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Geomorphology 52 (2003) 193–213
GEOMORPHOLOGY
www.elsevier.com/locate/geomorph

Using unsupported lead-210 measurements to investigate soil erosion and sediment delivery in a small Zambian catchment

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Received 8 January 2002; received in revised form 11 July 2002; accepted 8 August 2002

Abstract

Traditional techniques used to assemble information on rates of erosion and soil redistribution possess many important limitations. As a result, the use of environmental radionuclides, and more particularly ^{137}Cs measurements, has attracted increasing attention in recent years as a means of obtaining spatially distributed information on rates of erosion and deposition. The application of the ^{137}Cs approach is, however, hampered in some areas of the world where ^{137}Cs inventories are low and the low concentrations of ^{137}Cs found in soils and sediments cause problems for laboratory analysis. These problems will increase as time progresses due to the radioactive decay of the existing inventory, most of which was deposited as fallout ca. 40 years ago. This contribution explores the potential for using another fallout radionuclide, namely unsupported ^{210}Pb, as an alternative to ^{137}Cs, in the small (63 km^2) Upper Kaleya catchment in southern Zambia where ^{137}Cs inventories are already very low. The approach employed with unsupported ^{210}Pb is similar to that used for ^{137}Cs, although the essentially constant fallout of unsupported ^{210}Pb through time means that the resulting estimates of erosion and soil redistribution rates reflect a longer period of time (ca. 100 years rather than ca. 40 years). The estimates of erosion and deposition rates derived from the unsupported ^{210}Pb measurements are used to construct typical sediment budgets for the three main land-use types in the Upper Kaleya catchment, namely, commercial cultivation, communal cultivation and bush grazing. The results obtained from the unsupported ^{210}Pb are compared with equivalent results based on ^{137}Cs measurements provided by a previous investigation undertaken in the study catchment. The two sets of results are highly consistent. The study reported confirms the viability of using unsupported ^{210}Pb as an alternative to ^{137}Cs in this environment and demonstrates that conjunctive use of both radionuclides can provide additional information on the erosional history of a study area.

Keywords: Soil erosion; Unsupported ^{210}Pb; ^{137}Cs; Soil redistribution rates; Sediment delivery; Zambia

1. Introduction

Because of the many limitations associated with traditional techniques for documenting rates of soil erosion and sediment redistribution (cf. Stocking, 1987; Loughran, 1989), the use of environmental radionuclides, and more particularly caesium-137 (^{137}Cs) measurements, has attracted increasing attention as an alternative approach. The use of ^{137}Cs measurements affords a valuable means of assembling spatially distributed information on medium-term (ca. 40 years) rates of soil erosion and deposition on the

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doi:10.1016/S0169-555X(02)00244-1

basis of a single site visit (cf. Walling and Quine, 1995; Walling, 1998, 2002). Over the past decade, the ^{137}Cs approach has been successfully applied in a wide range of environments in many different areas of the world (Ritchie and McHenry, 1990; Walling, 1998). Such work has included a number of studies in Africa, where, for example, ^{137}Cs measurements have been used to quantify soil redistribution at both the field and catchment scale in Lesotho (Walling and Quine, 1992), to study local patterns of soil erosion in Niger (Chappell et al., 1998) and to investigate rates of soil loss and soil redistribution in areas of commercial and communal farming in Zimbabwe (Quine et al., 1993) and southern Zambia (Collins et al., 2001).

Despite the success of these and other studies, one important potential limitation hampering the application of the ^{137}Cs approach in some parts of the world, and particularly in Africa south of the Sahara, relates to the low ^{137}Cs inventories found in equatorial areas. Existing information on the global distribution of bomb fallout (e.g. Larsen, 1985; Walling, 1998, 2002; Garcia Agudo, 1998) indicates that ^{137}Cs inventories are considerably lower in equatorial regions than in the mid-latitudes, and in the Southern Hemisphere compared to the Northern Hemisphere. Thus, for example, ^{137}Cs reference inventories reported for southern Zambia by Collins et al. (2001) are almost an order of magnitude less than those found in the mid-latitudes of the Northern Hemisphere where the ^{137}Cs technique was originally developed. The low ^{137}Cs reference inventories associated with equatorial areas can introduce measurement problems associated with low ^{137}Cs concentrations close to the detection limit and the longer count times required to provide an acceptable degree of analytical precision (Walling and He, 1999a). Such problems will be further compounded in the future by the progressive reduction in ^{137}Cs activity caused by radioactive decay, and it may become impossible to apply the ^{137}Cs approach in such areas in the foreseeable future.

Due to these potential limitations on the use of ^{137}Cs to investigate soil erosion in some areas of the world, there is a need to investigate the use of alternative environmental radionuclides including lead-210 (^{210}Pb). Lead-210, a naturally occurring radionuclide (half-life 22.2 years), is a product of the ^{238}U decay series, derived via a series of other short-lived radionuclides from the decay of gaseous ^{222}Rn (half-life 3.8 days), the daughter of ^{226}Ra (half-life 1622 years). The ^{210}Pb content of soils and rocks produced by the natural in situ decay of ^{226}Ra is termed 'supported' ^{210}Pb because it is in equilibrium with its parent. However, upward diffusion of a small proportion of the ^{222}Rn produced naturally in soils and rocks releases ^{222}Rn to the atmosphere, and the subsequent fallout of ^{210}Pb provides an input to surface soils and sediments which is not in equilibrium with ^{226}Ra. Such fallout ^{210}Pb is commonly termed 'excess' or 'unsupported' ^{210}Pb (Robbins, 1978). Because of its natural origin, the deposition of fallout ^{210}Pb has, unlike that of ^{137}Cs, been essentially constant through time (Nozaki et al., 1978; Crickmore et al., 1990). Relatively little is currently known about the global distribution of fallout ^{210}Pb, although in a review of existing data on ^{210}Pb deposition fluxes for different areas of the world, Appleby and Oldfield (1992) indicate that such fluxes are greater over the land than over the ocean and lower over the western margins of continental land masses, due to the predominant west to east trajectory of air mass movement. The same authors report an average deposition flux for the world of 118 Bq m^{-2} year^{-1}, with values generally lying in the range 50–150 Bq m^{-2} year^{-1}. Since the global patterns of ^{137}Cs and unsupported ^{210}Pb fallout are influenced by different controls, it is unlikely that both ^{137}Cs and unsupported ^{210}Pb inventories will be low in a particular location.

Due to its strong affinity for soil and sediment particles (Van Hoof and Andren, 1989; He and Walling, 1996), fallout ^{210}Pb reaching the soil surface is readily adsorbed and its subsequent vertical and lateral redistribution are primarily controlled by erosion, transport and deposition processes. Consequently, unsupported ^{210}Pb, like ^{137}Cs, offers the potential for use as a tracer in estimating rates of soil redistribution.

To date, there have been few attempts to exploit this potential, although Walling and He (1999a) report the successful use of unsupported ^{210}Pb measurements to estimate soil erosion rates within a cultivated field in Devon, UK, and document procedures for deriving such estimates. This paper explores this potential further by testing the approach in an equatorial area where ^{137}Cs inventories currently approach the limit for the viable application of the ^{137}Cs technique and where it may in the foreseeable future be impossible to use ^{137}Cs measurements to estimate rates of soil redistribution. It presents the results of employing unsupported ^{210}Pb measurements to quantify longer-term (~ 100 years) soil redistribution rates on cultivated and grazing land within areas of communal and commercial farming in the Upper Kaleya River basin (63 km^2) in southern Zambia. Since ^{137}Cs measurements had already been used to estimate medium-term (ca. 40 years) soil redistribution rates at the same sites (cf. Collins et al., 2001), it is possible to compare the two sets of results to provide further validation of those based on unsupported ^{210}Pb measurements. In addition, by applying both fallout radionuclides, it was hoped to investigate the potential for obtaining additional information on the erosional history of the study site. The work reported was undertaken as a subsidiary component of a collaborative research programme aimed at developing an integrated approach to catchment sediment budgeting (Walling and Collins, 2000; Walling et al., 2001). To the authors' knowledge, it affords the first test of the viability of the ^{210}Pb approach for documenting soil erosion and deposition rates within an African environment.

2. Study area

The Upper Kaleya River basin (63 km^2) is located near Mazabuka in the southern province of Zambia (16°11′S, 28°02′E), ca. 65 km southwest of the capital city Lusaka (see Fig. 1). The climate is characterised by distinct wet (November–March) and dry (April–October) seasons, and the mean annual precipitation is ca. 800–900 mm. Altitudes range from 1240 m at the catchment outlet to 1400 m in headwater areas, and slopes are typically in the range 2–5°. Localised steeper slopes (7–8°) are found in the central and upper portions of the study area. The geology comprises Kaleya limestone and the Mazuma calc-silicate formation, of the Mazabuka Group, which belongs to either the Katanga system or assemblages between the older Pre-Cambrian and the Katanga (Smith, 1963). Soils are typically ferruginous tropical laterites including red clays, but patchy skeletal soils and regolith are also found on the steeper slopes (Brammer, 1976). Communal agriculture, with the cultivation of maize, sunflowers, groundnuts and cotton, occupies 68.9% of the study catchment, with bush grazing and commercial farming for coffee, mange-tout peas, chillies, wheat and passion fruit occupying the remaining 29.0% and 2.1%, respectively (see Fig. 1).

3. Methodology

3.1. The procedure

The procedure used to test the use of unsupported ^{210}Pb measurements for quantifying longer-term (~ 100 years) rates of soil redistribution associated with cultivated and uncultivated land within the study area involved the following elements (see Fig. 2):

(a) Reconnaissance soil coring to confirm the viability of the approach (i.e. unsupported ^{210}Pb activities and inventories of sufficient magnitude) and the validity of the assumptions (e.g. surface adsorption confirmed by exponential depth distributions in undisturbed soils).
(b) Collection of soil cores from an undisturbed location with minimal slope and no evidence of erosion or deposition to establish the unsupported ^{210}Pb reference inventory.
(c) Collection of soil cores to document the spatial variation of unsupported ^{210}Pb inventories within cultivated and uncultivated land in the communal and commercial sectors.
(d) Derivation of estimates of erosion and deposition rates for the sampling points by comparing their measured unsupported ^{210}Pb inventories with the local reference inventory and using a theoretical conversion model to convert the values for the percentage decrease or increase in the inventory to equivalent estimates of the erosion or deposition rate.

