	–0.48	–2.5
Sediment delivery ratio	0.83	0.80

Based on Walling *et al.* (1999)

resulting pattern has been mapped in Fig. 3(b) and the data have been summarized in Table 1, which presents values for the range of soil redistribution rates, the mean erosion rate for the eroding areas, the mean deposition rate for the depositional areas, the net soil loss from the field and its sediment delivery ratio. The latter value is of considerable importance in establishing a sediment budget, since it provides an estimate of the relative proportions of the mobilized sediment which have been transported beyond the field or redeposited within the field. Such information is extremely difficult to obtain using conventional monitoring techniques.

The spatial distribution of ^{7}Be inventories within the study field measured at the end of the period of heavy rainfall in early January 1998 is presented in Fig. 3(c). The equivalent value for the local reference inventory was estimated to be 533 Bq m^{-2} and the pattern shown in Fig. 3(c) provides clear evidence of areas with both reduced and increased inventories, which in turn reflect both erosion and deposition within the field. In order to interpret this pattern in terms of soil redistribution rates associated with the period of heavy rainfall in early January, it is important to consider the extent to which it may reflect spatial variability inherited from previous erosion events. In this case, however, the preceding autumn and early winter had been relatively dry and there was no evidence of surface erosion having occurred during the previous 6 months. In this situation, the inventory within the field immediately prior to the period of heavy rainfall in early January 1998 could be expected to be spatially uniform, since it would reflect the wet fallout associated with rainfall over the preceding months. Any spatial variability produced by soil redistribution associated with erosion events much earlier in 1997 would have been rendered insignificant by radioactive decay, due to the short half-life of ^{7}Be. It is therefore possible to assume that the spatial variability in ^{7}Be inventories within the study field evident in Fig. 3(c) reflects soil redistribution associated with the period of heavy rainfall in early January 1998. By relating the increase or decrease in inventory to the reference inventory and knowing the depth distribution of ^{7}Be in uneroded soil within the field, it is possible to estimate the soil redistribution rates (cf. Walling *et al.*, 1999). The resulting pattern of soil redistribution rates is presented in Fig. 3(d) and summary data, equivalent to that provided for the ^{137}Cs measurements, are listed in Table 1.

The soil redistribution rates (kg m^{-2}) associated with the period of heavy rainfall in early January 1998 estimated from the ^{7}Be measurements are substantially higher than the equivalent longer-term mean annual soil redistribution rates estimated using the ^{137}Cs measurements. However, the sediment delivery ratios are closely similar, indicating that ~80% of the eroded sediment was transported out of the field. The high sediment redistribution rates associated with the period of heavy rainfall in early January 1998 reflect both the extreme nature of this period of rainfall and, perhaps more importantly, the condition of the field, which having been compacted by the maize harvesting machinery and left bare after the harvest, was particularly susceptible to surface runoff and erosion. Such results underscore the potential significance of a small number of extreme events and the incidence of particular land use conditions in controlling erosion from the study field.

In the example presented above, a large number of cores was used to establish the pattern of soil redistribution within the study field. It is clearly impossible to extend sampling at this intensity to more than a few fields and, if a sediment budget is to be constructed for a larger area, it will be necessary to design a sampling strategy which focuses on representative areas and permits extrapolation of the results to a wider area. An example of this approach is provided by the work of the author and his co-workers in the 63 km^2 Upper Kaleya catchment in Zambia, which aimed explicitly to develop a reconnaissance level approach to the establishment of a catchment sediment budget (cf. Walling *et al.*, 2001, 2003). In this study, measurements of both ^{137}Cs and unsupported ^{210}Pb were included, in an attempt to exploit the potential for using unsupported ^{210}Pb to obtain additional information on longer-term trends in erosion and sediment delivery within the catchment. Thus, the information on medium-term erosion rates over the past ~40 years provided by the ^{137}Cs measurements can be compared with that provided by the unsupported ^{210}Pb measurements, which will reflect a longer period extending back 100 years.

