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A tracer budget quantifying soil redistribution on hillslopes after forest harvesting

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Abstract

Managing the impacts of erosion after forest harvesting requires knowledge of erosion sources; rates of sediment transport and storage; as well as losses from the system. We construct a tracer-based (^{137}Cs) sediment budget to quantify these parameters. The budget shows significant redistribution, storage and transport of sediment between landscape elements and identifies the snig tracks and log landings as the major impact sites in the catchment. Annual sediment losses from them were estimated to be 25 ± 11 and 101 ± 15 t ha^{-1} year^{-1}, respectively, however, it is probable that most of this is due to mechanical displacement of soil at the time of harvesting. The budget showed greatest net transport of material occurring from snig tracks; representing some 11 ± 4% of the ^{137}Cs budget. Of the latter amount, 18%, 28% and 43% was accounted for within the cross banks, filter strip and General Harvest Area (GHA), respectively. The ^{137}Cs budget also showed the GHA to be a significant sediment trap. The filter strip played a fundamental role in the trapping of material generated from the snig tracks, the mass delivery to them from this source was calculated to be 1.7 ± 0.6 kg m^2 year^{-1}. Careful management of these remains critical. Overall we could account for 97 ± 10% of ^{137}Cs. This retention suggests that (within errors) the overall runoff management system of dispersing flow (and sediment) from the highly compacted snig tracks, by cross banks, into the less compacted (and larger area) GHA and filter strips has effectively retained surface soil and sediment mobilised as a result of harvesting at this site. Crown Copyright © 2002 Published by Elsevier Science B.V. All rights reserved.

Keywords: Sediment budget; ^{137}Cs; Tracers; Soil redistribution; Forest harvesting; Snig track erosion; Filter strips

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1. Introduction

1.1. Quantifying sediment redistribution after harvesting

The impact of harvesting operations on sediment delivery has generally been assessed by measurement of the suspended solids flux at the stream or catchment outlet (Tebo, 1955; Gilmour, 1971; Olive and Reiger, 1985; Anderson and Potts, 1987; Borg et al., 1987; Cornish and Binns, 1987). However, these measurements reveal little about the source of the eroded material, or about the specific impact of harvesting on sediment redistribution upstream of that point (Croke et al., 1999a). Sediment budgets can be used to construct a more complete picture of sediment generation and distribution within catchments (Dietrich and Dunne, 1978; Kelsey et al., 1981; Reid et al., 1981; Trimble, 1983; Roberts and Church, 1986). However, these budgets require quantification of the inputs, outputs and storages associated with the study area. To address this, a range of studies has been undertaken to quantify rates of material transport from, or deposition within, various landscape elements. For example, rates of loss from roads (e.g., Reid and Dunne, 1984; Fahey and Coker, 1989; Montgommery, 1994) snig tracks, (Croke et al., 1999a) snig tracks and cross banks (Hairsine et al., 2001) as well as slope erosion rates and channel storage (Best et al., 1995; Madej, 1995).

In spite of these studies, data gaps invariably exist for some of these terms, while others simply remain difficult to quantify. For example, very little is known about sediment redistribution at the hillslope scale in forestry environments, particularly in terms of the relative transfer and storage of material that occurs between erosion units such as haulage tracks and storage sites such as the cross banks and filter strips. The sediment budget remains incomplete where there is a lack of detail about these processes.

In this paper, we use the redistribution of fallout ^{137}Cs as the basis of a sediment budget. We use our budget to examine the issue of onslope sediment redistribution processes; in particular to quantify the transfer and storage of material that occurs within and between different elements of harvested slopes. The budget can also be used to quantify the losses of sediment from the catchment after harvesting, and thus assess the overall effectiveness of erosion mitigation practices at the study area.

1.2. Using tracer budgets

We use fallout ^{137}Cs as the basis of our work. Its utility in soil erosion studies was first investigated by Rogowski and Tamura (1965), who demonstrated a linear relationship between soil loss and ^{137}Cs depletion. This basic relationship has been used by many subsequent authors to characterise geomorphic processes (see references in review by Ritchie and McHenry, 1990). In particular, sediment budgets based on tracers such as ^{137}Cs can also provide a powerful framework for soil redistribution studies (Ritchie et al., 1974; Walling et al., 1986, 1996; Loughran et al., 1992; Quine et al., 1994; Owens et al., 1997).

In its simplest form, constructing a tracer-based budget requires the measurement of three terms: (1) the initial inventories, that is the amount of the tracer present in the soil prior to disturbance; (2) the spatial extent of the landscape elements of interest; and (3) the amount of tracer left in each landscape element after disturbance. The product of the initial tracer inventories and surface area gives the total amount of each tracer initially contained within each element. Comparison of the tracer amounts remaining in the hillslope elements with that calculated to be present prior to disturbance reveals net losses from the system. Differences in radionuclide amounts between the landscape elements (both before and after harvesting) reflect the relative flow of material between elements; revealing much about the transfer and storage occurring on hillslopes after harvesting. In the study presented here, the landscape elements are defined as the log landing areas (otherwise known as log dumps) snig tracks, (otherwise known as skid tracks or skid trails), the General forest Harvesting Area (hereafter termed the GHA), and filter strip areas retained for wildlife and riparian protection. In total they account for the entire surface area of the study catchment.

There are three factors that may complicate the application (or interpretation) of ^{137}Cs to tracer-based sediment budget studies. The first is the initial spatial variability (coefficient of variation, CV) of ^{137}Cs in soils, which in reference areas may be as large as 40% (Fredericks et al., 1988; Sutherland, 1994; Wallbrink et al., 1994). The second is the affinity of ^{137}Cs to particles of different sizes (Wallbrink et al., 1999). The third is accounting for natural erosion that has occurred in the period between the initial fallout of ^{137}Cs and the beginning of the disturbance regime. These factors are addressed specifically within the 'experimental design'. The radionuclide used as the basis for this study is described below.

1.3. Fallout caesium-137

Caesium is an alkali metal with a chemistry similar to that of sodium, potassium and other elements of Group I in the Periodic Table. Caesium-137 has a half life of $\sim$ 30 years and is found in the environment as a consequence of above ground nuclear weapons testing between 1945 and 1974 (Wise, 1980; Longmore, 1982; Ritchie and McHenry, 1990). Fallout in the northern hemisphere was about an order of magnitude higher than in the southern hemisphere (Ritchie and McHenry, 1990). Fallout of this nuclide was strongly related to local patterns and rates of precipitation (Davis, 1963; Longmore, 1982; Basher et al., 1995). Upon reaching the soil surface, ^{137}Cs not only complexes strongly with the soil particles, but appears to occur in an almost non-exchangeable form. Lomenick and Tamura (1965) reported that they removed less than 1% of ^{137}Cs from sediment by a variety of salt, base, oxidizing, weak acid, and water solutions. In undisturbed soils ^{137}Cs is generally found with a maximum concentration slightly below the soil surface ($\sim$ 10 mm), from where concentrations decrease exponentially to below detection limits at $\sim$ 200- to 250-mm depth (Wallbrink and Murray, 1993; Zhang et al., 1994; Basher et al., 1995; Wallbrink et al., 1999).