3.2. Data collection

3.2.1. Soil coring and laboratory analysis

A motorised percussion corer equipped with a steel core tube (internal diameter 6.9 cm) was used to collect soil cores to depths of ca. 40 cm. The core tube was extracted from the ground using a hand-operated

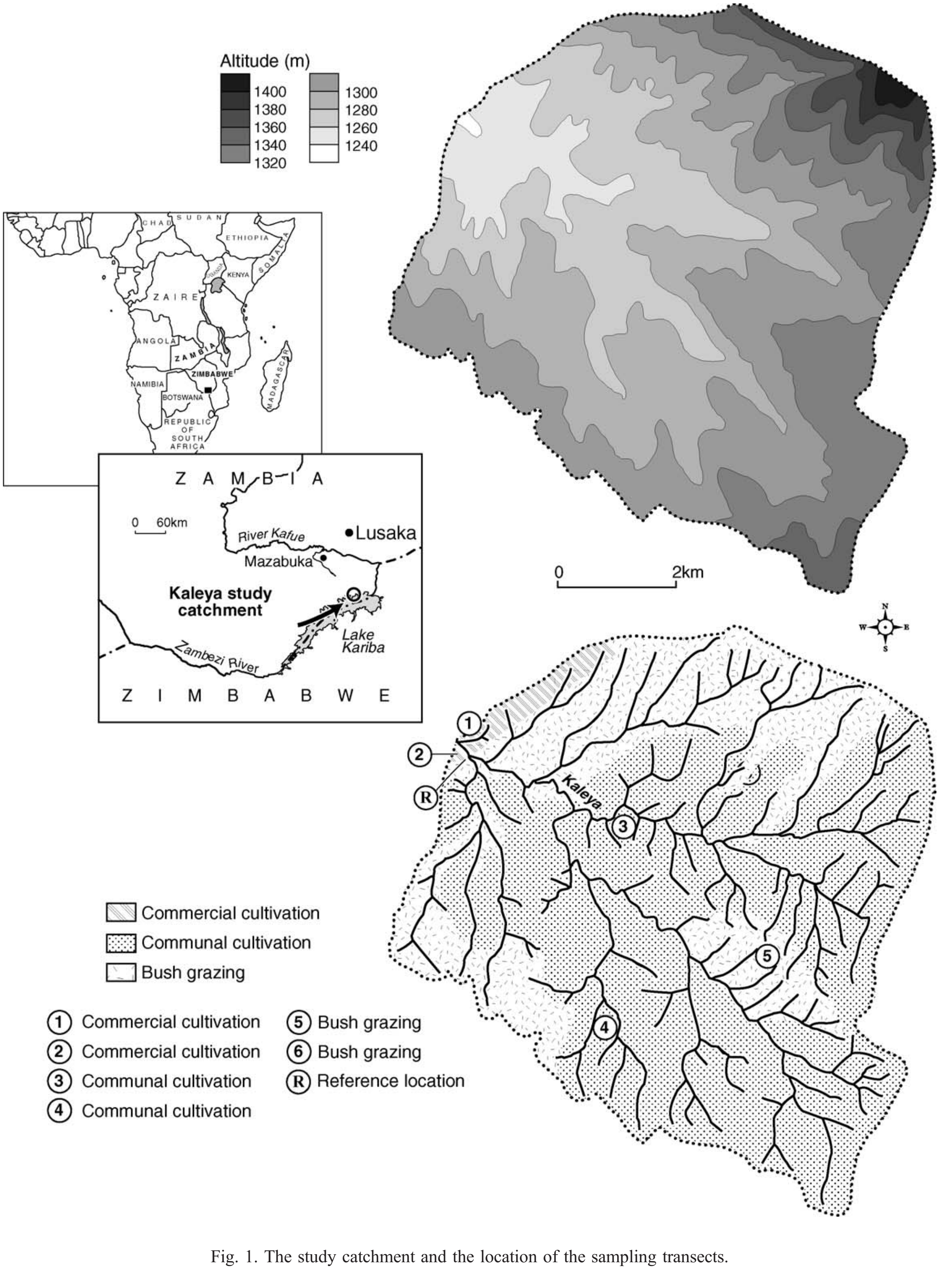

Fig. 1. The study catchment and the location of the sampling transects.

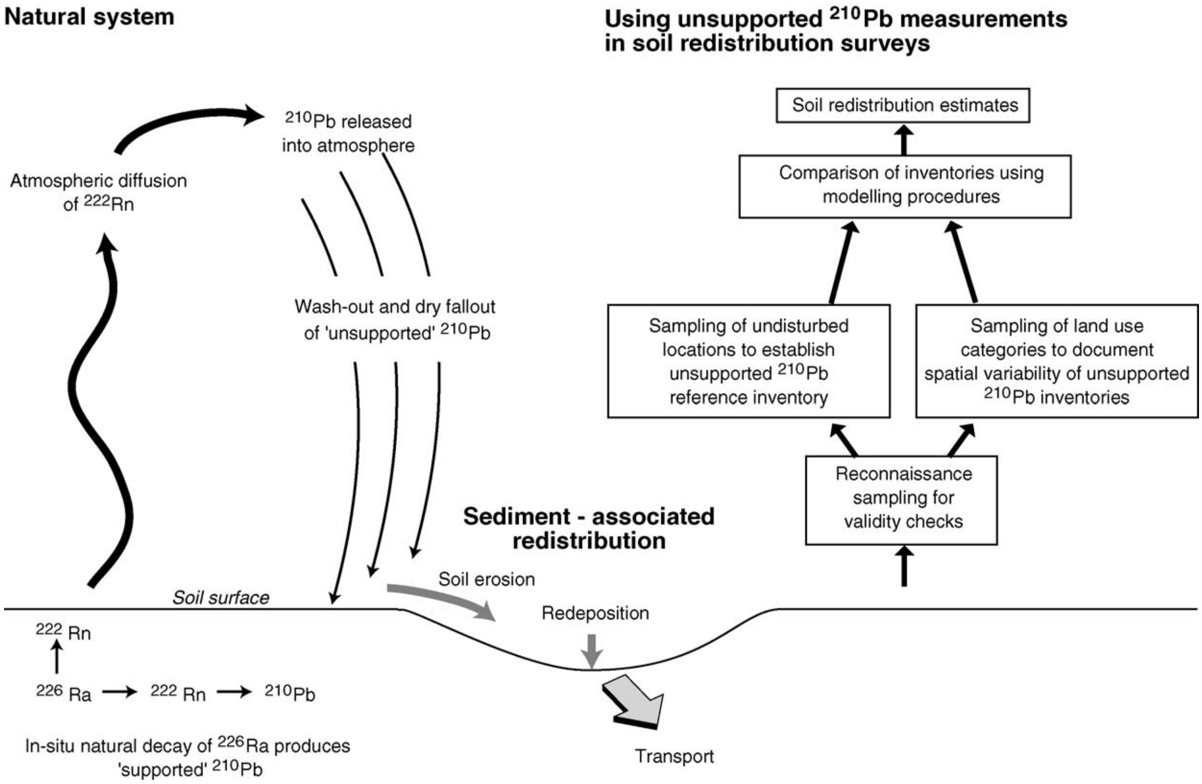

Fig. 2. Using unsupported ^{210}Pb measurements to estimate soil erosion and deposition rates.

winch. Most cores were treated as bulk samples, but some were sectioned into 2-cm depth increments. The resulting samples were returned to the local project laboratory in Lusaka to be air-dried, disaggregated and sieved through a 2-mm mesh. The < 2 mm fractions of the processed samples were subsequently sent to the University of Exeter, UK, for ^{210}Pb analysis by gamma spectrometry.

Measurements of the unsupported ^{210}Pb concentration in the soil samples were undertaken using a high-resolution low background N-type HPGe detector (EG&G/ORTEC LOAX) coupled to an amplifier and multichannel analyser. Each sample was placed in a plastic Marinelli beaker, the lid of which was sealed with PVC tape at least 21 days prior to analysis, in order to ensure equilibrium between ^{226}Ra and its daughter ^{222}Rn. Count times were typically 80,000 s, providing an analytical precision of ca. ± 13% at the 95% level of confidence. Standard solutions were used to produce calibration standards with a representative bulk density and geometry for determining the efficiency of the detection system. The total ^{210}Pb concentration was determined from the 46.5-keV gamma ray for ^{210}Pb and the ^{226}Ra concentration from the 351.9-keV gamma ray for ^{214}Pb, a short-lived daughter of ^{226}Ra. The unsupported ^{210}Pb concentration was calculated by subtracting the ^{226}Ra-supported ^{210}Pb concentration from the total ^{210}Pb concentration (Joshi, 1987). In addition, a Malvern Mastersizer MS20 laser diffraction granulometer was used to measure the absolute grain size composition of the < 2 mm fraction of each bulk soil sample, following pretreatment with hydrogen peroxide to remove organic matter and chemical dispersion with sodium hexametaphosphate. These particle size data were used in the models employed to derive estimates of soil redistribution rates from the measurements of unsupported ^{210}Pb inventories, in order to take account of the grain size selectivity of sediment mobilisation and deposition.