Figure 4 presents a map of the Upper Kaleya catchment, which is located near Mazabuka in southern Zambia. The local climate is characterized by distinct wet (November–March) and dry (April–October) seasons and the mean annual precipitation lies in the range 800–900 mm. Catchment land use is dominated by three main zones (cf. Fig. 4). Communal agriculture occupies 69% of the catchment and is characterized by the cultivation of maize, cotton, groundnuts and sunflowers. Bush grazing of communal lands extends over 29% of the catchment, and the remaining 2% is devoted to commercial farming, with emphasis on crops such as coffee, potatoes, wheat and vegetables. In order to obtain representative information on soil redistribution rates within each of the three land use types, two typical areas of each land use were selected (cf. Fig. 4). Within each of these areas, a pair of parallel downslope transects was established, with the individual transect lines being 30–40 m apart. Cores were collected at approximately equidistant points (~10–12 m intervals) along the transects, with replicate cores being collected at each sampling point and bulked into a single bag. All cores were collected to depths of ~45 cm using a motorized percussion corer equipped with a 6.9 cm internal diameter core tube. A total of 206 bulk cores were collected from these transects. Additional samples were collected from a reference site, in order to establish the local reference inventories for both ^{137}Cs and unsupported ^{210}Pb.

Conversion models were used to derive estimates of the erosion or deposition rates for the individual cores based on both their ^{137}Cs and unsupported ^{210}Pb inventories, and these values were in turn used to establish the gross and net erosion rates for the four individual transects associated with each land use. Combination of the results from the four transects provided a means of establishing the sediment budget for each land use zone. The results are presented in Fig. 5. These results indicate that the gross erosion rates associated with the areas of commercial and communal cultivation are similar (~7 t ha^{-1} $year^{-1}$), whereas those for bush grazing are somewhat lower (~4 t ha^{-1} $year^{-1}$). Greater differences are apparent between the estimates of net soil loss from the individual land use zones, with the highest values being associated with the areas of commercial cultivation and the lowest with the areas of communal cultivation. The small size and scattered nature of the individual plots within the areas of communal cultivation limits the downslope transfer of sediment and results in a low sediment delivery ratio, whereas the dense network of paths crossing the areas of bush grazing promotes sediment transfer and results in a much higher sediment delivery ratio (>80%).

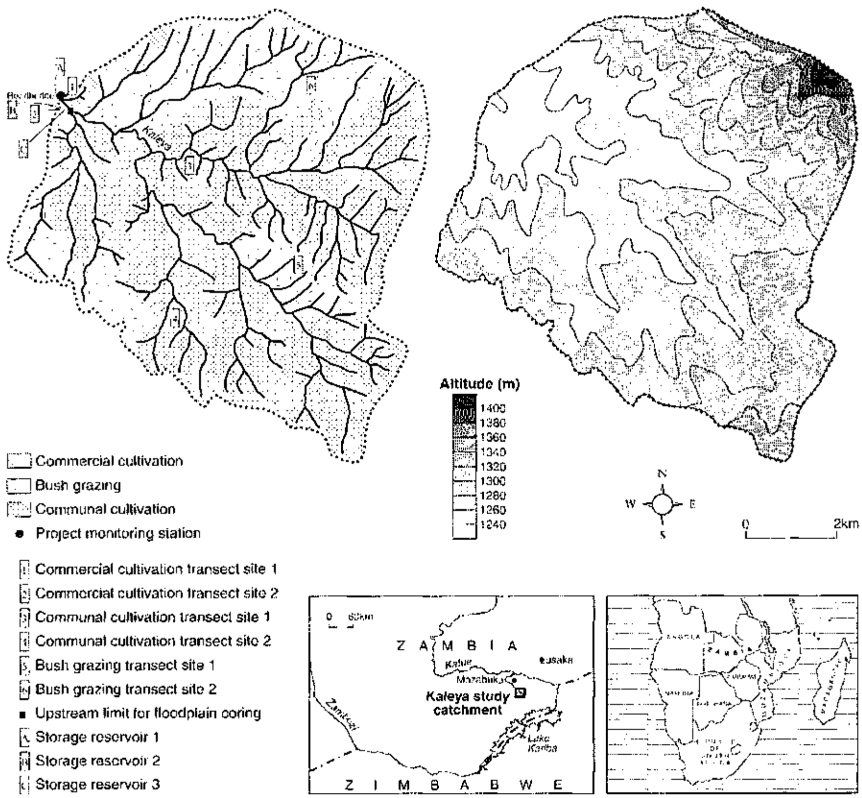

Fig. 4 The Upper Kaleya catchment and its location.

