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Assessing soil remobilisation in catchments using a 137 Cs-sediment hillslope model

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Abstract

Soil redistribution studies are important, especially in water supply catchments, because the rate at which denudation is occurring has implications for offsite water quality. However, the extent to which soil is redistributed within the landscape can be difficult to determine. This challenge can be overcome using fallout caesium-137 (^{137}Cs). This paper describes the rates of soil loss and remobilisation in two sub-catchments within the Sydney Basin region, namely Kembla and Kentish Creeks, which drain to the Cordeaux reservoir. The total inventories of ^{137}Cs in catchment soils were determined, a 137Cs-regression equation and a theoretical diffusion and migration model were used to established relationships between ^{137}Cs inventories and the rates of soil loss. These relationships revealed relatively low occurrence of soil loss in Kentish Creek, but two slopes in the Kembla Creek sub-catchment had losses that appear to be moderate. However, there was no clear evidence to suggest whether slopes in upper and lower reaches of catchments had specific patterns of soil remobilisation. Qualitative categorisation of the slope elements using a ^{137}Cs-sediment hillslope model can be a useful sentinel for land users and decision makers even if absolute rates of soil loss or gain are not certain. The findings suggest that sediments mobilised in the study sub-catchments are not likely to impact significantly on the water quality in the Cordeaux reservoir.

Keywords

Assessing, Soil, Remobilisation, Catchments, using, 137, sediment, Hillslope, Model, GeoQUEST

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Assessing Soil Remobilisation in Catchments using a ^{137}Cs-sediment Hillslope Model

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ABSTRACT *Soil redistribution studies are important, especially in water supply catchments, because the rate at which denudation is occurring has implications for offsite water quality. However, the extent to which soil is redistributed within the landscape can be difficult to determine. This challenge can be overcome using fallout caesium-137 (^{137}Cs). This paper describes the rates of soil loss and remobilisation in two sub-catchments within the Sydney Basin region, namely Kembla and Kentish Creeks, which drain to the Cordeaux reservoir. The total inventories of ^{137}Cs in catchment soils were determined, a ^{137}Cs-regression equation and a theoretical diffusion and migration model were used to established relationships between ^{137}Cs inventories and the rates of soil loss. These relationships revealed relatively low occurrence of soil loss in Kentish Creek, but two slopes in the Kembla Creek sub-catchment had losses that appear to be moderate. However, there was no clear evidence to suggest whether slopes in upper and lower reaches of catchments had specific patterns of soil remobilisation. Qualitative categorisation of the slope elements using a ^{137}Cs-sediment hillslope model can be a useful sentinel for land users and decision makers even if absolute rates of soil loss or gain are not certain. The findings suggest that sediments mobilised in the study sub-catchments are not likely to impact significantly on the water quality in the Cordeaux reservoir.*

KEY WORDS *Soil remobilisation; caesium-137; sediment budget; hillslope model; erosion; soil loss rates.*

Introduction

Designing the most appropriate soil conservation and land management strategies requires an understanding of the sources and, where possible, rates of soil loss. Rates of soil loss can be obtained from measurements at the hillslope scale from consolidated soil material or, using a number of offsite approaches, from sediment accumulation in farm dams, and from lakes and reservoirs that receive the sediment. A few monitoring techniques exist to accurately determine the sources of erosion and sediment production, which are important information required to help counter offsite impacts. Furthermore, the usefulness of the technique can be diminished by spatial and temporal sampling problems and resource limitations. While offsite measurements are useful to understand current conditions of sediment delivery to water storages, the measurements are often retrospective and would be too late if appropriate water quality strategies were required to counter the impacts of sediment input. Thus there is a need for approaches that can determine onsite estimates of soil loss.

A number of approaches, such as farm dams (Neil & Fogarty 1991; Mahmoudzadeh 1997; Erskine *et al.* 2003), soil pits and sediment settling trays (Humphreys & Mitchell 1983) and runoff plots (Blong *et al.* 1982) have been employed to investigate soil loss in Australia. Soil loss from hillslopes can be estimated using farm dams, which are created to store water for use on farms. The usefulness of farm dams in determining soil loss at the hillslope scale is based on the concept of calculating total volumes of sediment, which have been washed in from surrounding hillslopes and accumulated as a function of time (Neil & Fogarty 1991; Mahmoudzadeh 1997; Erskine *et al.* 2003). Volume can be determined as a product of the sediment thickness and surface area of the farm dam. This can then be used to express sediment yield as a function of the age of the dam. Farm dams are useful because the date of their creation is usually known and rates of soil loss can be extrapolated for various land uses. However, this approach does not fully explain onsite denudation (Mahmoudzadeh 1997).

The use of soil pits and settling trays has shown varying causes of erosion in the Sydney Basin region in southeast New South Wales (NSW), along hillslopes in catchment areas such as the Cordeaux, Cattai, Oxford Falls and Lidsdale, and Blackheath (Humphreys & Mitchell 1983). Humphreys and Mitchell (1983) evaluated the relative importance of obvious soil turnover mechanisms in operation, in comparison with surface processes of rainwash, soil creep and the burial of clasts of soil. Turnover mechanisms involved estimating the amount of soil mobilised in ant mounds and by upturned trees, whereas processes such as rainwash and soil creep were investigated by examining profiles of soil pits, and by collecting dyed sand in sediment settling trays. It was observed that soil loss as a result of rainwash ranged from 0.04 t/ha/year to almost 7 t/ha/year, whereas the turnover mechanism ranged from 0.03 t/ha/year to almost 45 t/ha/year. Negligible material was generated by soil creep. The estimated net soil losses from slopes caused by different agents of erosion are summarised in Table 1. The values

TABLE 1. Hillslope soil loss and its contributing causes throughout NSW

Agent/cause	*Rate (t/ha/year)*
Cicadas	0.03–0.20
Rainsplash	0.04–0.68 (mean =0.32)
Earthworms	1.33
Fallen trees	0.20–1.34
Slopewash	0.07–6.97 (mean =0.86)
Termites	5.7–8.4
Lyre birds	44.7

Source: Humphreys and Mitchell (1983).

obtained for soil loss due to rainwash processes are in general agreement with values of 2.5–8 t/ha/year recorded using other techniques, such as runoff plots, in the catchment of the Narrabeen Lagoon (Blong *et al.* 1982). Whereas these approaches give insights of onsite erosion, repeated site visits are required before the extent of soil loss can be determined.

The challenges of repeated site visits can be overcome by investigating the movement of caesium-137 (^{137}Cs) within the landscape. ^{137}Cs (half-life = 30 years) is a radionuclide deposited worldwide as a result of atmospheric nuclear weapons testing between 1952 and 1964 (Loughran *et al.* 1989; Ritchie & McHenry 1990).

The use of ^{137}Cs measurements affords a valuable means of assembling spatially distributed information on medium-term (*c.*40 years) rates of soil erosion. The ^{137}Cs approach has been applied successfully in a wide range of environments in many different areas of the world over the past decade (Ritchie & McHenry 1990; Wallbrink & Murray 1996; Walling *et al.* 1996; Ferro *et al.* 1998). These studies indicated that soil loss rates can be assessed with high accuracy on the basis of a single site visit. The approach is based on the key assumption that the local fallout distribution is uniform and that subsequent remobilisation of ^{137}Cs reflects sediment movement.

