QUANTIFYING THE FINE SEDIMENT BUDGETS OF RIVER BASINS

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INTRODUCTION

Traditionally, interest in erosion and sediment transport problems in river basins has focussed on issues relating to on-site soil degradation and loss of productivity and downstream impacts associated with reservoir sedimentation and the resulting loss of storage, siltation of water intakes, irrigation canals, navigation channels and harbours, as well as the problems for water use caused by high levels of suspended solids (see Sundborg, 1982). Against this background, erosion and sediment transport and the development of soil conservation programmes attracted little attention in countries such as the UK, where soil erosion was perceived to be of limited significance and both suspended sediment loads and concentrations were low by world standards. As a result, little effort has been directed to monitoring sediment loads in UK rivers and, in contrast to many other countries, such as the USA, Canada, Russia and Germany, there is no national sediment monitoring network. In consequence, relatively little is known about the fine sediment dynamics of UK rivers.

More recently, a growing recognition of the important role of fine sediment in the transfer, storage and fate of sediment-associated nutrients and contaminants, including phosphorus, pesticides, heavy metals and persistent organic pollutants (cf. Allan, 1986) and in the more general degradation of aquatic habitats, including the siltation of fish spawning gravels, (cf. Clark et al., 1985; Sear, 1993; Soulsby et al., 2001) has focussed attention on the wide-ranging environmental and ecological significance of fine sediment and the wider need to incorporate effective sediment control strategies into catchment management. In the UK, for example, recent concern for the improvement of river water quality and the ecological status of aquatic habitats prompted by the EC Water Framework and Habitats Directives has now identified fine sediment as a key contributor to diffuse source pollution and the degradation of aquatic habitats and emphasised the need to control sediment mobilisation and delivery to water courses, even though rates of soil loss and specific suspended sediment yields are low by world standards. This growth of interest in fine sediment in UK rivers has served to emphasise the inadequacy of both existing knowledge and current monitoring programmes and practices and these deficiencies are now being increasingly recognised.

Monitoring Suspended Sediment Loads and Concentrations

Information on the suspended sediment load of a river, which can be used to calculate an equivalent value of specific suspended sediment yield for the associated river basin, and on suspended sediment concentrations, must be seen as a key requirement in any attempt to understand the fine sediment dynamics of a river basin, by quantifying the amounts of sediment involved. It should, however, be recognised that such information is difficult to assemble and requires a carefully designed monitoring programme aimed explicitly at generating reliable suspended sediment load data. In many countries (such as the UK), most of the available information on suspended sediment transport is limited to data provided by standard water quality sampling programmes based on regular sampling at weekly or even monthly intervals. The key factor underlying the problem of deriving accurate estimates of fine sediment flux is that significant suspended sediment transport is primarily associated with storm events, when both concentrations and flows are high. Typically, in a medium-sized UK river more than 80% of the annual suspended sediment load will be transported in less than 2% of the time i.e. typically ca. 7 days or 170 hours and it is very unlikely that such periods will be covered by regular periodic sampling programmes. As a result, the various extrapolation and interpolation procedures that have been developed to estimate suspended sediment loads from infrequent samples (see Walling and Webb, 1985, 1988; Phillips et al., 2000) are lacking in both accuracy and precision. An indication of the potential magnitude of the resulting errors in sediment load estimates is provided by a recent study undertaken in the catchments of the Hampshire Avon and the Wye (Webb and Walling, unpublished), where it was possible to compare the various estimates of annual suspended sediment load that might have been provided by typical infrequent sampling programmes with accurate estimates of the annual sediment load obtained using continuous records of suspended sediment concentration derived from recording turbidity meters. By combining these records with the continuous flow records, it was possible to establish both the 'true' load and to derive replicate datasets representative of typical infrequent periodic sampling programmes, that could be used with various load calculation procedures to generate what could be seen as 'typical' load estimates. The results demonstrated that loads were generally underestimated by infrequent sampling, with median errors of ca. 40% or even more being common. However, the range associated with the replicate estimates was also large and typically ca. 50%, although the replicate estimates of annual load for a given sampling site could vary by as much as 600%, depending on the time that the samples were collected. The considerable variability in the replicate estimates of annual load emphasises the lack of precision associated with such estimates and thus the limited potential to apply simple correction factors to take account of over- or under-estimation. The above results serve to emphasise both the inadequacy of any estimates of suspended sediment yield derived from infrequent samples and the need for intensive sampling programmes or recording turbidity meters calibrated to provide sediment concentration, if accurate estimates of suspended sediment yield are required. Against this background, only limited information concerning fine sediment transport by UK rivers is currently available.