2. Study site

2.1. Physical characteristics

The study area is a small basin ($\sim$ 12 ha) within compartment 1708 in the Bondi State Forest, approximately 24 km south of Bombala in the south east of NSW, Australia (Fig. 1).

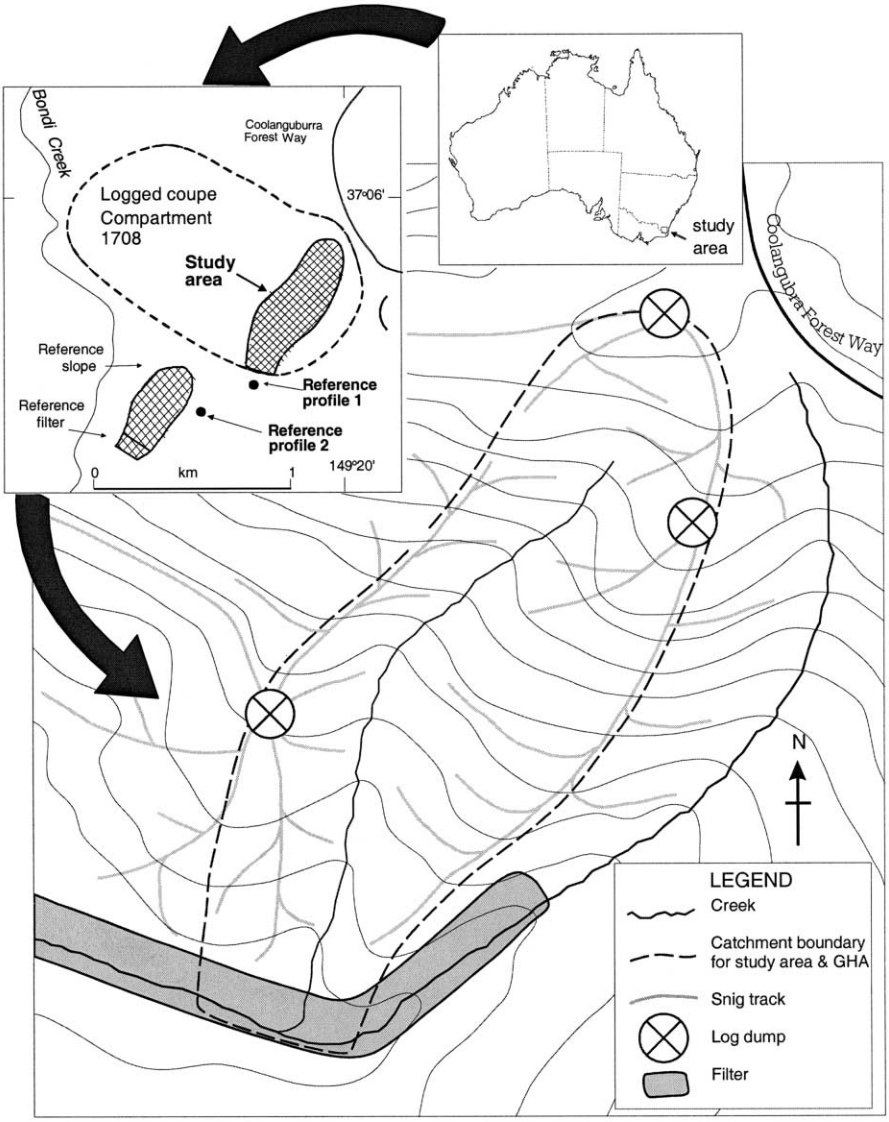

Fig. 1. Location map and diagram of study area within compartment 1708, situated in Bondi State Forest, NSW.

The site is underlain by granite–ademellite (Beams, 1980). The typical soil is a light yellow-coloured, uniformly coarse-textured and weakly structured Orthic Tenosol, (Isbell, 1996) or a Uc 5.22, Northcote (1979). The particle size composition of these soils is 67% sand (< 63 μm), 16% silt (63–40 μm) and 17% clay (< 40 μm) by weight. The study site is 650 m above sea level and has a mean annual rainfall of ~ 1000 mm. Rainfall is variable and episodic. Lowest rainfall occurs from midwinter to early spring; the highest rainfall from midsummer to early winter (State Forests of NSW, 1994). The dominant tree species at the study site are Silvertop Ash (*Eucalyptus sieberi*), Grey Gum (*E. cypellocarpa*) and Brown Barrel (*E. fastigata*). Regrowth vegetation consists mainly of smaller eucalypts, acacias and Musk Daisy Bush (*Olearia argophylla*) (Croke et al., 1997). A well-incised ephemeral stream, ~ 300 m long, is the major drainage feature of the study area. It has vertical banks of about 1 m at its headwaters, and about four m at the outlet.

2.2. Logging treatment for compartment 1708

Timber extraction occurred in compartment 1708 from May to October 1990. The trees were cut by chainsaw, the branches and crowns were left in situ. The logs were dragged (or snigged) by bulldozers along snig tracks to log landings where they were stripped of bark, and loaded onto log trucks for transport out of the coupe via the compartment road (Fig. 1). The compartment, and our study area within it, was harvested under the Standard Erosion Mitigation Guidelines (CaLM, 1993). This prescribed, among other conditions, that drainage lines in catchments greater than 40 ha are protected by a 20-m wide filter strip of undisturbed vegetation. Trees were harvested from this zone, but only if they could be felled and snigged without machinery entering the zone.

After harvesting, the in-situ crowns, leaves, and slash were burnt to reduce the residual fuel load available for wildfire. The aim is to reduce the fine fuel available for fires by ~ 80% (State Forests of NSW, 1994). In Australia, burning also facilitates regeneration of trees by releasing nutrients and initiating seed release. The log landings usually become compacted during harvesting. In order to promote regeneration of these areas, the log landings are treated by first blading off the surface soil and placing this in large mounds adjacent to the landing area. Some of this material is then spread back onto the landing after harvesting. The surface is then ripped and the log landings commonly replanted with *E. nitens*.

2.3. Erosion mitigation strategies

Treatments to reduce post-harvest sediment loss principally involved the construction of cross banks on the snig tracks and the provision of the filter strip along the major drainage line. The cross banks were typically spaced between 20 and 30 m, depending on the slope. The control on the spacing is to reduce contributing area for runoff generation, thus reducing water velocity and discharge volume, (Croke et al., 1999b). The cross banks are designed to divert water and sediment from the snig tracks onto the GHA. The cross banks are constructed by blading material off the snig track (with a bulldozer) to form a mound of material at right angles to the predominant direction of water flow on the track. In this paper, the term ‘cross banks’ also includes the mounds of material, sometimes called ‘windrows’ that are created alongside the snig tracks during their construction.