3.2.2. Reconnaissance measurements

In order to confirm the viability and validate the assumptions of the approach, pilot sectioned soil cores were collected from potential reference locations (i.e. undisturbed sites with minimal slope and no evidence of erosion or deposition) and from areas representative of each main land use within the Kaleya study catchment. Fig. 3 shows the depth distributions of unsupported ^{210}Pb for a representative selection of these cores. In all cases, the unsupported ^{210}Pb activities were well above the detection limit. Equally, in most cases, the unsupported ^{210}Pb activity is found in the upper 16 cm of the soil profile. The depth distribution of unsupported ^{210}Pb associated with a stable undisturbed soil profile is illustrated in Fig. 3a. In common with the depth profiles reported for such sites by studies in other parts of the world, the unsupported ^{210}Pb content of this soil core exhibits peak concentrations at the surface and an approximate exponential decline with depth (cf. He and Walling, 1997). In contrast, the unsupported ^{210}Pb concentrations associated with cores collected from cultivated soils in areas of communal and commercial agriculture in the study catchment (see Fig. 3b and c, respectively) are essentially uniform throughout the plough layer, reflecting the mixing caused by tillage. The shape of these profiles conforms to that expected for cultivated soils (cf. He and Walling, 1997) and provides clear evidence of the greater tillage depths associated with the use of modern farming machinery in the commercial sector. The total inventory for the soil core illustrated in Fig. 3b is 42% lower than the inventory measured for the undisturbed core (Fig. 3a), indicating that a significant proportion of the fallout input of unsupported ^{210}Pb has been lost from this particular site through erosion. In contrast, the inventory for the

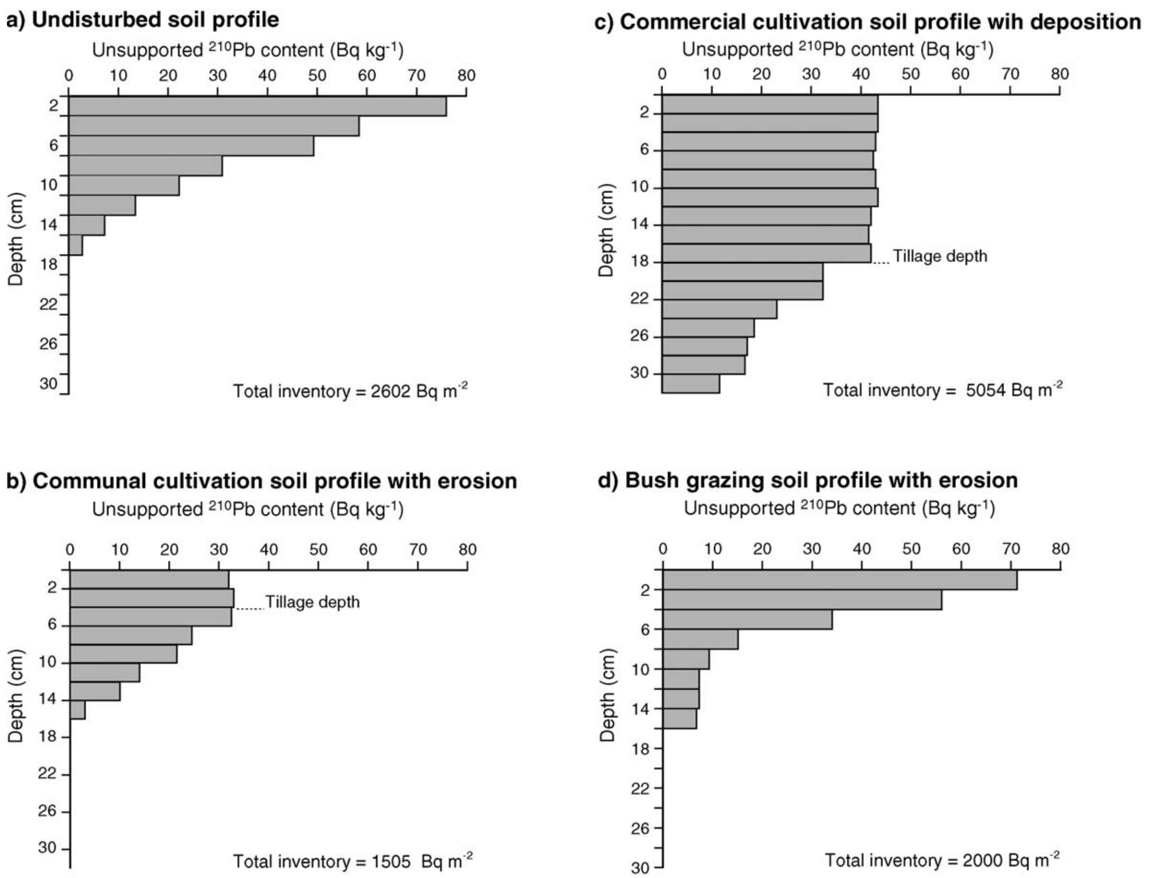

Fig. 3. Representative unsupported ^{210}Pb depth distributions for soil cores collected from the study catchment.

soil core shown in Fig. 3c is almost double of that for the undisturbed soil profile, indicating that this site has experienced significant deposition. Fig. 3d shows the unsupported ^{210}Pb depth profile for a soil core collected from an area of bush grazing. Here, as would be expected (cf. He and Walling, 1997), the ^{210}Pb content declines approximately exponentially with depth. This confirms that the soil has not been mixed by cultivation. The total inventory for this soil core is 23% lower than the corresponding value for the undisturbed soil profile (Fig. 3a), indicating that this particular site has experienced soil loss.

The readily detectable levels of unsupported ^{210}Pb within the soils of the Kaleya study catchment shown by the reconnaissance measurements confirmed the viability of measuring unsupported ^{210}Pb concentrations in soils and sediment from the study area and of documenting the spatial variability of unsupported ^{210}Pb inventories resulting from erosion and soil redistribution. Furthermore, the ^{210}Pb depth distributions for an undisturbed location and for both cultivated and uncultivated areas conformed to those expected, based on other studies. As a result, the samples collected during a comprehensive soil coring programme undertaken in the study catchment for ^{137}Cs measurements (cf. Collins et al., 2001) were also analysed for unsupported ^{210}Pb in order to provide a detailed assessment of longer-term soil redistribution rates.

3.2.3. Catchment sampling

The comprehensive soil coring campaign employed previously for ^{137}Cs measurements encompassed the collection of cores from an undisturbed reference location and from representative areas under each primary land use. The reference location (cf. Fig. 1) was selected using the guidelines conventionally applied in ^{137}Cs investigations (cf. Walling and Quine, 1993). These guidelines identify a number of requirements, including, for example, that the site should be undisturbed and characterised by minimal slope and therefore by negligible erosion or deposition. Following the selection of the reference location, a total of 12 cores had been retrieved from the corners of a 12 × 12 m square, with three replicates collected at each corner.

The sampling programme used to assemble information on the magnitude and spatial variability of ^{137}Cs inventories characterising cultivated and uncultivated land within the study catchment had been stratified to encompass areas of both commercial and communal farming. Two parallel downslope transects were established within each of two representative topographic locations for each primary land use (see Fig. 1). Two replicate soil cores were collected from each of the sampling points along the individual transects, and the two cores were bulked. The transects were typically 30–40 m apart, and the sampling points along the transects were at a spacing of ca. 10–12 m. A detailed topographic survey was undertaken for each transect using a theodolite and staff. This sampling programme yielded a total of 206 bulked soil samples. In addition, a total of 33 complementary sectioned cores were collected from the upper, middle and lower sections of the slope transects.

3.3. Data processing and analysis

3.3.1. The approach

As with ^{137}Cs measurements, the estimation of longer-term (~ 100 years) soil redistribution rates from unsupported ^{210}Pb measurements is based on a comparison between the unsupported ^{210}Pb inventories for individual sampling points and the local reference inventory. A reduced inventory relative to the reference inventory is indicative of erosion, while an increased inventory is indicative of deposition. A computer-based conversion model, which converts the percentage loss or gain in the unsupported ^{210}Pb inventory, relative to the reference value, to an equivalent rate of soil loss or deposition was used to estimate the erosion or deposition rate.

3.3.2. The unsupported ^{210}Pb reference inventory

A summary of the unsupported ^{210}Pb inventories associated with the 12 soil cores collected from the undisturbed location within the Kaleya study catchment is presented in Table 1. These statistics indicate that the use of an inventory value based on a single core could be unrepresentative because the unsupported ^{210}Pb inventories at this reference location are spatially variable. Similar findings have been reported by a number of studies using ^{137}Cs measurements to investigate soil erosion (cf. Wallbrink et al., 1994). On the basis of these data, the unsupported ^{210}Pb reference inventory for the study area was estimated to be 2670 ± 402 Bq m^{-2} at the 95% level of confidence (i.e $\bar{x} \pm 2$SE). Assuming a constant depo-

Table 1
Summary of the unsupported ^{210}Pb inventories measured for the soil cores collected from the reference location in the Kaleya study catchment

	N	Mean (Bq m^{-2})	Standard deviation (Bq m^{-2})	CV (%)	SE mean (Bq m^{-2})
All cores	12	2670	696	26	201
Sampling points	4	2670	608	23	304

sition flux, this reference inventory represents an annual atmospheric ^{210}Pb deposition rate of $\sim$ 83 Bq m^{-2} year^{-1}. This value is consistent with the global range of annual deposition fluxes of unsupported ^{210}Pb of 50–150 Bq m^{-2} year^{-1} reported by Appleby and Oldfield (1992). Its position within the lower portion of the range may reflect the relatively low annual precipitation (ca. 800–900 mm) of the study area, and the existence of an extended dry season during which fallout, which occurs primarily in association with rainfall, will be minimal.

