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USING ENVIRONMENTAL RADIONUCLIDES TO TRACE SEDIMENT MOBILISATION AND DELIVERY IN RIVER BASINS AS AN AID TO CATCHMENT MANAGEMENT

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Abstract: Although sediment problems have long been recognised in those areas of the world with high rates of soil loss and high specific sediment yields, recent years have seen a growing recognition of the wider significance of fine sediment as an environmental problem. This has resulted in a growing need to incorporate sediment control strategies into catchment management programmes. The design of effective sediment control strategies is, however, heavily dependent upon a sound understanding and knowledge of sediment mobilisation and delivery within the catchment system and thus the overall sediment budget. Traditional monitoring techniques are generally unable to provide information on the internal functioning of the sediment budget. Against this background, there is a need for new approaches to data collection and tracing techniques are being increasingly used. Environmental radionuclides provide a particularly useful tool for tracing sediment mobilisation and delivery and this paper provides further information on the basis for their use and on their potential applications. Selected examples of their application to documenting rates of soil loss and sediment delivery from cultivated fields, sediment source tracing, and documenting rates of floodplain sedimentation and longer-term changes in suspended sediment properties are provided.

Keywords: Environmental radionuclides, Caesium-137, Lead-210, Beryllium-7, Erosion, Deposition, Sediment sources, Floodplain sedimentation, Sediment-associated nutrients, Phosphorus

1 INTRODUCTION

The transport of fine sediment by rivers has long been recognised as a major problem in those regions of the world where erosion rates and suspended sediment yields are high. There, loss of reservoir storage through sedimentation and the siltation of canals and water distribution networks can introduce serious problems for sustainable water resource development, irrigation systems and hydropower production, and the siltation of river channels and harbours can threaten the viability of navigable waterways (cf. Sundborg, 1982). 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 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*., 1995) 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, the transport of fine sediment by rivers was essentially ignored as a potential problem until fairly recently, since reservoir sedimentation did not represent a significant constraint on water resource development. Recent concern for the improvement of river water quality and aquatic habitats prompted by the EC Water Framework and Habitats Directives has, however, 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 relatively low by world standards.

Against this background, sediment control strategies now represent a vital component of catchment management strategies in many areas of the world, including both those with high sediment yields, which were traditionally seen as experiencing sediment problems, and those where specific sediment yields are an order of magnitude or more lower, but fine sediment represents a key source of diffuse source pollution and an important environmental problem. 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 limited understanding of the delivery and storage of fine sediment within drainage basin systems. The linkages between sediment mobilisation, transport, deposition, storage and sediment yield at the basin outlet can be highly complex , especially in situations where sediment storage equals or exceeds sediment export (cf. Trimble, 1983; Phillips, 1992; Walling, 2000).

The sediment budget concept affords a valuable framework for assembling the detailed information required to elucidate and interpret the drainage basin sediment delivery system (Golosov *et al*., 1992; Reid and Dunne, 1996; Walling, 2000, Walling *et al*., 2001). 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, catchment monitoring programmes have traditionally placed emphasis on monitoring sediment loads and yields, particularly at catchment outlets, and the need to elucidate catchment sediment budgets has introduced a requirement for information on other components of the sediment budget, including sediment mobilisation and storage, which are spatially variable and considerably more difficult to document. Furthermore, it can be suggested that the problems and uncertainties associated with constructing a meaningful sediment budget for a catchment frequently increase in those areas of the world where erosion rates and sediment yields are relatively low. In such areas, the primary sediment sources within a drainage basin may not be readily discernible, significant soil erosion is frequently restricted to small areas, and sediment deposition and storage may be difficult to identify and document.

Faced with demands for new information to underpin the development of sediment control and management strategies, there is a need for new approaches to obtaining the data required for establishing catchment sediment budgets. The potential for using environmental radionuclides as sediment tracers, both as an alternative to traditional monitoring techniques and to complement such techniques, has been increasingly recognised and exploited and arguably it is possible to identify a general shift away from the traditional emphasis on *monitoring* towards a growing emphasis on *tracing*. Key advantages of the use of environmental radionuclides as sediment tracers include the potential for assembling retrospective (medium-term) information on the basis of a limited programme of contemporary measurements, the possibility of using essentially 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 mobilisation 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 catchment sediment budget investigations, to underpin the design and implementation of sediment control and management strategies within catchment management programmes.

2 ENVIRONMENTAL RADIONUCLIDES

The term '*environmental radionuclide*' is 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. For applications relating to sediment tracing, most work to date has focussed on a particular group of environmental radionuclides, namely, *fallout radionuclides*, or radionuclides which reach the land surface as 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 on reaching the catchment surface as fallout, they accumulate at or near the surface and afford a means of tracing sediment mobilisation and deposition by documenting the post fallout redistribution of the radionuclide tracer, which will directly reflect the mobilisation, transport and deposition of soil and sediment particles. In essence, therefore, it is possible to view the fallout as equivalent to the artificial application of a sediment tracer to the land surface of a study area. Observation of the subsequent redistribution of the radionuclide provides a basis for establishing rates and patterns of sediment transfer and establishing the magnitude and relative importance of sediment storage within the landscape.