At undisturbed sites, where neither erosional nor depositional processes have occurred, measurement of ^{137}Cs input is commonly carried out; the original fallout activity may still be determined and is known as the reference value for caesium-137, $^{137}Cs_{ref}$. This reference site is generally along a plateau, for example on a ridge. Where soil erosion occurs, ^{137}Cs will also have been lost, leading to an inventory less than $^{137}Cs_{ref}$. Conversely, where deposition takes place, an increase in ^{137}Cs inventory will be found. Total $^{137}Cs_{ref}$ inventory measurements have been used for calculating soil erosion rates (He & Walling 1997), for determining patterns of soil loss and sediment accumulation (Walling *et al.* 1996) and for testing estimates generated from soil erosion models (Ferro *et al.* 1998; Walling *et al.* 2003b).

Loughran *et al.* (1989) demonstrate that for soils in which net erosion is occurring, the changes in ^{137}Cs content in the soil, relative to the ^{137}Cs content of a stable reference site, are highly correlated with those expected for the observed soil loss. Furthermore, Loughran *et al.* (2004) demonstrate that net deposition can be determined using the reverse mode of this approach. The studies showed that appropriate soil loss/gain, since bomb-testing began until the time of sampling, can be established with only a single site visit. In NSW, a State-wide survey of erosion using ^{137}Cs measurements revealed that in the southern tablelands of the Sydney Basin average annual soil loss caused by grazing ranged from 0.05 to 0.54 t/ha; some areas experienced soil losses up to 5.13 t/ha (Loughran & Elliott 1996). The survey measured ^{137}Cs levels along selected toposequences across a range of soils and land uses, and showed that the ^{137}Cs reference ($^{137}Cs_{ref}$) value was related to average annual rainfall. A reference site is generally along a plateau, and is relatively stable, with no net erosion or deposition.

Other soil loss investigations using ^{137}Cs have been undertaken in recent years (Croke *et al.* 1999; Wallbrink *et al.* 2002; Croke & Nethery 2006); however, these investigations involve estimating soil loss after a known period of land cover change or land-use intensification. The difference in the ^{137}Cs inventory over the two time periods can be used to estimate the change in soil loss. Thus this method has limited application for the current investigation since it requires at least two site visits to establish rates of soil loss.

A major objective of this paper is to determine the spatial variation of soil loss along slopes in a water supply catchment from a single site visit. This can be achieved by assessing the amount of soil remobilised along hillslopes. Previous investigations undertaken in catchments with similar physiography (Humphreys & Mitchell 1983) provide valuable insights about measurements of soil movement on hillslopes in this study area.

Regional setting

The study area, the Cordeaux catchment, is *c.*91.5 km^2 and is located between the Avon and Cataract catchments in the Sydney Metropolitan Special Area (see Figure 1). It is a small headwater catchment geologically dominated by Hawkesbury Sandstone and also includes alluvium, basalts and basinites, the Mittagong Formation and Narrabeen Group, including Stanwell Park Claystone (Sherwin & Holmes 1986). Relief is variable, with a gradual decrease in elevation from southeast to northwest; elevation ranges from more than 500 to 250 m in valley-bottom areas. Soil landscapes within the catchment are based on a combination of their underlying lithology, the processes by which they were formed, and the position occupied on landforms. Soil depth varies from 3 to 4 cm in the northern section of the catchment, and from 6 to 10 cm in the southern parts. There is, therefore, concern about the loss of such skeletal soils to the reservoir downstream.

The catchment experiences a warm temperate climate, but variations in temperature and rainfall are common because of orographic effects from onshore winds and closeness to the coast, as well as variations in elevation. Mean annual rainfall follows a broad decline from 1800 to 1350 mm with decrease in elevation from south to north. The catchment drains to an artificial lake, which has acted as a water storage for supply to Sydney and surrounding townships since 1935 (SCA 2001). Drainage in the Cordeaux catchment is highly branched and contains a large number of first- and second-order streams. Vegetation and land cover (see Table 2) are largely unmodified, dominated by dry sclerophyll woodland on sandstone ridges and slopes, and moister sclerophyll forest along the streams and drainage lines (SCA 2001). A few cleared areas exist where the land was used for agricultural purposes prior to 1954.

The catchment was selected from among three that are part of an ongoing collaborative research effort among several universities and the SCA. Particular focus is on assessing the degree of sheet and rill erosion and identifying locations where soil loss is likely to be serious, so that knowledge of these rates can guide development of policies and strategies to control erosion.

Methodology

Soil coring methods followed those of Loughran *et al.* (1989, 1993) and Loughran & Elliott (1996). Fieldwork was undertaken between 18 April and 6 May 2004, including extensive preliminary reconnaissance to determine suitable sample locations. While it is preferable to have a depth distribution for reference sites to ensure that the entire ^{137}Cs profile has been sampled (Walling *et al.* 2002), field reconnaissance indicated very shallow soils (6–10 cm) underlain by sandstone