3. Experimental design and research methods

3.1. Requirements for constructing a tracer-based budget

As described in the Introduction, the approach is based on determining the total amount of ^{137}Cs within the study area, and also within each of its individual landscape elements, both before and after harvesting. Thus, in order to construct a tracer budget we need to determine: (a) the surface areas of each landscape element, (b) the activity (Bq kg^{-1}) of ^{137}Cs within each of these elements, (c) a reference inventory for ^{137}Cs from an undisturbed coupe, with similar soils, aspect, and rainfall to the logged study area. Because natural 'background' erosion has occurred within the study area over the last ~ 30 years (the period since the main ^{137}Cs fallout input), we also need to determine: (d) the amount of natural ^{137}Cs (and thus soil) redistribution which is not due to forest harvesting. Finally, in order to better interpret the ^{137}Cs tracer budget we also investigate: (e) the dependence of ^{137}Cs activity on particle size and organic matter content.

3.2. Calculating the surface area of the different landscape elements and cross bank volumes

The areas of the different landscape elements within the study area were determined by digitising a 1:25,000 aerial photo using ARC/INFO GIS. Coverages were created for the snig tracks, log landings, and the buffer strip. The area of the GHA was obtained by subtracting the sum of the areas of the log landings, snig tracks and filter strip from the total study area. Digitising aerial photos creates errors due to optical distortion and relief displacement (Avery, 1968; Curran, 1985). Image stretching can also occur when geo-referencing points on the digitised aerial photo are aligned with the same points in the ARC/INFO coverage. The four location points formed a rectangle around the study area with sides of ~ 1000 m. The final image was stretched about 49 m when overlaid onto the contour map representing an error of ~ 5% for the entire image. This is used as an upper estimate of the uncertainty associated with the area of each of the landscape elements, and these uncertainty values are given in Table 1. The approximate mass of soil within the cross banks was calculated by multiplying their 'average' width, depth and height

Table 1
Landscape element proportions determined by ARC/INFO analysis of 1:25,000 digitised aerial photograph

Landscape element	Area (m^2)	Fraction of study area (%)
Snig track	$22{,}200^{a}_{1100}$	18
Log landing	3150_{160}	3
Filter strip	6850_{300}	6
General harvest area	$91{,}700_{4600}$	74
Total	$123{,}900_{6200}$	101[b]

Uncertainties are given in the least significant figure and represent those derived from landscape element area calculations, as per discussion in text.

[a] Snig track area calculated by multiplying total snig track length by survey width of 4.5 m.

[b] Due to rounding of area calculations.

dimensions by their total number and their mean bulk density. A similar approach was adopted for the snig track 'windrows' adjacent to the snig tracks.

3.3. Measuring the ^{137}CS inventory within the landscape elements and reference area

The distribution of ^{137}Cs can be quite heterogeneous in forests (Wallbrink and Murray, 1996a) and a large number of cores can be required to characterise it properly in these environments. Sutherland (1994) presents formula demonstrating that the number of samples required to determine a mean ^{137}Cs reference inventory with an allowable error of 10% at the 90% confidence limit was about 11, if the sample Coefficient of Variation (CV) (Relative Standard Deviation) was $\sim$ 20%. Wallbrink et al. (1994) show that the CV for ^{137}Cs reference inventories within Australia is $\sim$ 40%. Applying the formula of Sutherland under the same confidence limits but with the higher Australian CV, suggests that a minimum number of samples required to determine a mean reference inventory is about 45. Related to this, Wallbrink et al. (1994) showed that ^{137}Cs inventories of bulked soil cores from reference sites were less variable than those of the individual cores that comprised them. Further, bulking allowed an accurate estimate of the mean reference inventory (and standard error) to be determined with fewer analyses, although the standard deviation of the individual cores was more difficult to obtain. Consequently, because our budget calculations only required the mean (and standard error) of the various reference site and landscape element terms, bulking of material from soil cores was used in this study. This also allowed us to take many cores from each location (providing confidence that the distribution of ^{137}Cs had been adequately sampled) without increasing the demand on analytical facilities.

For example, the ^{137}Cs 'reference' inventory was measured in an unlogged, undisturbed native forest adjacent to the study area. It had similar slopes (10–20°), surface area ($\sim$ 11 ha), aspect (SSW), rainfall and vegetation (Fig. 1). Eighty individual cores were taken in a grid pattern from within the reference slope. For each core a metal ring of 100-mm diameter and height 20 mm was first hammered into the ground. The ring prevented the hole from collapsing and defined a specific surface area and volume of soil, which was then excavated by auger. The first ring was then extracted and a second ring of height of 50 mm was then located in the soil, and the enclosed material from 20- to 50-mm depth extracted. Depth increments within the ranges 50–100, 100–200 and 200–300 mm were then taken using a narrower auger of 80 mm diameter. A further 30 samples were also taken using the same method from within a strip of forest of equivalent width to the study area filter strip at the base of this slope. Samples from corresponding depth units were bulked together in groups of 10 for each of these elements. Each bulked group was then counted as a single sample. Analysing the separate depth units allowed a better discrimination of nuclide distributions with depth. The procedure in the above situation provided eight independent inventory analyses for the reference area, and three for the reference filter strip.

The same procedure for soil coring was also used to obtain 80 cores from the GHA (with the largest surface area), and 30 each from the log landings, snig tracks and cross banks, respectively. Individual depth increments from these cores were mixed together in groups of 10 as above. Forty-five cores were taken from the filter strip at the base of the harvested slope. Depth increments here were mixed together in groups of 15, again each bulked group was counted as a single sample. The snig tracks, filter strips and cross banks were sampled linearly (20-m spacing), the remainder on a grid (sample spacing at each 10×10 m). In this study, the sediment trapped in front of the cross banks adjacent to the log landing was included in the sampling of the log landing. A similar procedure was adopted for the cross banks on the snig tracks. The sediment tongues that developed from the cross banks and extended into the GHA, were included in the sampling of the GHA.

3.4. Determining depth distributions of ^{137}Cs and ^{210}Pb$_{ex}$ for soil loss calculations

In this study, the amount of soil lost from the snig tracks and log landings was quantified using the ^{210}Pb$_{ex}$ to ^{137}Cs inventory ratio method described in Wallbrink and Murray (1996a). Briefly, fallout ^{210}Pb is also known as ^{210}Pb excess (^{210}Pb$_{ex}$) and is generated from the decay of ^{222}Rn in the atmosphere. It has a half-life of $\sim$ 22 years and is continually precipitated on the soil surface by rainfall. It is usually defined as the excess of ^{210}Pb activity over that produced in situ by its parent ^{226}Ra. Maximum concentrations of ^{210}Pb$_{ex}$ in soils are usually found at the surface; decreasing approximately exponentially with depth, and reaching typically undetectable levels at depths of $\sim$ 100 mm (Wallbrink et al., 1999). The soil loss estimation procedure requires detailed depth distribution curves for both ^{137}Cs and ^{210}Pb$_{ex}$ to be determined at reference sites. This was undertaken by excavating soil profiles at two sites within the adjacent undisturbed forest area, approximately 100 and 250 m, respectively from the coupe boundary (Fig. 1). Detailed depth penetration information at these sites was obtained by cutting thin layers of soil sequentially down through the soil with a known surface area (0.16 m^2). One-millimeter layers were taken down to 2-m depth. Two-millimeter layers were then taken to 24-mm depth; 10-mm layers from 24 to 104 mm; 20-mm layers from 104 down to 204 mm, and 50-mm layers from 204 mm down to the final depth of 304 mm. This was undertaken using apparatus described in Wallbrink and Murray (1996b).

