Use of fallout radionuclides in studies of erosion and sedimentation

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Unless otherwise stated, the findings presented in this thesis are based on field work and analyses carried out by the author.

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Abstract:

The ^{137}Cs technique assumes that the initial distribution of ^{137}Cs is spatially uniform. However if it is non-uniform, then estimates of soil loss could be in error. Consequently the variability (relative standard deviation, *rsd*) of ^{137}Cs (Bq m^{-2}) is measured in various 'reference' areas in SE Australia and found to be ~40%, irrespective of scale and/or landform type. The areal concentrations of ^{137}Cs and ^{210}Pb$_{ex}$ were found to be correlated ($r^2 = 0.7$), suggesting that ~70% of their distribution could be explained by similarities in their initial fallout and relocation prior to sorption within the soil matrix. Because of this correlation, the *rsd* of the ratio between them was less than that of either nuclide alone. A method is developed using this ratio and its measured depth dependency, to calculate soil loss with better precision than was possible with either ^{137}Cs or ^{210}Pb$_{ex}$.

The fallout nuclides ^{7}Be, ^{210}Pb$_{ex}$ and ^{137}Cs were found to have different depth penetrations in undisturbed soils. These differences gave rise to unique concentrations (Bq kg^{-1}) and ratios that could be used to characterise sediment source, and by inference, erosion mechanism. This method was used to show that the dominant (~97%) contribution to fine-grained sediment flux in a small first order stream at Whiteheads Ck., NSW, was from subsoil erosion of gully walls. The contribution of material from the sheet/rill component was very small (~3%). The method also predicted that subsoil material dominated the fine grained sediment flux in rivers at the much larger scale (13,500 km^2) of the mid-Murrumbidgee catchments, NSW. This finding was confirmed by incorporating the ^{137}Cs and ^{210}Pb$_{ex}$ concentrations of the three catchment sources; subsoils, cultivated lands and uncultivated lands into a simple mixing model. This predicted the contribution from subsoils to be ~89%. The combined contribution from the uncultivated and cultivated sources was estimated to be ~11%, the majority of this (by a factor of ~6) was from cultivated sources. These contributions were dependent on residence time of the sediments, which was estimated to be less than 6 years in the mid-Murrumbidgee. Profile reference curves of ^{137}Cs and ^{210}Pb$_{ex}$ were also constructed for the cultivated and uncultivated source areas of the mid-Murrumbidgee, and predicted the depths of contribution to be $<3 \pm 10$ mm from the surface of the cultivated areas, and $<2 \pm 6$ mm from the surface of the uncultivated lands. The same method was also applied to a small plot on Black Mountain, ACT, and was able to predict the depths from which artificially generated sediments were derived, and estimate the proportional amounts of material from these different depths.

The low amounts of ^{137}Cs (Bq kg^{-1}) observed in fine grained sediments of Whiteheads Ck. and the mid-Murrumbidgee, were also observed in rivers draining some ~18% (1,400,000 km^2) of the Australian land surface. A two component mixing model estimated that the subsoil contribution to these was $90 \pm 10\%$. The actual amount varied depending on assumptions about (i) the mean depth of gullies/channels, and (ii) the ^{137}Cs concentration on surface soils. The substantial contribution by subsoils to stream sediments suggests that, by comparison, surface eroded material must be inefficiently delivered to streamlines. This is probably due to the low relief of the continent, sediment exhaustion of surface sources and the generally low rainfall and runoff in this country. Alternatively, subsoil material is rarely supply limited, and their channel and gully sources are extremely well connected to streamlines. The benefit of these tracer based results is that they directly estimate the contribution amounts from various erosion sources, to the sediments themselves. To the author's knowledge, this has not been attempted for the Murrumbidgee, or any other Australian river, before.

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Man must cherish the earth, for he will live by it forever
(Anon)

for

(Barbara, Holly, Audie and Ted)

Chapter 1: Introduction

Chapter 1: Introduction

1.1 Perspective

It is difficult to accept when things of value suffer neglect. Two such commodities in Australia today are soil and water. Their neglect is not by intent, or idleness, on behalf of their caretakers; rather, these people are distracted by the realities of generating an income from the land, and the cycle of debt that often goes with it. In this scenario, it is the soil and water, as well as the caretaker (often a farmer), that must work harder to sustain life. The consequences of the resulting pressure on soil and water often go unnoticed, simply because it occurs in small amounts, and over a time scale long compared to our lives. The cumulative impacts become apparent however, when fundamental change occurs: fish and birds disappear, crop productivity declines, or algal blooms flourish. At this point the links between cause and effect come into sharp focus.

What we need are indicators, to gauge degradation of soil and water as it occurs. In this way, the consequences of degradation can be quantified; better still they can be balanced against the perceived benefit from whatever action or process is responsible for them. This thesis sets out to provide indicators in the form of methods for measuring the erosion of soils, and the impact of erosion on downstream environments.

In the following sections of this chapter the general structure of the thesis is outlined. Broader issues in erosion are first discussed and two themes are identified, for which a more detailed understanding is required: i) measuring soil erosion; and ii) methods for tracing and quantifying the sources of suspended sediments in rivers. The aims and objectives of the research required to address these themes are presented in the context of hypotheses that are constructed and tested within each chapter. The results of all these chapters are then synthesised in a discussion chapter, preceding the general conclusions at the end of the thesis.

1.2 Land degradation

1.2.1 The problem

Degradation of soil resources is a serious problem in most countries, acknowledged in government reports, soil erosion reviews, and environmental reports (Anon., 1978; Brown, 1984; Clarke *et al.*, 1985; CSCAFF, 1984; Department of Environment, Sport and Territories, 1996). If soil removal from slopes exceeds the rate at which it is regenerated, fertility will decline and agricultural productivity will decrease. Loss of soil is of particular concern where soil is scarce, and Edwards (1987; 1991) argues that the rates of soil formation in Australia are so low, that effectively no soil loss can be sustained at all. In addition some of the soil removed from slopes eventually enters the fluvial system, where it adversely impacts on downstream environments (Walling, 1988). Impacts include higher levels of turbidity and associated effects on drinking water quality (Smalls, 1980), the decreased capacity and utility of water storages (Wasson, 1987; Walling, 1988), contamination of reservoirs (Foster *et al.*, 1991), and disruption to the ecology of aquatic plants and fish (Cullen, 1991). Nutrients such as phosphorus are also associated with sediments and may be an important factor in changes to aquatic ecology, such as the increased occurrence of algal blooms (Sharpley and Menzel, 1987; Oliver, 1993).

If we aim to conserve soil and wish to evaluate the effects of different conservation practices, then the amount of soil loss, and the rate at which it occurs, must be quantified. Also, if the downstream impacts of soil erosion are to be minimised, it is important to know from which parts of the landscape the material has originated so that management can be targetted. These issues are discussed separately below.

1.2.2 Erosion of soil from slopes - how much and at what rate?

There are several methods available for determining the rates and amounts of soil removed from slopes. These include: visual observation, survey by theodolite, measurement using erosion pins, erosion plots, aerial photography, and use of a rainfall simulator (de Ploey and Gabriels, 1980; Iverson, 1980; Evans *et al.*, 1994; Connolly *et al.*, 1996; Gascuel-Odoux *et al.*, 1996). The results from small scale erosion plot measurements have also been included in various predictive soil models such as the USLE, MUSLE, and SOILOSS (Rosewell, 1993). Radioactive caesium-137 (^{137}Cs) provides another approach to estimating soil loss which can provide a retrospective, time integrated view. This radionuclide was primarily deposited in Australia as fallout from atmospheric testing of nuclear weapons (Walton, 1963; McCallan *et al.*, 1980). It binds to soil particles upon deposition, and so translocation can be used as a measure of soil movement.

Rogowski and Tamura (1965) provide an early demonstration of its use. They showed a good correlation between soil loss and residual ^{137}Cs content in a prepared soil plot. The nuclide has been subsequently employed in many other locations and continents (Ritchie *et al.*, 1974; Brown *et al.*, 1981; Longmore *et al.*, 1983; Fredericks *et al.*, 1988; Elliott *et al.*, 1990; Walling and Quine, 1991; Sutherland, 1992; Loughran *et al.*, 1992; Quine *et* *al.*, 1993; 1994; Basher *et al.*, 1995; Loughran and Elliot, 1996). Reviews of the method are given in Wise (1980) and Ritchie and McHenry (1990). A usual assumption of this technique is that the initial distribution of ^{137}Cs is uniform across the landscape of interest (Longmore *et al.*, 1983). However, it has been suggested (Fredericks *et al.*, 1988; Sutherland 1991; 1994; Owens and Walling, 1996) that initial spatial variability of ^{137}Cs in some circumstances can be considerable. This could affect those models that estimate erosion and deposition by comparing ^{137}Cs residuals with only a single or limited number of known 'reference' values. Therefore in order to better use fallout tracers, the initial distribution of ^{137}Cs in 'reference' type locations needs to be characterised. Methods to take this effect into account must then be developed.

1.2.3 Downstream impacts - sediment sources

At the small scale, the links between soil erosion and its effect on downstream water quality are often clear. However, if the aim is to reduce sediment delivery to waterways it is unlikely that all the potential sources and forms of soil erosion can be treated simultaneously, as resources to undertake remediation work are limited. Nevertheless, by properly targetting the primary source(s) of either the particulate material, or the associated nutrient or pollutant, resources can be best allocated. It is then important to know whether the material is derived from surface soils, or from deeper in the profile.

There is often a marked contrast in nutrient status between the surface and the underlying soil mantle (Kirkby, 1980; Norrish and Rosser, 1983). A large contribution of soil from below the soil surface (with low concentrations of phosphorus) may contribute more to total phosphorus flux in a river system than a lesser amount from the soil surface (with higher phosphorus concentrations). Clearly, in this case, if a reduction in total phosphorus flux is the objective, remediation efforts should be directed towards subsoil sources. Other 'imported' pollutants such as heavy metals (in fertilisers), pesticides, and atmospheric contaminants are often associated only with surface soils (Novotny *et al.*, 1986; Foster and Dearing, 1987). The determination of the depth from which sediment originates has been undertaken using the mineral magnetic attributes of soils and sediments (Walling *et al.*, 1979; Foster *et al.*, 1996); a combination of magnetic with other diagnostic properties such as ^{137}Cs (Peart and Walling, 1986; Walling *et al.*, 1993; Hutchinson, 1995); ^{137}Cs alone (Ritchie *et al.*, 1974; Peart and Walling, 1986; 1988; Hasholt and Walling, 1992); ^{137}Cs and ^{7}Be (Burch *et al.*, 1988; Olley *et al.*, 1993); and ^{137}Cs, $^{210}\mathrm{Pb}_{ex}$ and ^{7}Be (Wallbrink and Murray, 1990; 1993; Wallbrink *et al.*, 1991; Walling and Woodward, 1992). A numerical mixing model has also been developed using ^{226}Ra, ^{137}Cs and ^{210}Pb (He and Owens, 1995). However, apart from the last example, these approaches tend to be descriptive. Consequently, methods need to be developed that can better report depth sources (to thereby identify erosion processes), precisely calculate the depths from which sediment has come, and more accurately determine the contributions from different sources.

1.3 This thesis - aims and outline

Thus far, two themes involving fallout radionuclide methods in studies of erosion and sedimentation have been identified, in which a knowledge gap is believed to exist. This thesis aims to close these gaps by improving these methods. The research required to do this provides the context for the five central chapters. All of these have been reviewed prior to publication in (Chapters 2, 3, 4), or submission to (Chapters 5, 6), an international journal. The first theme (Chapters 2 and 3) deals with variability of fallout

^{137}Cs and Lead-210. (Fallout ^{210}Pb, or atmospherically derived ^{210}Pb, is that fraction of the nuclide which is in excess of the parent Radium-226 (^{226}Ra), referred to hereafter as ^{210}Pb$_{ex}$). A method is described which makes use of the relationship between these two, to more accurately quantify soil erosion from slopes. The second theme (Chapters 4, 5, and 6) focuses on establishing the different depth distributions and concentrations of ^{137}Cs, ^{210}Pb$_{ex}$, and Beryllium-7 (^{7}Be) in disturbed and undisturbed soils. These differences are then used to quantify the depth sources, and relative contributions, of sediments in active transport. These general aims are examined in the form of hypotheses. Most of the hypotheses are constructed to examine the detail of a specific process at a particular site, or to challenge a preconceived view. Some, however, simply represent a starting point with which to begin the understanding of a particular process. A summary of all of them is set out below.

1.3.1 Theme 1: Measuring soil erosion

The first set of hypotheses is as follows:

(1) ^{137}Cs is uniformly distributed within reference areas;

(2) The distribution of ^{137}Cs at the small (1 ha) scale is associated with soil movement alone; and

(3) The variability of ^{137}Cs within tree covered areas will be higher than in pasture.

These are tested and discussed in Chapter 2: "Reference site variability" in which the variability in areal concentrations of ^{137}Cs in 'reference' locations, in and around the ACT region, is measured at the scale of 1 ha and 300 km^2. The variability of ^{137}Cs in pastureland and forested environments is also examined. The results of this chapter provide the framework for the next set of hypotheses, which are:

(4) The ratio of the soil inventories of $^{210}Pb_{ex}$ and ^{137}Cs has a lower overall variability in 'reference' areas than the areal concentrations of either nuclide alone;

(5) The ratio of $^{210}Pb_{ex}$ to ^{137}Cs in forested reference areas is different from those in harvested areas; and.

(6) Soil loss from a 'normal' impact logged plot will exceed that from a 'minimal' impact logged plot.

These are addressed in Chapter 3: "Determining soil loss using the inventory ratio of fallout ^{210}Pb to ^{137}Cs" in which the suggestion that fallout ^{210}Pb may be subject to the same causes of variability as ^{137}Cs is examined. It is proposed that the ratio between $^{210}Pb_{ex}$ and ^{137}Cs may provide a more robust description of cumulative soil erosion across the landscape than ^{137}Cs alone. The method is based on the simultaneous use of $^{210}Pb_{ex}$ and ^{137}Cs, and is tested by attempting to determine the different amounts of soil loss from plots treated by 'normal' and 'minimal' impact harvesting practices in a eucalypt forest.

1.3.2 Theme 2: Tracing and quantifying the sources of suspended sediments

The origin of suspended sediment material within fluvial systems is now considered, and a second group of hypotheses are tested:

(7) The tracers ^{137}Cs, $^{210}Pb_{ex}$, and ^{7}Be have different initial soil depth distributions in undisturbed soils;

(8) Eroded material from soil surfaces will have high fallout nuclide concentrations, and subsoil derived material will have low fallout radionuclide concentrations; and

(9) The net contribution from surface erosion will exceed that from gully erosion.

In Chapter 4, "Determining erosion processes using fallout radionuclides", it is proposed that differences in the concentrations of ^{137}Cs, $^{210}Pb_{ex}$, and ^{7}Be in soils and sediments can be used to describe erosion type, and thus the source depth of sediments. This is examined within a first-order gullied catchment in which the relative contributions from surface sheet erosion and subsoil gully erosion to catchment sediment flux are unknown. The aim of the experimental work is to first determine whether differences in radionuclide tracer concentrations occur between top-soil and subsoil derived material. These differences are then used to infer sediment source(s) and calculate their relative contributions. An attempt is also made to improve the precision of this method by testing the next group of hypotheses. These are as follows:

(10) The combined depth profiles of ^{137}Cs, $^{210}Pb_{ex}$, *and* ^{7}Be *provide unique concentration labels on soil material from different depths within those profiles;*

(11) The concentration labels of ^{137}Cs, $^{210}Pb_{ex}$, *and* ^{7}Be *on sediments from these profiles can be used to determine their depth origin(s) in the soil profile;* and

(12) Surface material derived from shallow rill and overland flow will exceed that from sides and floors of deeper rills and gullies.

These hypotheses are tested in Chapter 5, "Quantifying depth sources using fallout radionuclides", in which soil profile reference curves are constructed using the measured depth dependencies of ^{137}Cs, ^{210}Pb$_{ex}$, and ^{7}Be in a small (~150 m^{2}) plot. The plot surface is subject to separate treatments of increasing surface disruption, and rainfall is applied to the plot between each treatment condition. The tracer concentrations on the sediments generated from each treatment are then compared with the profile reference curves to retrospectively determine their original location in the soil profile. The problem of selective transportation of finer particles is addressed by normalising soil concentration data to the particle size range of the transported sediments. The results of the tracer based research at the small scale are then examined at the large scale of the Murrumbidgee River, by testing the following hypotheses:

(13) Different land use and land form types will produce distinct fallout radionuclide concentrations in eroded sediment;

(14) Surface material from the heavily cultivated lands of the mid-Murrumbidgee will dominate the flux of material in the channels draining this area; and

(15) The average residence times of fine particulates in this large catchment will be longer than 10 years.

This final set of hypotheses is tested in Chapter 6, "Determining the origin of suspended sediment sources at large scales". In this chapter, the different concentrations of $^{210}Pb_{ex}$ and ^{137}Cs in various potential catchment sources (uncultivated land, cultivated land and channel/gully banks) are used in a simple mixing model to calculate their relative contributions to sediments in the middle region of the Murrumbidgee River. Large parts of this catchment are actively cultivated for the production of wheat and cereals; there are also numerous channels and gullies. *In situ* ^{210}Pb labelling of sediment by direct fallout is also considered and used to estimate boundaries on the residence time of suspended sediment material in the major tributaries of this river.

In Chapter 7 the thesis is drawn together by applying the methods developed in each chapter to data from the other study sites. Some general trends in the predictions of the methods and the results from the different locations are noted. The significance of these results to regional processes and patterns of sediment production are then examined. These findings are extrapolated to catchments of similar morphodynamics elsewhere in Australia. In Chapter 8 the major conclusions of this thesis are summarised, various deficiencies in the research and methodology are discussed, and options for further research to improve and test the findings are proposed.