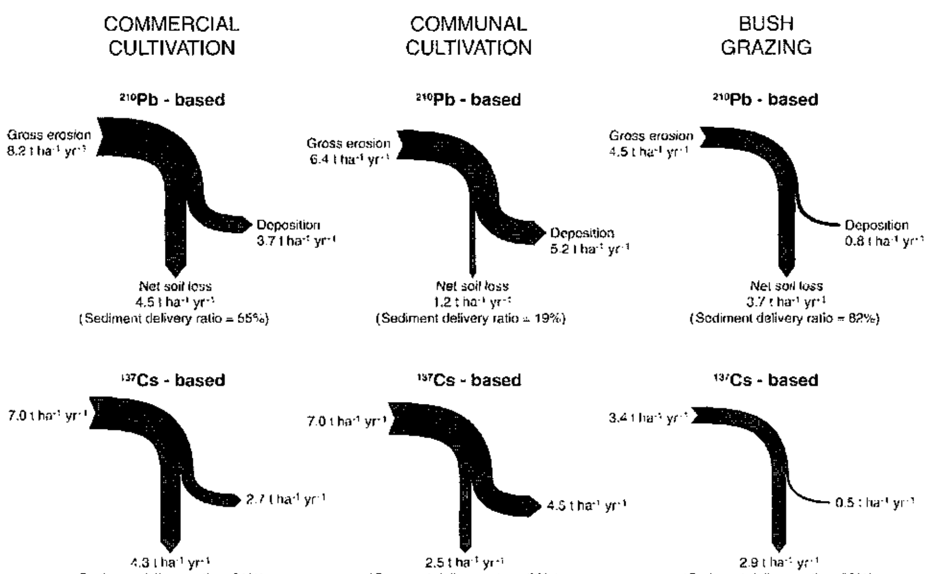

Fig. 5 A comparison of typical sediment budgets for land under commercial cultivation, communal cultivation and bush grazing in the Upper Kaleya catchment, Zambia derived using unsupported ^{210}Pb and ^{137}Cs measurements.

Comparison of the budgets derived using the ^{137}Cs and unsupported ^{210}Pb measurements provides a basis for assessing longer-term changes in erosion and sediment delivery within the Upper Kaleya catchment. The results provided by the ^{137}Cs measurements will be biased towards the present, relative to those derived from the unsupported ^{210}Pb measurements. Comparison of the budgets derived from the ^{137}Cs and unsupported ^{210}Pb measurements suggest that erosion rates have remained relatively constant within the areas of commercial cultivation, which is consistent with the early introduction of soil conservation measures into such farming areas in the early part of the 20th century. In contrast, both gross and net soil loss from areas of communal cultivation appear to have increased towards the present. This is consistent with the gradual breakdown of soil conservation programmes, introduced by the colonial government in the first part of the 20th century, since independence. In the case of bush grazing, the trend appears to be reversed, with both gross and net erosion rates declining towards the present. However, this is consistent with the known reduction in livestock numbers in recent years, due to rural depopulation, which has reduced grazing pressure.

ASSESSING FLOOD PLAIN SEDIMENTATION RATES AND ASSOCIATED CONVEYANCE LOSSES

Overbank deposition on river flood plains during flood events represents an important potential sink for suspended sediment transported through a river system and recent studies have demonstrated that such conveyance losses can be as high as ~40% of the suspended sediment load delivered to the main channel system (cf. Walling & Owens, 2002). In view of the potential importance of such conveyance losses in the overall sediment budget for a catchment, there is a need to quantify rates of overbank sedimentation on river flood plains and the use of environmental radionuclides has been shown to offer considerable potential in this context. As an example, Fig. 6 shows how ^{137}Cs and unsupported ^{210}Pb measurements have been used to document overbank sedimentation rates along a short reach of the flood plain of the River Severn near Buildwas in Shropshire, UK. In this study, 124 sediment cores were collected at the intersections of a 25 m × 25 m grid using a motorized percussion corer equipped with a 6.9 cm internal diameter core tube. Cores were collected to a depth of ~70 cm to ensure that they included the complete ^{137}Cs and unsupported ^{210}Pb profiles. Measurements of the ^{137}Cs and unsupported ^{210}Pb inventories of the cores were used to estimate the mean annual sedimentation rates at the coring points using the procedures documented by Walling & He (1997) and He & Walling (1996). These estimates have in turn been used to map the patterns of sedimentation within the reach shown in Fig. 6. These patterns reflect the interaction of a number of controls on the sedimentation rate, including depth of inundation, distance from the channel and the local microtopography. Again it is also possible to compare the sedimentation rate estimates derived from the ^{137}Cs measurements, which relate to the past ~40 years, with those based on the unsupported ^{210}Pb measurements, which relate to the past ~100 years, in order to assess longer-term changes in sedimentation at this location. The mean annual sedimentation rate at this site over the past 40 years is 0.28 g cm^{-2} $year^{-1}$, whereas the equivalent rate for the past 100 years is 0.33 g cm^{-2} $year^{-1}$. This suggests that rates of