The radionuclide that has been most widely used as a sediment tracer is caesium-137 (^{137}Cs) (cf. Ritchie and 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 and adjacent regions.

Use of other fallout radionuclides as sediment tracers has primarily focussed on unsupported or excess lead-210 (^{210}Pb) and beryllium-7 (^{7}Be). These two radionuclides differ from ^{137}Cs in two important respects. Firstly, 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. 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 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 ($Bq \cdot m^{-2}$)) will exhibit different temporal behaviour. In the case of unsupported ^{210}Pb, the essentially constant fallout 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 would have been zero prior to the onset of fallout in the mid 1950s. It will then have increased through to the 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, when fallout is limited, the inventory will rapidly decline due to decay, only to increase again as a result of rainfall and associated fallout. A range of techniques are available for measuring ^{137}Cs, ^{7}Be and ^{210}Pb activities, but the most widely used is gamma spectrometry (see Wallbrink *et al*., 2002). By using appropriate high purity germanium (HPGe) detectors, it is generally possible to measure all three radionuclides simultaneously. The long count times (e.g. 6-24 hours) commonly required for accurate measurements must, nevertheless, be seen as a significant limitation, in that the number of samples that can be analysed may be restricted.

3 USING FALLOUT RADIONUCLIDES TO TRACE SEDIMENT MOBILISATION AND DELIVERY

Fig. 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. At undisturbed pasture sites, the radionuclides are typically concentrated close to the surface, and concentrations decline exponentially with depth. 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 radionuclides. The vertical distribution of ^{7}Be differs significantly from that of the other two radionuclides and this 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. If an undisturbed site, that has been influenced by neither erosion nor deposition, can be identified, measurement of the total inventory of the individual radionuclides at that site ($Bq{\cdot}m^{-2}$) will provide an estimate of the local fallout input. Such sites are commonly referred to as *reference sites* (cf. Loughran *et al*., 2002) and are normally located in areas with limited relief, and particularly on interfluves.

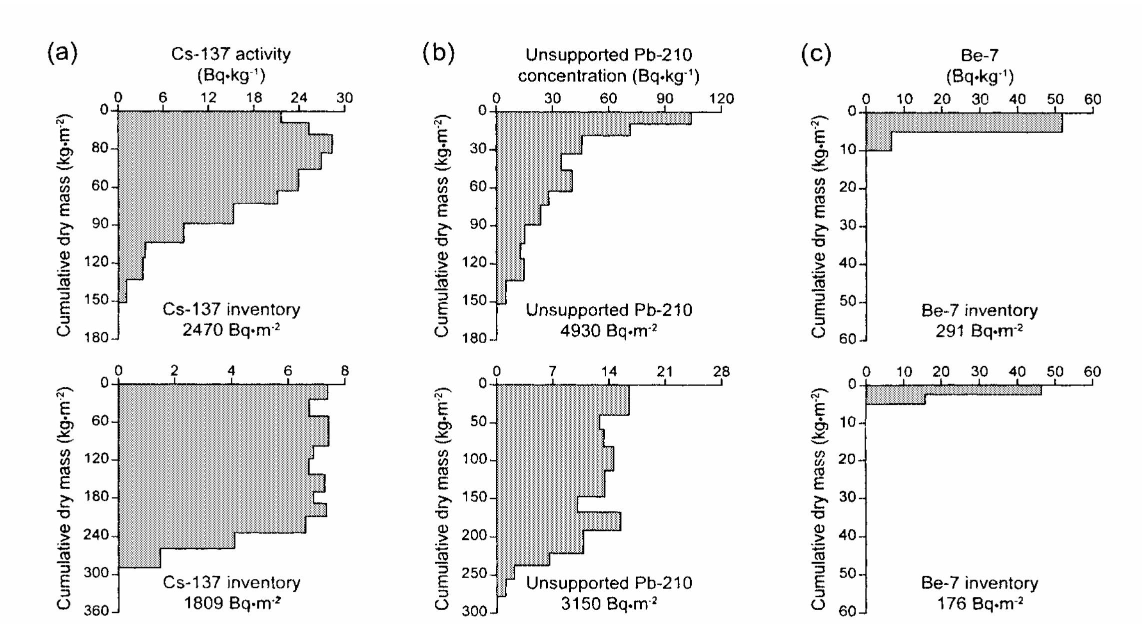


Fig. 1 Typical Depth Distributions of ^{137}Cs, Unsupported ^{210}Pb and ^{7}Be Concentrations in Undisturbed Pasture (upper) and Cultivated Soils (lower) in Devon, UK

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 to produce near uniform concentrations. The reduced inventories, relative to the undisturbed pasture sites, reflect removal of soil containing both radionuclides by soil erosion. The contrasting behaviour of ^{7}Be again reflects the short half-life of this radionuclide. This results in the presence of ^{7}Be being limited to a thin surface layer, which reflects the recent fallout to the surface. As with ^{137}Cs and unsupported ^{210}Pb, the reduced total inventory associated with the cultivated soil reflects loss of ^{7}Be in association with eroded soil. Such erosional loss is further reflected by the reduced depth to which ^{7}Be is found in the cultivated soil, relative to the pasture soil.