Theme 1:

Measuring soil erosion

Chapter 2: Reference site variability

Chapter 2: Reference site variability

2.1 Perspective

Soil erosion is a serious problem in Australia (see Section 1.2.2). One method to quantify the amount of erosion and the rate at which it is occurring, incorporates measurements of radioactive fallout ^{137}Cs. However, the use of this method is complicated by the potential for initial variability in its distribution. This variability may be a function of surface cover type (ie. the presence or absence of trees). This chapter describes measurements of ^{137}Cs in a small plot at Yarramundi Reach, ACT. Soil redistribution is first estimated using 'conventional' ^{137}Cs analysis. These estimates are then examined in the light of the initial variability of ^{137}Cs explicitly measured in reference areas both at different scales within the same region, and at different locations in southeastern Australia. Reference areas with and without tree cover are considered. Finally, the work in this chapter is linked to the second chapter by presenting preliminary evidence that the variability of ^{137}Cs can be partly accounted for by using it in conjunction with ^{210}Pb$_{ex}$.

Therefore, hypotheses tested in this chapter are that: *(1)^{137}Cs is uniformly distributed within reference areas; (2) The distribution of ^{137}Cs at the small (1 ha) scale is associated with soil movement alone; and (3) The variability of ^{137}Cs within tree covered areas will be higher than in pasture.*

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2.2 Introduction

The fallout radionuclide ^{137}Cs has been used extensively to quantify patterns of soil accumulation and movement within landscapes (McCallan *et al.* 1980; Campbell *et al.* 1982; Loughran *et al.* 1990; Quine and Walling 1991). The technique involves measuring the total input of ^{137}Cs at a 'reference' area and then comparing it to values observed from disturbed locations. Areas with lower areal concentrations than the reference value are considered to have undergone erosion and those with higher concentrations have experienced soil accumulation. Reviews of this technique are given in Ritchie and McHenry (1990) and Sutherland (1991).

A central assumption of this technique is that local fallout is uniformly distributed (Walling and Quine, 1992). However this assumption has rarely been tested (see Fredericks *et al.*, 1988 and Sutherland, 1994) and little consideration is given to the inherent natural variability of ^{137}Cs in undisturbed 'reference' areas.

In this study, ^{137}Cs areal concentrations have been measured over a 1- ha hillslope plot. The variability in the ^{137}Cs concentrations are first interpreted in terms of soil redistribution, assuming a uniform ^{137}Cs deposition. The variability of ^{137}Cs concentrations due to deposition effects are then assessed by examining ^{137}Cs concentrations at eight adjacent sites which fulfil the 'reference site' criteria used by others, (Ritchie *et al.* 1974; McCallan *et al.* 1980; Campbell *et al.* 1982). These data are then used to determine whether or not the variability in ^{137}Cs concentrations at the 1- ha hillslope site could be explained in terms of fallout variability.

A method of reducing the variability of ^{137}Cs due to fallout effects, utilising the covariability of $^{210}Pb_{ex}$ is then suggested.

2.3 Site Descriptions

The experimental work was undertaken in the city of Canberra which lies within the Australian Capital Territory (latitude 34°S, see Figure 2.3.1) and at St Helens, Tasmania (latitude 42°S). The long term average rainfall in Canberra is ~630 mm which falls evenly across the city; annual deviations from this can be up to 100 mm.

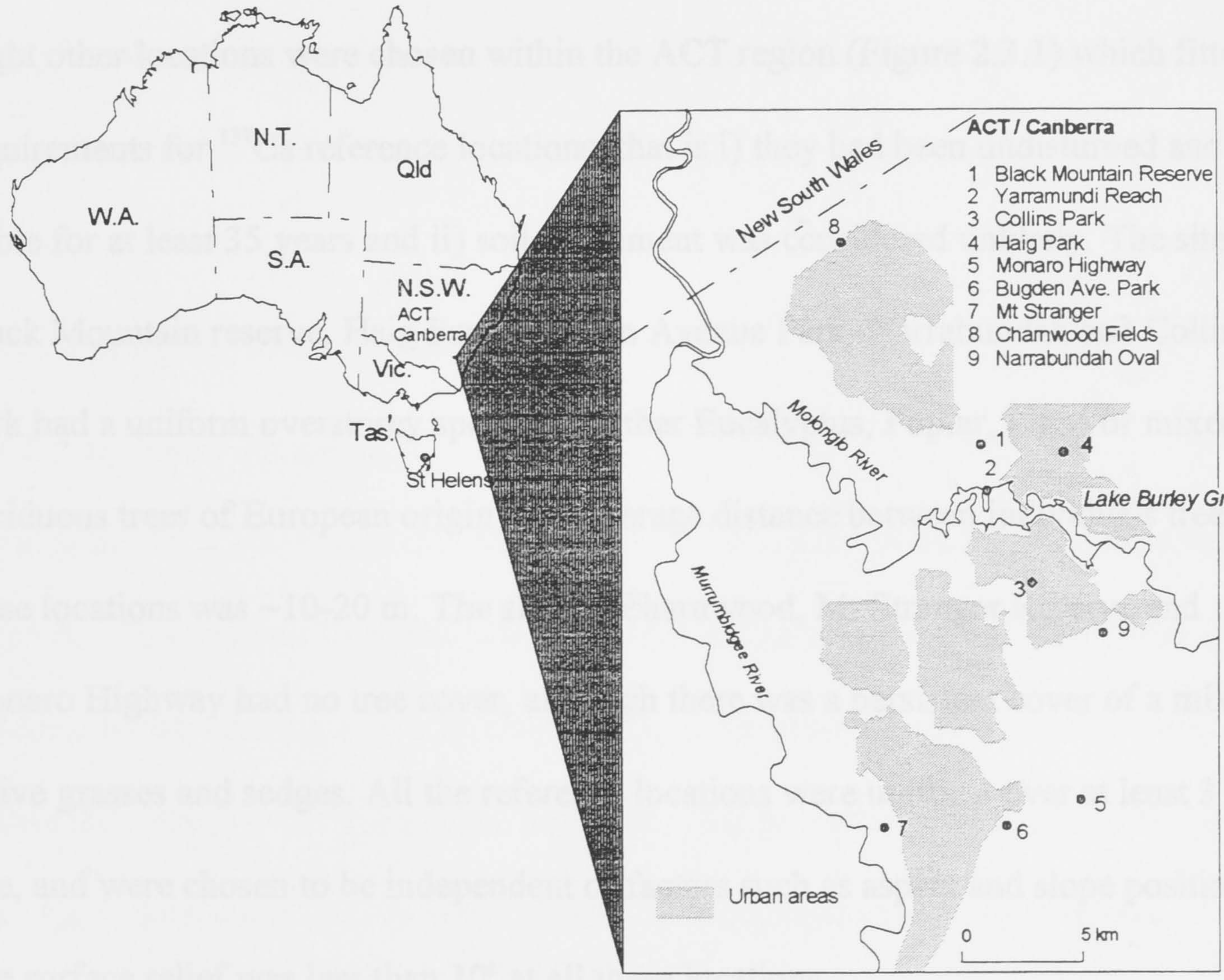

Figure 2.3.1 Location diagram of Yarramundi Reach test plot and eight reference sites sampled within the A.C.T., Australia.

2.3.1 Yarramundi Reach, ACT

The 1 ha test plot was at Yarramundi Reach on the western shores of Lake Burley Griffin which lies centrally within the city. The maximum vertical relief at the site was ~5 m. Adjacent to this plot, approximately 50 m distant, is a commercial *Pinus radiata* plantation. Based on data presented by Longmore *et al.* (1983) for Brisbane, the net fallout of ^{137}Cs in Canberra is expected to be ~640 Bq m^{-2}, decay corrected to 1991, although this makes no allowance for the 10° latitude difference between them.

2.3.2 ACT region reference sites

Eight other locations were chosen within the ACT region (Figure 2.3.1) which fitted the requirements for ^{137}Cs reference locations, that is i) they had been undisturbed and stable for at least 35 years and ii) soil movement was considered unlikely. The sites at Black Mountain reserve, Haig Park, Bugden Avenue Park, Narrabundah and Collins Park had a uniform overstorey species of either Eucalyptus, Poplar, Pinus or mixed deciduous trees of European origin. The average distance between individuals trees in these locations was ~10-20 m. The sites at Charnwood, Mt Stranger Reserve and Monaro Highway had no tree cover, although there was a persistent cover of a mix of native grasses and sedges. All the reference locations were uniform over at least 3 ha in size, and were chosen to be independent of factors such as aspect and slope position. The surface relief was less than 10° at all these locations.

Soils in the ACT region fall within the following six major soil groups as defined by Northcote, (1979), Uniform Coarse textured, Uniform fine-textured, Massive earths, Structured red and brown earths, Texture contrast and Alluvial. These are differentiated mainly on the basis of texture, Gunn et al., (1969). The reference sites were based on a selection of these different soil types.

2.3.3 St Helens, Tasmania

The St Helens forest (latitude 42°S) is dry sclerophyll and consists predominantly of *Eucalyptus seberii.* The understorey is largely nonexistent due to frequent burning. The average distance between individual trees is ~20 m. This overstorey and landform type is persistent over an area of ~100 ha. The soils here are described as yellow podzolics which have been formed on adamellite granites and are classified as Uc2.21 (Northcote, 1979; W. Nielson, 1994, pers. comm.). Two reference plots were selected within this forest, their surface relief was less than 15°

2.4 Soil sampling and analysis

2.4.1 Yarramundi Reach, ACT

An area of 100 x 100 m was divided into a 10 x 10 m grid. A series of 50 samples were taken for ^{137}Cs analysis from soil pits excavated within every second cell. These were obtained by hammering a 20 x 20 cm frame into the ground and excavating the soil within this area (0.04 m^2) to a depth of 25 cm. This produced a mass of soil in the range 8-12 kg, depending on soil density. A detailed topographical survey was then undertaken of this site with vertical resolution of ~1cm.

2.4.2 ACT region reference sites

At each of the reference sites, 9 samples were taken by the method described above, except for Black Mountain in which 5 samples were taken. At Narrabundah, Monaro, Bugden Avenue and Black Mountain all the samples were analysed separately. At Charnwood, Haig Park, Collins Park and Mt Stranger Reserve the nine samples were randomly mixed into groups of three which were then analysed.

2.4.3 St Helens, Tasmania

At St Helens, 20 representative soil cores, each of surface area 0.008 m^2, were taken from 2 reference plots, representing a surface area of ~700 m^2. The cores were 300 mm deep and contained approximately 2.5 kg of soil. Upon returning to the laboratory the tubes were cut into 2 sections (0-5, 5-30 cm) which were measured independently to allow for greater precision in $^{210}Pb_{ex}$ concentrations. Fallout ^{210}Pb is preferentially retained nearer the surface than ^{137}Cs (Wallbrink and Murray, 1993).

The soil samples from Yarramundi Reach, ACT region and St Helens were each thoroughly homogenised and crushed in a ring grinder. A representative subsample of this material (250g) was then removed for radionuclide analysis. Note: A full description of the analytical methods used in this thesis is given in Appendix A1.

2.5 Results and discussion

2.5.1 Yarramundi Reach, ACT

A digital elevation model was created for the Yarramundi Reach site using the SURFER^{T} software package. The ^{137}Cs (Bq m^{-2}) spatial data were also contoured and the resultant *isocaes* surface (Longmore *et al.* 1983) overlayed onto the digital terrain model (Figure 2.5.1). The concentration of ^{137}Cs ranged from 300 to 1800 Bq m^{-2}, uncertainties were ~90 Bq m^{-2}. The areal concentrations of ^{137}Cs are highest at the bottom of the slope and lowest in the midslope areas. If the Longmore *et al.* (1983) scale of erosivity, (converted to Bq m^{-2}, i.e. <270 Bq m^{-2} severe erosion, 270-570 moderate erosion, 570-685 mild erosion, 685-860 mild accumulation, 860-1145 moderate accumulation, 1145-1430 heavy accumulation, >1430 Bq m^{-2} very heavy) is applied to our site, then at least moderate erosion should be observed on the western side of the site and very heavy accumulation on the bottom of the slope.

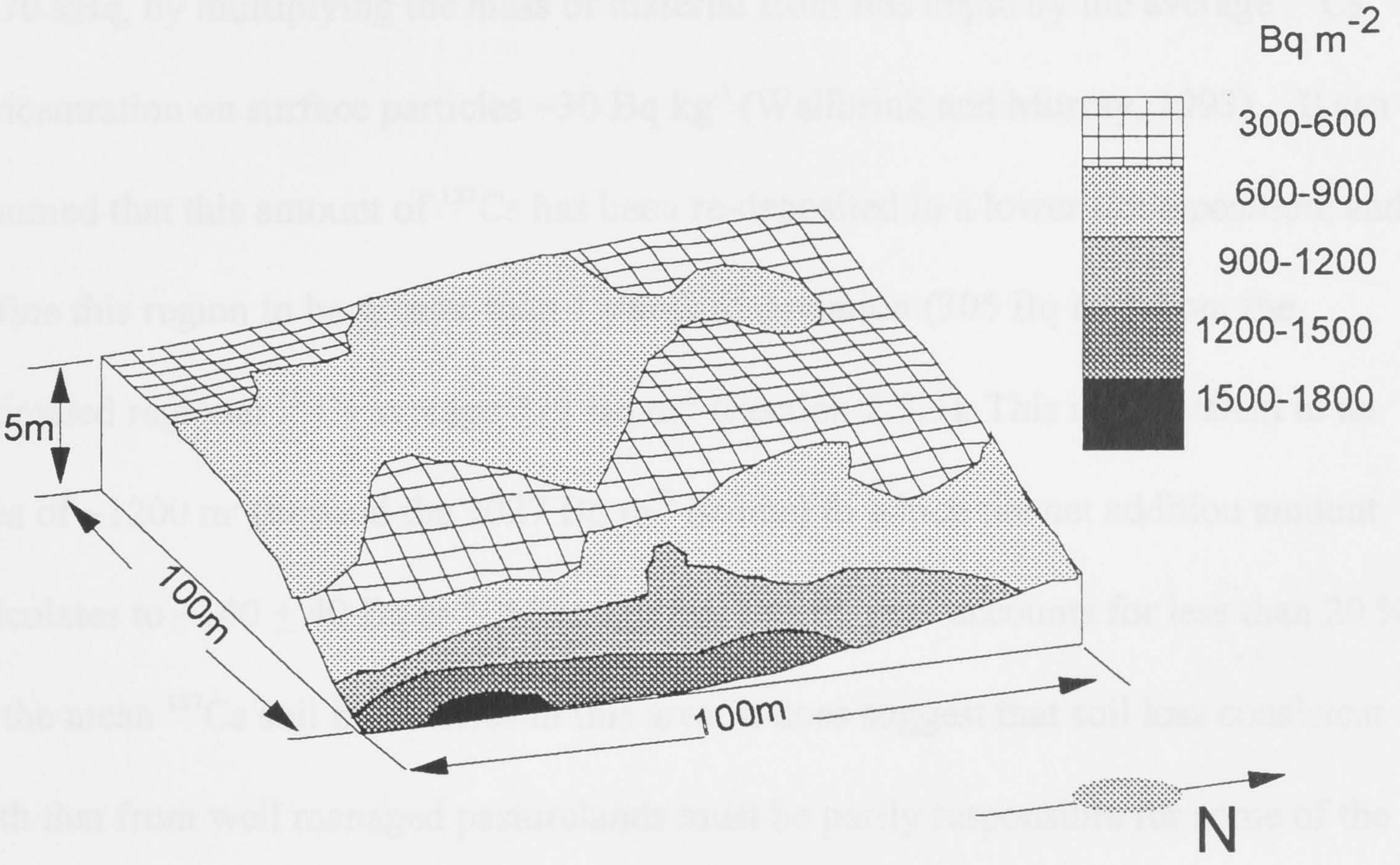

Figure 2.5.1 Map of ^{137}Cs areal concentrations and surface topography over a 1 ha site at Yarramundi Reach, ACT.

However, physical observation of the site, including examination of stratigraphy in soil pits, revealed no discernible evidence of soil movement or accumulation. There is also a semi permanent soil cover by native grasses which limits soil and particle movement. It is thus considered improbable that soil movement has occurred to the degree suggested by this simple application of Longmore's erosivity scale. However, a realistic estimate of soil loss from well managed native pastures, such as at Yarramundi, is considered to be ~19 ± 5 t km^{-2} yr^{-1} (R.J. Wasson, 1994, pers. com.). This value can be used to estimate the minimum influence of soil movement on the distribution of ^{137}Cs at this site. The analysis is begun by first converting the specific yield from pasturelands to a total average soil depth loss at Yarramundi Reach over the ^{137}Cs timescale (30 yr); which is equivalent to ~0.74 ± 0.2 mm using the measured bulk density of 1.3 g cm^{-3}. The total amount of ^{137}Cs redistributed by the movement of this soil then calculates to ~170 kBq, by multiplying the mass of material from this depth by the average ^{137}Cs concentration on surface particles ~30 Bq kg^{-1} (Wallbrink and Murray, 1993). It can be assumed that this amount of ^{137}Cs has been re-deposited in a lower slope position, and define this region to be greater than 1 standard deviation (305 Bq m^{-2}) from the estimated regional ^{137}Cs average 722 Bq m^{-2} (Section 2.5.3). This is equivalent to an area of ~1200 m^2 (beyond the 1027 Bq m^{-2} isoline) to which the net addition amount calculates to ~140 ± 40 Bq m^{-2}. Although this latter figure accounts for less than 20 % of the mean ^{137}Cs soil inventories in this area, it does suggest that soil loss consistent with that from well managed pasturelands must be partly responsible for some of the downslope trend of increasing ^{137}Cs areal activity in this plot. The remainder of the difference however, has yet to be accounted for.