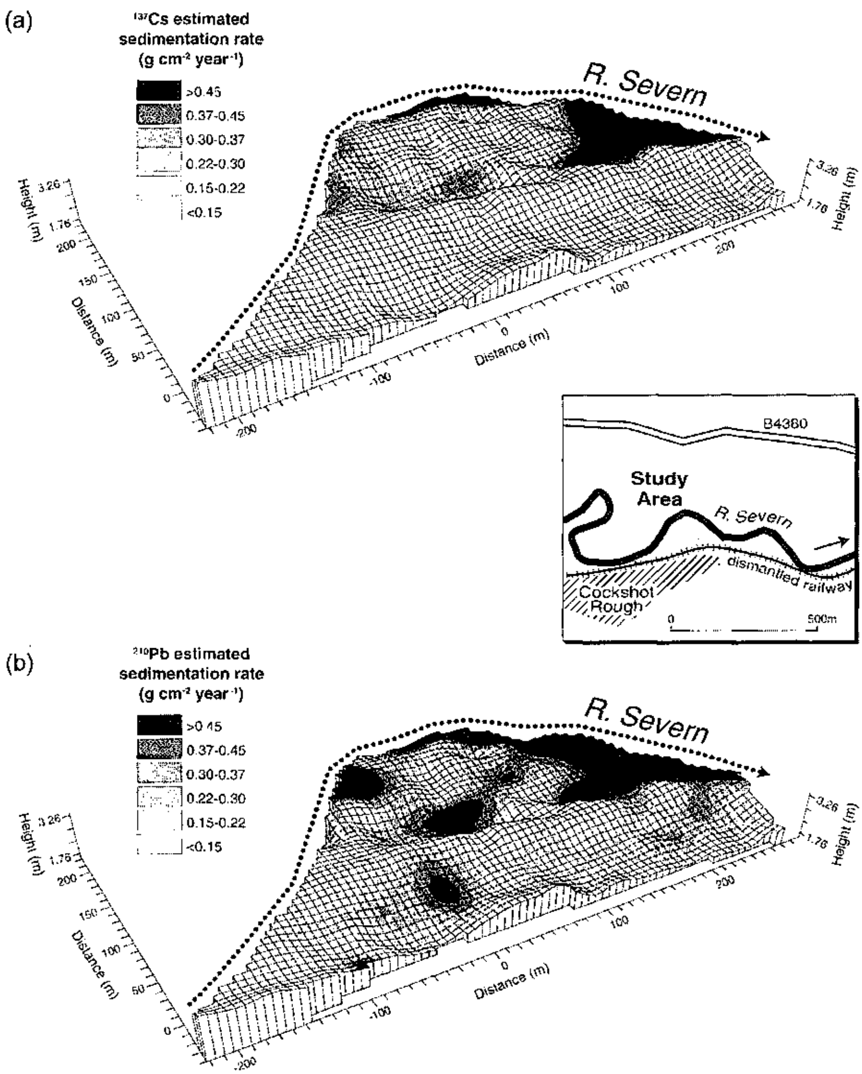

Fig. 6 The spatial distribution of overbank sedimentation rates within a small reach of the flood plain of the River Severn near Buildwas, Shropshire, UK derived from ^{137}Cs and unsupported ^{210}Pb measurements undertaken on flood plain cores.

overbank sedimentation have decreased slightly towards the present. This decrease could reflect changes in the magnitude and frequency of flood inundation, but it might also reflect a progressive reduction in inundation depths as the level of the flood plain surface gradually increased in height as a result of overbank deposition.

Although information on the magnitude and detailed pattern of overbank deposition rates within a specific river reach, as shown in Fig. 6, will be of value in some sediment budget investigations, in other investigations a more general estimate of the magnitude of the total conveyance loss associated with overbank deposition on the flood plains bordering the main channel system of a river basin may be required. In