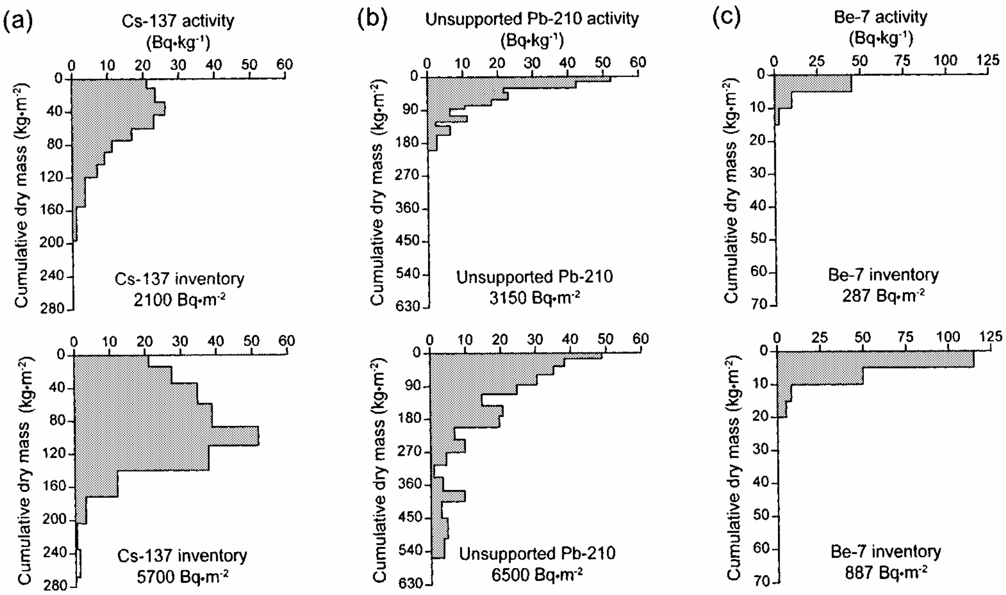


Fig. 2 Typical depth Distributions of ^{137}Cs, Unsupported ^{210}Pb and ^{7}Be in Overbank Sediments from River Floodplains in Devon, UK (lower) and in Adjacent Pasture Soils Above the Level of Inundation (upper)

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. 1. 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. 2, which compares the depth profiles of the three radionuclides in sediment cores collected from river floodplains 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 undisturbed by tillage mixing. In the case of ^{137}Cs, the depth profile provides clear evidence of the progressive accretion of the floodplain above the level marked by the maximum ^{137}Cs activity, which represents the floodplain surface in the mid 1960s. This accretion reflects overbank deposition of fine sediment containing ^{137}Cs that has been mobilised by erosion from the upstream catchment and which results in an increased total ^{137}Cs inventory relative to that associated with the core collected from the site above the level of inundation, which will have received only direct fallout inputs. The decline in radiocaesium activity towards the surface reflects the reduction and subsequent cessation of fallout after the mid 1960s, and a gradual reduction in the ^{137}Cs activity of sediment mobilised 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 evidenced by a more gradual exponential decline in unsupported ^{210}Pb activity with depth and the greater depth of the unsupported ^{210}Pb profile, when compared with the core collected from the site above the level of floodplain inundation, as well as an increased total inventory. In the case of ^{7}Be, its short half-life means that contrasts between the profiles from the floodplain area and the adjacent area above the level of inundation will only reflect very recent floodplain accretion. The ^{7}Be profile for the floodplain surface depicted in Fig. 2 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 ^{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 documenting sediment mobilisation and delivery within river basins. 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 the erosion or deposition rate (cf. Walling and He, 1999a,b). For ^{137}Cs measurements, the resulting estimates of erosion and deposition rates will reflect erosion and deposition occurring over the last ca. 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 ca. 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 ca. 6 months.

Similarly, the ^{137}Cs, unsupported ^{210}Pb and ^{7}Be depth distributions and inventories found on river floodplains and in other depositional environments (Fig. 2) afford a basis for estimating deposition rates. By collecting cores from such sites and determining their radionuclide profiles or, in simpler applications, comparing their total 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 ^{7}Be it is possible to obtain estimates of sedimentation rates associated with an individual events, whereas with ^{137}Cs the estimates will relate to a period of ca. 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 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, 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. Fallout radionuclide activities or 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 (cf. Fig. 1). 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 will be absent from channel banks and other subsurface sources.

More detailed examination of the potential application of environmental radionuclides in documenting sediment mobilisation and delivery in river basins 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, sediment source fingerprinting, floodplain sedimentation and reconstructing longer-term changes in the properties of the sediment transported by a river.