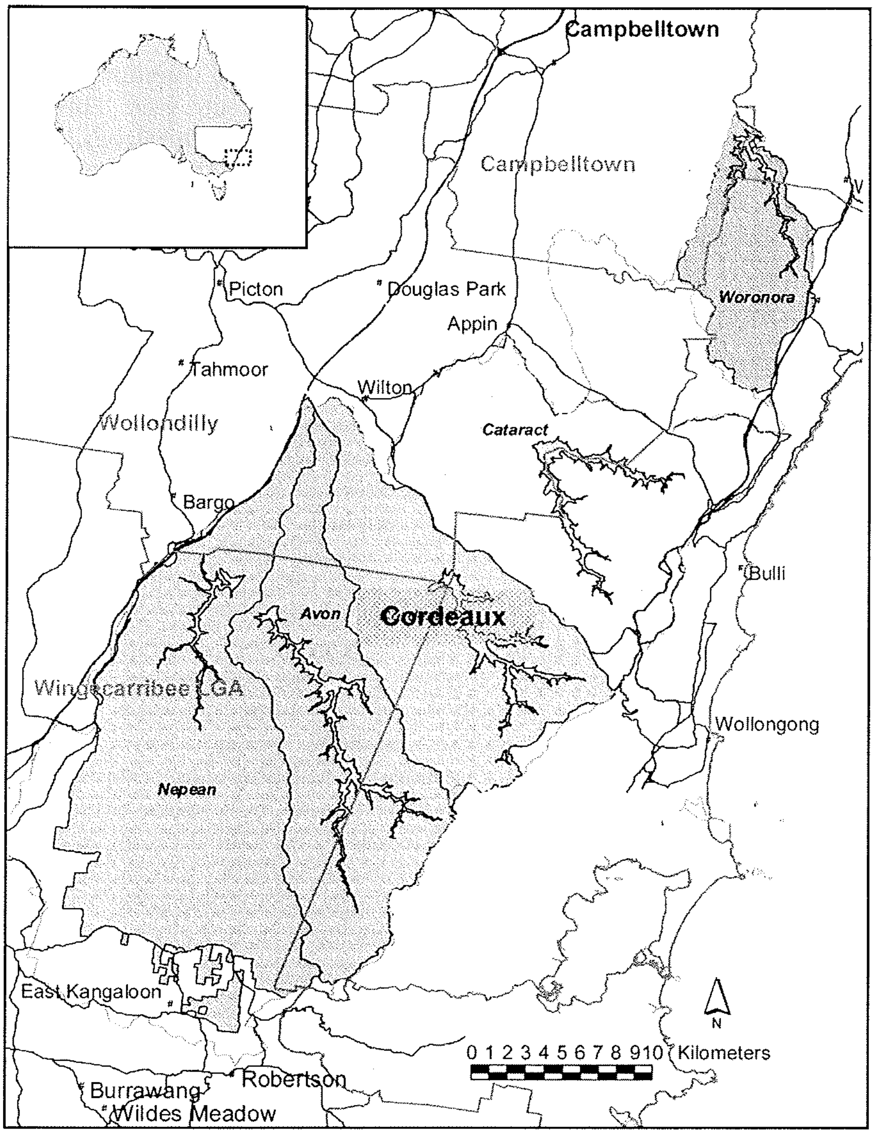

FIGURE 1. Location of the Cordeaux catchment within the Sydney Metropolitan Special Area (after SCA 2001).

bedrock. Thus, soil samples were collected using a 10 cm diameter steel corer, along eight transects, 80 m in length. The average depth of soil samples was 10 cm; replicate samples were taken within a 5 m width at each point. The distance between each transect point was 10 m, with stream widths, which were not sampled, varying from 7 to 10 m.

The upper and lower reaches of a catchment can be investigated since the two areas offer contrasting settings, which can then be compared. Soil cores along transect MB1–MB8 (north to south) were collected in the upper reaches of the Kembla Creek sub-catchment. The dominant lithology at this site is Stanwell Park Claystone and the soil landscape is Lower Mittagong (Sherwin & Holmes 1986).

TABLE 2. ^{137}Cs activity and physical properties for soil samples along slopes in the Cordeaux catchment. The depth of soil at the reference site is an approximation of the 12 bulked samples

Sample	*Soil depth (cm)*	*^{137}Cs activity (Bq/kg)*	*Silt+clay (%)*	*Loss on ignition (%)*	*Bulk density (g/cm^3)*
Reference ($^{137}Cs_{ref}$)	~10	9±1	6.05	32.31	1.55
Kembla Creek					
MB1	10	8±1	45.43	24.63	1.15
MB2	10	10±1	39.5	22.65	1.03
MB3	10	11±1	14.73	16.76	1.34
MB4	9	9±1	14.74	14.82	1.46
MB5	6	1±1	7.05	6.89	1.81
MB6	9	10±1	18.8	13.84	1.56
MB7	9	8v±1	41.84	19.35	1.28
MB8	8	8±1	18.25	7.04	1.66
MB9	10	5±1	42.31	16.02	1.53
MB10	9	4±1	63.44	14.83	1.30
MB11	7	6±1	23.63	17.65	1.29
MB12	6	5±1	50.51	13.16	1.59
MB13	8	1±1	31.15	7.21	1.73
MB14	10	5±1	30.55	15.87	1.30
MB15	10	3±1	44.93	14.47	1.54
MB16	10	8±1	53.66	17.77	1.39
Kentish Creek					
KH1	10	3±1	10.31	4.19	1.87
KH2	10	5±1	7.25	2.49	1.68
KH3	10	6±1	9.19	4.12	2.25
KH4	8	4±1	6.47	4.58	1.00
KH5	9	9±1	19.42	6.71	1.40
KH6	9	6±1	6.53	8.82	1.21
KH7	10	4±1	10.01	3.07	1.57
KH8	10	4±1	14.41	3.28	1.50
KH9	10	4±1	28.7	14.8	1.34
KH10	10	6±1	35.72	16.45	1.12
KH11	9	8±1	22.62	14.5	1.21
KH12	8	5±1	23.78	16.49	1.38
KH13	9	17±2	38.91	28.4	0.82
KH14	9	5±1	27.66	9.6	1.44
KH15	10	6±1	25.9	9.78	1.35
KH16	10	4±1	30.07	8.96	1.23

The vegetation is classified as regenerating and includes shrub (*Acacia mearnsii* and *A. melanoxylon*) and heath (*Banksia ericifolia*, *B. serrata* and *Hakea dactyloides*; Specht & Specht 1999). Transect MB9–MB16 (north to south) was also located on Stanwell Park Claystone within the Lower Mittagong soil landscape (Sherwin & Holmes 1986); however, it was in the lower reaches within Moist Forest vegetation. Kentish Creek was the other sub-catchment sampled and contains a predominantly sandstone lithology. A transect in the upper reaches, KH1–KH8 (north to south), is dominated by Woodland and a Penrose (variant-*a*) soil landscape, whereas transect KH9–KH16 (north to south) in the lower reaches is within Tall Open Forest vegetation (Specht & Specht 1999) on a Warragamba soil landscape (Sherwin & Holmes 1986). The location of the sample sites is shown in Figure 3. Sites were surveyed by levelling, and the position and elevation of every sample location were noted (see Figure 4).

Replicate samples were air-dried, sieved to remove materials that were larger than 2 mm, pooled and weighed to obtain 2–2.5 kg of soil. Samples prepared for gamma spectrometry were dried in an oven at 40°C. An average of 43 g of dried samples was packed with 7 g of sodium carbonate (Na_2CO_3), an inert material, into 65 mm Petri dishes and sealed with silicone sealant. Sample sources were counted according to similar procedures used by Alksnis *et al.* (1999) and Olley *et al.* (2001). The detector system energy was calibrated using a National Institute of Standards and Technology (NIST) traceable $^{154}Eu/^{155}Eu/^{125}Sb$ multi-nuclide standard source and the detector system efficiencies were determined using IAEA reference materials including RGU-1, RGTh-1, RGK-1 and Soil-6. ^{137}Cs activities in the samples were determined by gamma spectrometry, using the 662 keV peak after subtraction of the ^{214}Bi peak (609 keV) interference. Counting was conducted using an ORTEC LO-AX Series High-purity Germanium Low-energy photon spectrometer. Other soil analyses included particle size using a Malvern Mastersizer, carbon content using loss-on-ignition procedures at 550°C (Lewis & McConchie 1994) and dry bulk density by volumetric analysis (Charman & Murphy 2000). Altogether, 33 samples were analysed.

The Loughran regression model

To transform the inventory of ^{137}Cs to rates of soil loss or deposition, a regression equation was used to compute the percentage loss of ^{137}Cs at a sampling location, compared with the nearby reference site. The regression equation, developed for several sites within NSW, is given in equation (1) (Loughran & Elliott 1996).

$$Y = 17.49(1.0821)^x \; [n = 34; \; r = 0.84], \tag{1}$$

where Y is the net soil loss (kg/ha/year, which can be converted later to t/ha/year); and x is the percentage of ^{137}Cs loss/gain relative to a reference value of caesium ($^{137}Cs_{ref}$).