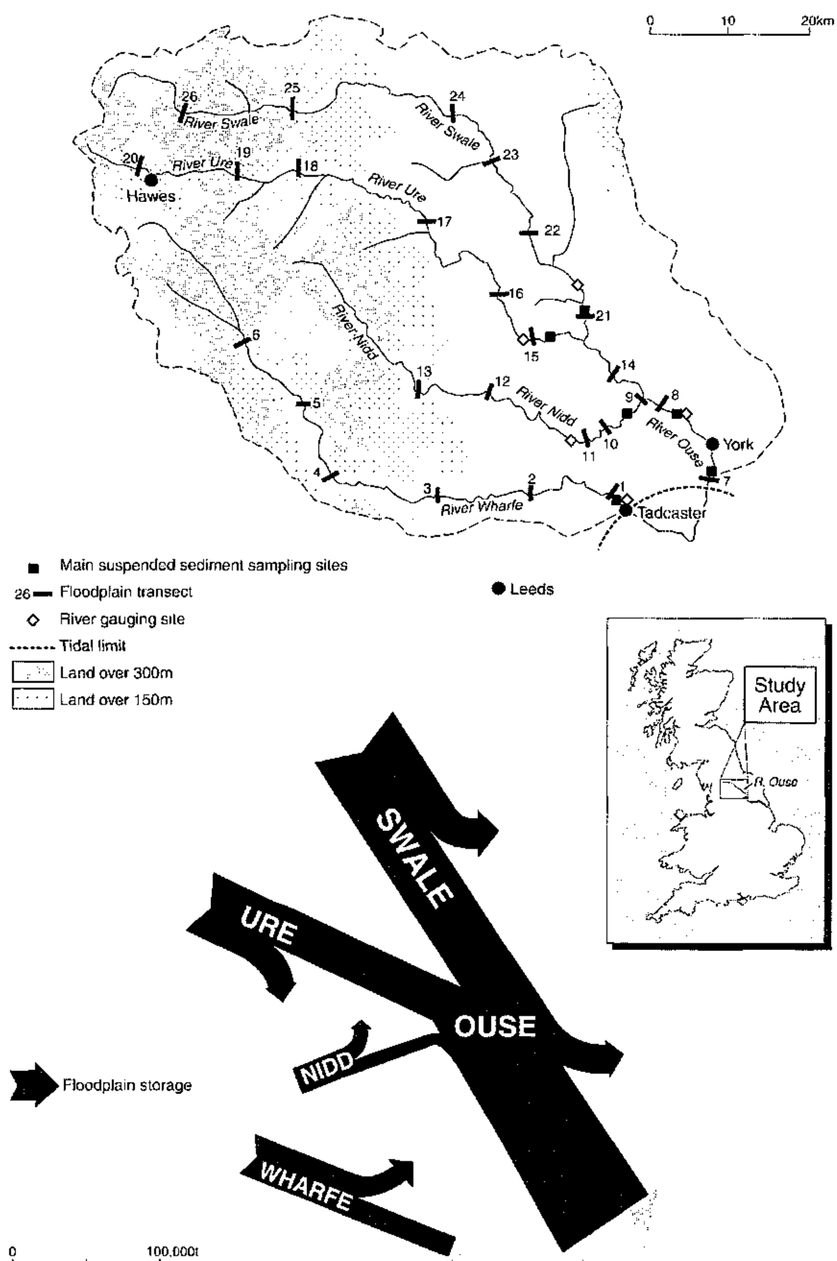

Fig. 7 The Catchment of the Yorkshire Ouse and its major tributaries in Yorkshire, UK, showing the location of the flood plain transects used for collection of cores for ^{137}Cs measurements and the sediment budget established for the main channel system within the catchment. (Based on Walling *et al.*, 1998.)

this case there will be a need to extrapolate the findings from individual reaches or cross-sections to the entire main channel system. An example of the potential of this approach is provided by the results of an investigation of the role of flood plain storage

Table 2 A comparison of estimates of total storage of sediment on the flood plains of the main channel systems of the River Ouse and its major tributaries and the River Wharfe with the estimated suspended sediment loads of these rivers.

River	Flood plain storage (t $year^{-1}$)	Annual suspended sediment load (t $year^{-1}$)	Total sediment delivered to channel (t $year^{-1}$)*	Conveyance loss (%)
Swale	19214	42352	61566	31.2
Nidd	7573	7719	15292	49.5
Ure	15125	28887	44012	34.4
Total Ouse	49041	75111	124152	39.5
Wharfe	10325	10816	21141	48.8

Based on Walling *et al.* (1998)

* The total amount of fine sediment delivered to the main channel system has been estimated as the sum of the suspended sediment load at the downstream monitoring station and flood plain storage.

in the suspended sediment budget of the main channel systems of the rivers Ouse and Wharfe in Yorkshire, UK, reported by Walling *et al.* (1998). In this case, more than 250 sediment cores were collected from 26 representative transects located along the main channel systems of the Yorkshire Ouse and the River Wharfe (cf. Fig. 7) for ^{137}Cs analysis. The estimates of average sedimentation rates for the individual transects obtained from these cores were extrapolated to the individual reaches between adjacent transects and the mean annual conveyance losses associated with overbank sedimentation on the flood plains bordering the main channel system were calculated. By comparing the magnitude of these losses with the mean annual suspended sediment loads of the rivers monitored at the primary gauging stations (Table 2), it was possible to establish the relative importance of flood plain storage in the sediment budget of the main channel system (cf. Fig. 7). In the case of the main River Ouse system, flood plain deposition accounts for ~40% of the total amount of suspended sediment delivered to the main channel system over the past 40 years and for the River Wharfe the equivalent value is ~49%.