2.5.2 Possible causes of initial non-uniform distribution of ^{137}Cs

Deposition of ^{137}Cs is known to occur predominantly in association with rainfall. There are two mechanisms that may contribute to a non-uniform distribution of ^{137}Cs at the time of deposition. The first of these is related to attributes of surface topography, which may influence fallout over a scale of one to hundreds of metres. Features such as hills, trees and shrubs affect the random pattern of rainfall by creating persistent perturbations in air flow (Gash, 1986). These perturbations create wind and rain shadows and thus, zones of relative ^{137}Cs depletion and enhancement. In this context the presence of the mature Pinus trees upwind of the Yarramundi plot should be noted. In contrast, concentrations of ^{137}Cs may be increased in the drip zone beneath canopies. It is therefore expected that the variability of ^{137}Cs will be greater within forested areas than areas without canopy cover. The second group of mechanisms which may influence ^{137}Cs deposition are related to soil factors which affect the local movement of rain water on and within the soil over a scale of 1 to hundreds of centimetres. These include differences in soil density, infiltration capacities and soil chemical properties. The net effects of these processes on ^{137}Cs fallout can be assessed by examining variability of ^{137}Cs at locations that fulfil the reference site criteria.

2.5.3 ACT region reference areas

The heterogeneity in ^{137}Cs distribution was estimated by determining the relative standard deviation (*rsd*, also known as coefficient of variation) of ^{137}Cs areal concentrations from a number of soil cores collected from the reference sites within the ACT, where it is thought that no significant soil loss or redistribution has occurred since 1950. The results have been condensed to show the number of samples taken, the average, standard deviations, relative standard deviations and the standard error from each reference location (Table 2.5.3.1). The corresponding values from Yarramundi Reach are also given. The mixed samples from the four sites (noted as * in Table 2.5.3.1) did not represent individual estimates and thus were not included in the calculation of the regional ^{137}Cs variability. This was calculated to be 722 Bq m^{-2} with standard deviation of 305 Bq m^{-2} using data from the other four non-mixed sites (denoted as #). The *rsd* of ^{137}Cs areal concentrations across these reference sites in the ACT was ~42 % and compares to ~38 % at Yarramundi Reach. There is a negligible difference in the net regional average if the Yarramundi data is included in the overall reference site average calculations (Table 2.5.3.1).

Table 2.5.3.1 Yarramundi Reach, ACT, ACT regional reference sites and St Helens, Tasmania, ^{137}Cs areal concentration data.

Location	Number of Samples	Average ^{137}Cs (Bq m^{-2})	Standard Deviation (σ)	Relative Standard Deviation	Standard Error
Yarramundi Reach	50	685	272	38	38
Bugden Avenue $^{(F\ \#)}$	9	509	204	40	68
Narrabundah $^{(F\ \#)}$	9	721	159	22	53
Monaro Hwy $^{(P\ \#)}$	9	797	329	41	109
Black Mountain $^{(F\ \#)}$	5	974	356	36	159
Charnwood $^{(P\ *)}$	3 (9)	573			88
Haig Park $^{(F\ *)}$	3 (9)	745			111
Collins Park $^{(F\ *)}$	3 (9)	546			69
Mt. Stranger $^{(P\ *)}$	3 (9)	655			32
# sites (excluding Yarramundi Reach)	32	722	305	42	54
# sites (including Yarramundi Reach)	82	699	288	41	32
St Helens					
^{137}Cs concentration	18	810	380	47	90
		^{210}Pb$_{ex}$/^{137}Cs			
^{210}Pb$_{ex}$/^{137}Cs ratio	18	2.2	0.6	27	0.14

Note: $^{\#}$ denotes all soil samples analysed separately
* denotes soil samples mixed into groups of 3
^{F} denotes forest cover, ^{P} denotes pasture

The ^{137}Cs data also show no systematic difference in average areal concentration between forested or pastured reference sites, with the lowest and highest variabilities occurring in the latter category. The total range of average areal values was from 546 (σ = 98) Bq m^{-2} at Collins Park to 974 (σ= 356) Bq m^{-2} at Black Mountain reserve.

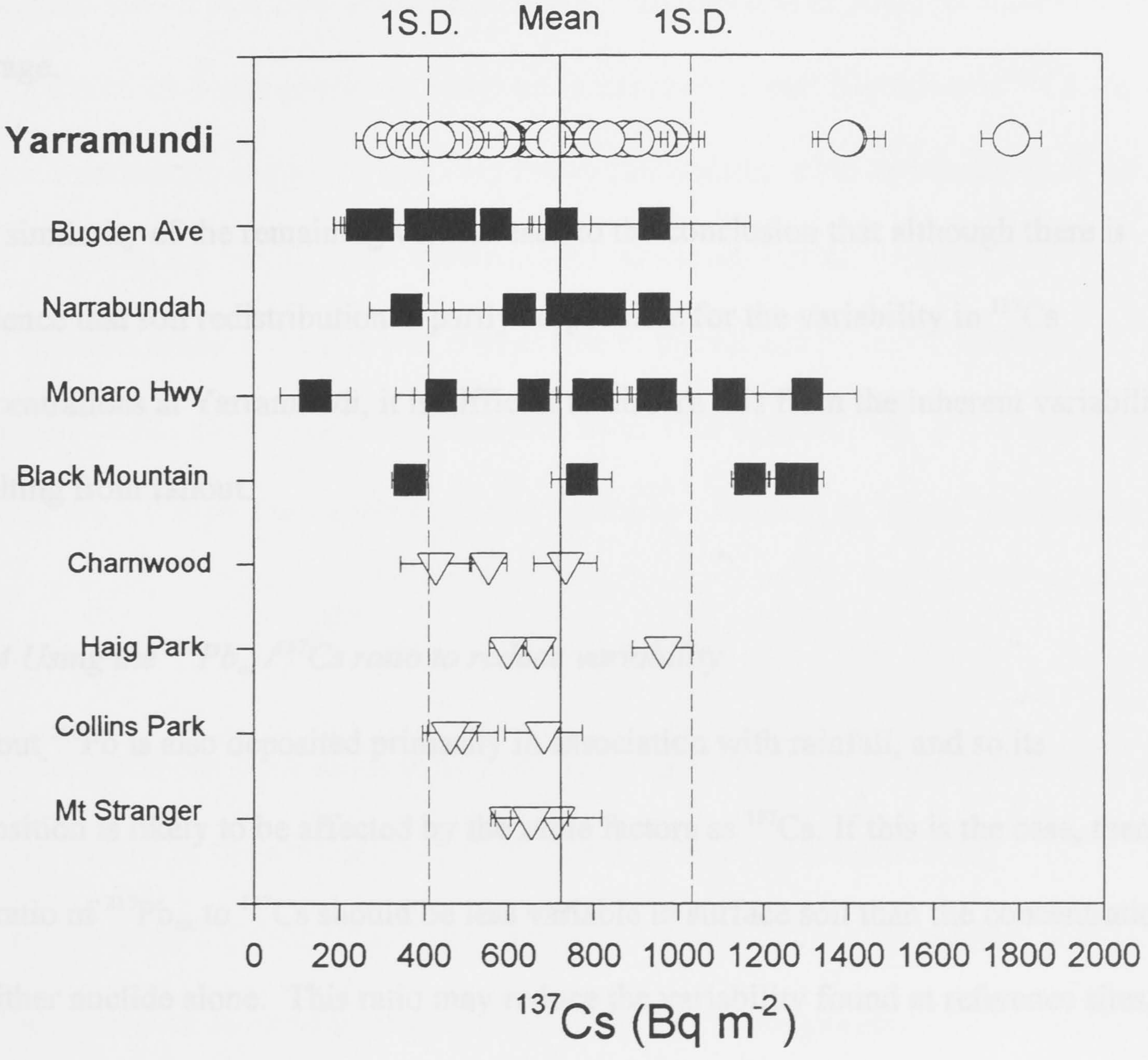

Figure 2.5.3.1 Spread in ^{137}Cs areal concentration data from Yarramundi Reach, ACT and ACT regional reference sites.

The data from all the sites have been plotted (Figure 2.5.3.1). The mean and 1 standard deviation reference lines (solid and dashed lines respectively) are derived from the non-mixed (black squares) reference site data. It can be seen that mixing of soil samples (open triangle) reduces the scatter of ^{137}Cs and allows accurate estimates of site averages to be made with fewer analyses, however information on real field variability is obscured. The standard errors from both the mixed and non-mixed reference areas (Table 2.5.3.1) are, as expected approximately equal. It is also evident that both the areal ^{137}Cs averages and the spread of concentration data from Yarramundi Reach are consistent with those from the reference areas. There is only one point from the Yarramundi Reach data set that is greater than 3 standard deviations of the regional average.

The similarity of the remaining values leads to the conclusion that although there is evidence that soil redistribution is partly responsible for the variability in ^{137}Cs concentrations at Yarramundi, it is difficult to discern this from the inherent variability resulting from fallout.

2.5.4 Using the $^{210}Pb_{ex}/^{137}Cs$ ratio to reduce variability

Fallout ^{210}Pb is also deposited primarily in association with rainfall, and so its deposition is likely to be affected by the same factors as ^{137}Cs. If this is the case, then the ratio of ^{210}Pb$_{ex}$ to ^{137}Cs should be less variable in surface soil than the concentrations of either nuclide alone. This ratio may reduce the variability found at reference sites, and its use has been examined at the St Helens site.

The average ^{137}Cs inventory at St Helens was 810 ($\sigma = 380$) Bq m^{-2} (n=18, see Table 2.5.3.1), with *rsd* of ~47 %. The ratio of ^{210}Pb$_{ex}$ to ^{137}Cs was then determined from the same samples to be 2.2 : 1 ($\sigma = 0.6$). This is an *rsd* of ~27 %, which represents a relative reduction in variability of ~40 %. Clearly, the ^{210}Pb$_{ex}$/^{137}Cs ratio is less variable at this site and suggests that it may be a more sensitive indicator of surface soil movement than measurements of ^{137}Cs concentrations alone. This technique is currently being pursued as a means of deriving more stable reference values elsewhere.

2.6 Conclusions

It has been shown that areal concentrations of ^{137}Cs varied with slope position in a 1-ha hillslope plot. Concentrations were highest in the lower areas. A published ^{137}Cs soil erosion relationship suggested that very heavy soil accumulation had occurred in the lower slopes in the last 30 years. However, it was shown that the variability in ^{137}Cs areal concentrations at the site was comparable to the variability in ^{137}Cs fallout determined from nearby uneroded reference sites. This suggests that, at the very least, some of the hillslope site variability could be due to variations in fallout distribution alone.

It is clear that more work needs to be undertaken to identify and quantify the cause of ^{137}Cs variability in the environment. Until this is done, areal concentration measurements of ^{137}Cs should only be used with caution. On the other hand, fallout variability may be reduced by taking the ratio of ^{137}Cs concentration to ^{210}Pb excess. This indicates that the ^{210}Pb$_{ex}$/^{137}Cs ratio may be a more reliable indicator of surface soil movement.

2.7 Summary and consideration of hypotheses

To determine soil movement based on redistribution of ^{137}Cs, 50 samples were taken at grid points on a 1- ha hillslope plot. There was a correlation between slope position and areal concentration of ^{137}Cs, suggesting that soil had been moved from upslope areas and deposited in lower slope positions. However, detailed sampling was undertaken at eight suitable nearby reference locations to determine ^{137}Cs fallout over the region. The distribution of ^{137}Cs in these locations was found to have a variability of ~40%. Therefore hypothesis *(1)^{137}Cs is uniformly distributed within reference areas* is rejected. The variability in areal concentrations at these locations was approximately equal to the variability observed in the test plot, suggesting that fallout variability alone could account for the observed differences in ^{137}Cs areal concentration. Thus it cannot be concluded that the ^{137}Cs distribution reflects only soil transport. Consequently, hypothesis *(2) The distribution of ^{137}Cs at the small (1 ha) scale is associated with soil movement alone* is also rejected at this site. At least part of the ^{137}Cs distribution could result directly from the pattern generated by fallout and immediate runoff. There appeared to be no discernible difference between the variability of ^{137}Cs in forested or pastureland sites, and the highest and the lowest variabilities occurred within forested areas. Consequently, hypothesis *(3) Variability of ^{137}Cs within tree covered areas will be higher than in pasture* is rejected. Finally, analysis of the inventories of ^{210}Pb$_{ex}$ and ^{137}Cs in forested soils at St Helens revealed that there was less variability in the ratio between them than for either nuclide alone. This approach may be a more robust indicator of landscape condition than measurements of ^{137}Cs in isolation. This finding is further explored in Chapter 3.

Chapter 3: Determining soil loss using the inventory ratio of $^{210}Pb_{ex}$ to ^{137}Cs

Chapter 3: Determining soil loss using the inventory ratio of $^{210}Pb_{ex}$ to ^{137}Cs

3.1 Perspective

In Chapter 2, the variability of the ratio between $^{210}Pb_{ex}$ and ^{137}Cs at St Helens was shown to be less than that for either radionuclide alone. This chapter explores the relationship between $^{210}Pb_{ex}$ and ^{137}Cs in more detail; in terms of the underlying processes responsible for their initial variability. A method for using the reduction in variability is also proposed, in which measurements of the ratio in disturbed areas are compared to the ratio as a function of depth in adjacent reference areas. In particular, measurements of ^{137}Cs and $^{210}Pb_{ex}$ are undertaken within 4 forest plots, of which two are controls and two have been harvested according to different practices, ie 'normal' and 'minimal' impact. Soil loss from the two harvested plots is first estimated using measurements of ^{137}Cs and $^{210}Pb_{ex}$ alone, and then compared to that determined from the ratio of the two.

Hypotheses tested in this chapter are, that: *(4) The ratio of the soil inventories of* $^{210}Pb_{ex}$ *and* ^{137}Cs *has a lower overall variability in 'reference' areas than the areal concentration of either nuclide alone; (5) The ratio of* $^{210}Pb_{ex}$ *to* ^{137}Cs *in forested reference areas is different from those in harvested areas;* and *(6) Soil loss from a 'normal' impact logged plot will exceed that from a 'minimal' impact logged plot.*

This chapter has appeared as: Wallbrink, P.J. and Murray, A.S., Determining soil loss using the inventory ratio of excess Lead-210 to Cesium-137, *Soil. Sci. Soc. Am. J.* 60, 1201-1208, 1996.

3.2 Introduction

3.2.1 Measuring soil loss

Redistribution of fallout ^{137}Cs has been widely used to determine patterns and rates of soil loss and sediment accumulation (Longmore *et al.*, 1983; Kachanoski, 1987; Sutherland and de Jong, 1990; Loughran *et al.*, 1990). Generally, an input or reference value of ^{137}Cs is determined and compared with soil inventories of ^{137}Cs in erosion or deposition sites; a review of this technique can be found in Ritchie and McHenry (1990). There are several methods available for calculating soil loss rates from ^{137}Cs measurements and these include: empirically derived relationships between the reduction in ^{137}Cs activity and soil loss such as those developed by Elliott *et al.* (1990) and Ritchie and McHenry (1975); theoretical models such as the proportional method of Kachanoski (1987); the gravimetric method (Brown *et al.*, 1981); profile distribution models (Zhang *et al.*, 1990); and mass balance models (Fredericks and Perrens, 1988). A review of these is given in Walling and Quine (1990). With the exception of Zhang *et al.* (1990) and Elliott *et al.* (1990) these methods have been developed for use on cultivated soils, and consequently assume mixing of ^{137}Cs at least to the depth of the plough layer. Generally they are intended to be used to calculate rates of loss in terms of an annual average. All these approaches rely on the assumption that the initial fallout deposition is uniform and that there is no immediate redistribution before the ^{137}Cs is trapped by soil particles. However, various authors (eg. Wallbrink *et al.*, 1994; Sutherland, 1994) have shown that the initial distribution of ^{137}Cs may not be uniform, indeed the variability can be large, up to 40 %. Many samples then need to be analysed to reduce the uncertainties on reference values, (Fredericks *et al.*, 1988;

Sutherland, 1991) and despite this it is possible or even probable that the input value will not be appropriate to the point values from suspected eroded or deposition sites.

3.2.2 Causes of variability in ^{137}Cs and $^{210}Pb_{ex}$

It is likely that the causes of variability in the ^{137}Cs areal concentrations similarly affect the initial areal concentration of other fallout radionuclides, such as ^{7}Be and ^{210}Pb$_{ex}$. Fallout of ^{137}Cs in the southern hemisphere essentially stopped in the mid 1970's (McCallan *et al.*, 1980; Longmore *et al.*, 1983) and since this time about half of the present soil inventory of fallout ^{210}Pb (half life 22 yr) has been precipitated. Nevertheless, both ^{137}Cs and fallout ^{210}Pb are deposited mainly in rainfall, and so will be similarly affected by micro-climate variations, such as rain-shadowing, and by differences in soil permeability. Thus it is possible that some initial correlation between the areal concentrations of these two fallout nuclides exists. Megumi *et al.* (1985) report a relationship between concentrations of these two in surface soils. Subsequent redistribution by soil movement may or may not affect the nuclides similarly, depending on their relative distribution with respect to depth and particle size. The degree to which these characteristics have changed over the last 40 years or so will also affect the correlation.

3.2.3 Aims of this work

In this chapter, soil losses from different logging treatments are measured using fallout radionuclides. These losses are first estimated by comparing ^{137}Cs and ^{210}Pb$_{ex}$ inventories with their measured depth distribution in reference profiles. Evidence is then presented of a good spatial correlation between areal concentrations of these two nuclides. This observation is then used to develop a new quantitative method for determining soil loss. The method is based on comparing areal averages of fallout $^{210}Pb/^{137}Cs$ inventory ratios from the logged sites with the activity ratio curve derived from depth dependent measurements of these two nuclides at two 'control' locations.