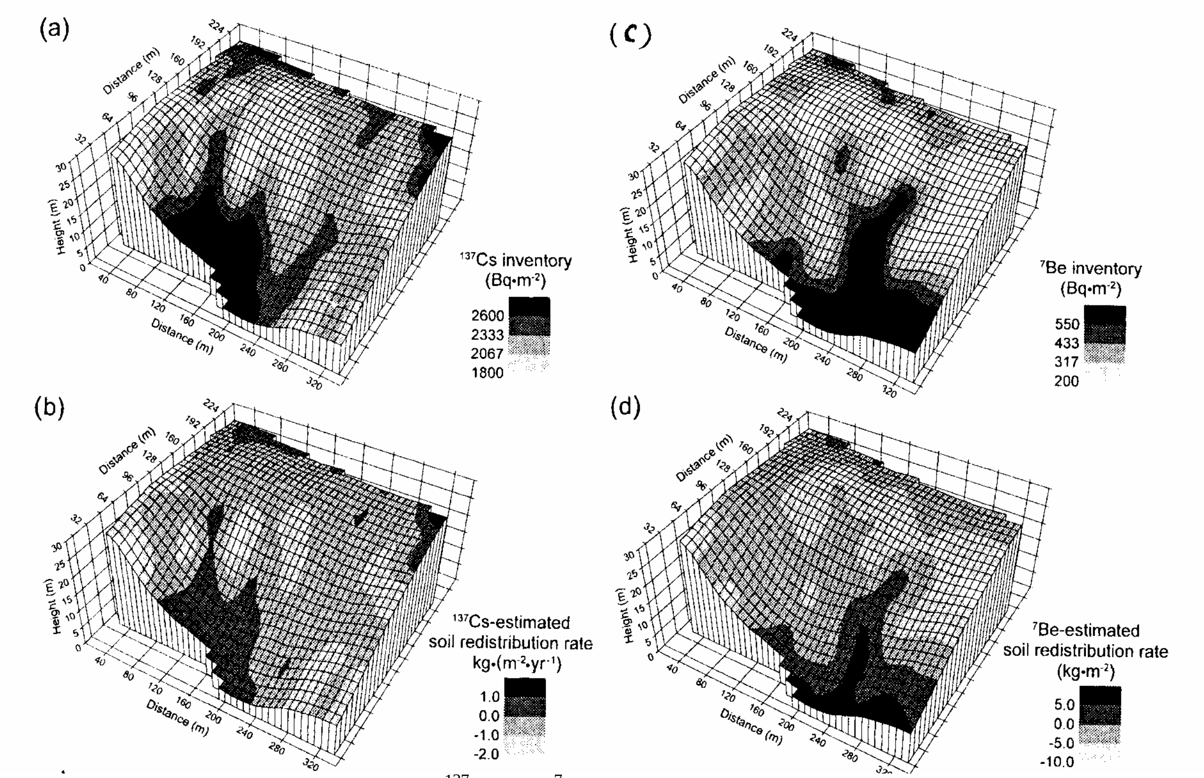


Fig. 3 The Spatial Distribution of ^{137}Cs and ^{7}Be Inventories Within a Field at Higher Walton Farm, Near Crediton Devon and of the Estimates of Soil Redistribution Rates Derived from These Measurements

4 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 and McHenry, 1990; Walling, 1998), both unsupported ^{210}Pb and ^{7}Be 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 ^{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 relative importance of erosion and deposition and thus the gross and net erosion and the sediment delivery ratio. Fig. 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). 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 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 ^{137}Cs and ^{7}Be 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 × 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 ^{137}Cs 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 ^{7}Be measurements were, in contrast, much shallower and were collected manually to depths of 3 cm-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 local reference inventory was estimated to be ~ 2,500 Bq·m^{-2} and the pattern of ^{137}Cs inventories therefore 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 ca. 40 years to be derived from the measured inventories. The resulting pattern has been mapped in Fig. 3b 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 considerable 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 to obtain using conventional monitoring techniques.

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}·y^{-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)

The spatial distribution of ^{7}Be inventories within the study field measured at the end of the period of heavy rainfall in early January1998 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 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 ^{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 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. 3(d) and summary data, equivalent to that provided for the ^{137}Cs measurements, are also listed in Table 1.

The soil redistribution rates ($kg \cdot m^{-2}$) 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 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.