Equation (1) is a revision of the original equation (Elliott *et al.* 1990) presented in earlier investigations, following corrections of the measured sediment yields using data from the Soil Conservation Service of NSW (Loughran & Elliott 1996) and has also been employed in southeast NSW (Elliott *et al.* 1997; Krause *et al.* 2003). To obtain x, the total inventory of ^{137}Cs within the soil samples must be known; this can be determined using equation (2) (Loughran & Elliott 1996):

$$\text{Total } ^{137}\text{Cs inventory} = \frac{^{137}\text{Cs concentration (Bq/kg)} * \text{sample mass (kg)}}{\text{Surface area of sample (m}^2\text{)}}. \tag{2}$$

Net deposition can be determined, in a similar way to Loughran *et al.* (2004), using equation (1) in reverse mode for samples that had more ^{137}Cs than the reference value.

Using this information on soil loss, Loughran *et al.* (1989) produced an idealised six-element ^{137}Cs-sediment hillslope model for Australian conditions. Correlating ^{137}Cs with topographic elements helped to determine the soil loss budget for an entire hillslope. Figure 2 illustrates the position of the idealised elements in relation to the activity of ^{137}Cs. The six elements, A to F, are identified and are associated with various slope facets, where: element A represents an undisturbed, stable site, usually level and on a crest or drainage divide where ^{137}Cs has accumulated from atmospheric fallout, the only loss being due to radioactive decay (^{137}Cs half-life = 30 years), thus providing a 'reference' value for ^{137}Cs. The other elements were assigned based on the ^{137}Cs inventory at the particular sample location. For example, element B is assigned to an eroded site, with minimal run-on and sediment transfer from upslope, whereas element D is assigned because a sample site has localised deposition of soil and ^{137}Cs can occur potentially at any position on the slope where the capacity for sediment transport is reduced (Loughran *et al.* 1989, 1993). All elements are assigned as illustrated in Figure 2.

Net soil loss at each sample point was calculated using equations (1) and (2). The mean net erosion for each transect was calculated from the average of losses, weighted for the sample spacing along the slopes using the ^{137}Cs-sediment hillslope method model developed by Loughran *et al.* (1989).

The Diffusion and Migration (D&M) model

The theoretical Diffusion and Migration (D&M) model for uncultivated sites (Walling *et al.* 2002) was used to determine whether soil loss data for each transect were comparable to global data. This D&M model is one component of the software developed by Walling *et al.* (2002) to determine the rates of erosion and deposition of soil, which, unlike the Loughran regression model, reflects the

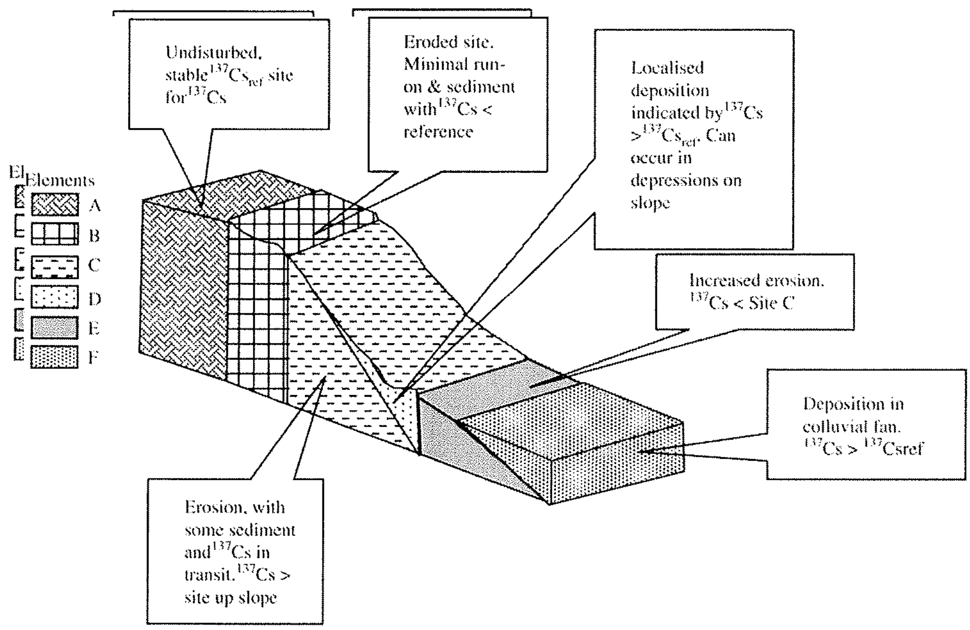

FIGURE 2. Six-element ^{137}Cs-sediment hillslope model (adapted from Loughran *et al.* 1989).

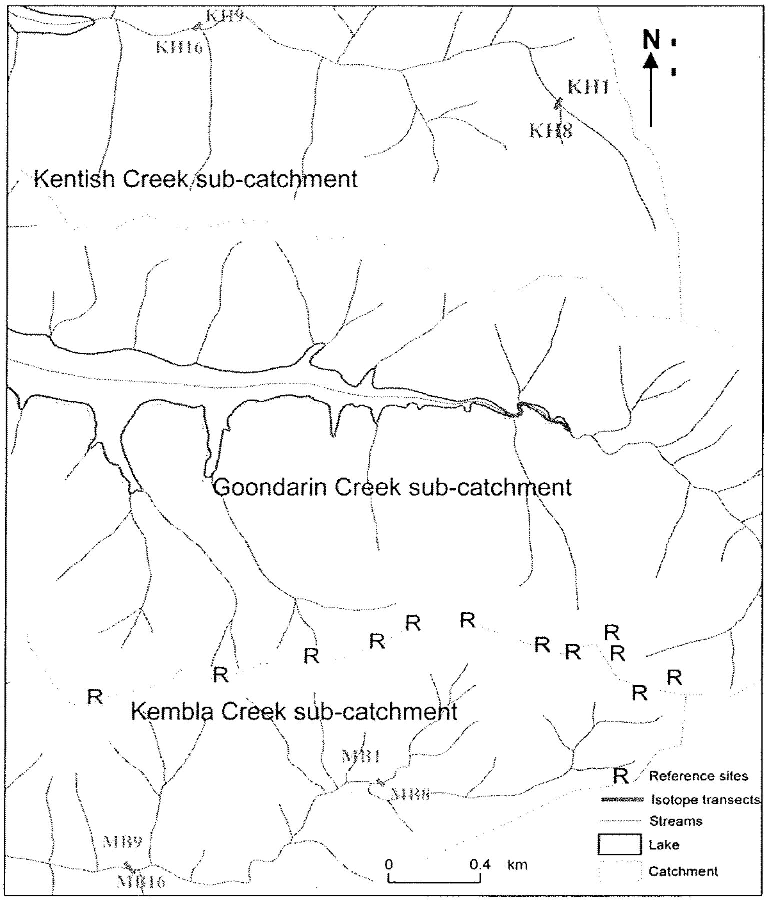

FIGURE 3. Location of soil samples in the Cordeaux catchment.

time-dependent ^{137}Cs fallout input and the progressive redistribution of ^{137}Cs in the soil after deposition from the atmosphere. The erosion rate R may be estimated from the reduction in the ^{137}Cs inventory and the ^{137}Cs concentration in the surface soil using equation (3) (Walling *et al.* 2002):

$$\int_0^t PRC_u(t')e^{-\lambda(t-t')}\mathrm{d}t' = {}^{137}Cs_{\mathrm{diff}}(t), \qquad (3)$$

where P is the particle size correction factor, which accounts for the selectivity of grain size during erosion and sedimentation processes; $C_u(t')$ = the variation of the ^{137}Cs concentration (Bq/kg) in surface soil with time t (years), and can be computed using equation (4); and $^{137}Cs_{\mathrm{diff}}(t)$ is the difference between the ^{137}Cs reference inventory, $^{137}Cs_{\mathrm{ref}}$, and the measured total ^{137}Cs inventory at each sampling point (Walling *et al.* 2002).

$$C_u(t) \approx \frac{I(t)}{H} + \int_0^{t-1} \frac{I(t')e^{-R/H}}{\sqrt{D\pi(t-t')}} e^{-V^2(t-t')/(4D)-\lambda(t-t')} \mathrm{d}t', \tag{4}$$

where D = the diffusion coefficient (kg^2/m^4/year); and V = the downward migration rate of ^{137}Cs in the soil profile (kg/m^2/year).