The catchment sediment budget as an integrating framework

Notwithstanding the problems of obtaining reliable sediment load data, most existing studies of suspended sediment delivery in areas such as the UK have focussed on measuring the load at the catchment outlet and estimating the sediment yield, in order to provide a spatially-lumped estimate of the rate of sediment generation from the catchment surface (i.e. t km^{-2} $year^{-1}$). However, recognition of the wider environmental significance of fine sediment mobilisation, transfer and storage has directed attention to the internal functioning of the catchment and the need to obtain information on sediment sources and mobilisation rates, sediment transfer pathways and the efficiency of such transfers, and the storage of mobilised sediment in both long-term and short-term sinks. These considerations can usefully be drawn together in the concept of the catchment sediment budget (cf. Dietrich and Dunne, 1978; Trimble, 1983; Walling et al., 2002), which identifies sources, transfers, sinks and outputs and quantifies the fluxes involved. Despite providing a valuable framework or tool for sediment studies in catchments, application of the sediment budget concept is hampered by the inability of traditional monitoring techniques to provide the information necessary to establish a meaningful sediment budget. In more recent years, this deficiency has been addressed by developing techniques for tracing sediment movement to complement existing monitoring techniques. This shift from monitoring to tracing, albeit with the former complementing, rather than replacing, the latter, has been identified as a new paradigm in sediment studies (cf. Walling, 2003) reflecting both new perspectives or questions, as well as technological and methodological advances in tracing techniques.

Using environmental radionuclides as tracers in sediment budget investigations

In applying sediment tracing techniques to assist in establishing and understanding the fine sediment budgets of catchments and river basins, the use of environmental radionuclides as sediment tracers has proved particularly effective and useful (cf. Walling, 2004). Most of such work has involved caesium-137 (Cs-137) (e.g. Zapata et al., 2002), but the potential for using excess lead-210 (excess Pb-210) and beryllium-7 (Be-7) has also been successfully demonstrated (cf. Walling and He, 1999a; Walling et al., 1999). By virtue of their different fallout histories and properties, these three environmental radionuclides together provide a valuable 'toolkit' for sediment tracing. All three radionuclides enter the soil-sediment system as fallout, mainly in association with precipitation, and the primary rationale for their use as sediment tracers is that the fallout arriving at the soil surface is rapidly and strongly adsorbed by the surface soil or sediment and its subsequent redistribution will reflect the redistribution of those particles by erosion, transport and deposition.

Cs-137 is an artificial radionuclide released into the stratosphere by the testing of atomic weapons in the late 1950s and early 1960s and the period of significant fallout was limited to the period extending from 1955 until the late 1970s, although the Chernobyl accident provided additional fallout inputs in

1986 in some areas of Europe and adjacent regions. With a half-life of 30 years, a substantial proportion of the original Cs-137 fallout remains in the environment. In contrast to Cs-137, Pb-210 is a naturally occurring radionuclide. It is a product of the uranium-238 decay series with a half-life of 22.2 years, that is derived from the decay of gaseous radon-222 (Rn-222), the daughter of radium-226 (Ra-226). Ra-226 exists naturally in soils and rocks and the lead-210 in soils generated in situ by the decay of Ra-226 is termed supported Pb-210 and is in equilibrium with Ra-226. However, upward diffusion of a small portion of the Rn-222 produced in the soil and rock introduces Pb-210 into the atmosphere and its subsequent fallout as Pb-210 provides an input of this radionuclide to surface soils and sediments that will not be in equilibrium with its parent Ra-226. Fallout Pb-210 is therefore commonly termed unsupported or excess Pb-210, when incorporated into soils and sediments, to distinguish it from the Pb-210 produced in situ by the decay of Ra-226. The amount of excess Pb-210 in a sample can be calculated by measuring both the Pb-210 and Ra-226 activities and subtracting the Ra-226-supported Pb-210 component from the total Pb-210 in the sample. With its half-life of 22.2 years, the activity of excess Pb-210 in soils and sediments will reflect fallout inputs and subsequent redistribution in association with soil and sediment particles over the past ca. 100years (i.e. ca. 4 half lives). Be-7 is again of natural origin, but this origin is cosmogenic and results from the bombardment of the earth's atmosphere by cosmic rays. Its half-life is only 53 days and is thus very much shorter than that of Cs-137 and excess Pb-210. This short half-life provides the basis for the use of Be-7 to trace sediment movement during individual events. All three of the above radionuclides can be measured relatively easily by gamma spectrometry (cf. Wallbrink et al., 2002), although the relatively long count times commonly involved (e.g.8 hours) and the cost of the measuring equipment must be seen as a potential limitation.