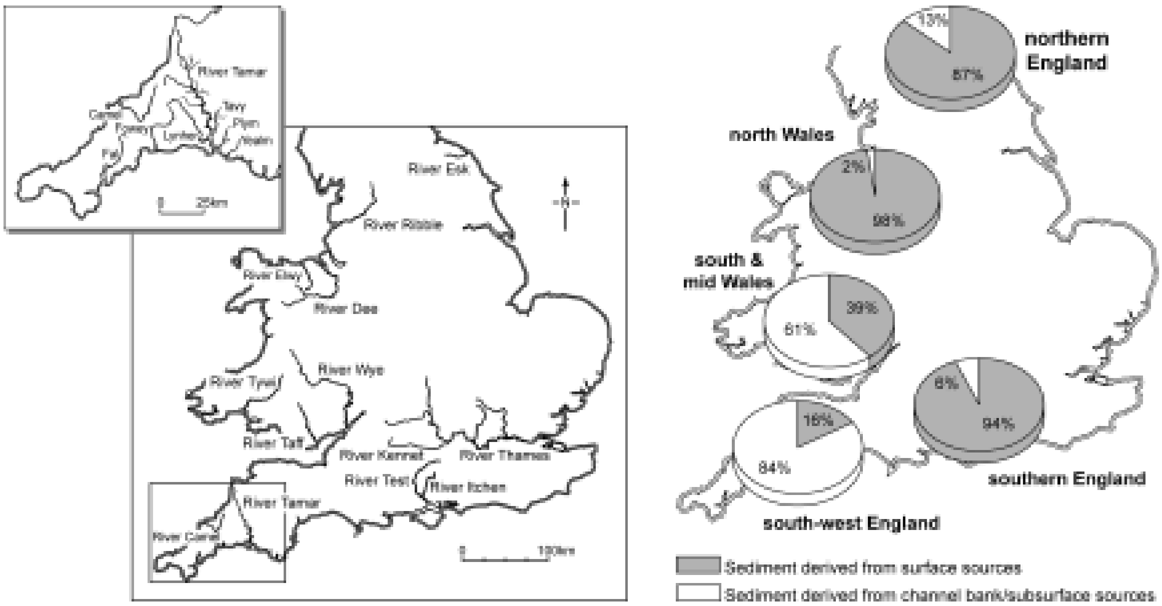
In the example presented above, a large number of cores were used to establish the pattern of soil redistribution within the study field. It is clearly impossible to extend sampling at this intensity to more than a few fields and, if a sediment budget is to be constructed for a larger area, it will be 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).

5 TRACING OR FINGERPRINTING SEDIMENT SOURCES

Another useful perspective on sediment mobilisation within river basins is provided by sediment source tracing or fingerprinting techniques. As outlined above, this approach is able to provide valuable information on the relative importance of different potential sources in contributing to the sediment load of a river. Such information can clearly provide a valuable basis for designing sediment control strategies, by indicating which sources 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 distinguish potential sources. Although most work of this type has focussed on fingerprinting the source of the suspended sediment load transported by a river, it can, for example, also be used to trace the source of fine sediment deposited on floodplains or accumulating within river gravels. An example of the latter application is provided below.

The siltation of spawning gravels has been increasingly 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 associated loads 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 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 his co-workers in collaboration with the Environment Agency (cf. Walling *et al*., 2003).

Samples of interstitial fine sediment were recovered from salmonid spawning gravels within a representative selection of rivers in England and Wales (Fig. 4(a)), during a national fieldwork programme conducted by the Environment Agency over the period 1999-2000. Sample collection was based on 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. 4(a). 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, firstly, surface and, secondly, 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 (^{137}Cs, unsupported ^{210}Pb, ^{226}Ra) measurements, coupled with information on the organic properties (C and N content) 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) (b)

Fig. 4 Fingerprinting the source of fine sediment accumulating in salmon spawning gravels, showing (a) the location of the sampled rivers and (b) regional contrasts in the source of the fine sediment

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. 4(b). Appreciable contrasts in the relative importance of surface and channel bank/subsurface sources between the regions are apparent in Fig. 4(b), 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 incised, a combination of livestock trampling of channel banks and erosion of unstable well-developed vertical 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 channel banks, result in surface soils providing the dominant source (94%) of the samples of 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 within the catchment more generally.

6 INVESTIGATING FLOODPLAIN SEDIMENTATION

Overbank deposition on river floodplains during flood events represents an important potential sink for suspended sediment transported through a river system and recent studies have demonstrated that such transmission losses can be as high as ca. 40% 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. There will, therefore, frequently be a need to quantify rates of overbank sedimentation on river floodplains and, in view of the difficulties of obtaining such information using conventional approaches, the use of environmental radionuclides has been shown to offer considerable potential.

As an example, Fig. 5 shows how ^{137}Cs and unsupported ^{210}Pb 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 × 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 ^{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 and He (1997) and He and Walling (1996). These estimates have in turn been used to map the patterns of sedimentation within the reach shown in Fig. 5. By comparing the estimates of sedimentation rate derived from the ^{137}Cs measurements, which relate to the past ca. 40 years, with those based on the unsupported ^{210}Pb 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 $g \cdot cm^{-2} \cdot yr^{-1}$, whereas the equivalent rate for the past 100 years is 0.33 $g \cdot cm^{-2}$ yr^{-1}. This suggests that rates of overbank sedimentation have changed little over the past 100 years.