3.5. Radioactivity in the filter strips resulting from natural erosion

As mentioned above, one factor in developing a budget is accounting for any natural sediment redistribution occurring at the site prior to harvesting. In this respect, the radionuclide content of the study area harvested filter strip comprises radioactivity from: (i) direct ^{137}Cs fallout, (ii) soil particles deposited as a function of natural pre-harvesting sediment redistribution processes, and (iii) soil particles trapped within the filter as a result of post-harvesting disturbance. The first component (item (i)) can be simply derived from the local reference inventory, and item (iii) is what we wish to derive. An estimate of item (ii) can be obtained from the amount by which the ^{137}Cs inventory (Bq m^{-2}) of the filter strip in the reference area was elevated above that of the upslope reference area. This value was then subtracted from the inventory for the study area filter strip (Bq m^{-2}) for use in the subsequent budget calculations.

3.6. Sampling and processing of soil material for particle size analyses

A separate suite of soil samples collected from the reference area was used for particle size analysis. Samples consisted of material taken by hand auger to a depth of 0–20 mm from 36 cores on a 100-m grid on the reference slope. All the material was mixed together, the total dry mass was ~ 5.5 kg. A 300-g sub-sample was taken from the dry mass and analysed separately. The remainder was slaked in water and mechanically agitated through sieves with apertures of 500, 250, 125, and 63 μm. Each sample was then washed again by hand in the appropriate sieve until only clear water emerged from the base of the sieves. The < 63-μm fraction was then separated into 63–40, 40–20, 20–10, 10–2, and < 2 μm fractions by settling. The material comprising each of these size fractions was placed in a dehydrator at 40 °C until dry. Water was added to the dry samples and the organic fraction floated off. This process was repeated three times. The organic material from each fraction was combined, dried and weighed. The mineral material from each particle size fraction was then also dried, weighed and ground. The individual fractions (including organic fractions) were then analysed by gamma spectrometry (described below) to determine their concentrations of ^{137}Cs (Bq kg^{-1}).

3.7. Preparation of soil samples

All soil samples were initially weighed, dried at 40 °C for 48 h, and weighed again to determine field moisture content, and bulk density. Typically, 300 g was extracted for analysis and the remainder archived. The samples were ashed in a muffle furnace at 400 °C for 48 h to determine the Loss on Ignition (LOI); then ground in a rock mill to a fine powder. This powder was then cast in a polyester resin matrix in either a 'cup' (~ 250 g), 'disk' (~ 30 g), or 'stick' (~ 10 g), geometry depending on the sample size. All of the above field samples were collected and analysed during 1996 and 1997. All data are time-corrected to 1996.

3.8. Gamma spectrometry methods for analysis of low level radioactivity

^{137}Cs is a gamma emitter at 662 keV and ^{210}Pb at 46 keV. Both are readily detectable by routine high resolution gamma spectrometry techniques (Hamada and Kruger, 1965). Analysis for these nuclides at the CSIRO laboratories follows the methods described by Murray et al. (1987). The detectors used in this study are 'n'-type closed ended co-axials. Detection limits are about ± 0.3 Bq kg^{-1} for ^{137}Cs and ± 3.0 Bq kg^{-1} for $^{210}Pb_{ex}$. Typical count times were in the order of ~ 84 ks. Independent checks on detector calibrations were undertaken by participating in International Atomic Energy Association (IAEA) intercomparisons.

4. Results

4.1. Detailed depth distributions of ^{137}Cs and $^{210}Pb_{ex}$

The ^{137}Cs and $^{210}Pb_{ex}$ activities (Bq kg^{-1}) associated with the two detailed depth profiles from the reference site are given in Fig. 2. As expected, maximum activities of

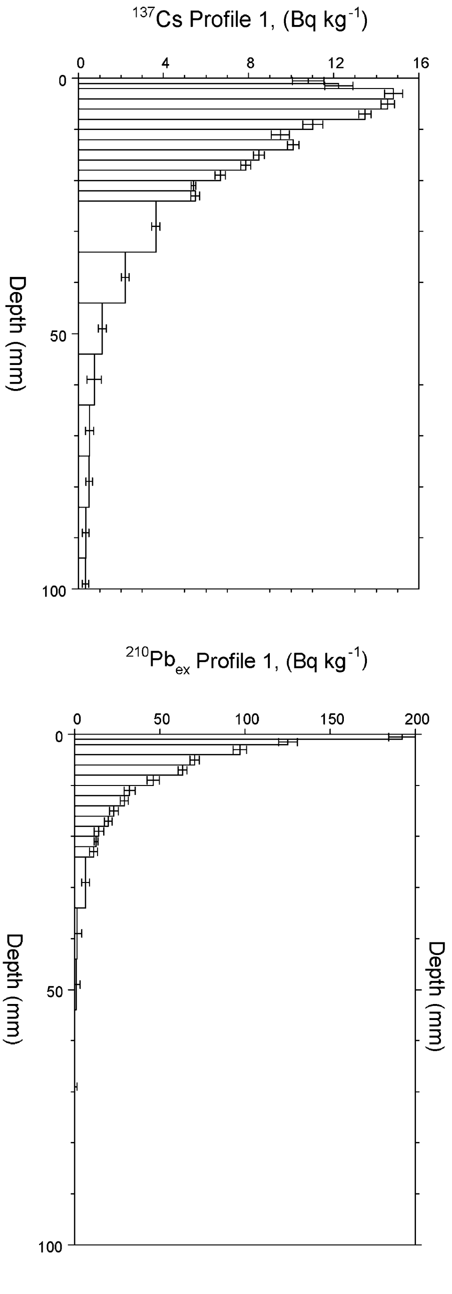

Fig. 2. ^{137}Cs and ^{210}Pb$_{ex}$ activity distribution with depth from profiles 1 and 2 within the reference area adjacent to compartment 1708.

$^{210}Pb_{ex}$ occur at the soil surface, while those of ^{137}Cs occur just below the soil surface. These are both consistent with other profiles described in the literature cited above. The cumulative inventory ratio values (Wallbrink and Murray, 1996a) for these are 3.05 ± 0.2 and 3.09 ± 0.3 for profiles 1 and 2, respectively.