3.3.3. The conversion model for cultivated land

Longer-term ($\sim$ 100 years) water-induced soil redistribution rates on cultivated (commercial and communal) land within the Kaleya study catchment were estimated using a theoretical mass balance model based on that developed previously at the University of Exeter (cf. Walling and He, 1999a), but also incorporating the role of tillage translocation in the redistribution of fallout inputs. The original model takes account of the essentially continuous atmospheric deposition of unsupported ^{210}Pb and its subsequent radioactive decay, as well as its redistribution in association with water-induced erosion or deposition. In addition, the model considers the particle size selectivity of sediment mobilisation and transport and the removal of freshly deposited fallout ^{210}Pb by erosion, before its incorporation into the tillage horizon. The basic form of the model for an eroding site can be expressed as:

$$\frac{\mathrm{d}A(t)}{\mathrm{d}t} = (1-\Gamma)I(t) - \left(\lambda + P\frac{R}{d}\right)A(t) \tag{1}$$

where $A(t)$ = cumulative ^{210}Pb inventory (Bq m^{-2}); R = erosion rate (kg m^{-2} year^{-1}); d = tillage depth (kg m^{-2}); λ = ^{210}Pb decay constant (year^{-1}); $I(t)$ = annual fallout ^{210}Pb deposition flux at time t (Bq m^{-2} year^{-1}); Γ = the proportion of the freshly deposited ^{210}Pb fallout input removed by water erosion before incorporation into the tillage layer; and P = a particle size correction factor to take account of differences between the grain size composition of the mobilised sediment and the original soil.

The tillage depth d, expressed as a mass depth, was typically 160–200 kg m^{-2} and 60–90 kg m^{-2} for commercial and communal areas, respectively (Collins et al., 2001). Γ was estimated using experimental data for the initial depth distribution of fallout ^{210}Pb in surface soil (represented by the relaxation mass depth H), and information on the local rainfall regime in relation to the timing of cultivation (represented by a proportion factor γ). On the basis of experiments reported by He and Walling (1997), the value of H was assumed to be ca. 4.0 kg m^{-2}. Values of γ are typically 0.5 (Walling and He, 1999a). Walling and He (1999a) provide further details on these aspects of the conversion model used for cultivated land. The particle size correction factor was calculated using the ratio of the specific surface area (m^2 g^{-1}) of mobilised sediment to that of the original soil (cf. He and Walling, 1996), represented by the relationship:

$$P = \left(\frac{S_{ms}}{S_{os}}\right)^{v} \tag{2}$$

where S_{ms} = specific surface area of mobilised sediment (m^2 g^{-1}); S_{os} = specific surface area of original soil (m^2 g^{-1}); and v = constant (typically 0.76 for unsupported ^{210}Pb in cultivated soils (cf. He and Walling, 1996). In the study catchment, P was typically >1.0 because transported sediment is generally enriched in fines compared with the original soil (cf. Collins et al., 2001).

Assuming a continuous input of fallout ^{210}Pb, solution of Eq. (1) gives:

$$A(t) = A(t_o)\mathrm{e}^{-\int_{t_o}^{t}(PR/d+\lambda)\mathrm{d}t'} + \int_{t_o}^{t}[1-P\gamma \times (1-\mathrm{e}^{-R/H})]I(t')\mathrm{e}^{-(PR/d+\lambda)(t-t')}\mathrm{d}t' \tag{3}$$

where t_o = year when cultivation started (taken to be 100 years prior to the sample collection date and assuming negligible erosion before t_o).

In situations where $A(t)$ at a sampling point is greater than the local reference inventory, deposition may be assumed. The excess unsupported ^{210}Pb inventory, defined as the measured total inventory $A(t)$ less the local reference inventory, can be attributed to the accumulation of fallout ^{210}Pb associated with the deposition of sediment eroded from the upslope area, which can be expressed as:

$$A_{c,ex} = \int_{t_o}^{t} R'C_d(t')e^{e-\lambda(t-t')}dt' \quad (4)$$

where $A_{c,\,ex}$ = excess ^{210}Pb inventory (Bq m^{-2}); R' = sediment deposition rate (kg m^{-2} year^{-1}); and $C_d(t')$ = ^{210}Pb concentration in deposited sediment (Bq kg^{-1}).

$C_d(t')$ represents the weighted mean unsupported ^{210}Pb concentration of soil and sediment particles mobilised from the upslope contributing area converging on the aggrading point, and can be calculated as:

$$C_d(t') = \frac{1}{\int_S R\mathrm{d}S} \int_S P'C_e(t')R\mathrm{d}S \quad (5)$$

where P' = a further particle size correction factor reflecting the ratio of the specific surface area of the deposited sediment to that of the mobilised sediment (cf. Eq. (2)) (P' is generally < 1.0 because of the preferential deposition of coarser particles in areas of soil accumulation); $C_e(t')$ = ^{210}Pb concentration in mobilised sediment (Bq kg^{-1}); and S = the upslope contributing area (m^2).

The role of tillage translocation in the lateral redistribution of unsupported ^{210}Pb was incorporated into the conversion model following the same procedure as that used by Walling and He (1999b) for a ^{137}Cs conversion model developed for slope transects. The reader is referred to Walling and He (1999b) for further details on the procedure and its assumptions, but a brief outline is provided below. The effect of tillage in redistributing soil can be represented by a downslope sediment flux. Following Govers et al. (1996), the downslope sediment flux F_Q (kg m^{-1} year^{-1}) from a unit contour length may be expressed as:

$$F_Q = \phi \sin\beta \quad (6)$$

where β (°) = the steepest slope angle; ϕ (kg m^{-1} year^{-1}) = a constant related to the tillage practice employed.

If, for a uniform slope (i.e. where there is no divergence or convergence of surface flow), a flow line down the slope is divided into several segments and each segment can be approximated by a straight line, then for the ith section (from the hilltop), the net soil redistribution induced by tillage R_t (kg m^{-2} year^{-1}) can be expressed as:

$$\begin{aligned} R_t &= (F_{Q,out} - F_{Q,in})/L_i \\ &= \phi(\sin\beta_i - \sin\beta_{i-1})/L_i = R_{t,out} - R_{t,in} \end{aligned} \quad (7)$$

where β_i and β_{i-1} are the slope angles of the ith and $(i-1)$th segments, L_i (m) is the slope length of the ith segment, and $R_{t,out}$ (kg m^{-2} year^{-1}) and $R_{t,in}$ (kg m^{-2} year^{-1}) are defined as:

$$R_{t,out} = \phi \sin\beta_i / L_i$$

$$R_{t,in} = \phi \sin\beta_{i-1} / L_i \quad (8)$$

Values of the parameter ϕ in Eq. (6) can be estimated from the results of experimental studies (cf. Govers et al., 1994).

For a point experiencing soil loss, the water erosion rate R_w (kg m^{-2} year^{-1}) can be expressed as:

$$R_w = R - (R_{t,out} - R_{t,in}) \quad (9)$$

where R is the net soil redistribution rate.

For a point experiencing deposition, the water-induced deposition rate R'_w (kg m^{-2} year^{-1}) can be expressed as:

$$R'_w = R - (R_{t,out} - R_{t,in}) \quad (10)$$

3.3.4. The conversion model for uncultivated grazing land

A modified version of the theoretical diffusion and migration conversion model developed at the University of Exeter for use with ^{137}Cs measurements (cf. Walling and He, 1999b) was employed to estimate longer-term water-induced soil redistribution rates on uncultivated grazing land within the Kaleya study catchment. This model assumes a constant fallout flux of unsupported ^{210}Pb and takes into account post-depositional redistribution processes and their influence on the ^{210}Pb depth distribution. A diffusion coefficient D (kg^2 m^{-4} year^{-1}) and a migration rate V (kg m^{-2} year^{-1}) are used to represent the net effect of the slow vertical redistribution of unsupported ^{210}Pb by physicochemical and biological processes. These parameters account for the evolution of the shape of the unsupported ^{210}Pb depth profile over time.

Initial estimates of D and V were obtained using the guidelines proposed by Walling and He (1999b) for ^{137}Cs depth profiles. These values were further optimised by comparing the shapes of measured unsupported ^{210}Pb depth profiles from undisturbed locations with the shape predicted by the model when combining these parameters with the estimated local fallout reference inventory and the ^{210}Pb decay constant (λ). The values of D and V were adjusted until a close match between the observed and predicted unsupported ^{210}Pb depth distributions was obtained.

At a location experiencing water-induced erosion, the rate of soil loss R (kg m^{-2} year^{-1}) can be estimated from the reduction in the unsupported ^{210}Pb inventory ($A_{u,ls}(t)$) compared with the local reference value, in combination with information on the ^{210}Pb content of the surface soil ($C_u(t')$), viz.:

$$\int_0^t PRC_u(t')e^{-\lambda(t-t')}dt' = A_{u,ls}(t) \tag{11}$$

Alternatively, at a location experiencing water-induced deposition, the rate of sedimentation R' (kg m^{-2} year^{-1}) can be estimated from the increase in the ^{210}Pb inventory ($A_{u,\,ex}(t)$) compared with the local reference value, coupled with information on the ^{210}Pb content of deposited sediment ($C_d(t')$), viz.:

$$R' = \frac{A_{u,ex}}{\int_0^t C_d(t')e^{-\lambda(t-t')}dt'} \tag{12}$$

where $C_d(t')$ can be calculated as the product of:

$$C_d(t') = \frac{1}{\int_S R\mathrm{d}S}\int_S P'\,PC_u(t')R\mathrm{d}S \tag{13}$$

4. Results

The conversion models were used to generate estimates of point soil redistribution rates (kg m^{-2} year^{-1}) from the unsupported ^{210}Pb inventories associated with individual soil cores collected along each slope transect. Assuming each transect represented a 1-m wide strip, these point estimates were used to calculate equivalent values of soil loss or deposition (kg year^{-1}) for individual slope segments, extending halfway to the adjacent coring points from the sampling point in each direction. The resulting values for each segment were summed to establish a sediment budget for the overall transect, comprising estimates (~ 100 years) of total erosion, total deposition, net soil loss and the sediment delivery ratio. Finally, the estimates of soil redistribution obtained for the individual transects were averaged to establish the mean sediment budget for each land use.