3.3 Site description

The St Helens State Forest (latitude 42°S, longitude 148°E, Figure 3.3.1) consists predominantly of *Eucalyptus seiberi.*

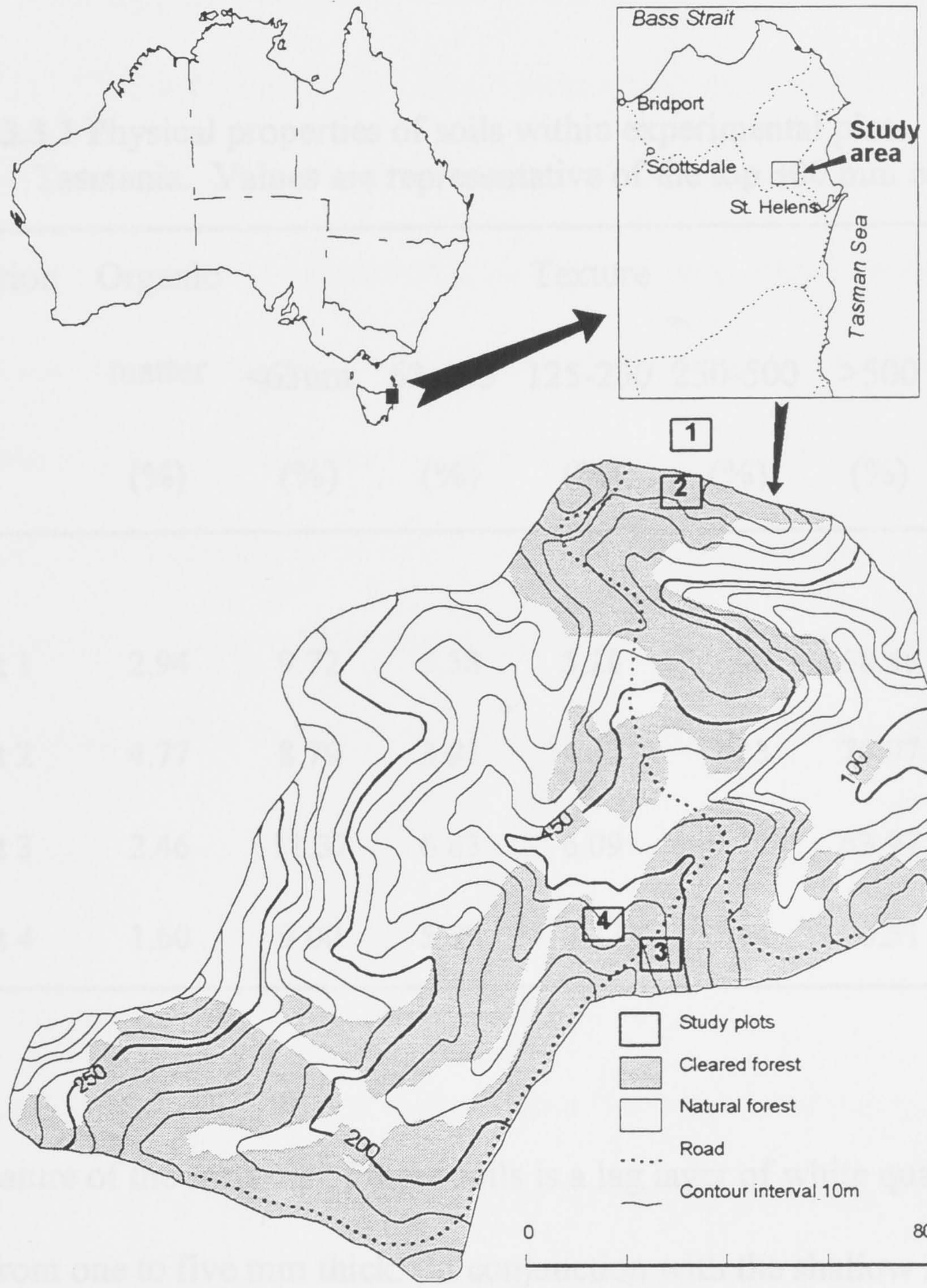

Figure 3.3.1 Location diagram of St Helens field study area, Tasmania, showing relative positions of experimental plots within the St Helens State Forest.

As outlined in Section 2.3.3, there is little shrub understorey because of frequent burning. The lithology is mainly granite, which is porphyritic, equigranular and coarse grained, with outcrops occurring on ridgetops and some slopes. There is a small component of biotite and a major component of muscovite adamellite. The soils are yellow podzolics and have shallow (2-15 cm) sandy A-horizons which overlay yellow gravely sandy/loams. They are classified according to the Australian soil taxonomy classification system as Uc2.21 (Northcote, 1979), this falls within the U.S. soil taxonomy suborder Aqualf. Some basic properties of the soil are listed in Table 3.3.1.

Table 3.3.1 Physical properties of soils within experimental plots, at St Helens, Tasmania. Values are representative of the top 300 mm only.

Location	Organic matter (%)	Texture <63um (%)	63-125 (%)	125-250 (%)	250-500 (%)	>500 (%)	Bulk density (g cm^{-3})
Plot 1	2.94	9.72	5.58	5.28	7.98	68.48	1.52
Plot 2	4.77	8.79	3.91	4.32	7.23	70.97	1.46
Plot 3	2.46	11.37	6.83	6.09	9.27	63.97	1.53
Plot 4	1.60	9.00	5.99	5.43	7.44	70.51	1.31

A major feature of the surfaces of these soils is a lag layer of white quartz grains of thickness from one to five mm thick. In conjunction with the shallow A horizons this suggests that natural erosion may be significant. All the plots had slopes less than 15°.

Annual rainfall is ~650 mm yr^{-1}, with a winter predominance, however high intensity storms (up to 200 mm hr^{-1}) may occur over the summer months.

3.4 Materials and methods

3.4.1 Logging treatments for each plot

In March and April 1991 the Tasmanian Forestry Commission logged parts of the St Helens forest as part of an experiment to examine erosion under different harvesting conditions. After logging operations ceased, four 350 m^2 plots were cordoned off (Figure 3.3.1). The unlogged Plots 1 and 4 were adjacent to each of the harvested areas and left as control sites. Plot 1 was just outside the catchment boundary, however it contained similar soils and vegetation type to the others. Plot 2 (normal impact) represented standard logging practice and had crown and stem removal from the site and soil banks to 60 cm high pushed up by bulldozers to reduce water flow over the disturbed soil. Plot 3 (minimal impact) was harvested less intensively and the crowns and waste were left on site, vehicle movement was kept to a minimum and no soil banks were installed. It was anticipated that soil loss from Plot 2 would exceed that of Plot 3.

3.4.2 Radionuclide concentration of soils within study plots

The radionuclide concentrations of the soil from the logged and unlogged plots were determined by dividing each plot into roughly 10 x 35 m^2 grid cells. Representative soil cores were then taken randomly from within each cell to a total of 9 or 10 cores per plot although obvious features such as trees and rock outcrops were avoided. The distance between cores was about 5 - 10 m and the coring tubes (diameter 100 mm, depth 300 mm) each contained approximately 2.5 kg of soil. Sampling occurred about 14 months after logging.

Detailed depth sampling was undertaken at the two 'control' sites to determine the initial distribution of the two nuclides. These profiles were of known area and acquired using the method described in Wallbrink and Murray (1996a). Briefly, a high speed router with a 25 mm diameter tungsten carbide bit was used to obtain samples from an area of 0.16 m^2, in depth increments of as little as one mm, down to 30 mm. Below this, the sample area was reduced to 0.04 m^2, and a scraper used to obtain depth increments of 10 mm down to 300 mm.

3.4.3 Analytical procedure

All samples were oven dried, ashed at 400^0 C, and analysed by gamma spectrometry for ^{137}Cs, ^{226}Ra and ^{210}Pb as described by Murray *et al.* (1987). They were counted for a minimum of 85 ksec. ^{210}Pb is generated from the decay of Radon-222 (^{222}Rn) both within the soil, and in the atmosphere and a review of its genesis is given in Wise (1980). Generally the fallout ^{210}Pb concentration, ($^{210}Pb_{ex}$) is calculated as the difference between the ^{210}Pb concentration and ^{226}Ra concentration in the soil (Appleby and Oldfield, 1978), and no allowance is made for the escape of ^{222}Rn gas. At St Helens the ^{210}Pb activity concentrations become a constant fraction of ^{226}Ra at depth (about 95%) ie. there is about a 5 % loss of ^{222}Rn activity to the atmosphere. Consequently, our values of $^{210}Pb_{ex}$ are calculated by subtracting 95% of the observed ^{226}Ra at each depth from the appropriate total ^{210}Pb concentration. Unless otherwise noted, all mean values are reported plus or minus one standard error.

3.5 Results

3.5.1 Radionuclide concentrations of soil cores from Plots 1, 2, 3, and 4

The radionuclide areal concentrations of the soil cores from all the plots are summarised in Table 3.5.1.1. The concentrations of $^{210}Pb_{ex}$ and ^{137}Cs measured in the detailed profiles from Plots 1 and 4 are shown as a function of depth in Figures 3.5.1.1a, b and 3.5.1.2a, b, respectively. The cumulative inventories of each nuclide have been calculated from depth by summing the areal concentrations within each increment towards the surface (Figures 3.5.1.1c and 3.5.1.2c; for clarity only the top 150 mm are shown). The deeper penetration of ^{137}Cs is clearly visible.

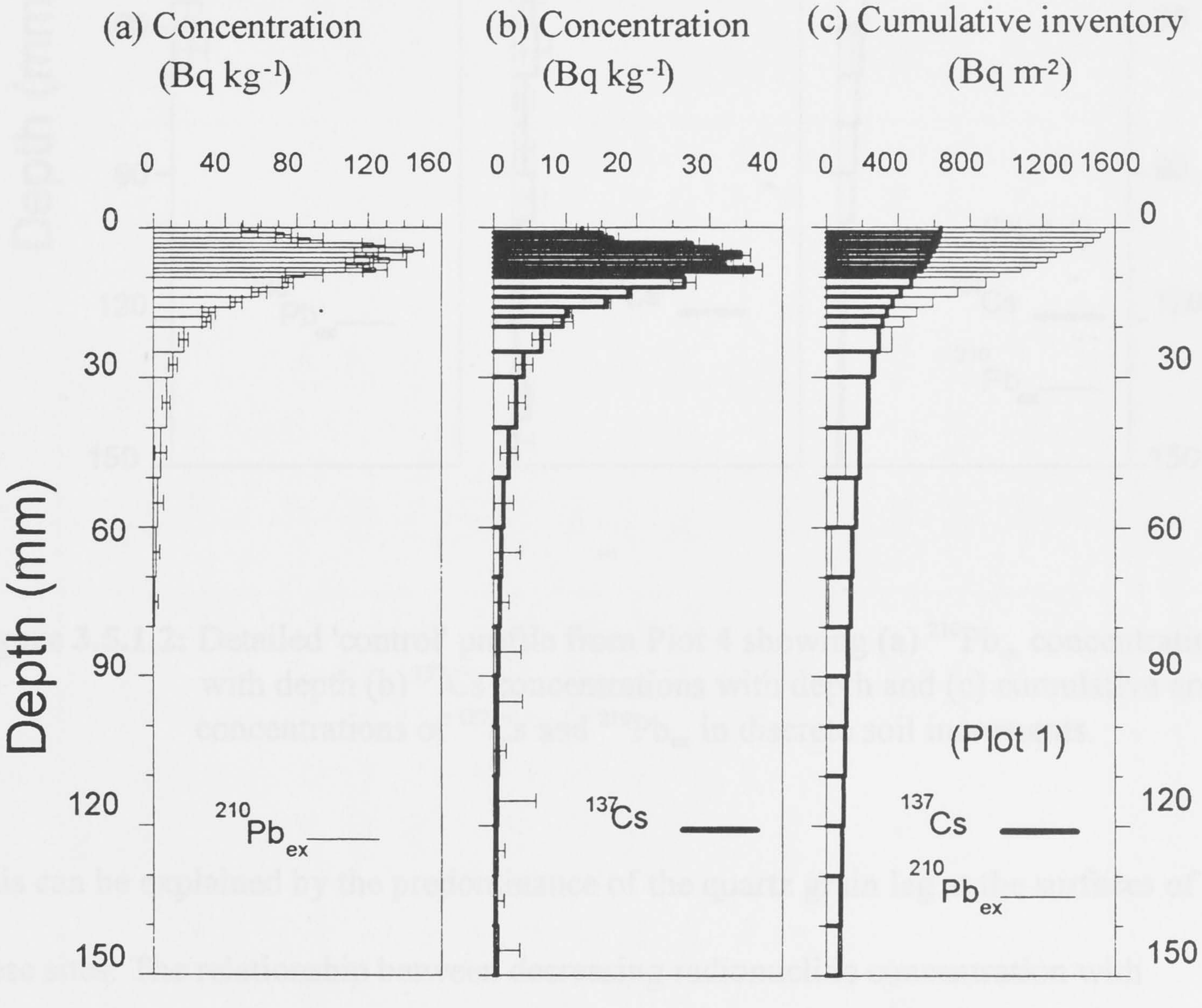

Figure 3.5.1.1: Detailed 'control' profile from Plot 1 showing (a) $^{210}Pb_{ex}$ concentrations with depth (b) ^{137}Cs concentrations and (c) cumulative areal concentrations of ^{137}Cs and $^{210}Pb_{ex}$ in discrete soil increments. Note: i) uncertainties are analytical only, ii) cumulative inventory for ^{137}Cs starts at 300 mm.

The subsurface maximum of ^{137}Cs is not unexpected and has been observed elsewhere (Walling and Woodward, 1992). The ^{210}Pb$_{ex}$ also has a subsurface peak at about 5 mm in these soils (Figures 3.5.1.1a and 3.5.1.2a).

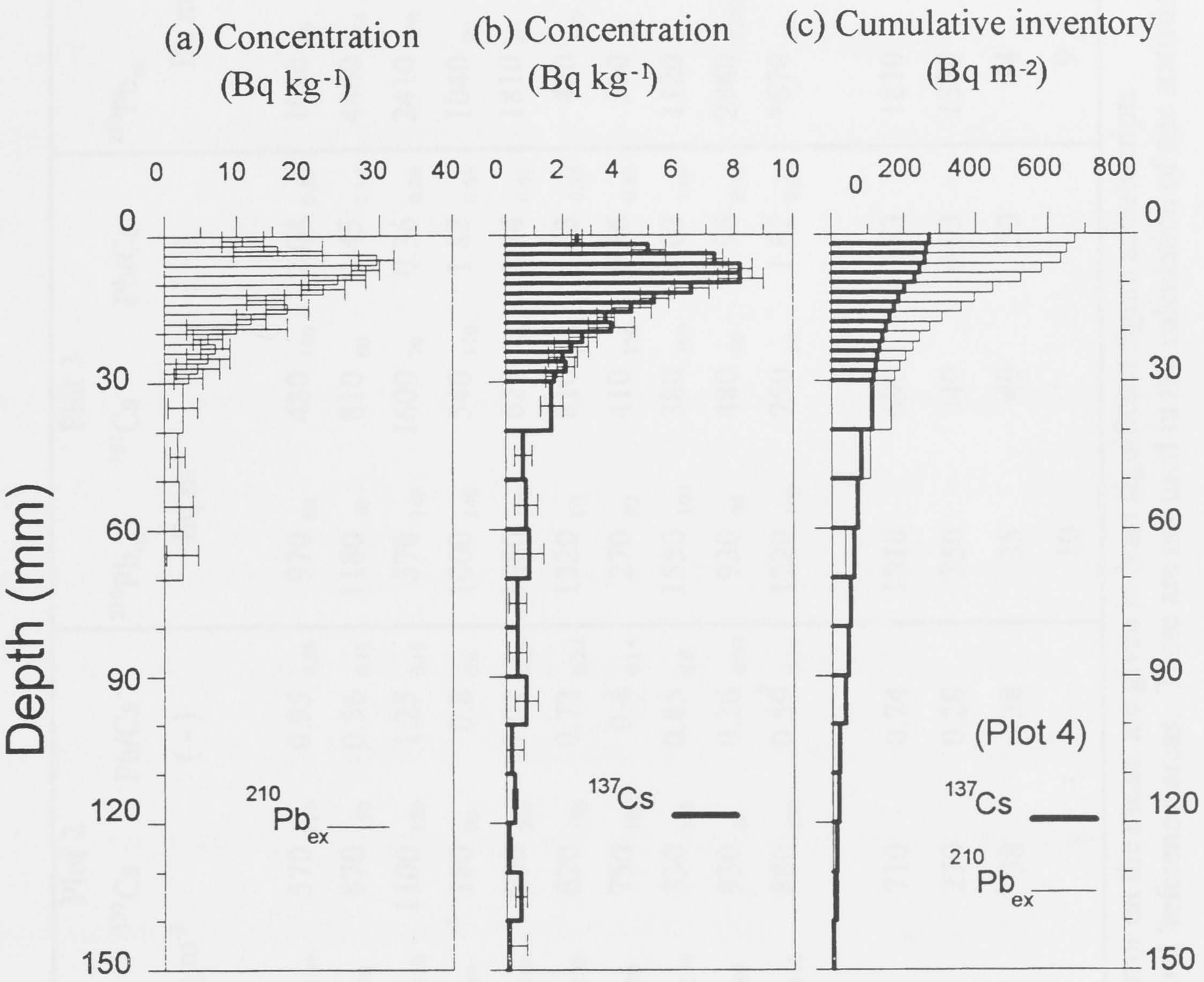

Figure 3.5.1.2: Detailed 'control' profile from Plot 4 showing (a) ^{210}Pb$_{ex}$ concentrations with depth (b) ^{137}Cs concentrations with depth and (c) cumulative areal concentrations of ^{137}Cs and ^{210}Pb$_{ex}$ in discrete soil increments.

This can be explained by the predominance of the quartz grain lag at the surfaces of these sites. The relationship between decreasing radionuclide concentration with increasing particle size has been well documented in granite soils (Olley *et al.*, 1996). Further evidence for this particle size dependence at St Helens is given in Wallbrink and Murray (1996a).