The basis for using these three environmental radionuclides as tracers is illustrated in Fig. 1, which illustrates their typical depth distributions in stable undisturbed pasture soils (a) and cultivated soils (b) and at depositional sites in uncultivated areas (c), in the UK. At undisturbed pasture sites (Fig. 1a), the radionuclides are typically concentrated close to the surface and concentrations decline exponentially with depth. The minor contrast between the Cs-137 and excess Pb-210 depth distributions reflects the continuous fallout input of excess Pb-210, which contrasts with that of Cs-137, which commenced in the mid 1950s and effectively ceased during the mid 1970s. Because of its shorter half-life, Be-7 is only found very near to the soil surface, where it is readily replenished by fresh fallout. Any of this radionuclide moving further down into the soil will soon disappear due to radioactive decay. The total radionuclide content of the soil profile per unit area is referred to as the inventory (Bq m^{-2}). With cultivated soils (Fig. 1b), tillage mixes Cs-137 and excess Pb-210 into the soil, producing near uniform concentrations within the plough layer. In many instances the total inventories associated with cultivated soils will be reduced relative to the undisturbed pasture areas, due to ongoing soil erosion, which will result in the loss of a proportion of the Cs-137 and excess Pb-210 contained within the plough layer. Removal of soil from the surface by erosion will cause soil from below the original plough layer containing no Cs-137 or excess Pb-210 to be incorporated into the plough layer, thereby reducing the concentrations within this layer. In the case of Be-7, however, mixing of the soil by cultivation would reduce the concentrations to very low levels, which would subsequently quickly decay and the near surface distribution will reflect recent fallout inputs to the soil surface. As with Cs-137 and excess Pb-210, the reduced Be-7 inventory associated with the cultivated soil reflects loss of Be-7 in association with the eroded soil, and such erosional loss is further reflected by the reduced depth to which Be-7 is found in the cultivated soil, relative to the pasture soil (cf. Fig. 1). At sites in the landscape where deposition occurs, both the depth distribution and total inventories of the three radionuclides will differ from those shown in Fig. 1a and 1b. Deposition of soil or sediment containing the radionuclides will cause both the depth to which the radionuclide is found and the total inventory to increase. This situation is illustrated in Fig. 1c, which depicts the depth profiles of the three radionuclides in sediment cores collected from a river floodplain in Devon, UK. In all cases, the soils are uncultivated and the profiles are therefore undisturbed by tillage mixing. Comparison of these depth profiles with those for undisturbed pasture in the same area unaffected by deposition, shown in Fig. 1a, provides a basis for identifying the effects of deposition on the profiles. In the case of Cs-137, the depth profile provides clear evidence of the progressive accretion of the floodplain above the level marked by the maximum Cs-137 activity, which represents the floodplain surface at the time of peak fallout in the mid 1960s. The response of the excess Pb-210 profile to progressive accretion shown in Fig. 1c differs from that shown by the Cs-137 depth profile, due to the continuous fallout input. In this case, progressive accretion is evidenced by a more gradual exponential decline in excess Pb-210 activity with depth and the greater depth of the excess Pb-210 profile, when compared with the core from undisturbed pasture, as well as an increased total inventory. In the case of Be-7, its short half-life means that contrasts between the profiles from the floodplain area and undisturbed pasture will only reflect very recent floodplain accretion. The Be-7 profile for the floodplain surface depicted in Fig. 1c was measured shortly after a sizeable flood had inundated the floodplain, causing significant deposition. The influence of this accretion is evident in both the increased inventory of the floodplain core and the great depth to which Be-7 is found in this core.