Equivalent results based on ^{137}Cs measurements were available for each transect (cf. Collins et al., 2001), and comparison of the two data sets provides a means of confirming the general validity of the estimates of soil redistribution obtained from the unsupported ^{210}Pb measurements. It must, however, be recognised that the two radionuclides are unlikely to provide identical results, due to the different time periods involved. In the case of ^{137}Cs measurements, the estimates of mean soil redistribution rate relate to the period extending from the onset of ^{137}Cs fallout in the mid-1950s (or the peak of ^{137}Cs fallout, i.e. 1964) to the time of sampling. In contrast, the essentially continuous input of ^{210}Pb fallout through time means that the contemporary unsupported ^{210}Pb inventory will reflect redistribution and thus loss and gain of unsupported ^{210}Pb over a longer period. The length of this period is, however, constrained by the half-life of ^{210}Pb (i.e. 22.2 years), because the effect of past changes in the unsupported ^{210}Pb inventory, caused by erosion and deposition, on the contemporary inventory, will progressively decline as the period of time elapsed increases. Following previous work, it has been assumed that the contemporary unsupported ^{210}Pb inventory will only be sensitive to erosion and deposition occurring within a period equivalent to four times the half-life, and thus the past 100 years. Furthermore, the progressive reduction in the effect of past changes in the inventory on the contemporary inventory, as the period of time elapsed since those changes increases, must also be taken into account when interpreting the impact of the erosional history of a study site on the magnitude of the measured unsupported ^{210}Pb inventory. This inventory will clearly be more sensitive to recent soil redistribution, and the estimate of the mean rate of soil redistribution for the past 100 years provided by the conversion model is thus likely to be biased towards the recent erosional history of the study site. In the case of ^{137}Cs, fallout inputs were effectively restricted to a relatively short period extending from the mid-1950s to the early 1970s. Estimates of mean soil redistribution rates for the past ca. 40 years derived using the appropriate conversion model are thus likely to be biased towards the main period of fallout because the contemporary inventory will have been particularly sensitive to soil redistribution rates during that period. Although clearly involving some uncertainty, comparison of the longer- and medium-term estimates of soil redistribution rates provided by the unsupported ^{210}Pb and ^{137}Cs measurements, respectively, can provide useful information on the erosional history of the study area and more particularly of medium- and longer-term changes in rates of soil loss. Estimates of longer-term water-induced soil redistribution rates derived from the unsupported ^{210}Pb measurements undertaken on the soil cores collected from the individual transects established within the areas of different land use are presented in Figs. 4, 7 and 10. The values indicated for the individual cores represent estimates of the total soil loss or gain from the slope segment represented by a strip 1 m wide extending halfway to the adjacent coring points from the sampling point in each direction. In some cases, there are minor inconsistencies in the results, in that deposition rates within the upper part of the transect cannot be accounted for by the erosion rates in the upslope segments. These apparent inconsistencies reflect the fact that the transects were selected to be representative of individual land use types and, in some instances, they did not extent to the divide. Run-on from upslope areas could result in deposition of sediment on the upper part of the transect. In such situations, the estimate of the net erosion rate for the transect and the associated value of sediment delivery ratio could underestimate the true values. However, in view of the considerable length of most transects, such errors are likely to be of limited significance. The results obtained for each land use type will be discussed in turn.

4.1. Areas of commercial cultivation

The estimates of soil redistribution rates for the cores collected from the transects used to represent commercial cultivation are presented in Fig. 4. Each transect comprises some segments evidencing erosion and some evidencing deposition. Thus, for example, in the case of transect 1A (see Fig. 4), total erosion for the 1-m wide strip is estimated at 171 kg year^{-1} and total deposition at 73 kg year^{-1}, yielding a net soil loss of 98 kg year^{-1} and a sediment delivery ratio of 57%. Sediment deposition generally represents a significant component of the sediment budget for the individual transects and, in consequence, sediment delivery ratios range from 40% to 61%. Net rates of soil loss, calculated as the net soil loss divided by the total surface area of the 1-m wide strip representing each individual transect, range from 0.30 to 0.54 kg m^{-2} year^{-1} (3.0–5.4 t ha^{-1} year^{-1}).

A comparison of the soil redistribution rates derived for the individual sampling points along these transects from unsupported ^{210}Pb measurements with those obtained from ^{137}Cs measurements on the same soil samples (cf. Collins et al., 2001), using a mass balance conversion model incorporating the effects of tillage translocation (cf. Walling and He, 1999b), is presented in Fig. 5. Although some scatter inevitably exists, Fig. 5 shows a clear positive relationship between the soil redistribution rates derived for the longer- (^{210}Pb—ca. 100 years) and medium-term (^{137}Cs—ca. 40 years) time scales. Sampling points consistently experience either erosion or deposition over both time scales, indicating that the detailed pattern of soil redistribution along these transects has remained essentially stable over time. Furthermore, there is no evidence of a systematic trend, such that maximum discrepancies between the two approaches are associated with either areas of high erosion rates (i.e. low inventories) or high deposition rates (i.e. maximum inventories), which could reflect measurement precision. It is, however, important to recognise that, notwithstanding the similarity in the pattern of soil redistribution over the different time periods, the absolute magnitudes of the estimate of erosion or deposition rate obtained for particular points using the ^{137}Cs and unsupported

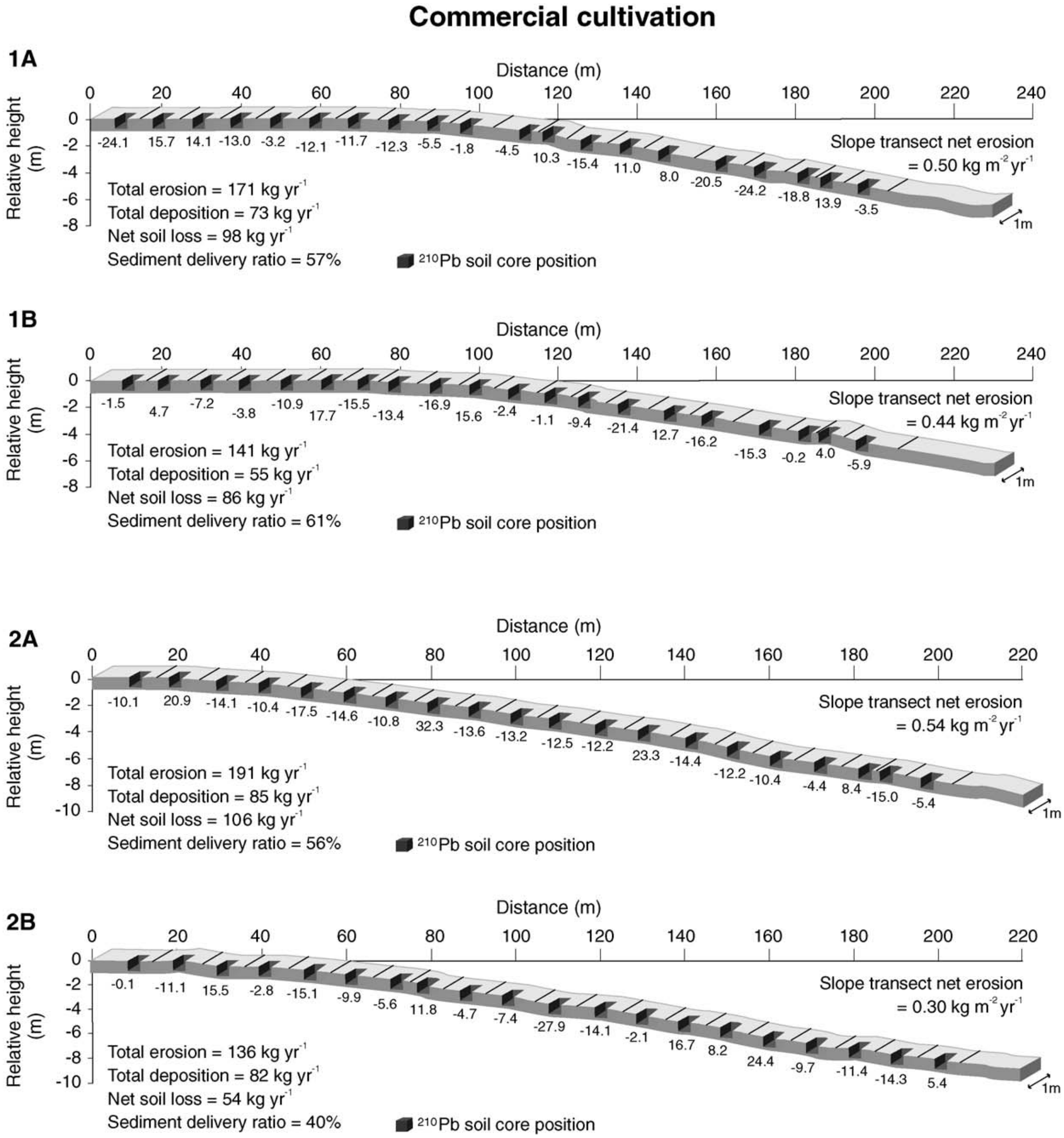

Fig. 4. Estimates of soil redistribution rates for individual slope segments derived from unsupported ^{210}Pb measurements for transects supporting commercial cultivation.

^{210}Pb measurements differ. Consequently, some points are seen to experience greater rates of erosion or deposition over the medium-term, as compared to the longer-term, and vice versa.