4.2. Particle size separation experiment

The relative affinity of ^{137}Cs for different particle size fractions is given in Table 2. There is a systematic trend for ^{137}Cs to be preferentially associated with smaller size particles. The highest activities (Bq kg^{-1}) are found on the < 2-μm material, the lowest on > 500-μm specimen. This trend is consistent with the results observed by He and Owens (1995) and Wallbrink et al. (1999) for ^{137}Cs, $^{210}Pb_{ex}$ and ^{7}Be. Of the total of 39.4 Bq, the highest proportion was retained in the 2–10 μm range (9.1%) and the lowest in the 40–63 μm range (1.6%).

4.3. Caesium-137 activities in the different landscape elements

Table 3 presents the ^{137}Cs inventories (Bq m^{-2}) for each of the landscape elements as well as those for the reference slope, and reference filter strip. The highest ^{137}Cs inventories were in the filter strip at the base of the harvested slope 860 ± 90 Bq m^{-2}. The lowest values were measured in the log landings and snig tracks, 150 ± 90 and 190 ± 90 Bq m^{-2}, respectively. The mean from the reference slope was 493 ± 25 Bq m^{-2} ($n=80$ cores, $n=8$ bulked samples). This was below that calculated for the GHA 525 ± 25 Bq m^{-2}. The difference between the ^{137}Cs reference value and the study area filter strip is ~ 360 Bq m^{-2}. This is significantly greater than the equivalent difference between the reference area slope and the reference area filter strip, i.e. 85 Bq m^{-2}. The latter value represents the amount by which background erosion has contributed tracer

Table 2
Affinity of ^{137}Cs, for different particle sizes at the study site

Size (μm)	Dry weight (g)	^{137}Cs activity (Bq kg^{-1})	Total ^{137}Cs activity (Bq)	^{137}Cs (%)	Cumulative activity by integrated size class (Bq kg^{-1})
< 2	90	57_{1}	$5.1_{0.1}$	$13.1_{0.4}$	$56.9_{0.7}$
2–10	262	35_{1}	$9.1_{0.1}$	$23.0_{0.7}$	$40.3_{1.1}$
10–20	143	20_{1}	$2.9_{0.1}$	$7.4_{0.3}$	$34.6_{1.9}$
20–40	124	16_{1}	$2.0_{0.1}$	$5.1_{0.2}$	$30.9_{2.6}$
40–63	147	11_{1}	$1.6_{0.1}$	$4.1_{0.1}$	$27.1_{3.2}$
63–125	406	10_{15}	$4.2_{0.4}$	$10.7_{1.1}$	$21.3_{4.7}$
125–250	508	$4.3_{0.4}$	$2.2_{0.2}$	$5.6_{0.6}$	$16.2_{5.2}$
250–500	911	$3.6_{0.4}$	$3.2_{0.4}$	$8.2_{0.9}$	$11.7_{5.1}$
> 500	1910	$1.6_{0.1}$	$3.1_{0.3}$	$7.9_{0.4}$	$7.4_{3.9}$
Organics	503	12_{0}	$5.9_{0.0}$	$14.9_{0.4}$	$7.9_{4.2}$
Total	5006		$39.4_{1.3}$	$100_{2.4}$	

Analytical uncertainties equivalent to 1 standard error are given as subscripts in the least significant figure.

Table 3
Inventories of ^{137}Cs for landscape elements within the study area, and adjacent reference slope near Bombala, NSW

Landscape element	Number of cores within sample (actual counted)	^{137}Cs (Bq m^{-2})
General Harvest Area (GHA)	10 (1)	590_{80}
	10 (1)	520_{40}
	10 (1)	560_{40}
	10 (1)	310_{70}
	10 (1)	540_{40}
	10 (1)	580_{40}
	10 (1)	610_{50}
	10 (1)	480_{40}
Mean	**80 (8)**	$\mathbf{525_{25}}$
Log landing	10 (1)	40_{130}
	10 (1)	220_{50}
	10 (1)	200_{61}
Mean	**30 (3)**	$\mathbf{150_{90}}$
Snig tracks	10 (1)	300_{40}
	10 (1)	210_{80}
	10 (1)	50_{80}
Mean	**30 (3)**	$\mathbf{190_{90}}$
Filter Strip	15 (1)	720_{60}
	15 (1)	1030_{70}
	15 (1)	830_{60}
Mean	**45 (3)**	$\mathbf{860_{90}}$ (775_{130})**!**
Reference slope	10 (1)	470_{30}
	10 (1)	400_{50}
	10 (1)	620_{50}
	10 (1)	530_{60}
	10 (1)	560_{60}
	10 (1)	490_{30}
	10 (1)	450_{40}
	10 (1)	430_{70}
Mean	**80 (8)**	$\mathbf{493_{25}}$
Reference filter strip	10 (1)	570_{40}
	10 (1)	500_{40}
	10 (1)	660_{40}
Mean	**30 (3)**	$\mathbf{580_{90}}$

Uncertainties equivalent to one standard error are given as subscripts. Uncertainties for means are derived from analysis of entire data set.

activity to the reference slope filter areas (described above) and has been subtracted from the study area filter strip inventory when incorporated into the ^{137}Cs tracer budget described below.

The cross banks do not have an inventory value in Table 3. This is because they have been created from material bladed off the snig tracks and log landings during and after harvesting. Consequently, the ^{137}Cs contained within them is distributed throughout the entire soil matrix of the bank and not just within the top 20 cm or so of the soil. Their total

Table 4
Total activity of ^{137}Cs contained within the cross banks of the study area in compartment 1708

Tracer	Total volume (m^3)	Total mass (kg)	Average activity (Bq kg^{-1})	Total activity (MBq)
^{137}Cs	450_{20}	678270_{33914}	$1.78_{0.89}$	$1.21_{0.6}$

Uncertainties equivalent to one standard error are given as subscripts.

activity has therefore been calculated by multiplying their average ^{137}Cs concentration (Bq kg^{-1}) by the total mass of soil (kg) contained within them. The concentration is derived from soil coring and the total mass from the measured volumes and bulk density (~ 1.52 Mg m^{-3}) described above. The total amount of ^{137}Cs (MBq) contained within the cross banks is given in Table 4.

4.4. Comparison of study period with long-term rainfall record

Table 5 presents the rainfall record for the township of Bombala, the nearest measurement site to the study area. The period of record is continuous for 117 years. It can be seen that the monthly means and medians over the 6-year study period are consistent with the longer term trends, as are the total annual amounts. The Median annual rainfall is 639 mm for the entire 117 years, and 613 mm for the study period. These are in close agreement. Also given is the maximum daily rainfall that has occurred within each month for both periods of record. It is interesting that the highest daily rainfall (108 mm) for all the July months in the 117-year record, occurred in 1991, the first year after harvesting. Total rainfall in this month was 178 mm, which followed 189 mm in the month of June, 1991.