SEDIMENT SOURCE FINGERPRINTING

A good example of the potential for using environmental radionuclides, and more particularly ^{137}Cs, to establish the relative importance of a number of potential sediment sources within a drainage basin is provided by the work of Zhang *et al.* (1997) undertaken in the 3.86 km^2 Zhaojia gully catchment located in the rolling loess plateau region of Zichan county, Shaanxi Province, China. The catchment can be divided into two geomorphological units representing the rolling plateau surface and the gully itself, which account for 53% and 47% of the area respectively (cf. Fig. 8). There was a need to establish the relative importance of the cultivated areas of the rolling plateau and the steep slopes of the main gully area, in contributing to the sediment output from the basin. Such information is clearly of importance in targeting soil conservation and sediment control measures. A check dam had been constructed across the channel in the middle portion of the catchment in early 1991 (cf. Fig. 8) and when the catchment was visited in October 1993 the deposits associated with two floods occurring during the period July–August 1993 were clearly distinguishable in sediment cores collected from the area of sediment storage. The total depth of sediment deposition associated with the two events was ~33 cm. Based on an analysis of three cores, the mean ^{137}Cs content of sediment deposited by the first flood was 0.74 Bq kg^{-1}, whereas that for the second flood was 1.06 Bq kg^{-1}.

In attempting to decipher the main sources of the deposited sediment, attention was directed to variations in the ^{137}Cs concentration of potential sediment sources within the catchment. Mean ^{137}Cs concentrations associated with these sources exhibited significant variations and offered an effective means of discriminating such sources. A summary of the ^{137}Cs content of potential source materials in the Zhaojia Gully catchment is provided in Table 3.

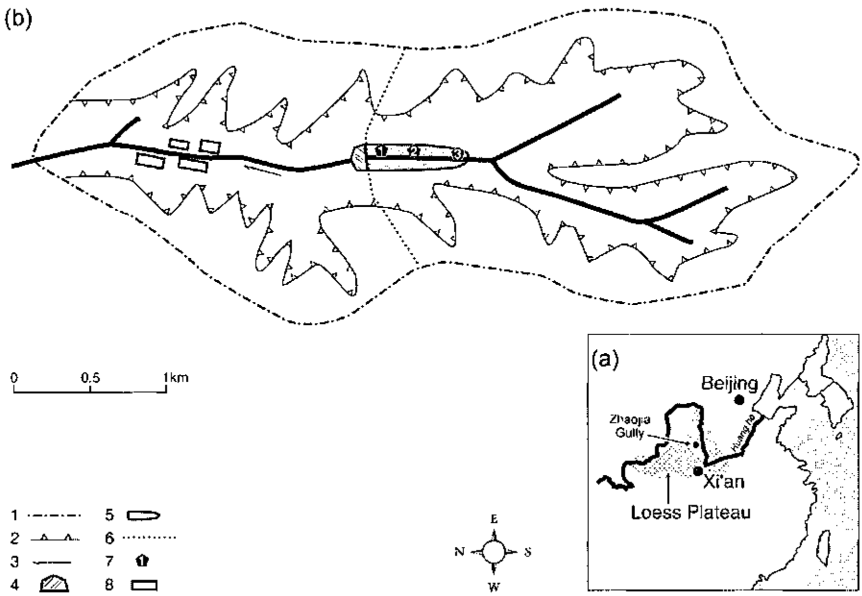

Fig. 8 The Zhaojia Gully catchment showing its location (a) and its main features (b). 1, catchment divide; 2, the boundary between the rolling plateau and the gully areas; 3, the main channel; 4, the check dam; 5, area of sediment deposition associated with the check dam; 6, divide of the catchment contributing to the sediment trap reservoir; 7, sampling sites for sediment deposits; 8, village. (Based on Zhang *et al.*, 1997)