The diffusion coefficient and the downward migration rate, which are used to characterise the shape of the ^{137}Cs profile with time, can be approximated using equations (5) and (6), respectively (Walling *et al.* 2002):

$$V \approx \frac{W_p}{t - 1963}, \tag{5}$$

$$D \approx \frac{(N_p - W_p)^2}{2(t - 1963)}, \tag{6}$$

where t is the year of sampling (year); W_p is the mass depth of the maximum ^{137}Cs concentration (kg/m^2); and N_p is the distance between the depth of the maximum ^{137}Cs concentration and the point where the ^{137}Cs concentration reduces to $1/e$ of the maximum concentration (kg/m^2).

A soil profile for the reference site was not obtained in the study catchment; however, data from the Nattai sub-catchment (English *et al.* 2005) were used as surrogates for W_p and N_p. This substitution is acceptable for locations that are within the same latitudinal region and which experience the same climatic conditions (Des Walling 2008, pers. comm.). In the Nattai sub-catchment, which is located about 20 km from the Cordeaux sub-catchment, ^{137}Cs concentrations in soils were barely detectable below the 5 cm depth, and the reference inventory was 493 Bq/m^2 (English *et al.* 2005).

The deposition rate, R', can be estimated from the ^{137}Cs concentration of deposited sediment and the excess ^{137}Cs inventory, using equation (7) (Walling *et al.* 2002):

$$R' = \frac{^{137}Cs_{\mathrm{ex}}}{\int_{t_0}^{t} C_d(t')e^{-\lambda(t-t')}\mathrm{d}t'} = \frac{137Cs_{\mathrm{meas}} - {}^{137}Cs_{\mathrm{ref}}}{\int_{t_0}^{t} C_d(t')e^{-\lambda(t-t')}\mathrm{d}t'}, \tag{7}$$

where $^{137}Cs_{ex}(t)$ is the excess ^{137}Cs inventory, which can be computed as the difference between total measured ^{137}Cs inventory, $^{137}Cs_{meas}$ and the inventory at the reference site; and $^{137}Cs_{ref}$; and $C_d(t')$ is the ^{137}Cs concentration of deposited sediment, which can be calculated from equation (8) (Walling *et al.* 2002):

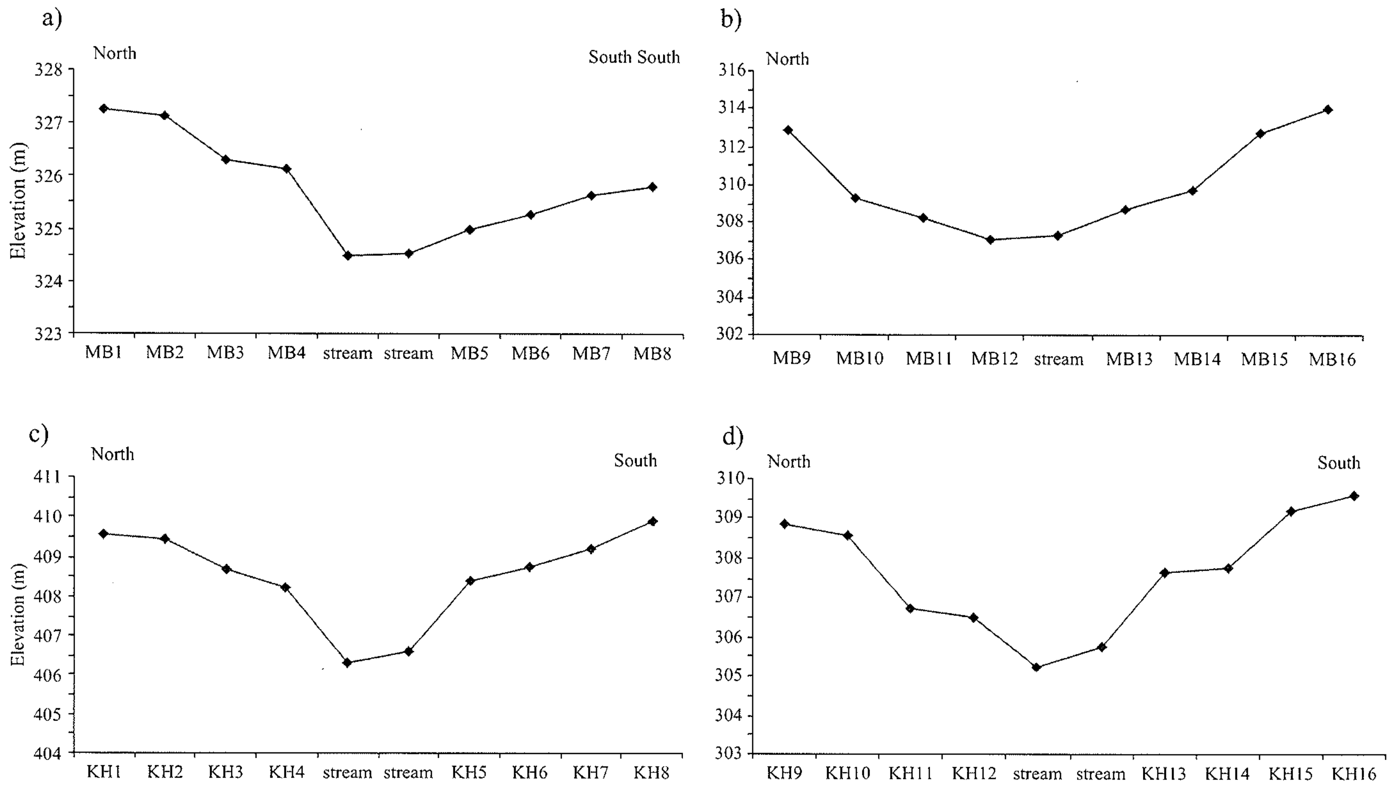

FIGURE 4. Profiles of transects taken in the sub-catchments of Kembla (MB1–MB16) and Kentish (KH1–KH16) Creeks. Sites labelled 'stream' were not sampled.

$$C_d(t') = \frac{1}{\int_S R\mathrm{d}S} \int_S P'PC_u(t')R\mathrm{d}S. \tag{8}$$

Variation of ^{137}Cs with physical properties of catchment soils

The soil samples were analysed for the levels of ^{137}Cs activity and for three physical properties of the soil, namely organic matter content, bulk density and particle fractions (above 63 μm = sand, and below 63 μm = silt + clay), respectively (see Table 2). This was undertaken to establish whether ^{137}Cs displayed an affinity towards different soil particles, because it has been shown that grain size composition of mobilised sediment can be enriched in fines compared with the original soil (Walling *et al.* 1996).