Table 5
Comparison of rainfall for 6-year study period with 117-year rainfall record for Bombala

Period (year)	Monthly rain (mm)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Ann
1990		36	116	24	108	47	9	19	72	55	61	20	25	590
1991		109	32	33	67	32	189	178	44	40	22	18	87	851
1992		144	54	20	51	21	63	20	41	94	76	88	76	748
1993		53	66	122	13	20	32	56	28	58	81	48	62	639
1994		25	109	39	87	22	28	5	10	25	57	122	33	562
1995		121	19	16	18	68	31	20	10	37	139	81	97	656
1996		61	42	37	32	61	16	47	20	54	35	106	44	554
1990–1996	Mean	78	63	41	54	39	52	49	32	52	67	69	60	657
1990–1996	Median	61	54	33	51	32	31	20	28	54	61	81	62	639
1885–2001	Mean	66	57	60	46	45	60	46	41	45	56	63	65	647
1885–2001	Median	59	46	47	33	30	38	30	31	38	49	53	55	613
1885–2001	Highest	334	390	204	166	283	377	247	157	166	175	279	207	1189
1885–2001	Lowest	0.3	0.0	0.5	0.8	1.6	2.8	0.0	1.8	2.5	2.5	0.3	1	308
1885–2001	highest daily	142	249	86	64	110	122	108	83	61	69	125	96	
1990–1996	highest daily	77	34	55.6	51	29.4	83	108	19.6	35.4	29.4	44	58	

These two monthly totals are well above the longer-term medians of 38 and 30 for June and July, respectively. This implies that the period of June and July 1991 in the first year after harvesting was significantly wetter than average at the study site.

5. Discussion

5.1. Combining tracer activity and landscape element data

As indicated above, construction of a tracer budget requires values for the total amount of ^{137}Cs (in Bq) within the catchment prior to harvesting, as well as within each landscape element before and after harvesting. This is obtained by multiplying the surface area (m^2) of the catchment, or landscape elements, (Table 1) by their measured areal inventories in Bq m^{-2} (Table 3). For example, the study area is 123 900 m^2 (12.4 ha). The ^{137}Cs reference inventory is 493 ± 25 Bq m^{-2}. Consequently, the total amount of ^{137}Cs contained within the study area (including filter strip) prior to logging is calculated to be 61.1 ± 4.3 MBq. The log landings occupy 3200 m^2 and contain 1.6 ± 0.1 MBq prior to harvesting. After harvesting the amount was only 0.5 ± 0.3 MBq, as the inventory was reduced to 150 ± 90 Bq m^{-2}. Similar calculations can be undertaken for each landscape element (Table 6). Note that the total activity contained within the cross banks is derived from Table 4.

Table 6 shows that the highest overall loss of ^{137}Cs (6.8 ± 2.1 MBq) occurred from the snig tracks, this is because the ^{137}Cs depletion associated with these tracks was significant and they occupy a large area (Table 1). This emphasises the importance of roads and tracks as sources of sediment, and is consistent with the work of others, (i.e., Megahan and Kidd, 1972; Grayson et al., 1993). The losses from the log landings and snig tracks (in MBq) represent 2% and 11% of the total initial ^{137}Cs budget, respectively. Gains of 2.9 ± 5.0 and 1.9 ± 0.7 MBq occurred in the GHA and filter strips, respectively, totalling some 8% of the initial ^{137}Cs fallout. A further 2% of the initial ^{137}Cs budget was stored within the cross banks and lateral windrows (1.2 ± 0.6 MBq). Total ^{137}Cs losses were 13 ± 4%, whilst total gains added to 10 ± 8%. About 1.8 MBq, or 3%, of the total initial amount was unaccounted for. Overall, some 97 ± 10% of the initial ^{137}Cs deposited at the site could be accounted for. The confidence limits used in these calculations represent the cumulative errors of the analytical precision of ^{137}Cs measurements, and the calculation of the inventories and areal coverages. These become additive, and thus larger, when adding or subtracting one component from another.

The same pattern is revealed when the absolute changes in pre- and post-harvesting tracer activity amount for each landscape element are compared (Table 6). The snig tracks and log landings lost 62 ± 30% and 69 ± 40% of their initial ^{137}Cs inventories. In contrast, the inventories associated with the GHA and filter strip increased by 6 ± 0.8% and 56 ± 8%, respectively.

5.2. Sediment redistribution based on ^{137}Cs tracer budgets

The ^{137}Cs data of (Tables 3, 4 and 6) has been expressed graphically in Fig. 3 to describe the flow of ^{137}Cs (and sediment) within and between the various landscape

Table 6
Tracer budget showing redistribution of material between different landscape elements before and after harvesting at the study area

Landscape element	Area (m^2)	Inventory (Bq m^{-2})	Number of cores in sample, (actual counted)	Total pre-harvest ^{137}Cs activity (MBq)	Total post-harvest ^{137}Cs activity (MBq)	Difference (MBq)	Losses and gains (% of total)	Absolute change within element (%)
Snig tracks	$22{,}200_{1110}$	190_{90}	30 (3)	$11.0_{0.8}$	$4.2_{2.0}$	$-6.8_{2.1}$	-11_{4}	-62_{30}
Log landings	3150_{160}	150_{90}	30 (3)	$1.6_{0.1}$	$0.5_{0.3}$	$-1.1_{0.3}$	$-2_{0.5}$	-69_{40}
GHA	$91{,}700_{4600}$	525_{30}	80 (8)	$45.2_{3.2}$	$48.1_{3.8}$	2.9_{5}	5_{8}	$6_{0.8}$
Filter strip	6850_{340}	775_{90}	45 (3)	$3.4_{0.2}$	$5.3_{0.7}$	$1.9_{0.7}$	$3_{1.0}$	56_{8}
Cross banks					$1.2_{0.6}$	$1.2_{0.6}$	2_{1}	
Reference area		493_{25}	80 (8)					
Total				$61.1_{4.3}$	$59.3_{4.4}$			
Unaccounted						$-1.8_{5.5}$		
Accounted					97 ± 10%			

Uncertainties are derived from the square root of the summation in quadrature of the surface area errors (Table 1) and analytical errors (Table 3). Rounding occurs in this table.