Table 3.5.1.1 Radionuclide concentrations and ratios of soil cores from St Helens, Tasmania

Core	Plot 1			Plot 2			Plot 3			Plot 4		
	$^{210}Pb_{ex}$	^{137}Cs	Pb/Cs	$^{210}Pb_{ex}$	^{137}Cs	Pb/Cs	$^{210}Pb_{ex}$	^{137}Cs	Pb/Cs	$^{210}Pb_{ex}$	^{137}Cs	Pb/Cs
	Bq m^{-2}		(-)	Bq m^{-2}		(-)	Bq m^{-2}		(-)	Bq m^{-2}		(-)
1	2040_{90}	650_{120}	$3.17_{0.62}$	540_{100}	570_{110}	$0.95_{0.25}$	970_{80}	480_{130}	$2.04_{0.58}$	1030_{80}	440_{90}	$2.33_{0.51}$
2	1360_{80}	670_{45}	$2.02_{0.18}$	490_{70}	870_{150}	$0.56_{0.13}$	1180_{70}	810_{60}	$1.45_{0.14}$	4860_{760}	1970_{100}	$2.47_{0.41}$
3	1170_{900}	600_{130}	$1.95_{0.61}$	1380_{120}	1100_{120}	$1.25_{0.18}$	570_{140}	1600_{70}	$0.36_{0.09}$	2410_{870}	770_{70}	$3.15_{1.17}$
4	1900_{100}	1230_{100}	$1.54_{0.16}$	140_{80}	180_{90}	$0.8_{0.6}$	1000_{140}	540_{170}	$1.83_{0.63}$	1040_{90}	610_{100}	$1.7_{0.31}$
5	1290_{70}	530_{40}	$2.44_{0.24}$	900_{100}	900_{200}	$1.01_{0.25}$	1080_{100}	620_{130}	$1.74_{0.41}$	1810_{80}	1010_{90}	$1.79_{0.17}$
6	3070_{900}	940_{70}	$3.26_{0.36}$	450_{110}	620_{130}	$0.72_{0.24}$	1320_{80}	810_{120}	$1.63_{0.25}$	480_{100}	370_{50}	$1.28_{0.33}$
7	2180_{100}	910_{90}	$2.4_{0.26}$	300_{70}	750_{190}	$0.4_{0.14}$	270_{90}	410_{150}	$0.66_{0.33}$	580_{70}	390_{90}	$1.47_{0.39}$
8	2430_{80}	830_{140}	$2.94_{0.53}$	280_{140}	330_{160}	$0.85_{0.6}$	1550_{100}	380_{190}	$4.03_{1.03}$	1120_{70}	520_{50}	$2.13_{0.24}$
9	1070_{100}	850_{130}	$1.25_{0.23}$	240_{80}	930_{80}	$0.26_{0.09}$	930_{90}	480_{90}	$1.96_{0.44}$	2940_{1280}	1270_{110}	$2.31_{1.03}$
10	$*1540_{80}$	$*640_{30}$	$*2.42_{0.16}$	520_{110}	880_{190}	$0.59_{0.18}$	1220_{100}	750_{90}	$1.62_{0.24}$	$*670_{20}$	$*270_{10}$	$*2.48_{0.1}$
Mean	1830	800	2.33	520	710	0.74	1010	690	1.73	1810	820	2.1
std.	630	200	0.66	350	270	0.28	350	340	0.93	1330	500	0.54
rsd	34	25	28	66	38	38	35	49	53	74	61	26
n =	9			10			10			9		

Note: Uncertainties, equivalent to one standard error on the mean, are given in least significant figure as subscripts

* Values from detailed profiles taken within control 'reference' areas. These are not used in the calculation of site statistics

3.5.2 Discrimination between forest logging treatments using ^{137}Cs

The methods available for relating ^{137}Cs measurements to soil loss given in Section 3.2.1 are not readily applicable to this study, because loss was essentially rapid (most of it presumably occurred over a period of less than 14 months), the soil was uncultivated and the nuclide is concentrated toward the soil surface. Nonetheless, it is possible to compare ^{137}Cs areal concentrations from the logged plots with those of the unlogged plots, and thus investigate the potential sensitivity of ^{137}Cs inventories in this area. The two unlogged plots (1 and 4) are statistically indistinguishable (using $p < 0.05$, one tailed test) and therefore the reference value can be taken as the average of all these cores, 810 ± 90 Bq m^{-2} (n=18, Relative Standard Deviation, *rsd*, 47%). This can be compared to the average areal concentrations of 710 ± 90 (n=10; *rsd* =38%) and 690 ± 110 Bq m^{-2} (n=10; *rsd* =49%) at Plots 2 and 3 respectively, giving relative ^{137}Cs depletions of 12±16 and 15±18 %. These can be converted to depth losses, of 6 ± 8 and 7 ± 8 mm respectively, by comparing them with the known depth distributions of ^{137}Cs areal concentration at their respective control sites, Plot 1 and 4 (Figure 3.5.1.1c and 3.5.1.2c). However, there is no statistical difference in ^{137}Cs areal concentrations, (using $p < 0.05$, one tailed test) between the unlogged control plots and either of these two logged plots. There is also no statistical difference between the two logged plots themselves (using $p < 0.05$, one tailed test). It is concluded that ^{137}Cs areal concentrations cannot be used to discriminate between the effects of the different plot treatments.

3.5.3 Discrimination between forest logging treatments using $^{210}Pb_{ex}$

It was thought that ^{210}Pb$_{ex}$ may be a more sensitive discriminator of surface erosion than ^{137}Cs within the upper sections of soils. This is because of its greater rate of change with depth over the top few centimetres, partially resulting from its constant fallout in rainfall and strong particle reactivity (Nozaki *et al.*, 1978; Matthews and Potipin, 1985). Therefore a similar analysis to that described above for ^{137}Cs was undertaken using ^{210}Pb$_{ex}$ areal concentrations. The control plots were again indistinguishable ($p < 0.05$, one tailed test) and the combined average areal concentration of ^{210}Pb$_{ex}$ from these was 1,820 ± 240 Bq m^{-2} (n=18; *rsd* =57%). This can be compared to the average inventory values of 520 ± 110 (n=10; *rsd* =66%) and 1010 ± 110 Bq m^{-2} (n=10; *rsd* =35%) from Plots 2 and 3. The logged plots 2 and 3 were statistically distinguishable both from one another and from the combined control plot value (using $p < 0.05$, one tailed test) and their inventory values represent reductions of 71 ± 15 % and 45 ± 15 % respectively. Clearly Plot 2 has lost a large fraction of the initial ^{210}Pb$_{ex}$ inventory, and Plot 3 has lost a smaller fraction. Comparing these inventory depletions with their respective reference depth profiles at control Plots 1 and 4, (Figures 3.5.1.1c and 3.5.1.2c) gives depth losses of ~20 ± 4 mm for Plot 2 and 11 ± 4 mm for Plot 3. Nevertheless, it is important to note that the inventory values from the two detailed profiles at Plots 1 and 4 (each taken over an area of at least 0.04 m^2) differ by a factor of about 2. This emphasises the possible error in assuming that the control inventories apply to the logged plots.

3.5.4 A new approach using $^{210}Pb_{ex}/^{137}Cs$ ratios

It was suggested in the introduction that areal concentrations of ^{137}Cs and ^{210}Pb$_{ex}$ may be related, and a clear correlation is observed between them in the total core inventories from the control Plots 1 and 4 (Figure 3.5.4.1). This correlation explains the overall reduction in variability in the inventory ratio ^{210}Pb$_{ex}$/^{137}Cs compared with that of the individual inventories at the control plots (Table 3.5.1.1). Thus an alternative approach is proposed here which is based on the additional observation that in undisturbed soils ^{137}Cs and ^{210}Pb$_{ex}$ also have different penetrations into the surface layers of soil.

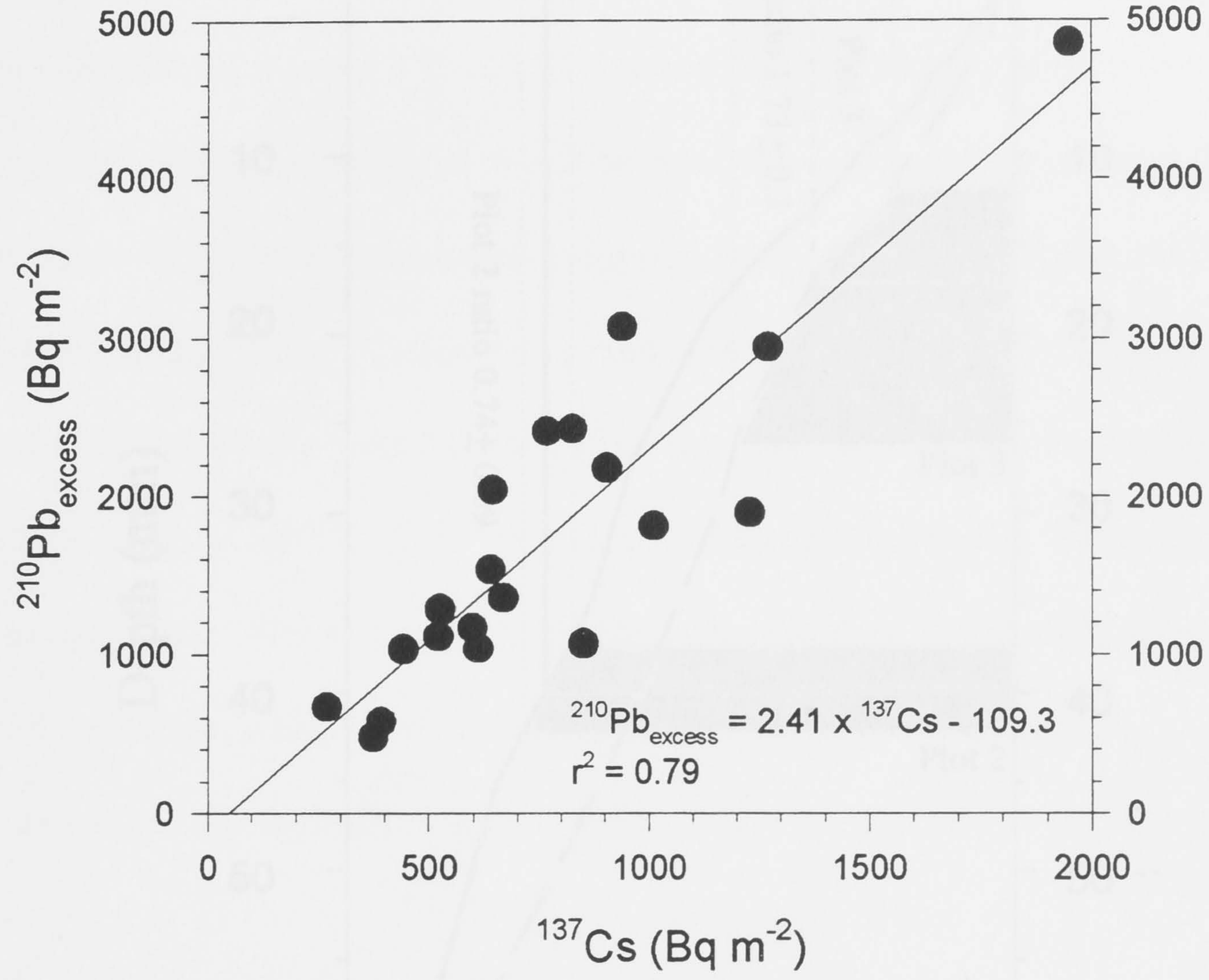

Figure 3.5.4.1: Relationship between ^{210}Pb$_{ex}$ and ^{137}Cs inventories from cores taken within unlogged 'control' Plots 1 and 4, St Helens, Tasmania.

The bulk of ^{210}Pb$_{ex}$ activity is retained closer to the surface than ^{137}Cs (Figures 3.5.1.1a and 3.5.1.2a). Thus the ratio of the inventory of ^{210}Pb$_{ex}$ (below a particular depth in the soil profile) to the inventory of ^{137}Cs below the same depth is unique. This ratio has finite values, and thus acts as a useful indicator, over the range of penetration of ^{210}Pb$_{ex}$, ie. over a few centimetres. The ratio of these inventories from the profiles at Plots 1 and 4 decrease monotonically with depth and provides unique values down to ~80 mm (Figure 3.5.4.2). The curves for the two profiles are similar, given the large differences in absolute areal concentrations between them (Figure 3.5.1.1c and 3.5.1.2c).

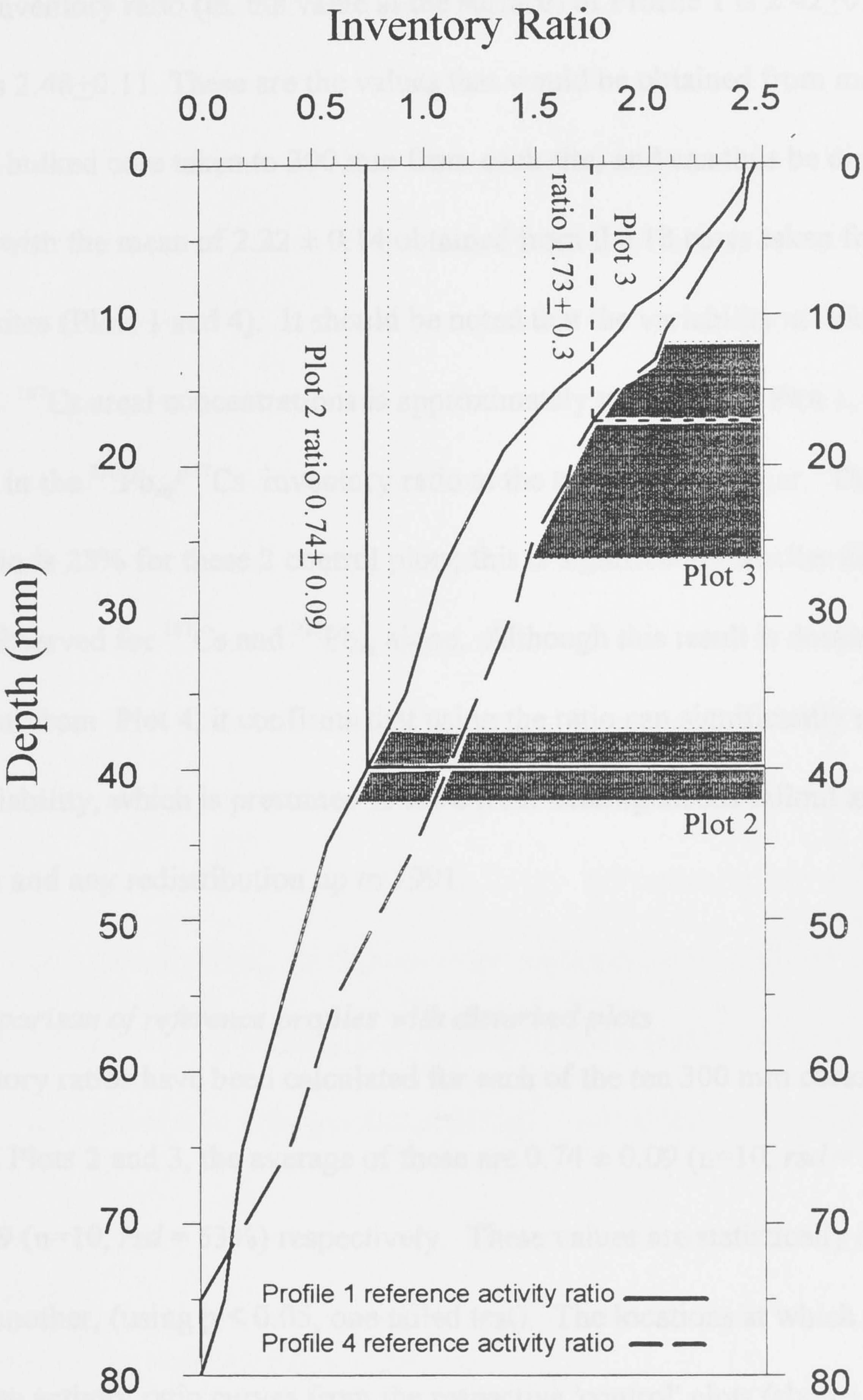

Figure 3.5.4.2 Activity ratio with depth from cumulative ^{137}Cs and ^{210}Pb$_{ex}$ areal concentrations at 'control' profiles, Plot 1 and 4. Graph shows intersection of Plot 2 and 3 average inventory ratio values with respective 'control' activity ratio curves and estimated depths of soil removal.

The total inventory ratio (ie. the value at the surface) of Profile 1 is 2.42±0.16 and of Profile 4 is 2.48±0.11. These are the values that would be obtained from measurements of a single bulked core taken to 300 mm from each site, and can thus be directly compared with the mean of 2.22 ± 0.14 obtained from the 18 cores taken from the 2 unlogged sites (Plots 1 and 4). It should be noted that the variability at Plot 4 in both $^{210}Pb_{ex}$ and ^{137}Cs areal concentrations is approximately twice that at Plot 1, although the variability in the $^{210}Pb_{ex}/^{137}Cs$ inventory ratio at the two sites is similar. The overall *rsd* for this ratio is 28% for these 2 control plots; this is significantly smaller than the 47% and 57% observed for ^{137}Cs and $^{210}Pb_{ex}$ alone. Although this result is dominated by the contribution from Plot 4, it confirms that using the ratio can significantly reduce the spatial variability, which is presumed to arise from heterogeneous fallout and infiltration and any redistribution up to 1991.

3.5.5 Comparison of reference profiles with disturbed plots

The inventory ratios have been calculated for each of the ten 300 mm cores taken from the logged Plots 2 and 3, the average of these are 0.74 ± 0.09 (n=10, *rsd* = 38%) and 1.73 ± 0.29 (n=10, *rsd* = 53%) respectively. These values are statistically different from one another, (using $p < 0.05$, one tailed test). The locations at which these ratios intersect the activity ratio curves from the respective 'control' plots (shown in Figure 3.5.4.2) correspond to the depth of soil removal required to leave behind these inventory ratio values. Average soil loss is calculated to be about 40 ± 5 mm from the heavily logged Plot 2, and 17 ± 5 mm from the more carefully managed Plot 3. The average measured bulk density of these soils was 1.1 g cm^{-3} through the top 100 mm, and so these depths from Plot 2 and 3 correspond to net soil losses of 440 ± 55 and 190

± 55 t ha^{-1} respectively. Note that the uncertainties given are relative to the measurement errors on the initial ratio values only.