If the average ^{137}Cs content of the sediment deposits associated with the two floods occurring in 1993 is compared with that of potential source material collected shortly after the flood (cf. Table 3), it can be seen to be similar to that of cultivated soil from the steep gully slopes, lower than that of soil from the cultivated land on the plateau crests and steeper lower slopes of the rolling plateau, and considerably lower than that of topsoil from the grass covered areas. It is also very substantially higher than that of surface material from steep bare slopes and from recent mass movement deposits within the gully area. In view of either their limited extent or their low erosion rates, it is unlikely that the cultivated land on the steep gully slopes or the grassland within the gully area, represent important sediment sources and the cultivated areas of the rolling

Table 3 The ^{137}Cs content of potential source materials in the Zhaojia Gully catchment from different topographic locations and under different land use conditions.

Zone	Source material and land type	% catchment area	No. of samples	Mean ^{137}Cs content (Bq kg^{-1})	Erosional behaviour
Gully (47%)	Surface soil on grassland of gully slopes	21	8	7.0	Slight sheet and rill erosion
	Ploughed soil on steep gully slopes	2	7	1.2	Very severe sheet and rill erosion
	Surface soil on bare slopes and mass movement deposits	19	7	0.02	Very severe sheet and rill erosion on bare slopes and active mass movements
	Village area, cultivated land on terrace and gully floor, tracks	5	–	–	
Rolling plateau (53%)	Cultivated soil of crest slopes	14	36	6.4	Moderately severe sheet and rill erosion
	Cultivated soil of lower slopes	36	32	3.4	Severe sheet and rill erosion
	Surface soil on grassland	3	–	–	

Based on Zhang *et al.* (1997).

plateau and the steep bare slopes and recent mass movement deposits within the gully area are likely to represent the primary sediment sources. Taking account of the relative extents of the cultivated areas on the crests and the steeper upper slopes of the rolling plateau and estimates of erosion rates obtained from these areas using ^{137}Cs measurements, a weighted average ^{137}Cs content for sediment mobilized from these cultivated areas of the rolling plateau of 3.9 Bq kg^{-1} was estimated. If it is also assumed that the average ^{137}Cs content of surface material from the steep bare gully slopes and recent mass movement deposits (0.02 Bq kg^{-1}) is representative of sediment from the gully area and the minor contributions from the cultivated areas and areas of grass cover within the gully are ignored, a simple mixing model can be used to estimate the relative contribution of sediment derived from the gully area and the cultivated areas of the rolling plateau to the flood deposits, i.e.:

$$C_d = C_p \times f_p + C_g \times f_g \tag{1}$$

where C_d represents the average ^{137}Cs content of the flood deposits (Bq kg^{-1}), C_p and C_g represent the mean ^{137}Cs content of the sediment derived from the rolling plateau and the gully area, respectively, and f_p and f_g represent the relative contributions (fraction) from the rolling plateau and the gully areas, respectively.

The results of applying equation (1) to the sediment deposits resulting from the 1993 floods, using the data presented above, are listed in Table 4. These indicate that during the first flood 18.6% of the sediment was derived from the rolling plateau and 81.4% from the gully area. During the second event, the equivalent values were 26.8% and 73.2%. Taking the two events together, the equivalent values were 23% and 77%.

Table 4 The relative contribution of sediment from the rolling plateau and gully areas of the Zhaojia Gully catchment during the floods of 1993.

Flood	Date of flood	Deposit thickness (cm)	^{137}Cs content of sediment deposits (Bq kg^{-1})	Relative contribution (%) Rolling plateau	Gully
2	21 August 1993	18.3	1.06	26.8	73.2
1	21 July–1 August 1993	14.7	0.74	18.6	81.4
Total	–	33	mean = 0.9	23.0	77.0

Based on Zhang *et al.* (1997).

These results emphasize the importance of the gully area and the associated gully erosion and mass movement processes as the main sediment source within the Zhaojia Gully catchment. Any strategy aimed at reducing sediment yields from this catchment should clearly focus on reducing sediment production within the gully area.

In this example, a comparison of the ^{137}Cs content of the suspended sediment and of potential suspended sediment sources has been used to establish the relative contributions from the two major potential sources. In other source fingerprinting studies it has proved advantageous to employ several sediment properties within a composite fingerprint, in order to increase the discrimination between potential sources and to consider more than two sources. A multi-component mixing model is then used to establish the relative contributions from the individual sources (cf. Walling *et al.*, 1993; Walling & Woodward, 1995; Collins *et al.*, 1997). Fallout radionuclides are commonly included in such composite fingerprints, since they are particularly valuable for discriminating surface and subsurface sources and surface material from areas under different land use (cf. Collins & Walling, 2002).