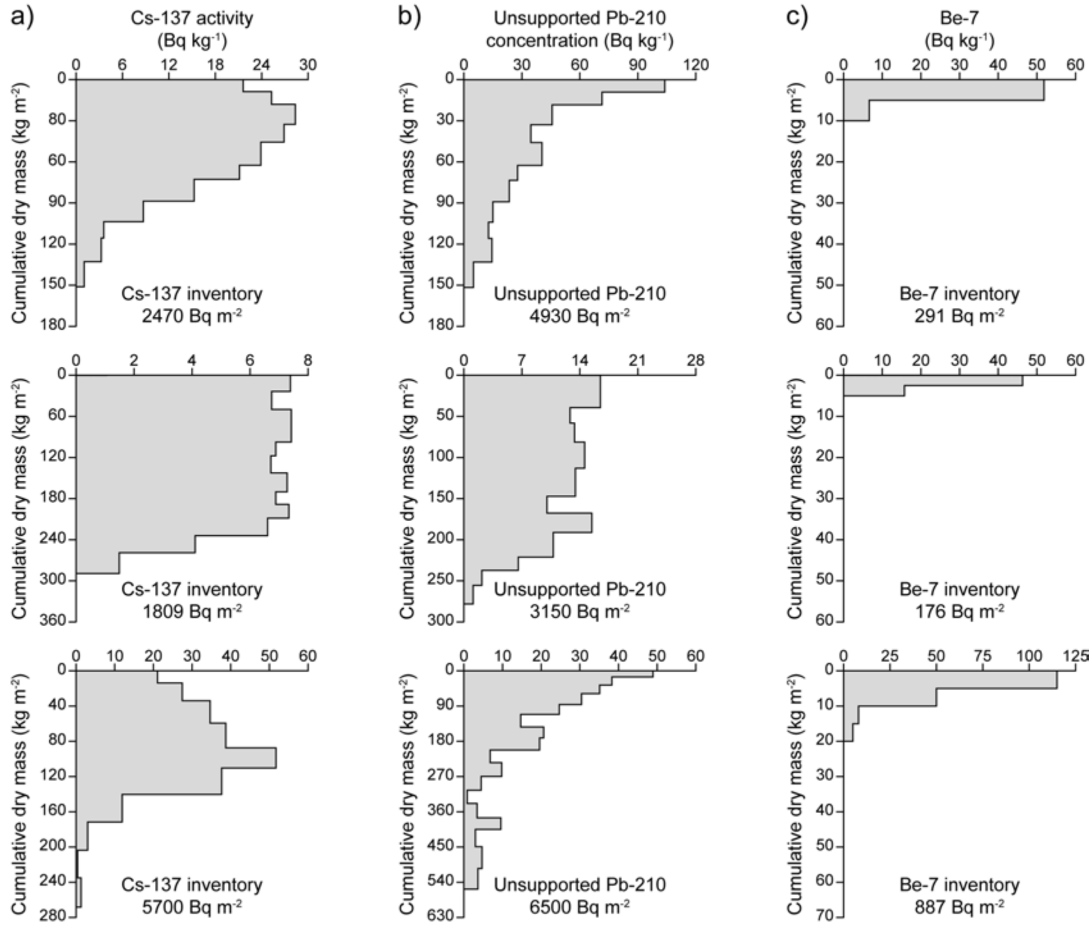

Fig. 1 Typical depth distributions of Cs-137, excess Pb-210 and Be-7 concentrations in undisturbed pasture (upper), cultivated soils (middle) and a depositional site under permanent pasture on a river floodplain in Devon, UK (lower).

The distinctive behaviour of Cs-137, excess Pb-210 and Be-7 at erosional and depositional sites, illustrated in Fig. 1 provides the basis for their use as tracers in documenting sediment mobilisation and delivery within river basins. Thus, for example, by collecting soil cores from a field, measuring their Cs-137, excess Pb-210 or Be-7 inventories and comparing these with the local reference inventory, representing the inventory expected in the absence of post-fallout remobilisation, 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 the erosion or deposition rate involved (cf. Walling and He, 1999a,b). For Cs-137 measurements, the resulting estimates of erosion and deposition rates will reflect erosion and deposition occurring over the last ca. 40-45 years (i.e. since the beginning of significant Cs-137 fallout), whereas for excess Pb-210 and Be-7 they will relate to longer and much shorter periods, respectively. With its half-life of 22.2 years and essentially continuous input, excess Pb-210 will provide estimates of erosion and deposition rates extending back over ca. 100 years (i.e. 4-5 half-lives), whereas for Be-7 the estimates could relate to a single event, when there has been little or no erosion in the preceding ca. 6 months. Similarly, the Cs-137, excess Pb-210 and Be-7 depth distributions and inventories found on river floodplains and in other depositional environments (e.g. Fig. 1c) afford a basis for estimating deposition rates. By collecting cores from such sites and determining their radionuclide depth distributions or, in simpler applications, comparing their total inventories with the local reference inventory, it is possible to establish both rates and patterns of sedimentation (cf. He and Walling, 1996; Walling and He, 1997; Blake et al., 2002). Again the time base of the estimates will vary according to the radionuclide involved. With Be-7 it is possible to obtain estimates of sedimentation rates associated with an individual events, whereas with Cs-137 the estimates will relate to a period of ca. 40-45 years and with excess Pb-210 the period involved will be still longer, although some workers have succeeded in breaking this down into shorter periods for which the associated deposition rate can be estimated.

The radionuclide behaviour illustrated in Fig. 1 can also be exploited in suspended sediment source tracing or 'fingerprinting' investigations. The fingerprinting approach (cf. Walling and Woodward, 1992,1995; 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 contrasts in grain size composition between the sediment and the potential sources. A key requirement of the approach is the need to identify a number of fingerprint properties that will clearly discriminate between several potential sources. Cs-137 and excess Pb-210 concentrations are particularly useful in this regard, since they 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 (i.e. cultivated and non-cultivated, (cf. Fig. 1). In the case of Be-7, significant concentrations of this radionuclide will only be found where the soil or sediment surface has been recently exposed to rainfall and thus Be-7 fallout, and the radionuclide will be absent from channel banks and other subsurface sources.

The potential application of environmental radionuclides in elucidating the fine sediment budget of a catchment can be further demonstrated by briefly considering several examples drawn from studies in UK river basins undertaken by the author and his co-workers in recent years. These include studies of soil erosion and sediment delivery from agricultural land, sediment source fingerprinting and the quantification of rates of overbank sedimentation on river floodplains. .

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 Cs-137 (cf. Ritchie and McHenry, 1990; Walling, 1998), both excess Pb-210 and Be-7 have also been used in similar applications (cf. Walling et al., 1999; Walling and He, 1999a). By collecting cores from a study site, measuring the Cs-137, excess Pb-210 or Be-7 inventories, comparing these with the reference inventories, and applying a conversion model, it is possible to derive point estimates of the erosion and deposition rates associated with the cores. By spatially integrating these values across the study site, it is possible to establish the relative importance of erosion and deposition and thus the gross and net erosion and the sediment delivery ratio. Fig. 2 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). In this case measurements of both Cs-137 and Be-7 activities were undertaken, with the former providing estimates of average rates of soil redistribution over the past ca. 40 years and the latter estimates of the erosion rates associated with a particular period of heavy rainfall (69 mm in 7 days) occurring in early January 1998.In this study, the soil cores used for the Cs-137 and Be-7 measurements were collected by two separate sampling campaigns, although they could have been collected together. In both cases the cores were collected at the intersections of a 20m x

20m grid, resulting in a suite of ca. 140 cores. Cores used to establish the local reference inventory were also collected from adjacent areas of undisturbed land. For the Cs-137 measurements, the cores were collected in August 1996, using a motorised percussion corer equipped with a 6.9 cm internal diameter steel core tube, which was inserted into the soil to a depth of ca. 60 cm. The cores used for the Be-7 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.