These estimates of soil loss are significantly higher than long term rates from logged slopes of about 0.2 t ha^{-1} yr^{-1} given in Megahan and Kidd (1972), although they are consistent with values for logging roads during use, of about 120 - 180 t ha^{-1} yr^{-1} given in Reid and Dunne (1984). They are also consistent with a one-off estimate of depth loss on deforested slopes of 41 mm due to storms associated with cyclone Hilda in New Zealand in 1990 (de Rose *et al.*, 1993). Various other estimates of loss within forests under a variety of conditions can be found (Roberts and Church, 1986). However these tend to be for northern hemisphere cool temperate climate conditions and are presented as annual figures and thus are difficult to compare with the short term losses discussed here. Possible discrepancies may in part arise from a difference in the experimental areas and time scales involved. In this study the plots were only ~350 m^2 and the results apply to the 14 months immediately post logging. Losses would therefore be expected to be larger than the annual averages derived from whole slopes reported elsewhere.

3.6 Discussion

3.6.1 Caveats on the radionuclide estimates of soil loss

Model predictions are no better than the model assumptions. In this case, it is assumed that erosion at the disturbed sites removes soil with nuclide labels described by the profiles at the undisturbed sites. This clearly cannot be exactly true, because the sites were disturbed, and this disturbance included some lateral transport of soil; although these criticisms apply to all techniques which use any form of 'reference' profile. Also any lateral movement of soil may involve the preferential removal of finer particles. As these fines carry more of the radionuclide label by mass than larger particles, their preferential loss may artificially increase the estimates of radionuclide depletion. It should also be noted that the material in the soil banks at Plot 2 were excavated from a smaller pit immediately upslope of these features and the possibility remains that some topsoil was also used. This may have left some areas between the soil banks depleted in ^{137}Cs and $^{210}Pb_{ex}$. Although the sampling process was intended to average out these effects, the calculations of loss should be regarded as upper estimates.

3.6.2 Evidence supporting the radionuclide estimates of soil loss

Ultimately the only reliable test of the approach is by independent measurement. This was partly achieved by measurements of soil loss in an adjacent plot of similar size and subject to the same intensive logging practice as Plot 2. In this plot, sediment was collected in a trough over a 2 month period immediately prior to the work described in this chapter. Assuming that the trough trap efficiency was close to 100% and that this 2 month period was representative of the 15 months since logging, the volume of collected sediment suggests a surface lowering of at least 30 mm over this latter period (J.C. Wilson, 1993, pers. comm.). This is broadly consistent with the radionuclide predictions. The estimates of soil loss by radionuclides and the collection trough are also consistent with visual observations at the time of sampling which found that the 2-15 cm thick dark organic-rich A horizon was still largely intact in Plot 3, whereas only remnants were visible in Plot 2. Mixing of the soil had also occurred in Plot 2, with areas of brown clay-rich B horizon exposed. These observations support the finding that Plot 3 retained more soil following harvesting than Plot 2.

3.6.3 Differential depth dependency of $^{210}Pb_{ex}$ and ^{137}Cs

The depth losses for Plots 2 and 3 respectively, determined by ^{137}Cs (6 ± 8 and 7 ± 8 mm) and ^{210}Pb$_{ex}$ (20 ± 6 and 11 ± 3 mm) were smaller than the values predicted from the ratio data (Figure 3.5.4.1) of 40 ± 6 mm (Plot 2) and 17 ± 5 mm (Plot 3). These discrepancies are thought to arise because of the uncertainties in the estimates of ^{137}Cs and ^{210}Pb$_{ex}$ inventories at the logged sites. Control inventories may be systematically different from those of the logged sites. In contrast, uncertainties arising from gross differences in initial inventories between locations do not affect the ratio technique; in which the controlling factors are the ratio between the inventories of the two nuclides and their rate of change with depth. These parameters appear to offer greater reliability in determining soil loss than either of the two nuclides alone. Further work is required however, to confirm that the ^{210}Pb$_{ex}$/^{137}Cs inventory ratio correlation with depth is widespread and that the relative penetration does not vary markedly across the landscape. The inventory ratio may also be useful in investigations of the rates of natural vertical soil mixing due to soil fauna, such as described by Lee (1985).

3.6.4 Causes of variability

In Chapter 2 it was demonstrated that ^{137}Cs variability in uneroded 'control' sites can be as much as 40%. It was speculated that variability arising from initial deposition may be due to a number of factors including: rainshadowing, interception, lateral throughflow and differential run-on / run-off resulting from soil hydrophobicity and compaction. The variability of areal fallout activity concentrations at the 'control' sites reported here was considerable (^{137}Cs: *rsd* = 47 %; ^{210}Pb$_{ex}$:*rsd* = 57 %). It is thought that this variability within the reference areas at St Helens may have two origins (i) variability resulting from initial deposition processes (such as described above) and (ii) long term post depositional redistribution with soil particle movement. There is evidence from this study that the first of these factors may be more important (see below).

3.6.5 The influence of sample surface area on spatial variability

It is surprising that the inventory ratios for the two detailed profiles at Plots 1 and 4 agree so well given that the *rsd* for Plots 1 and 4 is ~27%. However, this latter value is for 18 cores that each have a surface area of about 0.008 m^2. The detailed profiles have a surface area of 0.16 m^2, down to 30 mm, and 0.04 m^2 below that. The top 30 mm contains ~50% of the ^{137}Cs and 80 % of the $^{210}Pb_{ex}$. Now, if the spatial frequency of the variability in the ratio was high, for instance the same as the diameter of a 10 cm core, then the detailed profiles (which contain the areal equivalent of 20 cores, at least down to 30 mm), would have a smaller variability than individual cores. The uncertainty arising from spatial variability in the inventory ratio value for the detailed profiles would then be reduced to $(27\ \% / \sqrt{20}) = 6\%$. It may be reduced by more than this if the variability arises from local redistribution of the top few millimetres of soil over lateral distances of only a few centimetres. In this case, the scatter would not be random, and might be almost entirely removed by sampling over a scale that is large compared with the redistribution distance. Whether or not this is this case, this averaging clearly has not removed the variability in the concentration data, as the Plot 1 profile has total inventories of ^{137}Cs and $^{210}Pb_{ex}$ approximately 2.5 times greater than those at Plot 4. This suggests that the origin of this variability may lie in initial fallout distribution rather than significant subsequent lateral transport.

3.6.6 Possible influence of vegetation on spatial variability

The variability of ^{137}Cs and ^{210}Pb$_{ex}$ was greater in Plot 4 than Plot 1 (Table 3.5.1.1). This may be due to differences in their slope position. Plot 1 was approximately midslope and Plot 4 was closer to the slope bottom and there were slight soil moisture differences between them. This was evident in the more profuse vegetation at Plot 4, although the overstorey species were the same. Furthermore, Plot 1 had undergone more frequent litter reduction burns; there was little or no surface litter and the understorey was relatively sparse. All of these factors would result in a greater interception of fallout radionuclides at Plot 4, potentially leading to a more heterogeneous distribution.

3.6.7 Other factors affecting radionuclide variability

The decrease in variability in the inventory ratio at the 'control' sites (*rsd* = 28%) when compared with ^{210}Pb$_{ex}$ (*rsd* = 57%) and ^{137}Cs alone (*rsd* = 47 %) also deserves consideration. Both nuclides are deposited primarily by rainfall, and so if ^{137}Cs and ^{210}Pb were deposited at the same time the variability in the ratio would presumably be very small. However, half the ^{210}Pb$_{ex}$ inventory has been deposited after fallout of ^{137}Cs effectively stopped. Furthermore, because the two nuclides have different penetration profiles, the ratio in material eroded from the surface few millimetres is markedly different from the total inventory ratio. Thus local redistribution of sediment within the control site should greatly increase the variability in the ratio until ultimately the 'memory' of initial condition would be lost. In addition, changes in the micro-climate and micro-topography at the site since the 1960's would again reduce the degree of correlation expected between the two nuclides. However, a significant reduction in the *rsd* is observed between the ^{210}Pb$_{ex}$ and ^{137}Cs absolute concentrations and the ratio value, and a correlation persists to the present (see Figure 3.5.4.1). Therefore it must be concluded that the variability in the concentration data mainly results from deposition and initial redistribution (from mechanisms described earlier) prior to particle attachment, rather than particle redistribution over the 10-20 years prior to sampling. This also implies that these factors retained some constancy over this period.

3.6.8 Potential for further applications

The ability of $^{210}Pb_{ex}/^{137}Cs$ ratios to reduce variability in 'control' sites should be independent of differences in penetration depth. Therefore this approach for measuring erosion depth has the potential to be used in arable (ploughed) areas if the continuous fallout of ^{210}Pb and the ongoing loss of particulates due to the ploughing process, can be accounted for.

The method has been shown here to be useful for examining erosion following a single disturbance event which occurred in a short time period compared with the ^{210}Pb half life. This allows a direct comparison with the 'control' sites, because no significant fallout of ^{210}Pb had taken place after the disturbing event. However, the approach is capable of extension to continuous erosion processes, if the steady fallout of ^{210}Pb is taken into account. It is also worth noting that ^{7}Be (half life 53 days) is also found preferentially very near the surface (Wallbrink and Murray, 1993, 1996a - see Chapter 4) and so a similar approach using ^{7}Be to ^{137}Cs or ^{7}Be to $^{210}Pb_{ex}$ ratios could be applicable even on annually ploughed land.

3.7 Conclusions

Areal concentrations of ^{137}Cs were not able to distinguish either the control plots from the differently logged plots, or these logged plots from one another. On the other hand it was possible to discriminate, and then quantify, the effect of different plot treatments using areal concentrations of ^{210}Pb$_{ex}$. However, there were differences in the control site areal concentration values such that it could not be certain that the control inventories applied to the logged sites. It has been demonstrated that areal concentrations of the two nuclides are spatially correlated. Consequently, taking the ratio of these two nuclides removes some of this variability in areal concentrations, which is assumed to arise from deposition and initial redistribution processes. Furthermore, most of the ^{210}Pb$_{ex}$ activity was retained closer to the soil surface than ^{137}Cs and the areal concentration activity ratio between them decreases monotonically with depth. This inventory depth dependence is very similar at the two 'control' sites, despite more than a factor of 2 difference in areal concentrations. Thus, differences in the inventory ^{210}Pb$_{ex}$/^{137}Cs ratio between the sites could be quantified in terms of depth loss. The advantage of this ratio over using ^{210}Pb$_{ex}$ or ^{137}Cs areal concentrations alone is that the differences in ratio values between sites are less affected by possible differences in total fallout. It is concluded that this ratio is a sensitive technique for retrospectively comparing soil losses between sites following recent disturbance in forested or pastured sites. Indeed it may also have the potential to be applied in any land use where the soil surface has been otherwise undisturbed over the last one or two decades.

3.8 Summary and consideration of hypotheses

The anthropogenic nuclide ^{137}Cs is widely used to measure soil movement. However changes in ^{137}Cs areal concentrations may not result from soil redistribution alone. There can be considerable variability in ^{137}Cs fallout due to rain shadowing and small-scale runon-runoff processes at the time of deposition. Fallout ^{210}Pb is also affected by these processes and thus areal concentrations of the two are correlated. It has been shown that the ratio of fallout ^{210}Pb to ^{137}Cs reduces variability by up to a factor of two compared with concentrations alone, in undisturbed 'control' forest sites at St Helens, Tasmania. Therefore, hypothesis *(4) The ratio of the soil inventories of* $^{210}Pb_{ex}$ *and* ^{137}Cs *has a lower overall variability in 'reference' areas than the areal concentrations of either nuclide alone* is not rejected. In addition, these two nuclides penetrate to different depths, producing a varying activity ratio with depth, and this gave rise to a new method for determining soil loss. The average inventory ratios of fallout ^{210}Pb to ^{137}Cs from plots adjacent to the 'controls' were measured; these had undergone 'normal' and 'minimal impact' logging procedures. There was a clear difference between the average core inventory ratio at the 'control' reference locations (2.24 ± 0.14; n=18), compared to the 'normal' impact site (0.74 ± 0.09; n=10) and 'minimal impact' site (1.73 ± 0.29; n=10). Thus hypothesis *(5), The ratio of* $^{210}Pb_{ex}$ *to* ^{137}Cs *in forested reference areas is different from those in harvested areas,* is also not rejected at this site. The average depth of soil removed from the logged sites was calculated by comparing these ratios with the inventory activity ratio curve from their respective 'control' sites. This gave, 40 ± 6 mm, removal for the 'normal' site and 17 ± 5 mm removal for the 'minimal' site. Consequently, hypothesis *(6) Soil loss from a 'normal' impact logged plot will exceed that from a 'minimal' impact logged plot,* is not rejected. It is concluded that ratios of fallout nuclides may be able to provide a more sensitive method for investigating surface erosion than areal concentrations alone, particularly in landscapes where vertical soil mixing has not recently occurred.

Theme 2:

Tracing and quantifying the sources of suspended sediments

Chapter 4: Determining erosion processes using fallout nuclides

Chapter 4: Determining erosion processes using fallout nuclides

4.1 Perspective:

In Chapters 2 and 3, a method for measuring soil loss was developed which incorporated the measured spatial variability of ^{137}Cs and ^{210}Pb$_{ex}$. However, much of the impact of soil erosion occurs away from the point of initial soil particle detachment. Soil material can become entrained in runoff waters and then potentially transported large distances from the source. Thus, it is often not known from which location, or from what erosion process, sediment material is derived. This chapter investigates the potential for differences in the depth distributions of ^{7}Be, ^{210}Pb$_{ex}$, and ^{137}Cs to reveal the depth origins of, and erosion processes responsible for, material removed from i) the surface of a degraded pastureland; and ii) a gully wall and its upslope contributing area. This work was undertaken in a small first order stream at Whiteheads Creek, which is in the headwaters of the Wollondilly River, NSW.

Hypotheses tested in this chapter are: *(7) The tracers ^{137}Cs, ^{210}Pb$_{ex}$, and ^{7}Be have different initial depth distributions in undisturbed soils; (8) Eroded material from soil surfaces will have high fallout radionuclide concentrations and subsoil derived material will have low fallout radionuclide concentrations* and; *(9) The net contribution from surface erosion will exceed that from gully erosion.*

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4.2 Introduction

4.2.1 Background.

In the absence of detailed measurements of sediment transport on ground surfaces and within various parts of channels, detailed conclusions about the erosion mechanism(s) responsible for transported sediment cannot be reached. Source tracers have the potential to supply land and water managers with the kind of detailed information they need, without an expensive investment in monitoring equipment. In this chapter, I report the results of a field test of a technique involving three radionuclides that label surface soils to estimate the origins of transported sediment.

4.2.2 Fallout radionuclides ^{137}Cs, ^{7}Be, and ^{210}Pb

The use of ^{137}Cs as an indicator of soil erosion (Ritchie *et al.*, 1964; McCallan *et al.*, 1980; Lowrance *et al.*, 1988) and sediment delivery (Walling and Bradley, 1988) processes has been reported. Burch *et al.*, (1988) suggested that the different depth penetration characteristics of ^{137}Cs and the cosmogenic radionuclide ^{7}Be in soils could be used to determine the origins of sediments mobilised from hillslopes and so infer the nature of the erosion processes operating. Further evidence of this was presented by Wallbrink and Murray, 1990 and Wallbrink *et al.*, 1991.

The penetration characteristics of ^{137}Cs (half life ~30 years) in southern hemisphere undisturbed soils are well known (Campbell *et al.*, 1982; Norris and Perrens, 1988). At the study site, for example, the ^{137}Cs concentrations decrease to half the surface activity at a depth of about 30 mm (Figure 4.2.2.1a). As deposition of ^{137}Cs in the southern hemisphere effectively stopped in the mid-1970's (McCallan *et al.*, 1980; Longmore *et al.*, 1983), the present soil store is only affected by radioactive decay and soil movement.

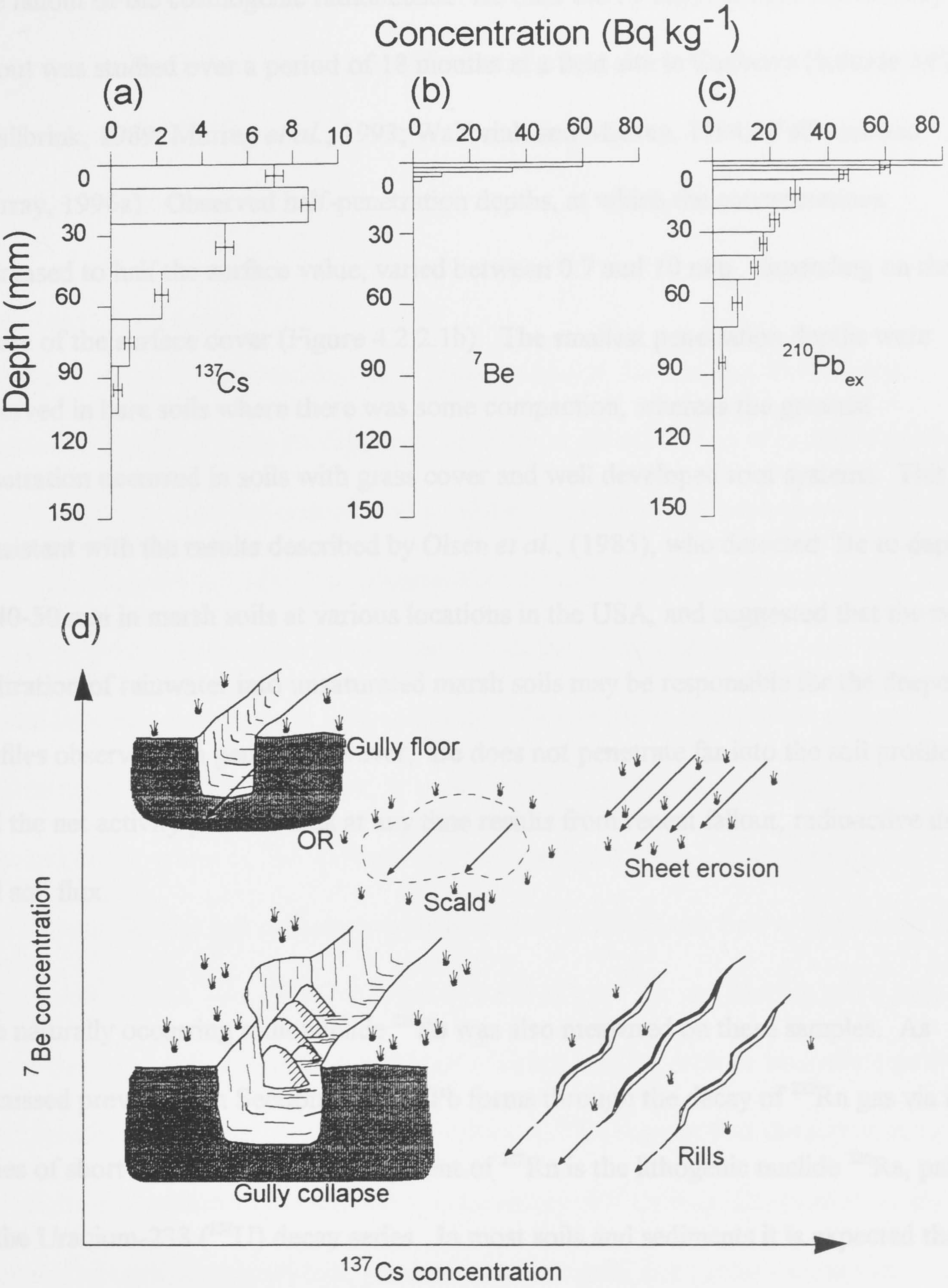

Figure 4.2.2.1 (a) Penetration profile of ^{137}Cs. (b) Penetration profile of ^{7}Be. (c) Penetration profile of ^{210}Pb$_{ex}$. (d) Hypothesised sediment labelling with fallout radionuclides.