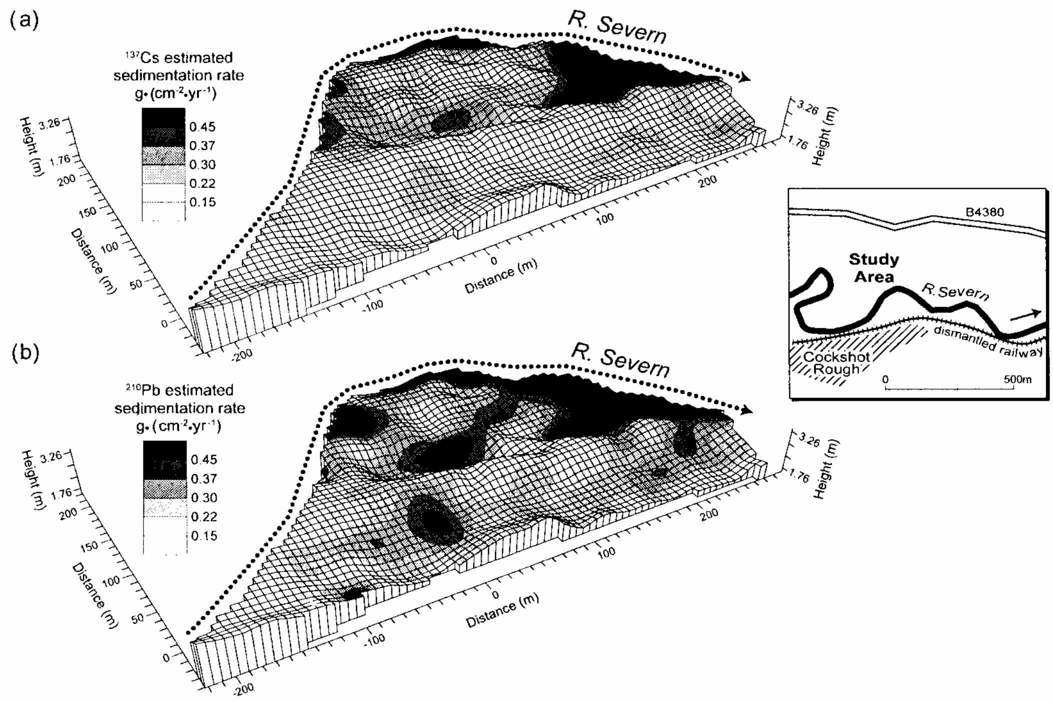


Fig. 5 The spatial distribution of overbank sedimentation rates within a small reach of the floodplain of the River Severn near Buildwas, Shropshire, UK derived from ^{137}Cs and unsupported ^{210}Pb measurements undertaken on floodplain cores

Although the example presented in Fig. 5 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, 1999) or to establish the magnitude of the longer-term transmission 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) .

7 RECONSTRUCTING LONGER-TERM CHANGES IN SEDIMENT PROPERTIES

Where post-depositional changes in sediment geochemistry can be assumed to be of limited importance, it is also possible to use downcore changes in the geochemistry of overbank deposits to reconstruct changes in the properties of the suspended sediment transported by a river. The overbank deposits effectively preserve a record of the suspended sediment transported by the river in the recent past (cf. Walling *et al.*, 2000). Work undertaken by the author on the main floodplain of the River Axe, which drains a 303 km^2 catchment in Devon, UK, aimed at reconstructing changes in the phosphorus content of the deposited sediment, provides a useful example of the potential of this approach. Floodplain cores were collected from three representative sites in the upper, middle and lower reaches of the river basin (cf. Fig. 6(a) and measurements of both the ^{137}Cs activity and the total-P content of the sectioned cores were undertaken, in order to establish the associated depth profiles. The total-P profiles shown in Fig. 6(b) indicate that the total-P content of sediment deposited on the river floodplains at the three sites has more than doubled in recent years in response to increased fertiliser application and livestock densities in the upstream catchments and increased effluent discharges from sewage works and related point sources. The ^{137}Cs measurements were used to establish an approximate chronology for these changes by defining the level of the floodplain surface in 1963, the year of peak ^{137}Cs fallout (Fig. 6(b)), and assuming a uniform annual sedimentation rate. This chronology was used to derive the plots of the changing total-P content of deposited sediment since 1950 presented in Fig. 6c. At all three sites, the total-P of deposited sediment has increased by more that 100% over the past 50 years, but higher concentrations are found in the middle and lower reaches of the catchment where agriculture is more intensive and effluent discharges are greater. Although not clearly defined, there is some evidence of a reduced total-P content in deposited sediment in the last 10 years in response to improved land management and treatment of waste water discharges. Such information can afford a valuable basis for defining baseline levels and assessing the impact of changing land management and waste water treatment in reducing the inputs of P and other sediment-associated nutrients and contaminants to river systems.