PERSPECTIVE

The case studies described above provide several examples of the potential for using fallout radionuclides as tracers in sediment budget investigations. However, in each case they focus on a particular component of the sediment budget. In many investigations the ultimate aim will be to establish the overall catchment sediment budget and it is important to recognize that the results obtained from studies of the individual components using fallout radionuclides can be combined to establish the overall sediment budget of a catchment. The use of the same radionuclide tracer in studies of the individual components will clearly facilitate this exercise.

The work of Walling *et al.* (2003) in the Upper Kaleya catchment in southern Zambia provides an example of how the results from investigations of several components of the catchment sediment budget can be combined. The use of ^{137}Cs measurements to establish sediment budgets for areas under different land use was described above (cf. Fig. 5). These results were combined with measurements of the annual suspended sediment flux at the catchment outlet, fingerprinting of the sediment load at the catchment outlet, estimation of conveyance losses associated with overbank sedimentation on the river flood plain, and measurements of sediment accumulation in a number of small reservoirs to construct the sediment budget shown in Fig. 9. By summing the estimates of mean annual suspended sediment load at the catchment outlet, flood plain conveyance loss and reservoir sedimentation it was possible to estimate the total amount of sediment delivered to the main channel system. The source fingerprinting results were used to apportion this load to surface erosion within the three main land use types and to channel erosion. Comparison of these load components with the outputs from the sediment budgets for the individual land use areas (cf. Fig. 5), scaled to the areas involved, provided an estimate of the conveyance losses associated with sediment delivery from the slopes to the channel network.

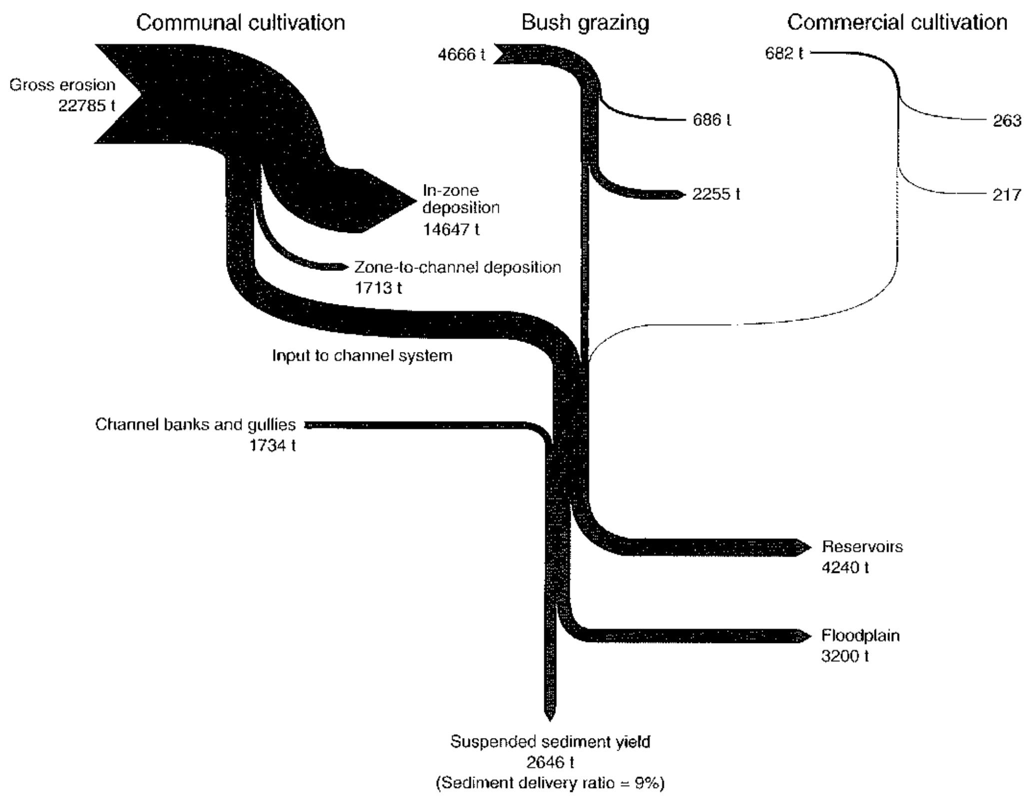

Fig. 9 The catchment sediment budget derived for the Upper Kaleya catchment, Zambia, by Walling *et al.* (2001).

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