Table 3 summarises the resulting statistical parameters; the analyses were performed at the 0.05 significance level. Soils in the Kentish Creek sub-catchment demonstrated a general tendency to retain ^{137}Cs; all the physical properties of the soil samples were strongly correlated to the radionuclide, with r ranging from 0.455 to 0.723 (p-value < 0.05). This can be related to the affinity of this radionuclide for smaller particle sizes and higher organic content. As seen in Table 2, the proportion of these two soil characteristics in the Kentish Creek sub-catchment is relatively low. In the Kembla Creek sub-catchment the silt + clay fractions did not appear to exhibit a preference for ^{137}Cs ($r = 0.143$, p-value $= 0.299$). This is expected since the silt + clay fractions in soil samples from the Kembla Creek are generally high. Thus there are more chances for ^{137}Cs to bind to the ubiquitous small claystone particles of Kembla Creek than with soils in the Kentish Creek sub-catchment, which is predominantly sandstone and contains fewer small particles. However, ^{137}Cs is present in such small quantities that particle-size effects on adsorption are rarely seen in soils from Australia (Wallbrink & Murray 1996) and the southern hemisphere (Collins *et al.* 2001; Walling *et al.* 2003a).

TABLE 3. Correlation coefficients for the variation of physical properties with ^{137}Cs for soils within the sub-catchments of Kembla Creek ($n = 16$) and Kentish Creek ($n = 16$)

	r	p-*value*
Kembla Creek, ^{137}Cs (Bq/kg)		
Physical properties		
LOI (%)	0.558	0.012
Bulk density (g/cm^3)	−0.560	0.012
Silt + clay (%)	−0.143	0.299
Kentish Creek, ^{137}Cs (Bq/kg)		
Physical properties		
LOI (%)	0.723	0.001
Bulk density (g/cm^3)	−0.455	0.038
Silt + clay (%)	0.485	0.028

TABLE 4. Total ^{137}Cs inventories and converted soil loss for points along slopes consisting of four transects: MB1–MB8, MB9–MB16, KH1–KH8 and KH9–KH16, in the Cordeaux catchment

Sample[a]	*^{137}Cs total activity (Bq/m^2)*	*Hillslope element*	*Loss/gain (t/ha/year)*[b]	*Loss/gain (t/ha/year)*[c]
Reference ($^{137}Cs_{ref}$)	465.76±51.75	–	–	–
Kembla Creek				
MB1	405.84±50.73	A/B	−0.05±0.020	−0.90±0.45
MB2	511.21±51.12	D	0.04±0.019	0.30±0.40
MB3	573.86±52.17	D	0.11±0.016	2.82±0.45
MB4	520.48±57.83	F	0.04±0.007	1.29±0.49
MB5	71.50±71.50	E	−13.91±0.001	−7.70±0.80
MB6	559.88±55.99	D	0.09±0.009	3.15±0.56
MB7	443.63±55.45	B	−0.03±0.010	−0.40±0.12
MB8	486.83±87.90	A	0.02±0.001	1.18±0.56
MB9	294.65±85.12	B	−0.32±0.001	−2.70±0.55
MB10	221.57±80.01	B/C	−1.09±0.001	−4.40±0.50
MB11	355.62±59.27	C	−0.11±0.006	−2.30±0.50
MB12	285.76±57.15	E	−0.37±0.008	−2.60±0.60
MB13	62.35±62.35	E	−16.25±0.003	−7.70±0.75
MB14	305.20±61.04	C	−0.27±0.004	−3.30±0.50
MB15	173.69±57.90	B	−2.46±0.007	−5.20±0.65
MB16	481.56±60.19	A	0.02±0.005	0.99±0.09
Kentish Creek				
KH1	200.20±66.73	B	−1.57±0.002	−4.60±0.70
KH2	332.27±66.45	C	−0.17±0.002	−2.40±0.60
KH3	410.54±68.42	D	−0.04±0.001	−0.40±0.30
KH4	281.89±70.47	E	−0.39±0.001	−5.10±0.45
KH5	631.82±70.20	F	0.29±0.001	1.12±0.51
KH6	352.38±58.73	C	−0.12±0.006	−2.50±0.45
KH7	252.61±63.15	B	−0.65±0.003	−3.80±0.60
KH8	296.31±74.08	B	−0.31±0.001	−4.00±0.55
KH9	233.15±58.29	B	−0.90±0.006	−4.40±0.45
KH10	333.41±55.57	C	−0.16±0.010	−2.90±0.50
KH11	402.25±50.28	D	−0.05±0.022	−0.70±0.23
KH12	283.72±56.74	E	−0.38±0.008	−3.20±0.55
KH13	703.09±82.72	F	0.97±0.000	4.66±0.39
KH14	311.19±62.24	E	−0.24±0.004	−3.00±0.50
KH15	351.79±58.63	C	−0.12±0.006	−2.10±0.50
KH16	244.17±61.04	B	−0.75±0.004	−4.50±0.50

Notes: [a] Sample locations run from north to south starting from 1 to 16 (see Figure 4). Soil loss/gain (t/ha/year) was calculated with 95 per cent confidence limits of using the [b]Loughran regression equation (Loughran & Elliott 1996) and the [c]D&M model (Walling *et al.* 2002). Negative numbers represent soil loss, whereas positive numbers represent gain.

Qualitative rating of landscape elements based on ^{137}Cs remobilisation along local slopes

Using the six-element ^{137}Cs-sediment hillslope model developed by Loughran *et al.* (1989) landscape elements along four transects were identified. These elements were identified by comparing the total inventories of ^{137}Cs (Bq/m^2) within soil samples as a function of the area sampled (column 2 of Table 4). Characteristic

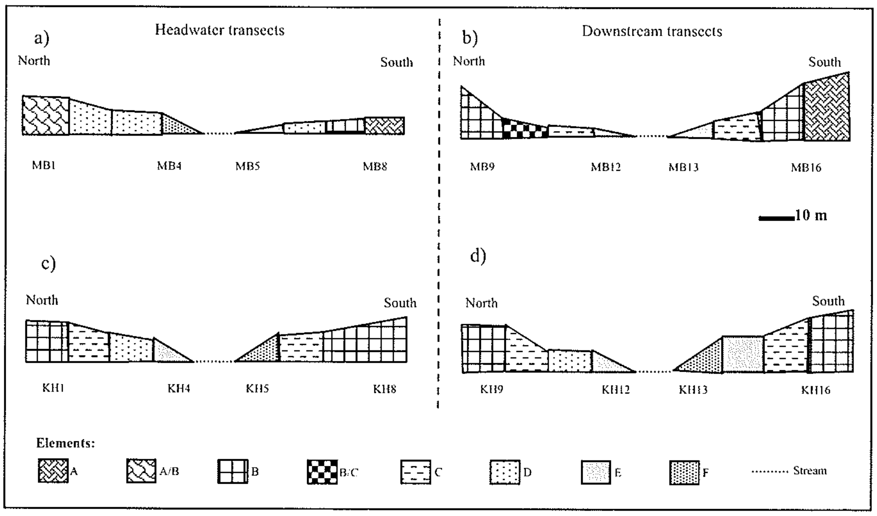

FIGURE 5. ^{137}Cs-sediment hillslope elements along transects in the sub-catchments of Kembla (MB1–MB16) and Kentish (KH1–KH16) Creeks. Slope facets are assigned corresponding elements based on the ^{137}Cs-sediment hillslope model developed by Loughran *et al.* (1989).

features of the hillslope elements (see Figure 2) were described for each site in the Cordeaux catchment. This qualitative assignment of landscape elements can be important in assessing and prioritising land degradation caused by erosion, and the representative slope diagrams function as visual aids.