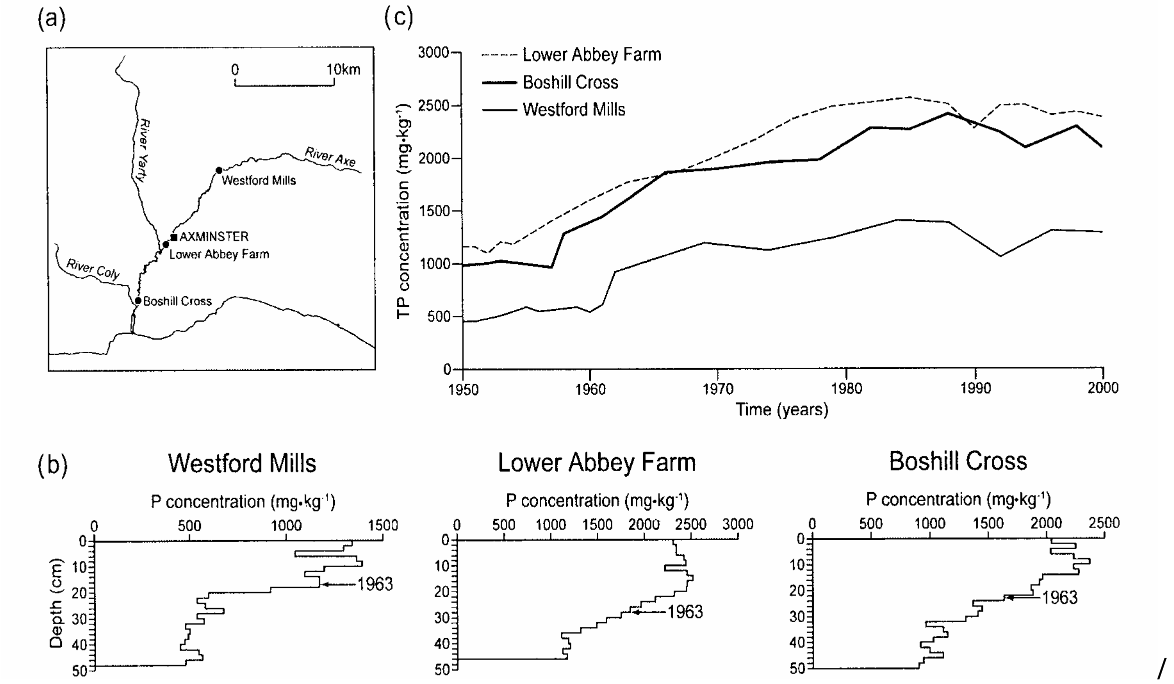


Fig. 6 The changing total-P content of fine sediment deposited on the floodplain of the River Axe over the past 50 years, showing (a) the location of the coring points, (b) the total-P depth distributions and (c) the reconstructed trends in the total-P content of fine sediment deposited on the floodplain over the past 50 years

8 PERSPECTIVE

The case studies described above provide several examples of the potential for using fallout radionuclides for tracing sediment mobilisation and delivery in river basins and providing a temporal perspective on the sediment delivery. In each case, however, 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 recognise 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.

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REFERENCES

Allan, R.J. 1986. *The Role of Particulate Matter in the Fate of Contaminants in Aquatic Ecosystems.* Environment Canada Scientific Series No. 142, Inland Waters Directorate, Burlington, Ontario.

Blake, W.H., Walling, D.E. and He, Q. 2002, Using cosmogenic beryllium-7 as a tracer in sediment budget investigations. *Geografiska Annaler*, Vol. 84A, pp. 89-102.

Clark, E.H., Haverkamp, J.A. and Chapman, W. 1985, *Eroding Soils: The Off-farm Impacts*. The Conservation Foundation, Washington DC.

Collins, A.L., Walling, D.E. and Leeks, G.J.L. 1997, Sediment sources in the Upper Severn catchment: a fingerprinting approach. *Hydrol. Earth System Sci.* Vol. 1, pp. 509-521.

Golosov, V.N., Ivanova, N.N., Litvin, L.F. and Sidorchuk, A.Yu. 1992, Sediment budget in river basins and small river aggradation. *Geomorphologiya*, Vol.4, pp. 62-71 (in Russian)

He, Q. and Walling, D. E. 1996, Use of fallout Pb-210 measurements to investigate longer-term rates and patterns of overbank sediment deposition on the floodplains of lowland rivers. *Earth Surf. Processes and Landforms*, Vol. 21, pp.141-154.

Loughran, R.J., Wallbrink, P.J., Walling, D.E. and Appleby, P.G. 2002, Sampling methods. *Handbook for the Assessment of Soil Erosion and Sedimentation Using Environmental Radionuclides,* Zapata, F. ed., Kluwer, Dordrecht, pp.41-57.

Phillips, J.D. 1992, Delivery of upper-basin sediment to the lower Neuse River, North Carolina, USA. *Earth Surf. Processes and Landforms*, Vol. 17, pp. 699-709.

Reid, L.M. and Dunne, T. 1996, *Rapid Evaluation of Sediment Budgets*. GeoEcology Paperbacks, Catena Verlag, Germany.