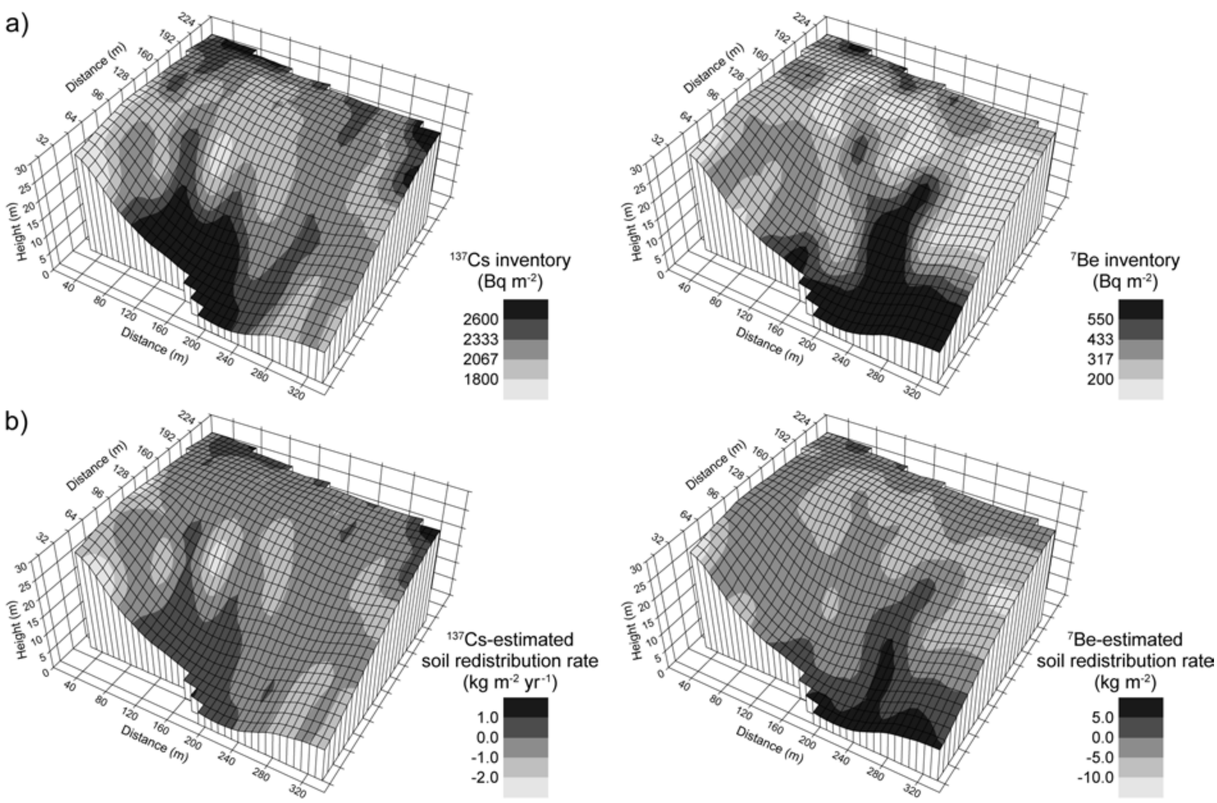

Fig. 2 The spatial distribution of Cs-137 and Be-7 inventories within a field at Higher Walton Farm, near Crediton, Devon and the estimates of soil redistribution rates derived from these measurements.

The pattern of Cs-137 inventories documented for the study field is presented in Fig. 2a. The local reference inventory was estimated to be ~ 2500 Bq m^{-2} and the pattern of Cs-137 inventories therefore shows clear evidence of both erosion (reduced inventories) and deposition (increased inventories). Use of a conversion model enabled estimates of mean annual soil redistribution rates over the past ca. 40 years to be derived from the measured inventories. The resulting pattern has been mapped in Fig. 2b and the data have been summarised 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 the sediment delivery ratio. The latter value is of particular importance, since it provides an estimate of the relative proportions of the mobilised sediment, which have been transported beyond the field or redeposited within the field. Such information is extremely difficult, of not impossible, to obtain using conventional monitoring techniques.

The spatial distribution of Be-7 inventories within the study field measured at the end of the period of heavy rainfall in early January1998 is presented in Fig. 2c. The equivalent value for the local reference inventory was estimated to be 533 Bq m^{-2} and the pattern shown in Fig. 2c provides clear evidence of areas with both reduced and increased inventories and thus of 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. It is therefore possible to assume that the spatial variability in Be-7 inventories within the study field evident in Fig. 2c 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 Be-7 at uneroded points 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. 2d and summary data, equivalent to that provided for the Cs-137 measurements, are also listed in Table 1.