A further comparison of the soil redistribution rates estimated for land supporting commercial cultivation using both unsupported ^{210}Pb and ^{137}Cs measurements is provided by the corresponding schematic mean

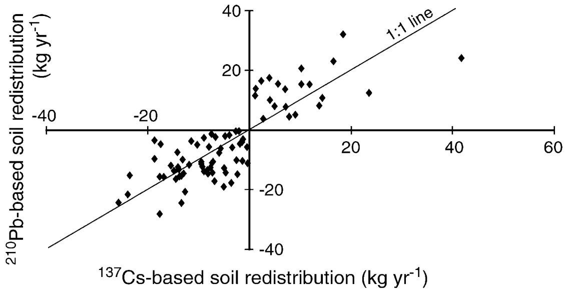

Fig. 5. The relationship between ^{210}Pb- and ^{137}Cs-derived estimates of soil redistribution rates for individual slope segments obtained for the soil cores collected from the transects located in areas of commercial cultivation.

sediment budgets presented in Fig. 6. Overall, the typical rates of net soil loss are similar for both time periods. Commercial farmers adopted soil conservation practices early in the 20th century, making full use of available technical expertise, planting advice and subsidies and have continued to do so since Zambia gained independence (Stocking, 1985). Consequently, due to the continuing priority accorded to conservation measures, the mean rates of net soil loss from commercial plantations are similar for both periods, being 4.5 t ha^{-1} year^{-1} over the longer-term and 4.3 t ha^{-1} year^{-1} over the medium-term. These results afford an interesting perspective on recent changes in cropping practices within the commercially farmed areas of the study catchment. Soil erosion was widespread during much of the 20th century because local commercial farms principally produced root crops, which exposed extensive areas of bare soil to rainsplash and sediment transport. In response to these problems, root crops have been replaced by coffee plantations over the past 15 years in an attempt to stabilise the soils and reduce erosion. Although any significant reduction in erosion rates associated with the more recent introduction of coffee as the principal cash crop could be expected to be reflected in a reduction of the medium-term (~ 40 years) erosion rates derived from ^{137}Cs measurements, when compared with the longer-term erosion rates estimated from ^{210}Pb measurements, the results of this investigation suggest that this change has had little effect on the time-averaged soil loss over this period.

4.2. Areas of communal cultivation

Fig. 7 presents the longer-term water-induced soil redistribution rates derived from the unsupported ^{210}Pb measurements on the soil cores collected from the transects used to represent areas of communal cultivation. For transect 4A (see Fig. 7), the conversion model predicts a total erosion of 142 kg year^{-1} and a total deposition of 90 kg year^{-1}, yielding a net soil loss of

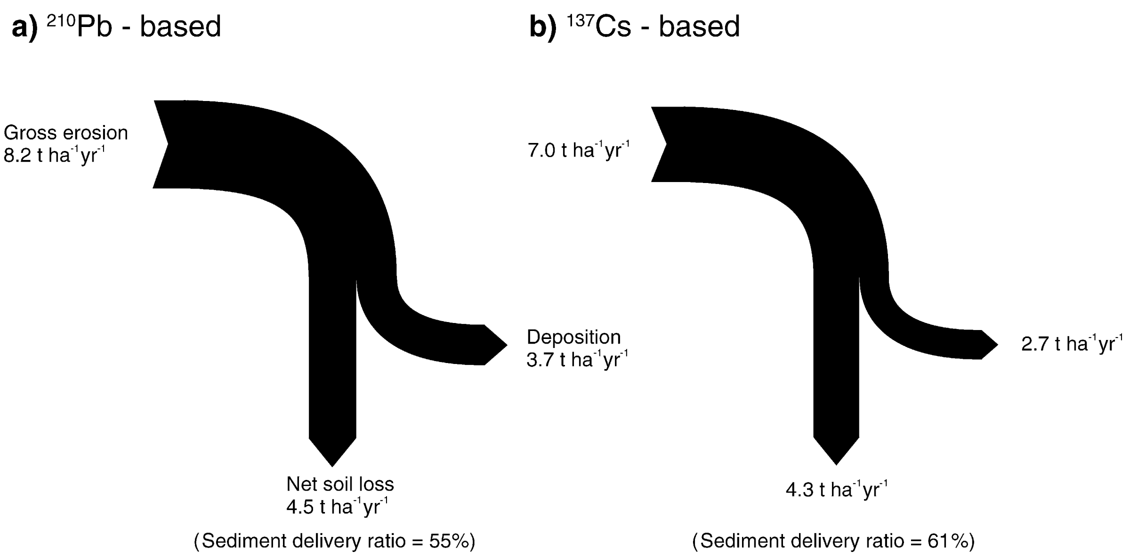

Fig. 6. A comparison of schematic mean sediment budgets for areas of commercial cultivation established using unsupported ^{210}Pb (a) and ^{137}Cs (b) measurements.

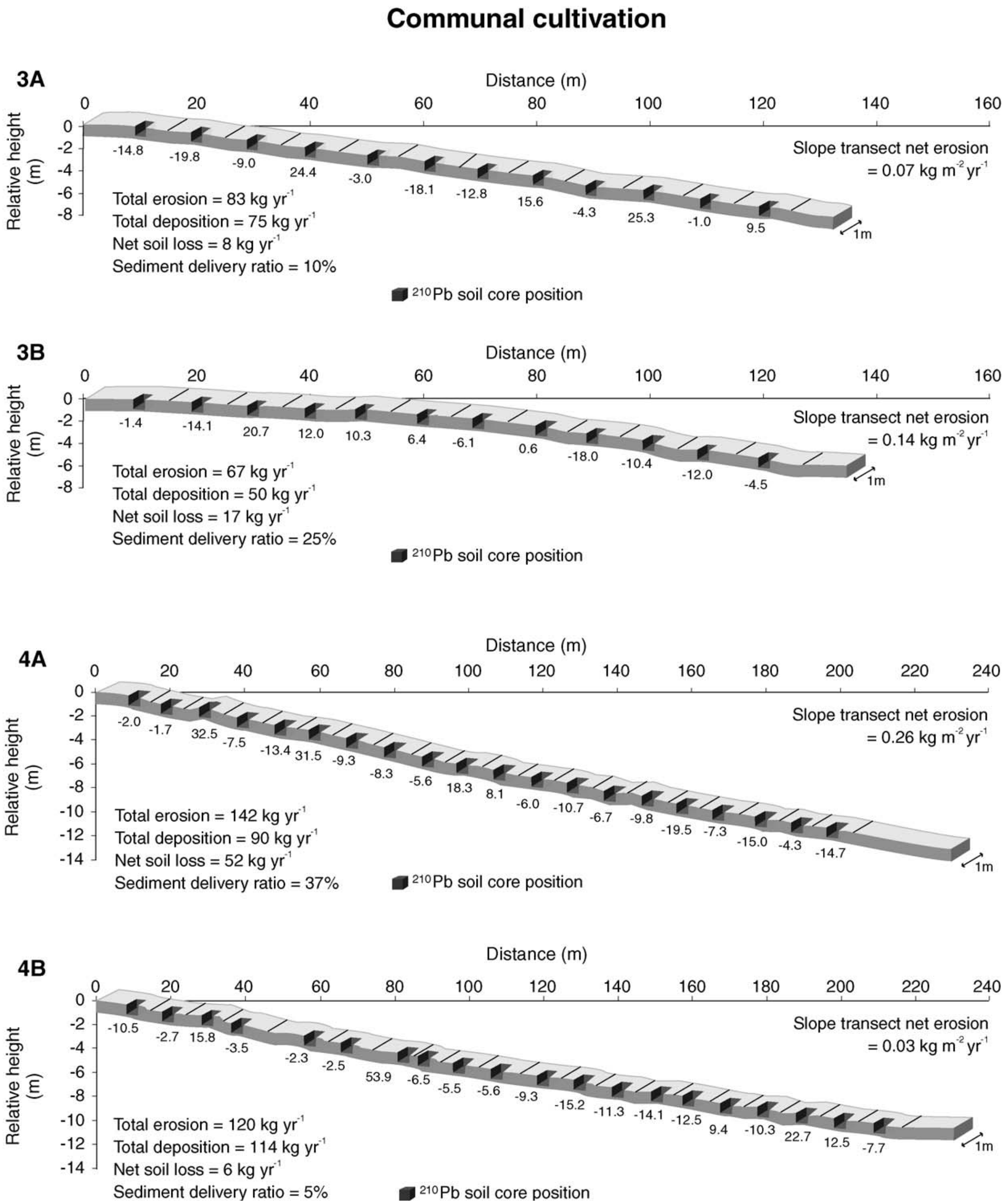

Fig. 7. Estimates of soil redistribution rates for individual slope segments derived from unsupported ^{210}Pb measurements for transects supporting communal cultivation.

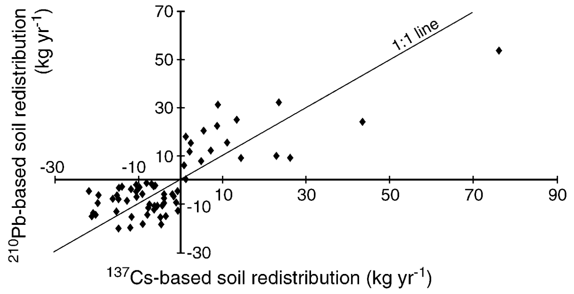

Fig. 8. The relationship between ^{210}Pb- and ^{137}Cs-derived estimates of soil redistribution rates for individual slope segments obtained for the soil cores collected from the transects located in areas of communal cultivation.

52 kg year^{-1} and a sediment delivery ratio of 37%. In contrast, for transect 3A (see Fig. 7), the total erosion is estimated at 83 kg year^{-1} and total deposition at 75 kg year^{-1}, yielding a net soil loss of 8 kg year^{-1} and a sediment delivery ratio of 10%. Net rates of soil loss range between 0.03 and 0.26 kg m^{-2} year^{-1} (0.3–2.6 t ha^{-1} year^{-1}) and are therefore significantly lower than the corresponding values for the area of commercial farming.

The scatter plot presented in Fig. 8 illustrates the relationship between the soil redistribution rates for the individual soil cores collected along these transects, derived from unsupported ^{210}Pb and ^{137}Cs measurements. The soil redistribution rates estimated from ^{137}Cs measurements (cf. Collins et al., 2001) were again derived using a mass balance conversion model incorporating the effects of tillage translocation (cf. Walling and He, 1999b). The overall patterns of soil redistribution associated with the two data sets are again similar, with individual points experiencing either erosion or deposition over both periods. The precise magnitude of the erosion or deposition rates estimated for individual sampling points differs between the two time scales, but there is no clear trend in the relative magnitude of mean soil redistribution rates over the two time scales. For some points, the ^{137}Cs-based values are greater than those estimated from the unsupported ^{210}Pb measurements, whereas for other points, the reverse is the case.