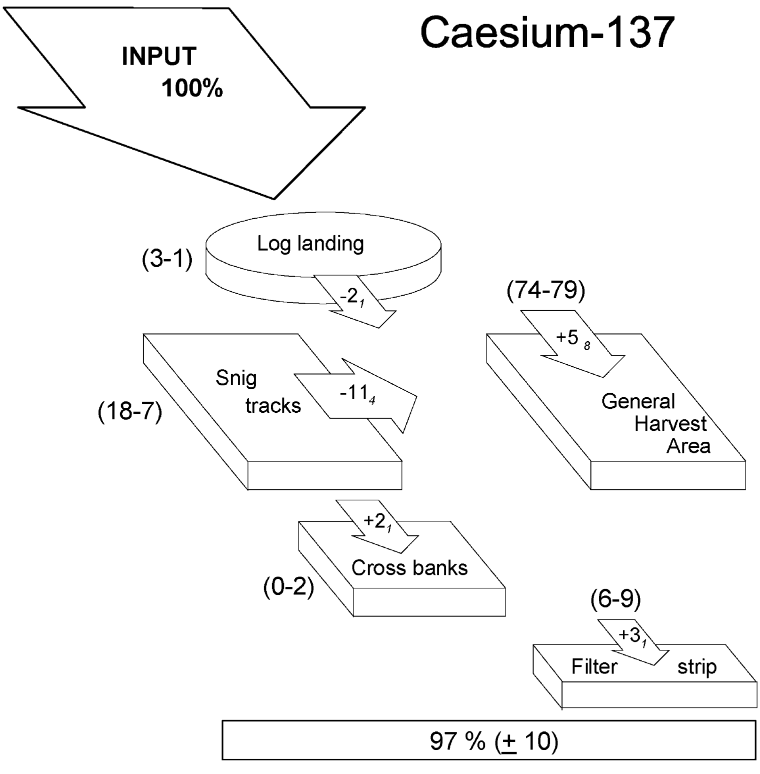

Fig. 3. Tracer budget for study area within compartment 1708, ~ 6 years after harvesting showing areas of erosion and deposition in various landscape elements, based on measurements of fallout ^{137}Cs. Note: Values within arrows represent tracer amounts either transported from or deposited within each element, as a fraction of total initial input. Values in parentheses represent the amount of activity contained within each element before and after harvesting as a percent of total inputs. Uncertainties are given as subscripts and are derived cumulatively from measurement errors and element area derivations.

elements of the study area. As discussed above, the sum of the tracer amounts within the landscape elements, can be compared to the amount calculated as initially present, and be used to quantify any net losses from the system. Fig. 3 shows that the ^{137}Cs budget balances within the confidence limits, i.e., 97 ± 10%. This is consistent with no net loss of material occurring from the study area. However, soil and sediment redistribution has clearly occurred within it, and the diagram illustrates the flow of material. As indicated above, losses totalling some 13 ± 4% of the total ^{137}Cs budget, have occurred from the snig tracks and log landings. This transport is illustrated as arrows leaving the respective landscape element. The placement of cross banks is designed to divert this eroded material onto high infiltration, high surface roughness areas such as the GHA and filter strip (Croke et al., 1999b). This has clearly occurred, as there is net deposition and storage of ^{137}Cs within the GHA, filter strip and cross banks. These deposition amounts are represented by arrows entering into these elements.

The total amount of ^{137}Cs transported from the snig tracks was 6.8 ± 2.1 MBq. Of this amount, 18% (1.2 ± 0.6 MBq) could be accounted for within the cross banks; 28% (1.9 ± 0.7 MBq) within the filter strip; and the remainder 43% within the GHA. The material contained within the crossbanks is presumably derived from mechanical action at the time of their construction. The amount of ^{137}Cs retained within the filter increased by ~ 60% (Table 6) from a background reference of 493 ± 25 to 775 ± 90 Bq m^{-2} in the ~ 6 years post-harvesting. This increase is due to harvesting associated erosion alone, as the natural erosion component has been taken into account.

5.3. Sediment delivery to the filter strip

Given the important role of the filters as a 'last line of defence' in mitigating offsite transport of material; the transport pathway, and size characteristics of material delivered to them is worth further consideration. For example, Croke et al. (1999a,b) use rainfall simulator experiments to show that runoff generation and sediment transport from within the GHA itself is negligible compared to that from snig tracks. They also demonstrate that coarse-grained (>63 μm) sediment transported into the GHA from the snig tracks is effectively deposited, either in front of the cross bank, or within 5 m of its entry to the GHA. Fine grained sediment diverted by the cross bank however was observed being transported in plumes within the GHA, some 15–20 m downslope from cross banks under extreme rainfall intensities (110 mm h^{-1}). The potential for similar incursions of sediment plumes from cross banks through the GHA to stream side vegetation has also been modelled probabilistically by Hairsine et al. (2001). Thus, combining the observations of Croke et al. (1999a,b) with the tracer measurements described above, leads us to conclude that the additional ^{137}Cs (and sediment) entering the filters is derived from snig tracks.

The rate of sediment accumulation in the Filter strip can be calculated from the available data. Firstly it has been shown that the material entering the Filters is probably derived as cross bank runoff from erosion of snig tracks. The overall loss of ^{137}Cs from the snig tracks has been calculated as 6.8 ± 2.1 MBq. Of this amount some 28 ± 13% (Table 6), or 1.9 ± 0.7 MBq can be accounted for within the Filter strip. From the rainfall simulator work of Croke et al. (1999a,b) we know that the predominant grain size delivered to these filters is fine grained, < 63 μm. The average ^{137}Cs concentration of material in this size range is 27 ± 2 Bq kg^{-1}, from Table 2. At this ^{137}Cs concentration, 1.9 ± 0.7 MBq is equivalent to a total of some 70 ± 25 t of material; or 11.7 ± 4 t year per year over the 6 years post-harvesting. The surface area of the filters is (6900 m^2, Table 1) thus representing an annual mass deposition to them, of 1.7 ± 0.6 kg m^{-2} $year^{-1}$. This figure however is an upper estimate, as it assumes that the depositing material is uniformally < 63 μm grain size. If depositing material is finer than this, i.e. < 20 μm then the mass deposition amount would be smaller. Nonetheless, these soils have high hydraulic conductivities (Moore et al., 1986; Croke et al., 1999b) and so the fine grained sediment delivered to them is likely to be deposited *into* the soil matrix with infiltrating runoff, as much as *onto* the soil surface. The close agreement between the pre- and post-budgets also suggests that, within uncertainties, little leakage of sediment has occurred from the system as a whole, and the filters in particular.

5.4. Loss of soil from log landings and snig tracks

The ^{137}Cs budget indicates net losses of material from the snig tracks and log landings. The amount of soil removed from these can be quantified by comparing their residual $^{210}Pb_{ex}$ to ^{137}Cs inventory ratios with that calculated for the detailed profiles in the reference area. Fig. 4 shows the inventory ratio curve calculated from the mean of the two detailed depth profiles. The method works by calculating the ratio of the cumulative inventories of $^{210}Pb_{ex}$ to ^{137}Cs with depth in a reference location and comparing this with the residual inventory ratio left behind in areas where soil has been removed. The point at which the disturbed area residual ratio intersects the reference inventory ratio curve represents the depth of soil that has been removed from the system. The soil depth loss amounts can then be converted to a total net loss in tonnes per hectare using the measured bulk density from the surface layers of the reference soil cores (1.12 Mg m^{-3}, $n = 10$).