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	Cs-137 (kg m^{-2} $year^{-1}$)	Be-7 (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	-2.0
Sediment delivery ratio	0.83	0.80

Based on Walling et al. (1999)

The soil redistribution rates (kg m^{-2}) associated with the period of heavy rainfall in early January 1998 estimated from the Be-7 measurements are substantially higher than the equivalent longer-term mean annual soil redistribution rates estimated using the Cs-137 measurements. However, the sediment delivery ratios are closely similar, indicating that ca. 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 were 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 (e.g. Walling et al., 2001). The data provided by such investigations could provide an important input to refining the national pattern of slope-channel connectivity ratios recently predicted for England and Wales by the Environment Agency (cf. Walling and Zhang, 2004).

Tracing or fingerprinting fine sediment sources.

Sediment source tracing or fingerprinting techniques can be used to provide information on the relative importance of a number of potential sources contributing to the sediment load of a river. Such information can clearly be of considerable value when designing sediment control strategies, since it will assist in identifying those sources which should be targeted for application of control measures. Although the approach can be used to establish the relative importance of different parts of a river basin (i.e. spatial sources), information on the relative importance of different source types (e.g. sheet and rill erosion, gully erosion and channel erosion) is frequently more useful in a management context. In the latter case, fallout radionuclides will commonly provide a key component of the composite fingerprint used to discriminate potential sources. Although most work of this type has focussed on fingerprinting the source of the suspended sediment load transported by a river (e.g. Walling and Woodward, 1995; Collins et al., 1997), it can, for example, also be used to trace the source of fine sediment deposited on floodplains or accumulating within river gravels (e.g. Bottrill et al., 2000; Walling et al., 2003). An example of the latter application is provided below.

The siltation of spawning gravels has frequently been identified as a key factor contributing to the declining success of salmon fisheries in British rivers. Concern for this problem has focussed attention on the need to reduce gravel siltation and thus to reduce fine sediment mobilisation and transport in impacted catchments, through the establishment of sediment control programmes. The development of effective sediment control programmes requires information on the likely sources of the fine sediment accumulating in spawning gravels, since these sources must be targeted if the control programme is to make optimum use of the available resources and prove effective. In an attempt to provide such information, a reconnaissance source fingerprinting survey of several representative rivers, located in different parts of Britain, was undertaken by the author and one of his co-workers in collaboration with the Environment Agency (cf. Walling et al., 2003). In this study, samples of interstitial fine sediment were recovered from salmonid spawning gravels for a representative selection of rivers in England and Wales (Fig. 3a), through a national fieldwork programme conducted by the Environment Agency over the period 1999-2000. Sample collection involved the use of retrievable basket samplers, which were installed in artificial redds constructed in spawning gravels at representative locations. Between one and five samplers were installed in each of the rivers identified on Fig. 3a. The basket samplers were filled with clean framework gravel (>6.4 mm) prior to their installation and they were retrieved ca. 3 months later. The gravel contained within the basket was subsequently wet sieved to recover the fine (<0.125mm) interstitial sediment that had accumulated within the gravel during the period of deployment and this fraction was used for sediment source fingerprinting. By virtue of the reconnaissance nature of the study, which included 18 catchments, sampling of potential source materials focussed on the broad distinction between (a) surface and (b) channel bank/subsurface sources, and a total of 672 source material samples were collected from the different study areas. These samples were sieved to <0.125 mm to facilitate direct comparison with the samples of fine interstitial sediment. The limited resources available to the study also precluded the use of an extensive range of fingerprint properties to discriminate the two potential sources and emphasis was placed on the use of radiometric (Cs-137, excess Pb-210 and Ra-226) measurements, coupled with information on the organic matter content (C and N) of the potential sources. Mean values of these properties were used to characterise the two potential sources in each of the river basins investigated.

A multicomponent mixing model, incorporating correction for the effects of contrasts in particle size and organic matter content between the samples of interstitial sediment and the source material, was used to estimate the relative contribution of surface and channel bank/ subsurface sources to the samples of fine sediment recovered from the spawning gravels. The results of these computations for different regions of England and Wales are presented in Fig. 3b. Appreciable contrasts in the relative importance of surface and channel bank/subsurface sources between the regions are apparent in Fig. 3b, with, for example, surface sources accounting for >90% of the fine interstitial sediment in north Wales and southern England, but only 16% and 39% in south-west England and south and mid Wales, respectively. These regional contrasts reflect the interaction of land use and both erosion processes and the efficiency of sediment transfer to the channel network. In south-west England, where stocking densities are high and river channels are frequently quite deeply incised, the combination of livestock trampling of channel margins and erosion of unstable channel banks mean that channel and subsurface sources are the dominant source of fine interstitial sediment. In contrast, the greater importance of arable farming, and more specifically soil erosion, on large cultivated fields with few boundaries to interrupt slope-channel connectivity, combined with the relative stability of the well-vegetated and relatively low channel banks, result in surface soils providing the dominant source of the fine interstitial sediment recovered from spawning gravels in southern England. Surface sources are also important in the upland areas of northern England and north Wales, where high rainfall and grazing pressure promote the erosion of surface soils under upland pasture or moorland and the steep topography combined with an absence of field boundaries result in the efficient routing of sediment to the river channel.