The fallout of the cosmogenic radionuclide ^{7}Be (half life 53 days) in both wet and dry fallout was studied over a period of 18 months at a field site in Canberra (latitude 34°S) (Wallbrink, 1989; Murray *et al.*, 1993; Wallbrink and Murray, 1994; Wallbrink and Murray, 1996a). Observed half-penetration depths, at which the concentrations decreased to half the surface value, varied between 0.7 and 10 mm , depending on the nature of the surface cover (Figure 4.2.2.1b). The smallest penetration depths were observed in bare soils where there was some compaction, whereas the greatest penetration occurred in soils with grass cover and well developed root systems. This is consistent with the results described by Olsen *et al.*, (1985), who detected ^{7}Be to depths of 40-50 mm in marsh soils at various locations in the USA, and suggested that the rapid infiltration of rainwater into unsaturated marsh soils may be responsible for the deeper profiles observed. In general however, ^{7}Be does not penetrate far into the soil profile, and the net activity per unit area at any time results from recent fallout, radioactive decay and soil flux.

The naturally occurring radionuclide ^{210}Pb was also measured on these samples. As discussed previously in Section 3.4.3, ^{210}Pb forms through the decay of ^{222}Rn gas via a series of short-lived daughters. The parent of ^{222}Rn is the lithogenic nuclide ^{226}Ra, part of the Uranium-238 (^{238}U) decay series. In most soils and sediments it is expected that ^{210}Pb will be in approximately secular equilibrium with ^{226}Ra. However, some ^{222}Rn diffuses into the atmosphere where it decays to ^{210}Pb. This then reaches the earth's surface by wet and dry precipitation (Koide *et al.*, 1972; Wise, 1980). It is referred to as 'unsupported' or 'excess' ^{210}Pb, and is trapped in surface soils (Krishnaswamy *et al.*, 1971). Its depth penetration has been reported down to 100-400 mm (Nozaki *et al.*, 1978; Olsen *et al.*, 1985) depending on atmospheric input and soil conditions. In

Australia, typical penetration half depths are in the order of ~30 mm (Figure 4.2.2.1c), and its distribution is best approximated by a negative exponential. However, it should be recognised that ^{210}Pb excess can exist at depth, supported by local excesses of ^{222}Rn in groundwater (Tanner, 1964; 1980).

4.2.3 Aims of this work

The aim of the study reported here is to present and test the idea that the different penetration depths of these nuclides may allow the original location of sediment particles in the soil profile to be determined, and so allow inferences to be made about erosional processes (Figure 4.2.2.1d). This is undertaken by first plotting the values against one another in crossplot form, the relative position of the points can then be used to reveal erosion process. For example, high values of ^{137}Cs and ^{7}Be in transported sediments indicate material derived from sheet or minor rill erosion, or both, whereas high levels of ^{137}Cs and lower values of ^{7}Be indicate that the material is from a more actively ruled environment, ie. that soil particles are being derived from slightly below (5-50 mm) the original soil surface. Very low or undetectable levels of ^{137}Cs and ^{7}Be suggest a subsoil source that has not been exposed to fallout of either nuclide, such as an eroding gully wall. Finally, high values of ^{7}Be and low values of ^{137}Cs suggest that the sediment is derived from a surface where the previous inventory of ^{137}Cs was either undetectably small or has been completely lost, yet there has been sufficient recent exposure to obtain a ^{7}Be label. This chapter describes the results of preliminary field tests of this technique. Concentrations of ^{7}Be, ^{137}Cs, ^{210}Pb, ^{226}Ra and Thorium-232 (^{232}Th) in suspended solids from artificially generated runoff are presented. Relationships between the antecedent soil surface conditions and the observed concentrations are examined. The potential applications of the technique are then discussed.

4.3 Experimental design

A rainfall simulator, producing uniform rain droplets over an area of several hundred square metres (Riley and Gardiner, 1991), was used to generate surface runoff in a grazed paddock near Goulburn, NSW (latitude 32°S) at intensities of 100-150 mm hr^{-1}.

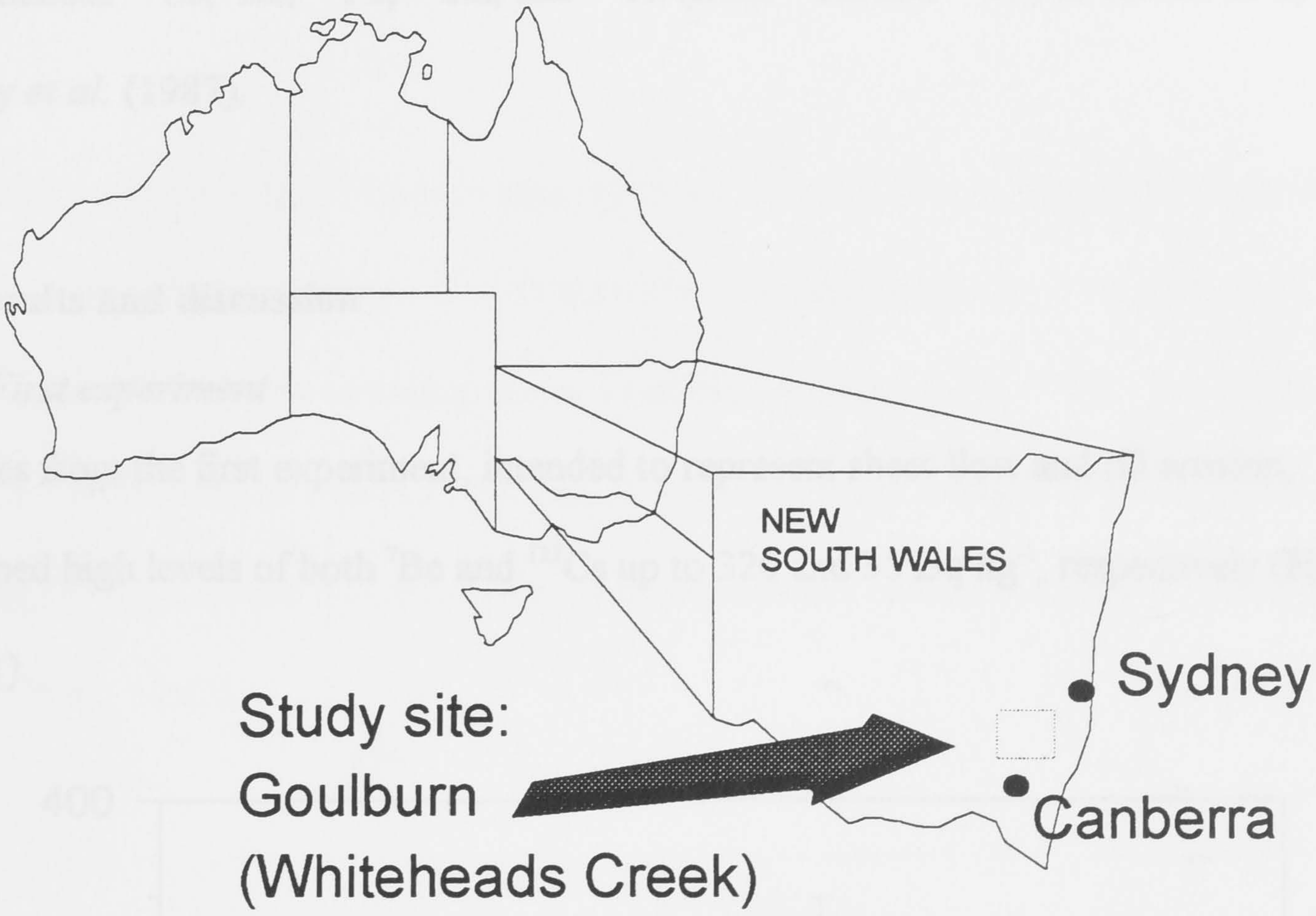

Figure 4.3.1 Site location diagram for rainfall simulator experiments at Whiteheads Creek, near Goulburn, NSW.

The first experiment involved applying rainfall to an area of 50 m^2 with a 5^0 slope, to produce sheet flow and rilling. In the second experiment, rainfall was directed onto a 4 m length of actively eroding gully wall face, and also onto the contributing upslope catchment (~70 m^2) which normally drained over the gully wall. The flow from these surfaces was kept separate by a temporary barrier erected at the edge of the gully, thus enabling the radionuclide signatures of the two components to be measured individually. The barrier was then removed and water from the upslope area allowed to flow over the gully wall. The combined flow was then sampled.

The runoff in all experiments was channelled into collection troughs and 10-40 g of suspended sediment extracted over 15 minute intervals using a continuous flow centrifuge (Alfa Laval, Model No. MAB103b). The resulting sediment samples were then analysed for their tracer properties by high resolution gamma spectrometry for the radionuclides ^{137}Cs, ^{7}Be, ^{210}Pb, ^{226}Ra, and ^{232}Th (from ^{228}Ra and ^{228}Th) as described by Murray *et al.* (1987).

4.4 Results and discussion

4.4.1 First experiment

Samples from the first experiment, intended to represent sheet flow and rill erosion, contained high levels of both ^{7}Be and ^{137}Cs up to 320 and 35 Bq kg^{-1}, respectively (Figure 4.4.1.1).

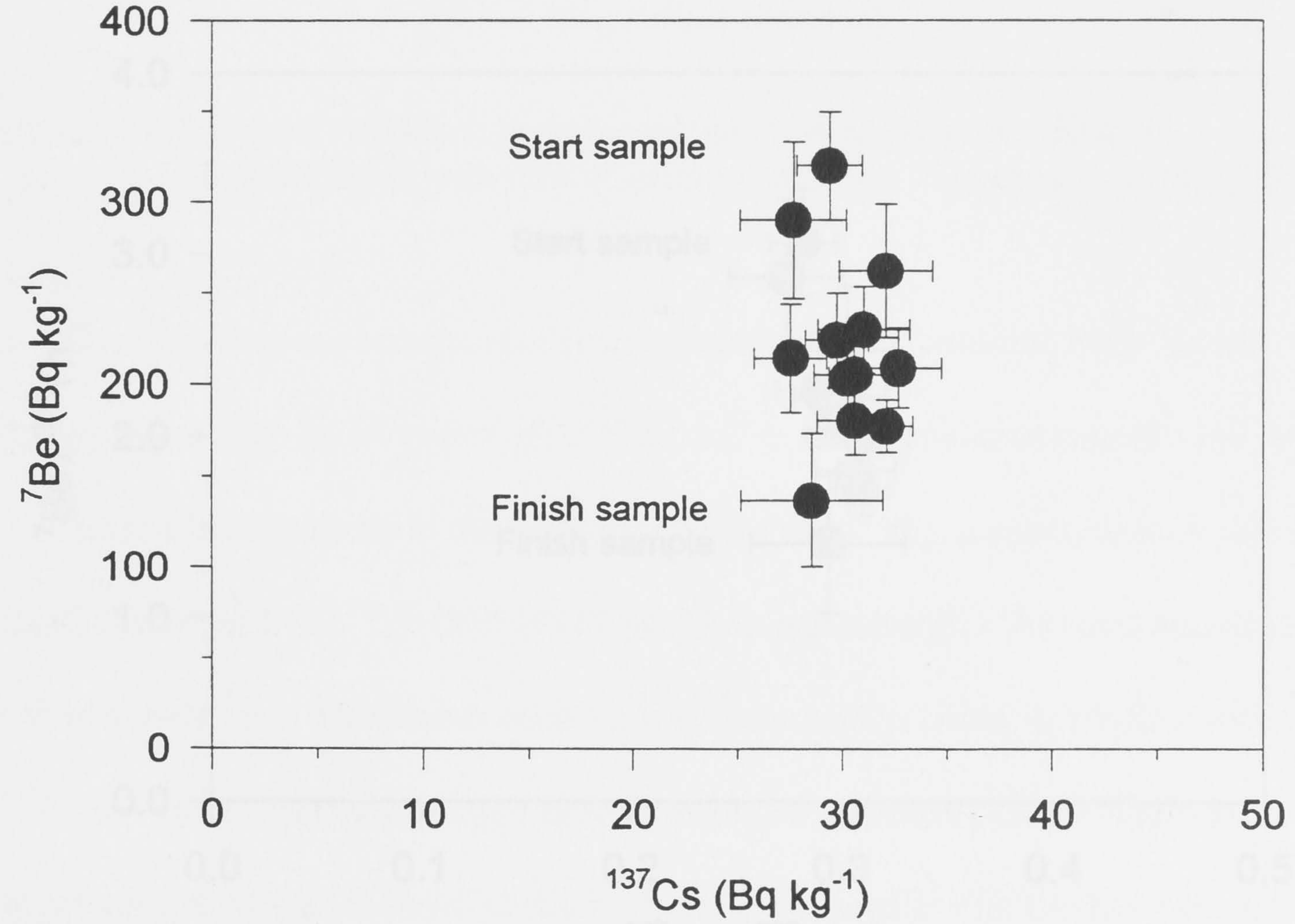

Figure 4.4.1.1 Suspended sediment radionuclide signatures from sheet erosion site, Whiteheads Creek, NSW.

The relatively high absolute activities, between four and five times the maximum concentration shown in Figure 4.2.2.1a and 4.2.2.11b (Page 56), reflect the smaller mean particle size in transported material compared with bulk soil (Meyer *et al.*, 1986), which results in increased relative particle surface area and therefore relative activity per unit weight (Scott, 1982; Olley *et al.*, 1990).

As rainfall progressed the ^{137}Cs activities remained effectively constant (relative standard deviation, *rsd* = 5.4%) whereas the ^{7}Be activities decreased steadily. This different behaviour is made clearer by taking the ratio of the two isotopes to ^{232}Th. Variability in the ^{137}Cs data is reduced (*rsd* = 3.7%; Figure 4.4.1.2); this is thought to be the result of reducing the dependence on particle size and geochemical effects (Murray *et al.*, 1990; Murray *et al.*, 1992).

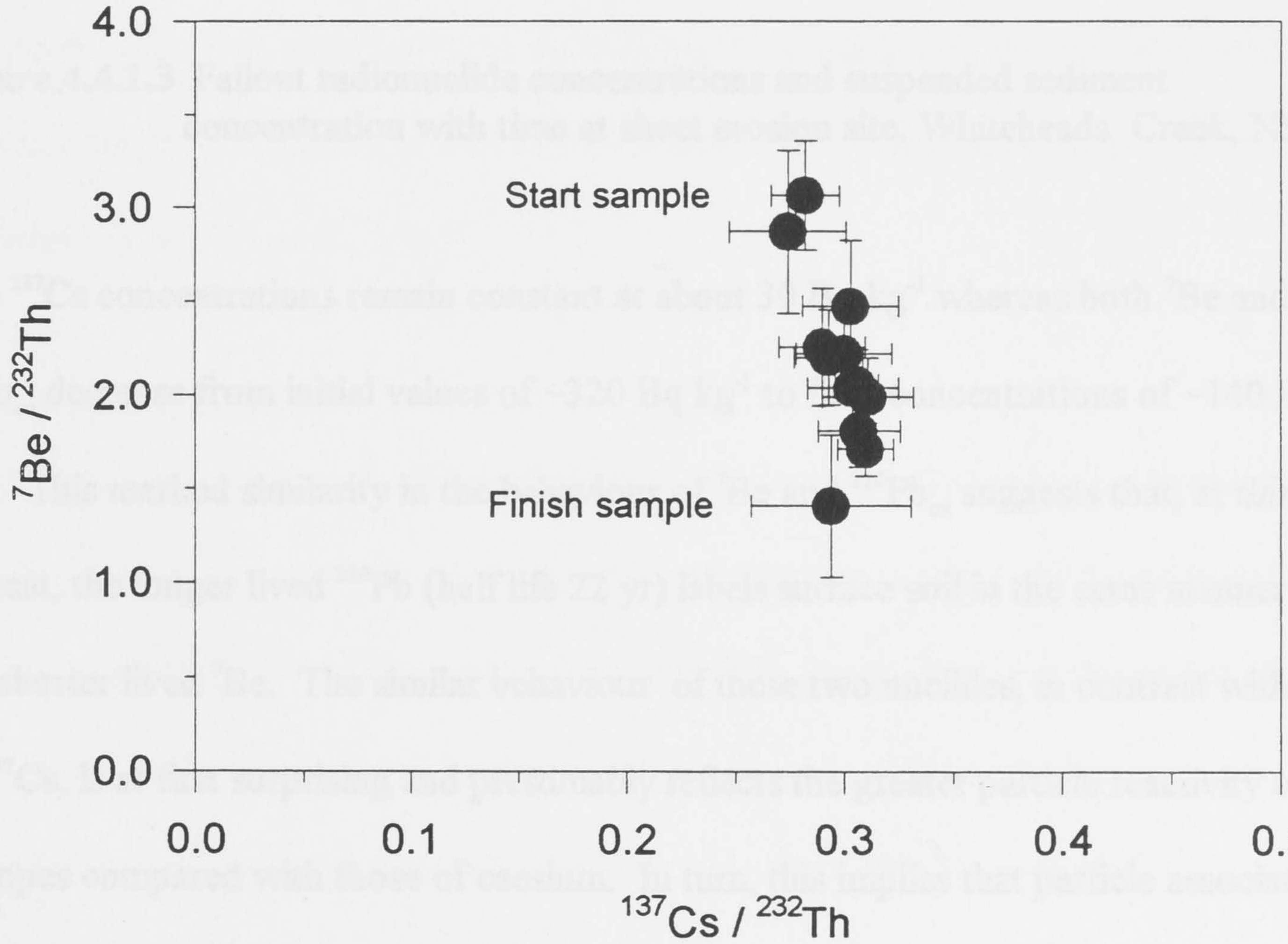

Figure 4.4.1.2 Reduction in ^{137}Cs variability by normalizing to ^{232}Th, in surface runoff material from sheet erosion site at Whiteheads Creek, NSW.