Schematic sediment budgets representing mean soil redistribution rates for areas of communal cultivation based on the unsupported ^{210}Pb and ^{137}Cs measurements are shown in Fig. 9. A number of contrasts between the values obtained from the measurements of the two radionuclides are apparent. For example, the longer-term (unsupported ^{210}Pb) estimates of both gross and net soil loss (6.4 and 1.2 t ha^{-1} year^{-1}, respectively) are somewhat lower than the medium-term estimates provided by the ^{137}Cs measurements (7.0 and 2.5 t ha^{-1} year^{-1}, respectively). Changing

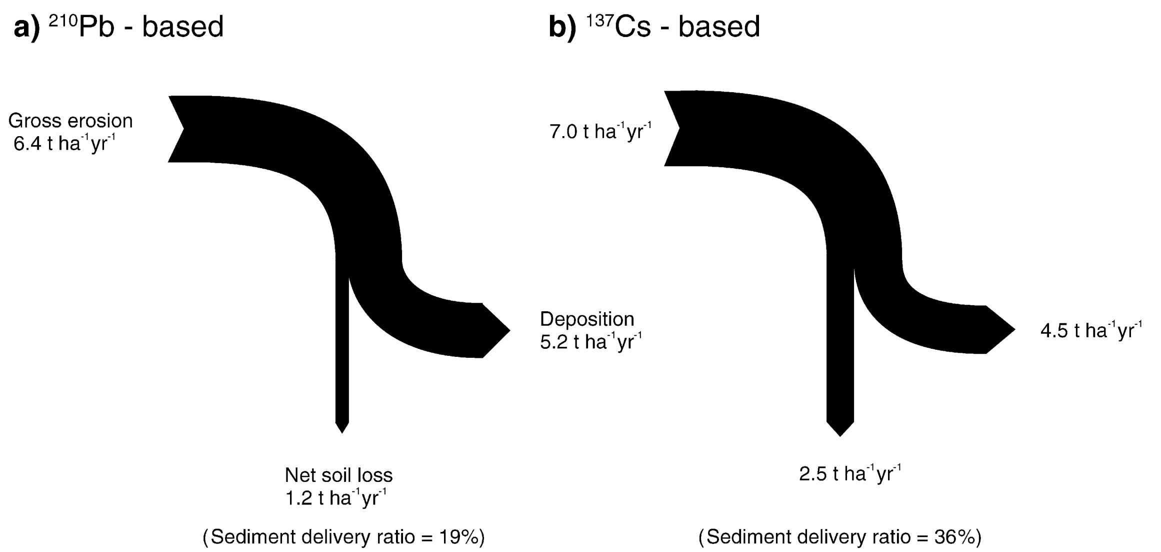

Fig. 9. A comparison of schematic mean sediment budgets for areas of communal cultivation established using unsupported ^{210}Pb (a) and ^{137}Cs (b) measurements.

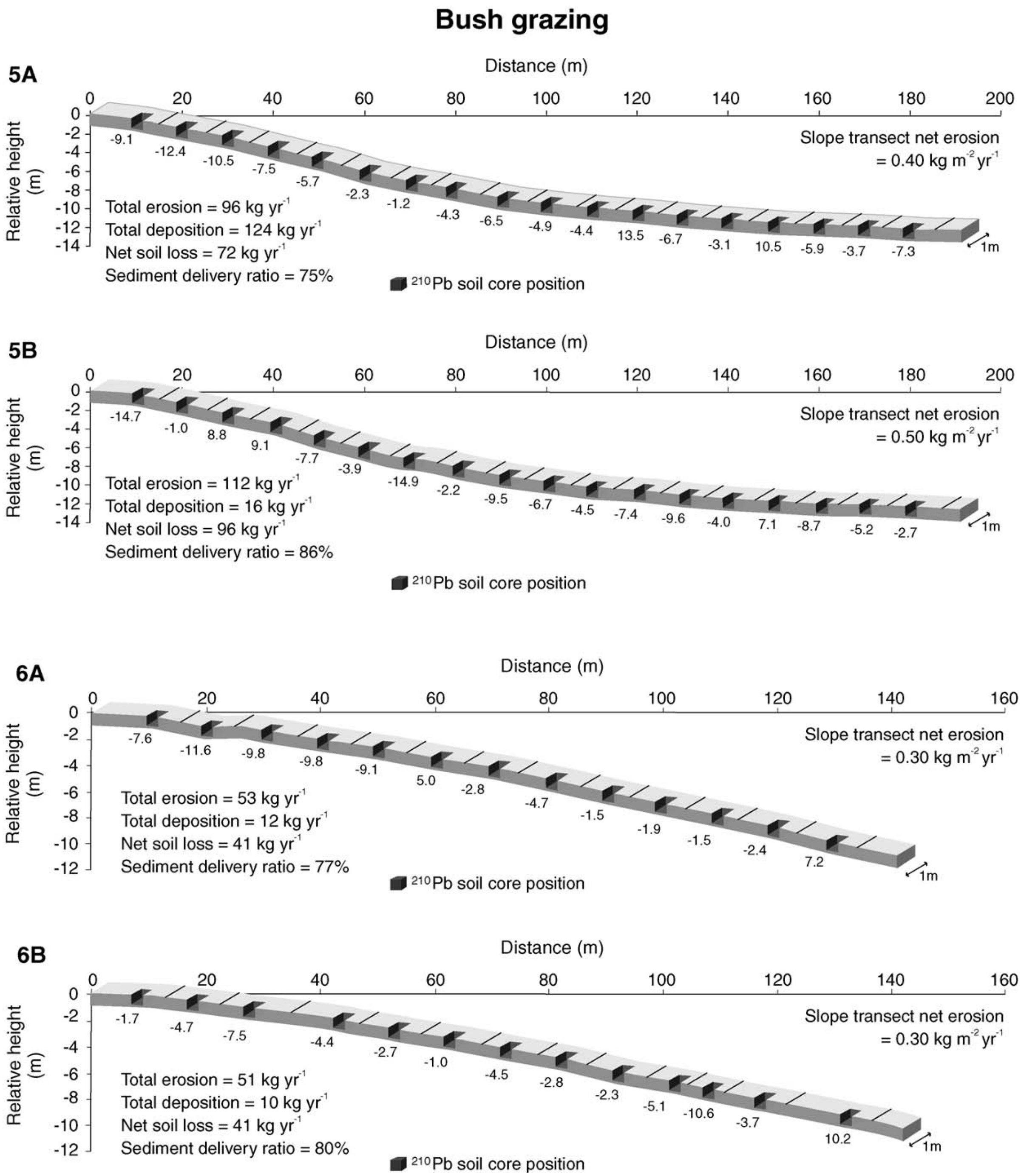

Fig. 10. Unsupported ^{210}Pb-based estimates of soil redistribution for transects supporting bush grazing cultivation.

land management practices are likely to represent an important explanatory factor. Because of the severe soil erosion widely reported in communal areas of southern Zambia during the early 20th century (Stone, 1978; Sturmheit, 1990), a major programme of soil conservation based on contour bunds and grass strips was introduced by the early 1930s (Stocking, 1985). This management programme is likely to have controlled soil loss during a considerable proportion of the time period encompassed by unsupported ^{210}Pb measurements. However, following independence in the 1960s, communal farmers abandoned soil conservation schemes because government policy diverted funds and trained manpower to alternative rural development initiatives or the mining sector (Noren and Norden, 1983), and because such programmes were unpopular following their legal enforcement under the colonial regime (Johnson, 1956; Robinson, 1978). The withdrawal of Zambia from the Southern African Regional Committee for the Conservation and Utilisation of the Soil (SARCCUS) compounded this situation (Stocking, 1985). Consequently, soil loss from areas beneath communal cultivation is likely to have increased over the second half of the 20th century. Any increase in soil erosion associated with the demise of soil protection schemes in the study area over the past 40 years can therefore be expected to have had a more profound effect on the medium-term ($\sim$ 40 years) erosion rates estimated from ^{137}Cs measurements than on the longer-term values estimated from the unsupported ^{210}Pb measurements. The medium-term net soil loss from such areas and the corresponding sediment delivery ratio are therefore double the equivalent longer-term estimates (see Fig. 9). Effective soil conservation measures can be expected to reduce both gross and net erosion rates and the sediment delivery ratio.

4.3. Areas of bush grazing

The estimates of longer-term water-induced soil redistribution rates derived for areas of bush grazing are shown in Fig. 10. Sediment delivery ratios are higher (75–86%) than those estimated for the areas under both communal and commercial cultivation (5–61%) because the multitude of paths and tracks linking grazing areas and the local villages increases the efficiency of sediment delivery from the local grazed hillslopes (cf. Lal, 1985; Elwell, 1990). In addition, the widespread surface crusting observed on uncultivated soils reduces infiltration and therefore increases surface runoff and sediment transport. Net rates of soil loss vary between 0.3 and 0.5 kg m^{-2} year^{-1} (3.0–5.0 t ha^{-1} year^{-1}) and are therefore similar to those estimated for areas of commercial cultivation, while exceeding those obtained for areas under communal cultivation.

Fig. 11 compares the estimates of soil redistribution rates derived from the measurements of unsupported ^{210}Pb and ^{137}Cs undertaken on the individual soil cores collected from the transects selected as representative of bush grazing. The ^{137}Cs-based estimates (cf. Collins et al., 2001) were derived using a theoretical profile distribution conversion model (the diffusion and migration model of Walling and He, 1999b). A clear positive relationship again exists between the two data sets, indicating that the overall pattern of soil redistribution has not changed substantially over the past 100 years. The rates of erosion or deposition at particular sampling points, nevertheless, provide evidence of a consistent trend over the two time scales. The estimates of both erosion and sediment deposition rates provided by the unsupported ^{210}Pb measurements are generally higher than those provided by the ^{137}Cs measurements (see Fig. 11).