The contrasts in the relative importance of surface and channel bank/subsurface sources in different regions of England and Wales outlined above have important implications for the design and implementation of effective sediment control strategies. Where bank erosion is the dominant source, attention should clearly focus on reduction of livestock trampling of channel margins and improvement of bank stability (e.g. by fencing and revegetation of channel margins). However, such measures are likely to be of little value in areas where surface sources are dominant and where emphasis should be placed on controlling sediment mobilisation and transfer from the catchment surface more generally.

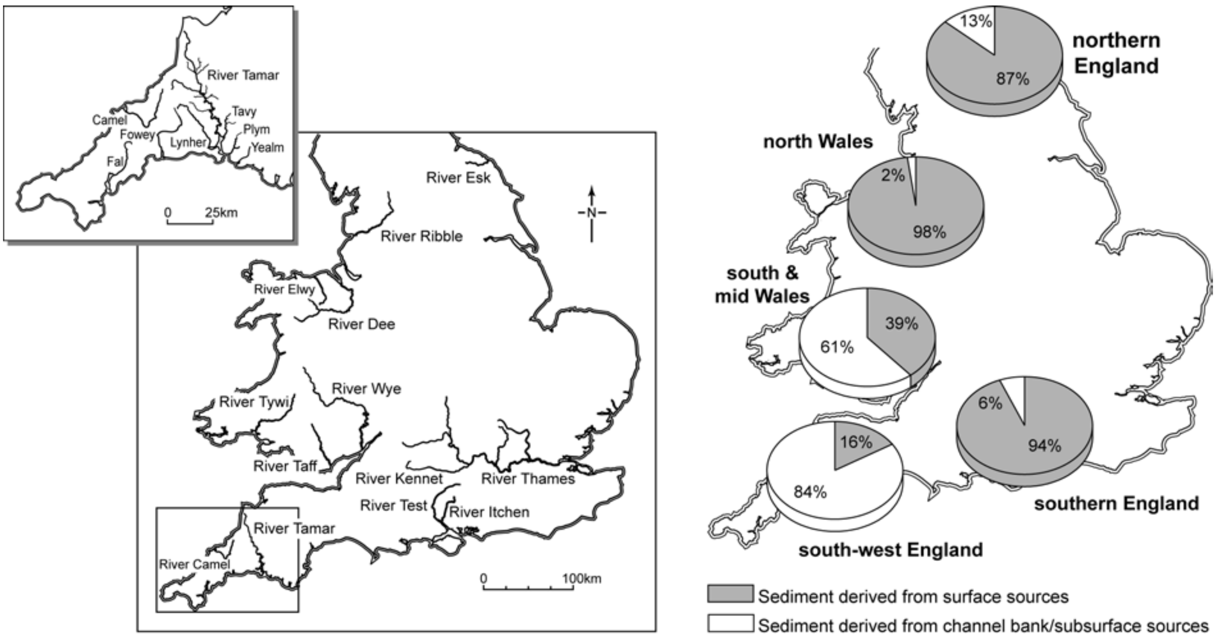

Fig. 3 Fingerprinting the source of fine sediment accumulating in salmon spawning gravels in UK rivers, showing (left) the location of the sampled rivers and (right) regional contrasts in the source of the fine sediment.

Quantifying overbank sedimentation rates on river floodplains

Overbank deposition on river floodplains during flood events can represent an important sink for suspended sediment transported through a river system, and recent studies have demonstrated that such transmission losses can account for as much as 40-50% of the suspended sediment load delivered to the main channel system (cf. Walling and Owens, 2002). Where the nutrient and contaminant content of the sediment is high, floodplains can represent significant nutrient and contaminant sinks, posing problems for their longer-term sustainable use. Equally, the progressive aggradation of river floodplains can result in reduced floodwater conveyance capacity and thus an increasing flood risk. Information on rates of overbank sedimentation on river floodplains is needed to investigate further their role as sediment sinks and, in view of the difficulties of obtaining such information using conventional approaches, the use of environmental radionuclides to establish deposition rates has been shown to offer considerable potential.

As an example, Fig. 4 shows how Cs-137 and excess Pb-210 measurements have been used to document overbank sedimentation rates along a short reach of the floodplain of the River Severn near

Buildwas in Shropshire, UK. In this study, 124 sediment cores were collected at the intersections of a 25m x 25m grid, using a motorised percussion corer equipped with a 6.9 cm internal diameter core tube. Cores were collected to a depth of ca. 70 cm to ensure that they included the complete Cs-137 and excess Pb-210 profiles. Measurements of the Cs-137 and excess Pb-210 inventories of the individual cores were used to estimate the mean annual sedimentation rates at the coring points using the procedures documented by Walling and He (1997) and He and Walling (1996). These estimates have been used to map the patterns of sedimentation within the reach shown in Fig. 4. By comparing the estimates of sedimentation rate derived from the Cs-137 measurements, which relate to the past ca. 40 years, with those based on the excess Pb-210 measurements, which relate to the past ca. 100 years, it is possible 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} $year^{-1}$ and the equivalent rate for the past 100 years is 0.33 g cm^{-2} $year^{-1}$. This suggests that rates of overbank sedimentation have changed little over the past 100 years.