All six ^{137}Cs-sediment elements (elements A–F) were recognised in this investigation, although a complete sequence was not observed on any one slope. However, the absence of a complete sequence of elements is not unique, as shown in eastern Australia (Loughran *et al.* 1993). Figure 5(a)–(d) display elements identified along the slope long-profiles for the corresponding sampling sites. Figure 5(a) and (c) are from the upper reaches of their respective catchments and Figure 5(b) and (d) are from the lower reaches. The valley floor system (shown as 'stream' in the figure) is not represented in the ^{137}Cs-sediment hillslope model and was not sampled. A number of elements were identified.

Sites MB8 and MB16 were on drainage divides and were considered stable with little erosion taking place. However, while both sites contained total ^{137}Cs inventories that were slightly higher than that of the $^{137}Cs_{ref}$ value, they were within the margin of error (see Table 4), and were considered to be equivalent to element A. Although site MB1 was also at the crest of a hillslope, it was categorised as element A/B because its total ^{137}Cs inventories were slightly lower than that of the $^{137}Cs_{ref}$ value and did not fall within the margin of error (see Table 4). Total ^{137}Cs inventories indicated that sites that fit element B include MB7, MB9, MB15, KH1, KH7, KH8, KH9 and KH16. At these sites ^{137}Cs inventories were noticeably less than the $^{137}Cs_{ref}$ value and thus represent eroded sites. Another mixed element, B/C, was identified at site MB10, because this site exhibited total ^{137}Cs inventory that was less than MB9 (element B) but was still higher than that at MB11.

Six sites (MB11, MB14, KH2, KH6, KH10 and KH15) were recognised as element C, which is an element of net soil and ^{137}Cs loss. However, ^{137}Cs levels in

TABLE 5. Estimated sediment yield to streams as a function of the proportion of slope along each transect. MB1–MB8 and KH1–KH8 are transects taken in the upper reaches of sub-catchments in the Kembla and Kentish Creeks, respectively. MB9–MB16 and KH9–KH16 are transects taken in the lower reaches of the sub-catchments

Location in sub-catchment[a]	*Slope system*	*Gross soil erosion*[b] *(t/ha/year)*	*Net soil loss or gain*[b] *(t/ha/year)*	*Gross soil erosion*[c] *(t/ha/year)*	*Net soil loss or gain*[c] *(t/ha/year)*
Kembla Creek—upper	MB1–MB4	−0.018			
	MB5–MB8	1.729	1.71 (SDR = 90%)	−0.138	0 (SDR = 3%)
Kembla Creek—lower	MB9–MB12	0.237			
	MB13–MB16	2.369	2.61 (SDR = 110%)	−0.438	−0.438 (SDR = 100%)
Kentish Creek—upper	KH1–KH4	0.272			
	KH5–KH8	0.098	0.37 (SDR = 177%)	−0.350	−0.325 (SDR = 93%)
Kentish Creek—lower	KH9–KH12	0.187			
	KH13–KH16	0.017	0.20 (SDR = 976%)	−0.325	−0.250 (SDR = 78%)

Notes: [a] Sample locations run from north to south starting from 1 to 16 (see Figure 4). [b] Using the regression model (Loughran & Elliott 1996). [c] Using the D&M model (Walling *et al.* 2002). Negative numbers represent soil loss, whereas positive numbers represent gain. SDR = sediment delivery ratio.

this element were higher than in the element upslope because sediment with adsorbed ^{137}Cs has temporarily accumulated in transit from element B. Sites MB2 and MB3 contained higher ^{137}Cs inventories than the $^{137}Cs_{ref}$ value and could represent localised deposition of soil and ^{137}Cs; thus they were identified as element D. MB6 had ^{137}Cs levels higher than the $^{137}Cs_{ref}$ value and was also identified as element D. Sites KH3 and KH11 had elevated ^{137}Cs inventories in comparison to nearby sites, but their ^{137}Cs inventories were not higher than the $^{137}Cs_{ref}$ value; nonetheless, since they seem to be at depressions (Figure 5(c) and (d), respectively) and were also categorised as element D. Element E was represented at six sites (MB5, MB12, MB13, KH4, KH12, and KH14). This element is often found on lower positions down slopes and can be distinguished from element C because its level of ^{137}Cs is noticeably less and it occurs on convex slopes. Element F was present at three sites: MB1, KH5, and KH13, and existed at the slope bases. They contained higher ^{137}Cs levels than the $^{137}Cs_{ref}$ value. Assigning it to element D would also be a reasonable option since element D is not constrained to any particular position along a slope.

Quantifying the rates of losses and gains in sub-catchment soils

There was high variability of ^{137}Cs along the four transects. Soil losses/gains for each point along the slopes in both sub-catchments are given in Table 4. In the Kembla Creek catchment the hillslope point-estimates of soil loss were greatest at slope bases using both the Loughran regression and D&M models. For instance, in the upper reach of the sub-catchment a minimum of 0.02 ($\pm$0.01) t/ha/year (Loughran regression model) or 1.18 ($\pm$0.56) t/ha/year (D&M model) was mobilised, while a maximum of 13.91 ($\pm$0.001) t/ha/year (Loughran regression model) or 7.70 ($\pm$0.80) t/ha/year (D&M model) was detached close to the stream (MB8 and MB5, respectively). Two anomalous transformations of ^{137}Cs to soil movement were detected: while the D&M model generally produced estimates that were higher than those generated by the Loughran model, the results obtained for MB5 and MB13 using the D&M model were actually lower than those generated by the Loughran regression model. In the lower reach of the Kembla sub-catchment the minimum soil mobilised was 0.02 ($\pm$0.005) t/ha/year (Loughran regression model) or 0.99 ($\pm$0.09) t/ha/year (D&M model), while the maximum was more than 16 ($\pm$0.003) t/ha/year (Loughran regression model) or 7.70 ($\pm$ 0.75) t/ha/year (D&M model; MB16 and MB13, respectively). In both the upper and lower reaches the minimum values corresponded to gains (deposition), relative to the $^{137}Cs_{ref}$ site, and the maximum values corresponded to losses (erosion).