A comparison of the mean soil redistribution rates estimated for land supporting bush grazing using measurements of unsupported ^{210}Pb and ^{137}Cs is provided by the schematic sediment budgets presented in Fig. 12. The mean gross and net soil erosion rates over the longer-term (4.5 and 3.7 t ha^{-1} year^{-1}, respectively) exceed the corresponding values (3.4 and

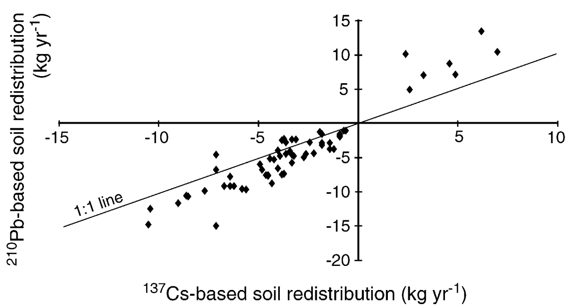

Fig. 11. The relationship between ^{210}Pb- and ^{137}Cs-derived estimates of soil redistribution rates for individual slope segments obtained for the soil cores collected from the transects located in areas of bush grazing.

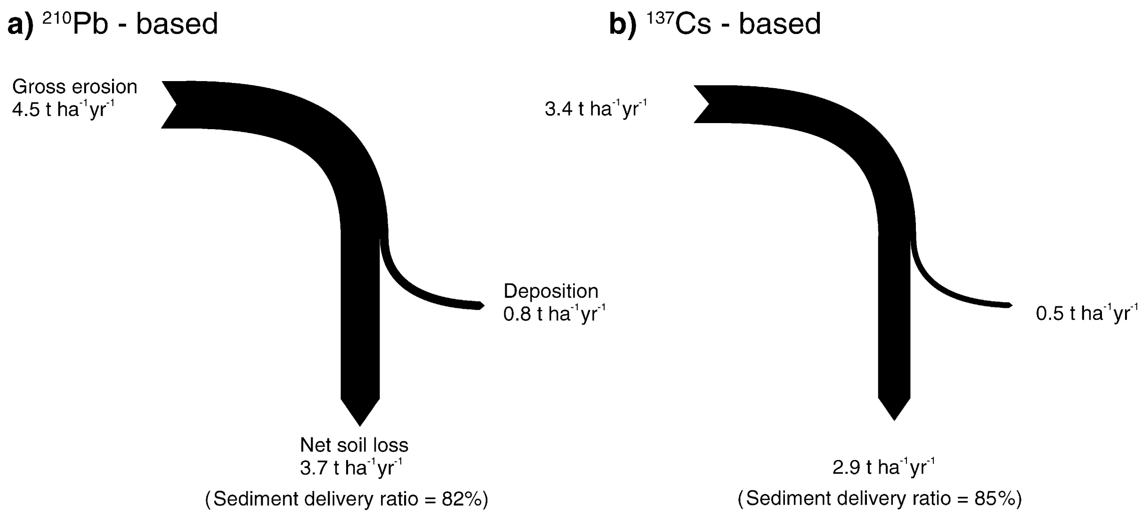

Fig. 12. A comparison of the typical sediment budgets for areas of bush grazing established using unsupported ^{210}Pb (a) and ^{137}Cs (b) measurements.

2.9 t ha^{-1} year^{-1}, respectively) for the medium-term. Overgrazing problems were common in the study catchment during the first half of the 20th century due to population pressures in the local native reserves (Stocking, 1985). However, since independence in the 1960s, increasing rural–urban migration and the associated reduction in stocking densities has gradually reduced such problems. Consequently, the erosion of soils under bush grazing could be expected to have decreased over the past 40 years and, in turn, this change can be assumed to have had a greater influence on the medium- rather than the longer-term estimates of mean soil redistribution rates. The similarity of the sediment delivery ratios for the two time periods indicates that the overall balance of erosion and deposition on slopes supporting bush grazing has, nevertheless, remained stable (see Fig. 12).

5. Perspective

Due to the many limitations associated with the measurement techniques traditionally used to assemble information on rates of soil erosion and sediment redistribution, the use of fallout radionuclides as a basis for estimating erosion and deposition rates would appear to offer considerable potential. Recent studies undertaken in many areas of the world have clearly confirmed the potential of ^{137}Cs measurements for this purpose. However, in the foreseeable future, the use of ^{137}Cs measurements is likely to be constrained by the global pattern of bomb fallout inventories and, more particularly, by the measurement problems associated with the resulting low concentrations of ^{137}Cs reported for soil and sediment samples found in some parts of the world, especially equatorial areas.

Against this background, this paper has reported the successful use of unsupported ^{210}Pb measurements to estimate longer-term (ca. 100 years) soil redistribution rates for areas under different land use in a small catchment in southern Zambia, where contemporary ^{137}Cs inventories are very low. The results indicate that the longer-term average rates of net soil loss for the principal land use types in the study catchment can be ranked in the order: commercial cultivation (4.5 t ha^{-1} year^{-1})>bush grazing (3.7 t ha^{-1} year^{-1})>commu-communal cultivation (1.2 t ha^{-1} year^{-1}). Furthermore, the results obtained are consistent with those previously derived using ^{137}Cs measurements on the same soil samples. This investigation therefore confirms the potential for using unsupported ^{210}Pb measurements as an alternative to ^{137}Cs measurements in areas where ^{137}Cs inventories are low.

Although useful as an alternative approach for assembling information on soil erosion and deposition, scope also clearly exists for combining measurements of unsupported ^{210}Pb and ^{137}Cs on the same soil cores. The conjunctive use of both radionuclides can afford a useful means of obtaining retrospective information on average rates of soil erosion and deposition over different time periods. Analysis of both radionuclides can be undertaken simultaneously and provides a framework for a more detailed assessment of the erosional history of a particular study area, with respect to changing land management practices and other aspects of environmental change.

The results presented above should, nevertheless, be seen as preliminary and need to be qualified in terms of a number of potential limitations. First, the conversion models used to convert measurements of unsupported ^{210}Pb inventories to estimates of soil redistribution rates for both cultivated and grazing areas assume that water is the principal erosive agent and take no account of the potential role of aeolian processes. However, available evidence suggests that aeolian processes are of limited importance in the study area. Secondly, because the ^{210}Pb-based estimates of soil erosion and deposition represent longer-term ($\sim$ 100 years) average values, it is clearly important to take account of the historical land use legacy of a study area when interpreting the results obtained. Two factors require attention in this respect, namely, changes in land use patterns through time and variations in land use intensity. Although detailed historical land use information was not available for the study area, contacts with local commercial farming families suggested that the general land use pattern had not changed significantly during the time period under scrutiny, because the demarcation of communal farming areas and commercial farms occurred in the late 19th century. However, recent epidemics of ‘corridor disease’ have reduced cattle numbers and grazing pressures in bush grazing areas, and commercial farmers have continued to improve their soil conservation measures and to introduce crop types providing extra ground cover. It seems likely that both of these recent changes in agricultural practices and land use intensity will have reduced contemporary erosion rates in the areas of commercial farming and bush grazing within the study catchment, compared to the longer-term average values represented by the estimates derived from the unsupported ^{210}Pb measurements. Thirdly, although the ^{210}Pb measurements have been used to establish schematic sediment budgets for the three different land use types in the study catchment, it should be recognised that the rates of net soil loss indicated by these budgets relate to a series of representative transects sampled within zones characterised by a particular land use, rather than to transects selected to encompass the entire slope profile from hilltop to channel network, irrespective of land use. Consequently, the schematic sediment budgets should not be used to estimate the amount of sediment reaching the channels of the Kaleya river system from areas of the study catchment under different land use. Additional conveyance losses are likely to occur between the transect locations and the river network (cf. Walling et al., 2001). Finally, further work is required to refine the interpretations based on a comparison of the erosion rate estimates provided by the unsupported ^{210}Pb and ^{137}Cs measurements. The two sets of estimates clearly have a different time base, with the ^{137}Cs measurements providing estimates of soil redistribution rates for the past ca. 40 years, whereas the estimates derived from unsupported ^{210}Pb measurements relate to the past ca. 100 years. However, it is also important to recognise that unsupported ^{210}Pb inventories are likely to be more sensitive to recent changes in soil redistribution rates than ^{137}Cs measurements, due to the essentially constant fallout inputs and the shorter half-life of ^{210}Pb, and that contemporary ^{137}Cs inventories are likely to have been most sensitive to soil redistribution rates associated with the time period corresponding to the main period of bomb-derived fallout.

Notwithstanding these limitations, the study reported confirms the potential for using unsupported ^{210}Pb measurements to estimate longer-term soil redistribution rates in an equatorial region of Africa, where ^{137}Cs inventories are very low compared to those in many other areas of the world. Furthermore, the results obtained highlight the potential for conjunctive use of both radionuclides to provide information on the erosional history of a study area over the past 100 years. More work is, however, required to refine the interpretation of combined measurements of both radionuclides.

Acknowledgements

The authors gratefully acknowledge the financial support of the UK DFID (Research Project R6868), the support of a Leverhulme Fellowship to DEW for work on catchment sediment budgets, cooperation of the local landowners in permitting access to the study area for soil sampling, the assistance of Michael Banda, Wisedom N'gandu, Timothy Malambo, Simon Wingrove and Simon Woods with soil core collection and topographic surveys and the help of Sue Rouillard in producing the diagrams. Soil samples were imported into the UK under MAFF Soil Licence No. PHL 43/2408(09/1997). The constructive comments and suggestions of three anonymous referees on an earlier version of the paper are gratefully acknowledged. The paper represents a contribution to the International Atomic Energy Agency Co-ordinated Research Project (CRP) 'Assessing the effectiveness of soil conservation techniques for sustainable watershed management using fallout radionuclides' through Technical Contract 12094.

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