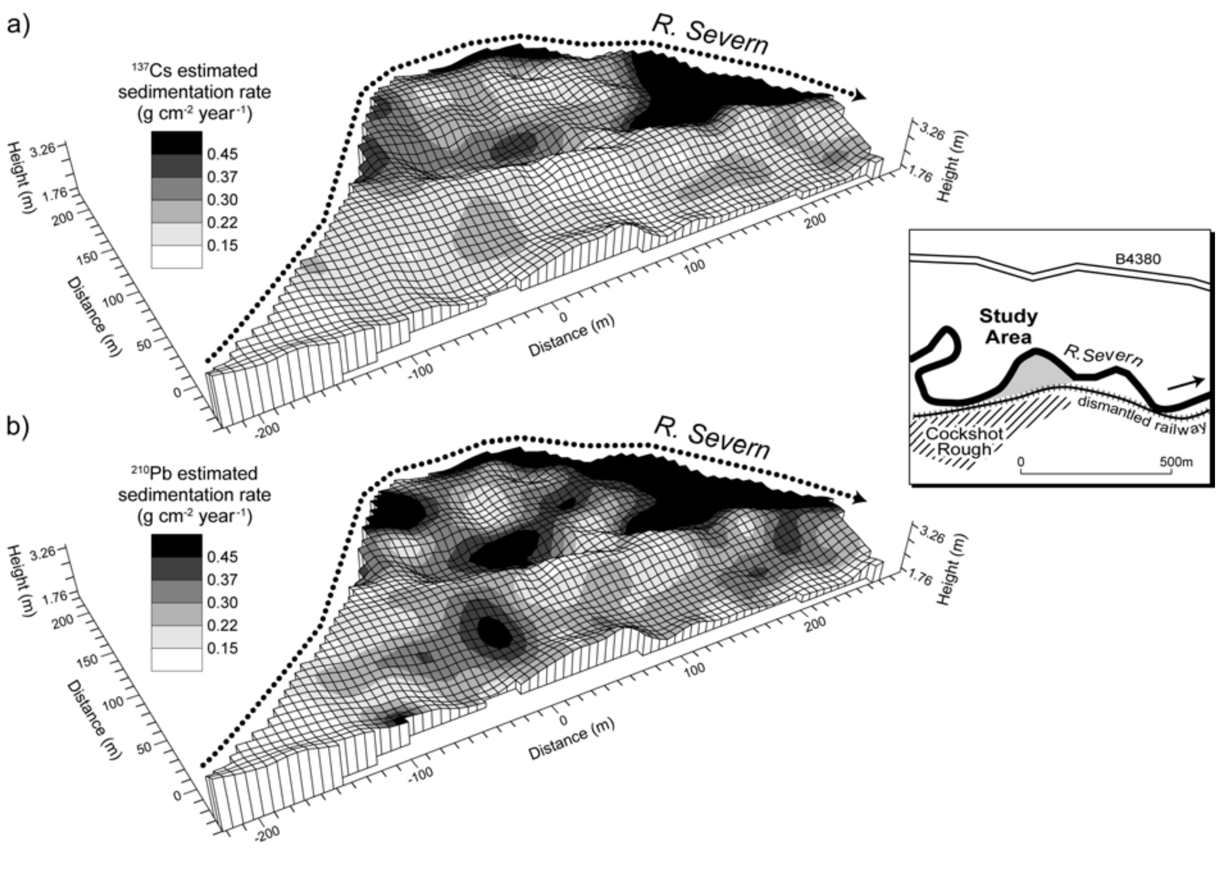

Fig. 4 The spatial distribution of overbank sedimentation rates within a small reach of the floodplain of the River Severn near Buildwas, Shropshire, UK derived from Cs-137 (a) and excess Pb-210 (b) measurements undertaken on floodplain cores.

Although the example presented in Fig. 4 represents a detailed investigation of an individual reach, it is equally possible to use the approach to obtain representative information on overbank sedimentation rates for a range of rivers within a region (e.g. Walling and He, 1999c) or to establish the magnitude of the longer-term delivery losses associated with overbank deposition on the floodplains bordering the main channels of a river basin. In the latter case there will be a need to extrapolate measurements from representative transects or small reaches to the entire floodplain area, in order to calculate the mass of sediment involved and to compare this with the suspended sediment flux at the basin outlet. (e.g. Walling et al., 1998). The potential for using measurements of both Cs-137 and excess Pb-210 to assess changes in overbank sedimentation rates over the past 100 years also offers considerable scope to investigate recent changes in floodplain sedimentation in response to changes in catchment land use (cf. Walling and He, 1999c).

The prospect

The case studies described above provide several examples of the potential for using fallout radionuclides as tracers in order to obtain information on the functioning of catchment sediment budgets, which can in turn be used to inform and support catchment management strategies. Each of the case studies focuses on a particular component of the sediment budget and it should be recognised that, in many investigations, the ultimate aim will be to elucidate the overall catchment sediment budget. The results obtained for the individual components of a sediment budget can be combined to establish the overall sediment budget of a catchment and the use of the same radionuclide tracers in studies of the individual components will clearly facilitate any such synthesis.

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