In the Kentish Creek sub-catchment the maximum amount of soil moved corresponded to an overall loss of 1.57 ($\pm$0.002) t/ha/year (Loughran regression model) or 4.60 ($\pm$0.70) t/ha/year (D&M model) in the upper reaches, while a minimum of 0.04 ($\pm$0.002) t/ha/year (Loughran regression model) or 0.40 ($\pm$ 0.30) t/ha/year (D&M model) was also lost in the upper reaches (KH1 and KH3, respectively).

Sediment budget calculation as a function of the area of slope elements

A budget for each local slope was determined and involved calculating net soil loss from the inventory of ^{137}Cs relative to the percentage representation of each element observed along transects. Each value of net soil loss/gain seen in Table 4 was weighted for the proportion of the 80 m transect represented by the sample. The proportion used was 12.5 per cent since sampling was undertaken equally at 10 m intervals. Weighted values were then summed to determine net soil loss for each hillslope. Adjustments were also made for sediment gains occurring at some sites and an overall sediment yield was derived for each transect. Table 5 shows sediment yields for each slope. These values represent averaged sediment movement from 1954 (first introduction of ^{137}Cs) to the time of sampling in 2004.

The net movements of soil along the slopes were expressed differently by the two models. The Loughran regression model implied that net deposition occurred on the slopes, whereas the D&M model indicated net soil loss (see Table 5). It was estimated that slopes in the upper reach of the Kembla Creek sub-catchment experienced four times more deposition than did slopes in the Kentish Creek sub-catchment using the Loughran regression model and an order of magnitude more deposition on slopes in the lower reach of the Kembla Creek sub-catchment than in the Kentish Creek sub-catchment. However, using the D&M model there was no significant net soil loss or deposition in the upper reach of the Kembla Creek sub-catchment, while net soil loss occurred on the slopes in the lower reach of the Kembla Creek sub-catchment, and this was 1.8 times that occurring on slopes in the lower reach of the Kentish Creek sub-catchment.

Discussion and conclusions

Using the six-element ^{137}Cs-sediment hillslope model all the elements were identified along the hillslopes investigated. These results indicate that soil materials that have been eroded are actively being redistributed in the environment. This active redistribution of eroded material in the environment is similar to the findings of Loughran *et al.* (1992, 1993), Loughran and Elliott (1996) and Shakesby *et al.* (2007).

Loughran *et al.* (1992, 1993) employed the six-element ^{137}Cs-sediment hillslope model and determined that landscapes in Australian catchments experience high movement of soil even if it were not detected at catchment outlets. In the 1993 investigation three catchments were assessed in relation to the soil loss redistributed and removed from hillslopes. Conditions in the catchments include open woodland with vineyards in South Creek near Lake Macquarie, NSW, lands under banana cultivation near Coffs Harbour, NSW, and farmland in Colbinabbin, northern Victoria. Net soil loss was estimated to be 0.019t/ha/year at South Creek, zero at Coffs Harbour and 0.4 t/ha/year at Colbinabbin. While these estimates were not tested, historical evidence suggested that they were within acceptable magnitudes. In a similar way, the hillslope model was used to estimate soil loss in the Maluna Creek catchment and this was compared with extrapolation of net sediment yield from 13 years of suspended sediment data (Loughran *et al.* 1992). Both estimates were within 10 per cent of each other and it was proposed that the hillslope model was an appropriate indicator of soil redistribution within catchments. Loughran and Elliott (1996) demonstrate that soil loss rates for the undisturbed conditions ranged from 0.5 to 8 t/ha/year. Although rates of soil loss were relatively high (in comparison to a tolerable soil loss value of 0.5 t/ha/year) there was no significant relationship with estimated sediment yield, thus indicating that high redistribution within the environment does not necessarily equate to the total sediment delivered.

In another study, soil loss was investigated in southeastern NSW, in the Nattai River catchment, which has similar characteristics to those of the Cordeaux catchment (Shakesby *et al.* 2007). In this instance soil erosion bridges were used to trap eroded material along slopes. The investigation demonstrated that even after extreme events, such as with post-fire rainfall, eroded soil does not always end up in streams or their outlets.

In the current investigation it was demonstrated that soil remobilisation is low in the predominantly unmodified sub-catchments and moderate in disturbed areas. Comparison of measurements of ^{137}Cs inventory at selected sites along toposequences of a hillslope to activity at a reference site allowed each site to be assigned one of six elements, which indicated the erosion status at the sampled site. The findings are significant because the rates are established for unmodified catchments, and thus the values can be used as benchmarks against which rates in other catchments can be compared. In this way catchments can be categorised relative to the level of erosion being experienced and targets set for monitoring and control. Furthermore, if land-use changes are suggested for the study catchments the measured rates can be used to check the actual impact of such changes. Thus the six-element ^{137}Cs-sediment hillslope model is useful for identifying relationships between slope facets in the study catchment and the concentrations of ^{137}Cs in order to detect the patterns of remobilised sediments in the catchment.

It was demonstrated that the ^{137}Cs inventory at a site relative to a reference site can indicate either erosion or deposition, and both processes were detected with comparable accuracy by the Loughran regression equation and D&M model for the point estimates. However, the estimates using the D&M model were generally higher, which might have resulted because this model was calibrated for use in soils with ^{137}Cs being detected to at least 15 cm deep (Des Walling, pers. comm. 2008), whereas soils in the catchment were up to 10 cm deep. Hence, the high diffusion coefficient appears to generate relatively higher estimates of sediment yield. Similar outcomes have been detected in shallow soils (16 cm depth) in Turkey (Haciyakupoglu *et al.* 2005) and 12 cm (Simms *et al.* 2006). It would appear that the D&M model could be more useful for transforming ^{137}Cs inventory to estimate soil loss or gain since the soil loss estimates obtained were comparable to those obtained in the same catchment by Humphreys and Mitchell (1983). This might be related to the fact that the Loughran regression model does not incorporate the temporal variation of ^{137}Cs fallout since the mid-1950s, which is likely to cause underestimation of soil loss.

The results presented in this investigation confirm the usefulness of ^{137}Cs measurements in categorising slope segments based on whether they experience net soil loss or deposition in the Cordeaux catchment. This work can provide reliable assessment of soil erosion status and hence provide a basis for developing a catchment management strategy to target land degradation and promote sustainable use of the land surrounding the reservoir. Other catchments, where water quality is of great importance, can benefit by investigating the extent to which soil particles are being remobilised before they are delivered to water bodies.

In conclusion, the findings of this research indicate that the application of the ^{137}Cs-sediment hillslope model can be used to assess the status of soil loss or deposition over medium-term time periods using a single site visit to catchments within the Sydney Basin region. However, there is a potential constraint with the application of ^{137}Cs to investigate soil loss. Because ^{137}Cs inventories currently approach the limit for viable application of the technique (Walling *et al.* 2003a), it is likely that, in the future, the use of ^{137}Cs measurements to estimate rates of soil redistribution may become impracticable. Thus other means of investigating soil loss at the decadal temporal scale need to be explored. Another limitation associated with this research included the use of surrogate data from a soil profile from a location in the same latitudinal region with similar climatological conditions, rather than from sites in the study catchment. Notwithstanding this possible limitation, the total inventory from both catchments compared well with those reported in other studies, and which suggested inventories of 420 Bq/m^2 (He & Walling 1997) and 400–500 Bq/m^2 (Krause *et al.* 2003) for southeast Australia.

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