Ritchie, J.C. and McHenry, J.R. 1990, Application of radioactive fallout cesium-137 for measuring soil erosion and sediment accumulation rates and patterns: a review. *J. Environ. Qual*, Vol. 19, pp. 215-233.

Sundborg, A. 1982, *Sedimentation Problems in River Basins*, UNESCO Studies and Reports in Hydrology No. 35, UNESCO Division of Water Sciences, Paris

Trimble, S.W. 1983, A sediment budget for Coon Creek basin in the Driftless Area, Wisconsin. 1853-1977. *Am. J. Sci*., Vol. 283, pp. 454-474.

Wallbrink, P.J., Walling, D.E. and He, Q. Radionuclide measurement using HPGe gamma spectrometry. *Handbook for the Assessment of Soil Erosion and Sedimentation Using Environmental Radionuclides,* Zapata, F. ed., Kluwer, Dordrecht, pp.67-96.

Walling, D. E. 1998, Use of ^{137}Cs and other fallout radionuclides in soil erosion investigations: Progress, problems and prospects. *Use of* ^{137}Cs *in the Study of Soil Erosion and Sedimentation,* International Atomic Energy Agency Publication IAEA-TECDOC-1028, IAEA, Vienna, pp. 39-64.

Walling, D. E. 2000, Linking land use, erosion and sediment yields in river basins. *Hydrobiologia*, Vol. 410, pp. 223-240.

Walling, D. E. 2002, Recent advances in the use of environmental radionuclides in soil erosion investigations. *Nuclear Techniques in Integrated Plant Nutrient, Water and Soil Management,* IAEA C&S Papers Series 11/C, IAEA, Vienna, pp. 279-301.

Walling, D.E., Collins, A.L. and McMellin, G.K. 2003, A reconnaissance survey of the source of interstitial fine sediment recovered from salmonid spawning gravels in England and Wales. *Hydrobiologia*, Vol. 497, pp. 91-108.

Walling, D.E., Collins, A.L., Sichingabula, H.M. and Leeks, G.J.L. 2001, Integrated assessment of catchment sediment budgets. *Land Degradation and Development*, Vol. 12, pp. 387-415.

Walling, D. E. and He, Q. 1997, Use of fallout ^{137}Cs in investigations of overbank sediment deposition on river floodplains. *Catena*, Vol. 29, pp. 263-82.

Walling, D.E. and He, Q. 1999a, Using fallout lead-210 measurements to estimate soil erosion on cultivated land, *Soil Sci. Soc. Amer. J.*, Vol. 63, pp. 1404-1412.

Walling, D.E. and He, Q. 1999b, Improved models for estimating soil erosion rates from ^{137}Cs measurements. *J. Environ. Qual.*, Vol. 28, pp. 611-622.

Walling, D.E. and He, Q. 1999c, Changing rates of overbank sedimentation on the floodplains of British rivers during the past 100 years. *Fluvial Processes and Environmental Change*, Brown, A.G. and Quine, T.A. eds. Wiley, Chichester, pp. 207-222.

Walling, D.E., He, Q. and Blake, W.H. 1999, Use of ^{7}Be and ^{137}Cs measurements to document short- and medium-term rates of water-induced soil erosion on agricultural land. *Wat. Resour. Res*., Vol. 35, pp. 3865-3874.

Walling, D.E., He, Q. and Blake, W. H. 2000, River flood plains as phosphorus sinks. *The Role of Erosion and Sediment Transport in Nutrient and Contaminant Transfer* (Proc. Waterloo Symp., July 2000), IAHS Publ. 263, IAHS Press, Wallingford, UK, pp. 211-218.

Walling, D.E. and Owens, P.N. 2002, The role of floodplain sedimentation in catchment sediment and contaminant budgets. *The Structure and Management Implications of Fluvial Sedimentary Systems (*Proc. Alice Springs Symp., Sept. 2002), IAHS Publ. 276, IAHS Press, Wallingford, UK, pp. 407-416.

Walling, D. E., Owens, P. N. and Leeks, G. J. L. 1998, The role of channel and floodplain storage in the suspended sediment budget of the River Ouse, Yorkshire, UK. *Geomorphology,* Vol. 22, pp. 225-242.

Walling, D.E. and Woodward, J.C. 1992, Use of radiometric fingerprints to derive information on suspended sediment sources. *Erosion and Sediment Transport Monitoring Programmes in River Basins* (Proc. Oslo Symp. August, 1992), IAHS Pub. 210, IAHS Press, Wallingford, UK, pp. 153-164.

Walling, D. E. and Woodward, J.C. 1995, Tracing sources of suspended sediment in river basins. A case study of the River Culm, Devon, UK. *Marine and Freshwater Research,* Vol. 46, pp. 327-336.

Zapata, F. ed. 2002, *Handbook for the Assessment of Soil Erosion and Sedimentation Using Environmental Radionuclides.* Kluwer, Dordrecht.