The mean residual inventory ratios for the log landings and snig tracks (calculated by dividing the average $^{210}Pb_{ex}$ and ^{137}Cs inventories within each of them) are 0.40 ± 0.18 and 1.68 ± 0.51 (Table 7). These are plotted in Fig. 4, and are equivalent to soil depth losses of 54 ± 8 and 13.5 ± 6 mm, respectively. It should be noted that our tracer-based values give total cumulative losses, including mechanical removal of soil by bulldozer blading, export on truck wheels, tyres and chassis, as well as losses due to wind and water erosion. The majority of this depth loss presumably occurs during or immediately after

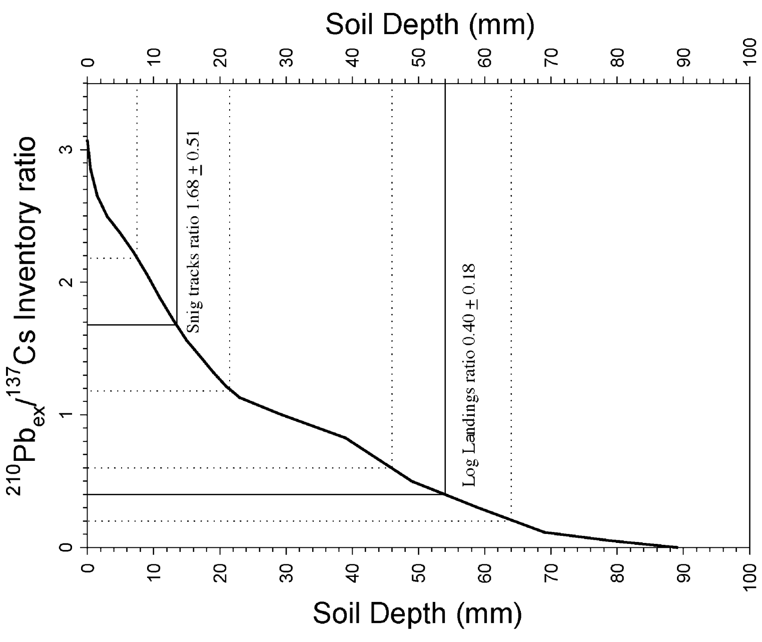

Fig. 4. Calculated depth loss for log landings and snig tracks using $^{210}Pb_{ex}/^{137}$Cs cumulative inventory ratio profile calculated from average of detailed depth profiles 1 and 2, given in Fig. 2.

Table 7
$^{210}Pb_{ex}/^{137}Cs$ inventory ratio values for reference profiles, log landings and snig tracks

Landscape element	Number of cores in sample, (actual counted)	$^{137}Cs/^{210}Pb_{ex}$ inventory (ratio value)
Reference profiles	2 (detailed profiles)	$3.07_{0.23}$
Log landings	30 (3)	$0.40_{0.18}$
Snig tracks	30 (3)	$1.68_{0.51}$

harvesting; probably resulting from construction of these features. It should also be noted that the period June–July 1991 represented above average months of rainfall for this region. The method also assumes that the soil density in the reference area is similar to that in the disturbed areas. Compaction of the disturbed log landings and snig tracks would effect this comparison to some degree, and any depth losses calculated would be upper estimates. This is especially so if the entire soil profile is compacted, although as indicated above, much of the overlaying 'eroded' material is scraped off mechanically and so density variations become less important. Nonetheless, these depth losses can be recalculated to annual losses of 25 ± 11 t ha^{-1} year^{-1} for the snig tracks and 101 ± 15 t ha^{-1} year^{-1} for the log landings, using the $\sim$ 6-year gap between harvesting and the measured soil density. These snig track values can also be partially compared to those of 4–5.8 t ha^{-1} measured from single high rainfall intensity events using a rainfall simulator on snig tracks within the study area. This also suggests that the water derived component of erosion may be small compared to mechanical impacts at this site.

5.5. Comparison of the ^{137}Cs budget, eroded sediment and the measured ^{137}Cs depth distributions

It is also worthwhile examining whether the estimates of soil losses predicted above are consistent with the ^{137}Cs budget, the concentrations of ^{137}Cs eroded sediment and the measured ^{137}Cs depth distributions. For example, the total loss of ^{137}Cs from the log landings and snig tracks was calculated to be 1.1 ± 0.3 and 6.8 ± 2.1 MBq (Table 6), respectively. The corresponding depth losses for these were 54 ± 8 and 13.5 ± 6 mm, giving a total soil depletion mass of 191 ± 28 and 355 ± 109 t for each of them, after accounting for their different surface areas. Dividing the total ^{137}Cs loss (Bq) by the mass loss (kg) provides estimated concentrations on eroded material from the log landings and snig tracks of $\sim 6 \pm 1.6$ and $\sim 19 \pm 6$ Bq kg^{-1}, respectively. If erosion is assumed to be integrated to the same erosion depths as above, then similar concentrations on eroded sediment can be independently estimated from the known depth profiles. For example, using the data of Fig. 1, the average ^{137}Cs concentrations of material eroded to depths of 54 and 13.5 mm would be 5.8 ± 1.5 and 13 ± 2.1 Bq kg^{-1}, respectively. Within uncertainties, and allowing for particle size effects (Table 2), these are consistent with the calculated concentrations on eroded material from the log landings and snig tracks from above (i.e., $\sim 6 \pm 1.6$ and $\sim 19 \pm 6$ Bq kg^{-1}). Overall, these calculations provide some confidence that the estimates of depth loss are internally consistent with the ^{137}Cs budget and the predicted ^{137}Cs concentrations on eroded material.

6. Conclusions

Managing the impacts of forest erosion requires knowledge of the erosion sources, the rates of soil movement from them, its transport and storage, as well as any losses from the system. We have presented a new approach using a tracer-based sediment budget that enabled us to quantify many of these parameters. The ^{137}Cs budget showed no net loss of soil material from within the study area after harvesting within uncertainties (97 $\pm$ 10%). However, it did reveal that significant redistribution, storage and transport of sediment had occurred between landscape elements. There was a net transport of material from the snig tracks and log landings (11 and 2% of total ^{137}Cs), net losses from them were calculated to be 25 $\pm$ 11 and 101 $\pm$ 15 t ha^{-1} year^{-1}, respectively. Although this is presented as an average rate, the majority of this may have been displaced due to mechanical erosion. There were also 2 months of well above average rainfall in the period 1 year after harvesting. Overall however, these are identified as the major erosion impact sites in the catchment. The erosion rate was highest in the log landings; the greatest net transport occurred from snig tracks. Of the tracer activity transported from the snig tracks 18%, 28% and 43% was accounted for within the cross banks, filter strip and GHA, respectively.

The ^{137}Cs budget showed the GHA to be a significant region of sediment trapping. The filter strip also played a fundamental role in the trapping of material generated from the snig tracks; the mass delivery to them from this source was calculated to be 1.7 $\pm$ 0.6 kg m^2 year^{-1}. Careful management of these remains critical. Our work suggests that (within errors) the overall runoff management system of dispersing flow from the highly compacted snig tracks, by cross banks, into the less compacted (and larger area) GHA and filter strips has effectively retained soil and sediment mobilised as a result of harvesting at this site. This confirms the utility, and necessity, of having filter strips to trap and store inflowing sediment, they remain a very important part of any erosion mitigation strategy for retaining sediment mobilised from upslope sources.

In summary, the use of tracer-based budgets can provide detailed information on rates, storages and transfers of transported material. These terms are difficult to measure by traditional means. In particular we have quantified the redistribution of material within and between different landscape elements on slopes following harvesting and used this to assess the effectiveness of erosion mitigation controls at the study site.

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