The high and constant concentrations of ^{137}Cs and high concentrations of ^{7}Be are consistent with the predictions of Figure 4.2.2.1d and indicate sheet or rill erosion, or both. Radionuclide concentration data and suspended solids concentrations are presented in Figure 4.4.1.3, plotted against time.

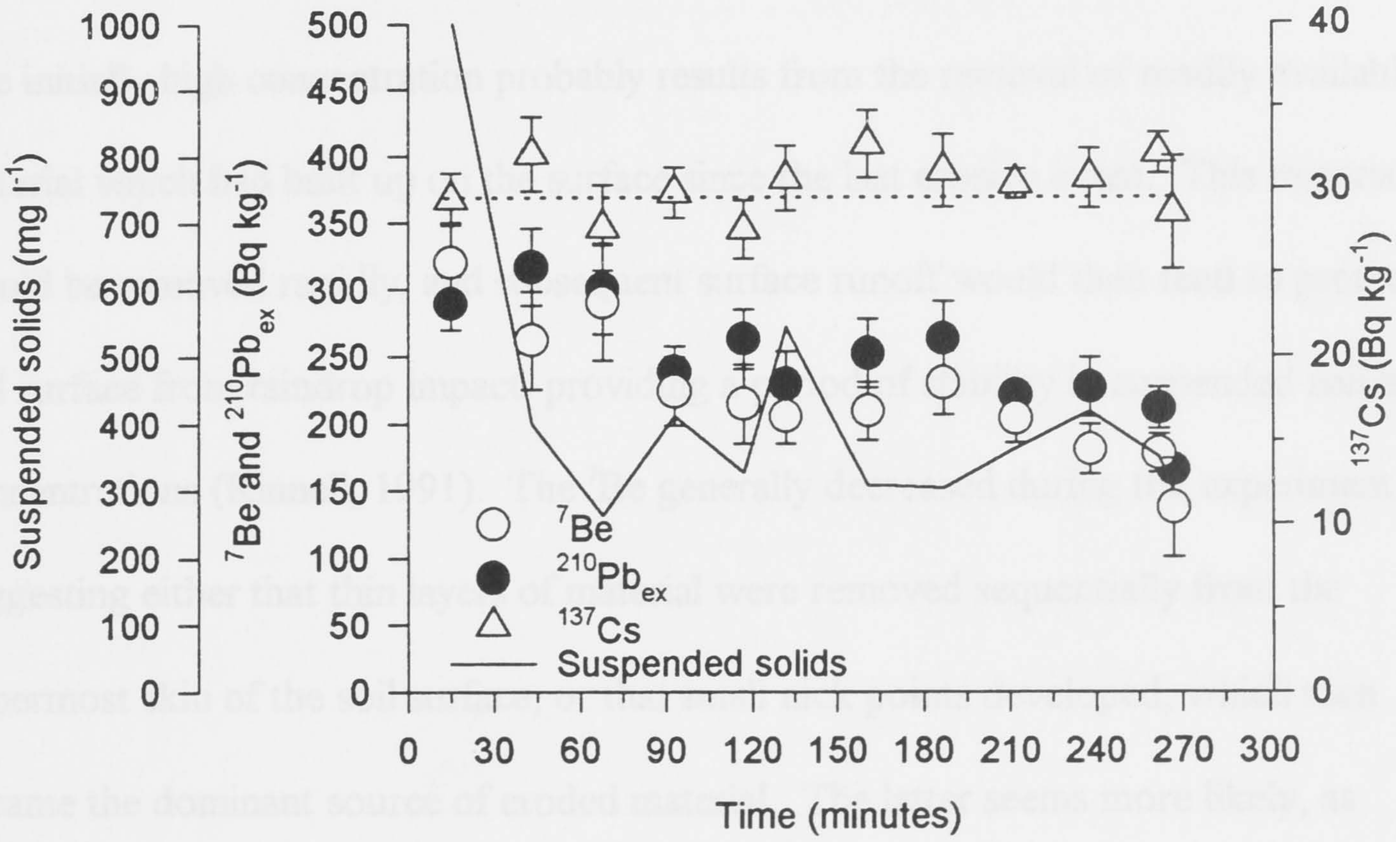

Figure 4.4.1.3 Fallout radionuclide concentrations and suspended sediment concentration with time at sheet erosion site, Whiteheads Creek, NSW.

The ^{137}Cs concentrations remain constant at about 30 Bq kg^{-1} whereas both ^{7}Be and ^{210}Pb$_{ex}$ decrease from initial values of ~320 Bq kg^{-1} to final concentrations of ~140 Bq kg^{-1}. This marked similarity in the behaviour of ^{7}Be and ^{210}Pb$_{ex}$ suggests that, at this site at least, the longer lived ^{210}Pb (half life 22 yr) labels surface soil in the same manner as the shorter lived ^{7}Be. The similar behaviour of these two nuclides, in contrast with that of ^{137}Cs, is at first surprising and presumably reflects the greater particle reactivity of lead isotopes compared with those of caesium. In turn, this implies that particle associated translocation down profile is less important than the initial trapping of these elements. Subsequent profiles at Black Mountain, ACT (Section 5.4) suggests that ^{210}Pb$_{ex}$ may preferentially accumulate in the top few millimetres of the soil profile, thus leading to high concentrations in this region. Such high values would have been averaged out by the relatively coarse sampling interval of Nozaki *et al.*, (1978) and Olsen *et al.* (1985).

The concentration of suspended solids decreases rapidly, then fluctuates about a mean. The initially high concentration probably results from the removal of readily available material which had built up on the surface since the last erosive event. This material would be removed rapidly, and subsequent surface runoff would then tend to protect the soil surface from raindrop impact, providing a period of stability in suspended solids concentrations (Kinnell, 1991). The ^{7}Be generally decreased during the experiment, suggesting either that thin layers of material were removed sequentially from the uppermost skin of the soil surface, or that small nick points developed, which then became the dominant source of eroded material. The latter seems more likely, as competent rills were observed to be developing from nick points during the experiment. This fits the general pattern of initiation of sediment movement from slopes as described elsewhere (Leopold *et al.*, 1964).

4.4.2 Second experiment

The second experiment was designed to compare the suspended sediment from a sheet eroding site with a gully wall. Sediment derived from the eroding gully wall contained no detectable ^{137}Cs or ^{7}Be (Figure 4.4.2.1). No ^{210}Pb$_{ex}$ was present in this material either (Figure 4.4.2.2). Analysis of the overland flow component from the upslope contributing area revealed high values of ^{7}Be, ^{137}Cs and ^{210}Pb$_{ex}$ as expected.

When the upslope runoff was allowed to run into the gully and combine with the runoff from the gully wall, no detectable change occurred in the sediment signature (Figures 4.4.2.1 and 4.4.2.2). The high values obtained from the upslope area were diluted to below detection limits by the larger volume of low concentration material from the gully.

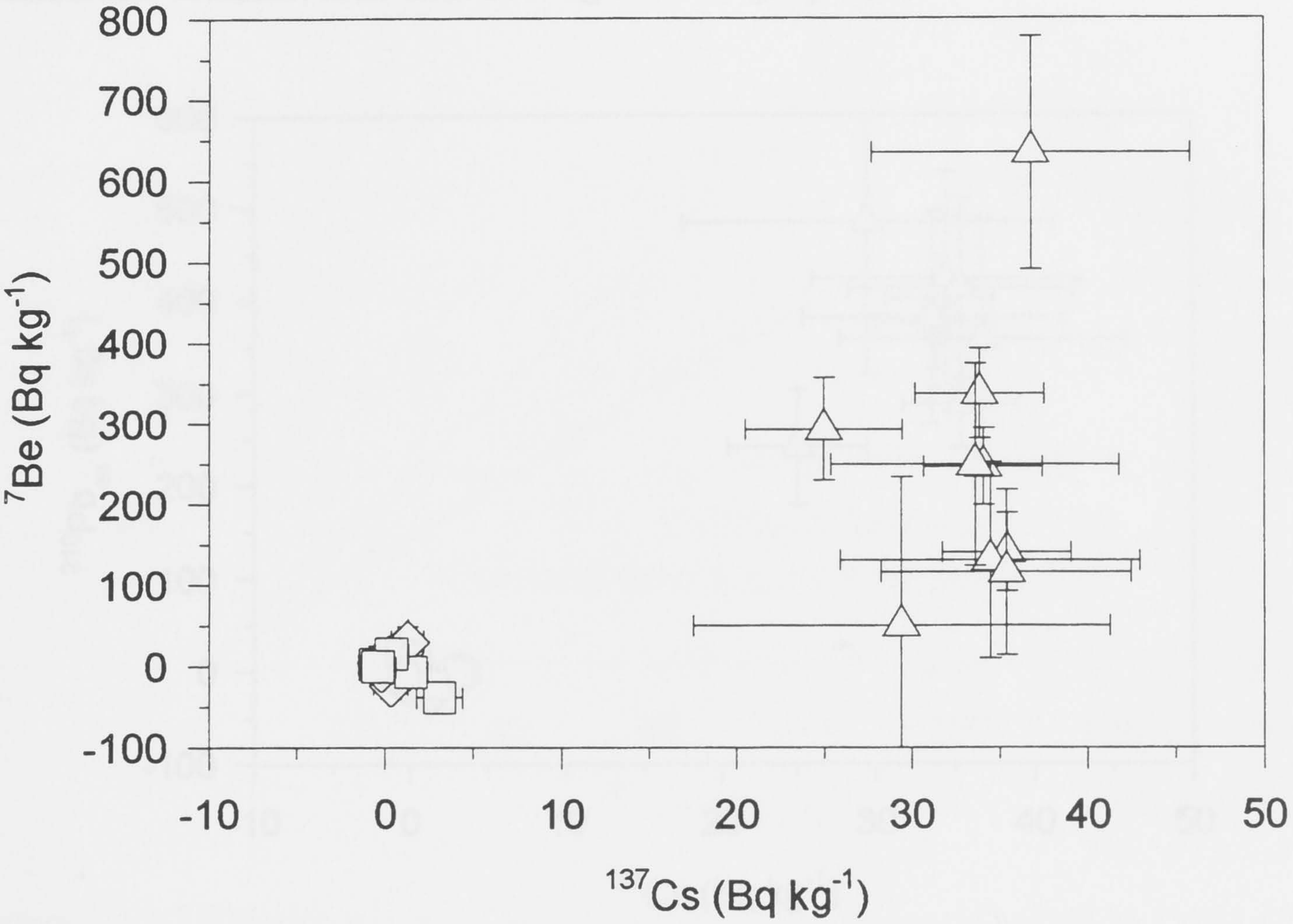

Figure 4.4.2.1 Radionuclide concentrations of ^{7}Be and ^{137}Cs in overland (triangles), gully wall (diamonds), and combined flow (squares) suspended sediment at gully erosion site, Whiteheads Creek, NSW.

Using the radionuclide concentration data, the maximum possible upslope sediment contribution is calculated to be <3%. This result is supported by the suspended solids concentration data. The average suspended solids concentration in the overland flow is 73 ± 12 mg L^{-1}; that in the gully wall flow is 26 ± 2 g L^{-1}. This is a 356 times increase in mass per unit volume. Given that the rainfall intensity per unit area was similar on both the gully wall and the upslope catchment, these values may be taken to be approximately proportional to sediment yield. This suggests that the upslope catchment was providing about 0.3% of the areal yield of the gully wall. This is likely to be the upper limit of the contribution from overland flow; the actual contribution is much less than this because the observed concentration in the combined runoff is 21.1 ± 0.8 g L^{-1}. As the overland flow and gully wall runoff are comparable, the overland flow must have picked up a considerable sediment load after flowing over the gully wall.

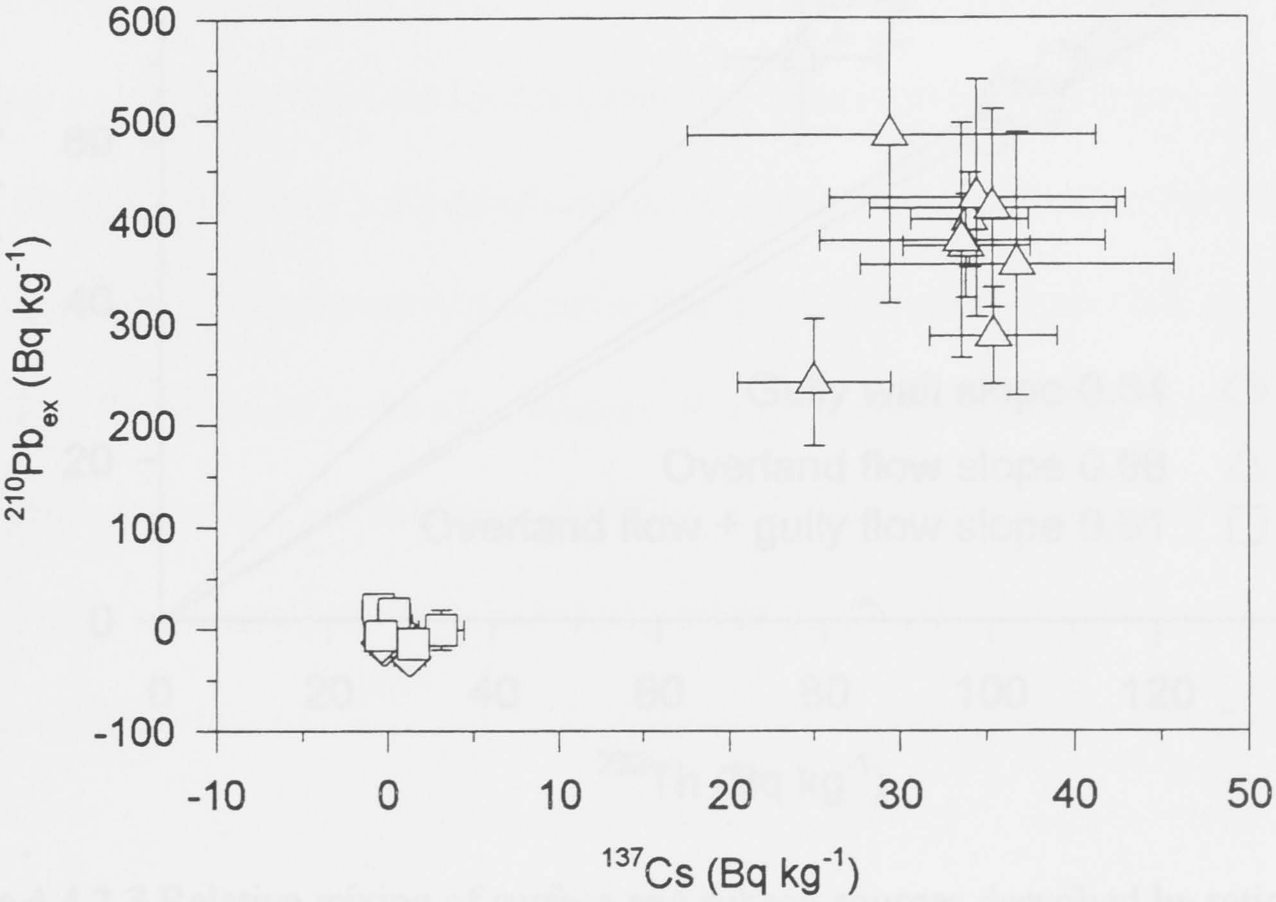

Figure 4.4.2.2 Radionuclide concentrations of $^{210}Pb_{ex}$ and ^{137}Cs in overland, gully wall and combined flow suspended sediment at gully erosion site, Whiteheads Creek, NSW. Note: symbols as per Figure 4.4.2.1.

Support for these estimates of relative contribution is available from analyses of ^{226}Ra and ^{232}Th, members of the ^{238}U and ^{232}Th natural decay series. Figure 4.4.2.3 shows the relationship between ^{226}Ra and ^{232}Th concentration in the suspended sediments derived from the second experiment. All three data sets are consistent with the linear correlation described by Murray *et al.* (1992) in fluvial sediments. For the purposes of this analysis linear relationships with zero intercepts are assumed. Material derived from the overland flow site lies on a line with a slope of 0.96. The subsoil material derived from the gully wall and the mixture of gully and overland flow have slopes of 0.64 and 0.61, respectively. ^{226}Ra/^{232}Th ratios from sediments have been used as indicators or 'fingerprints' of source areas (Murray *et al.*, 1990, 1993).

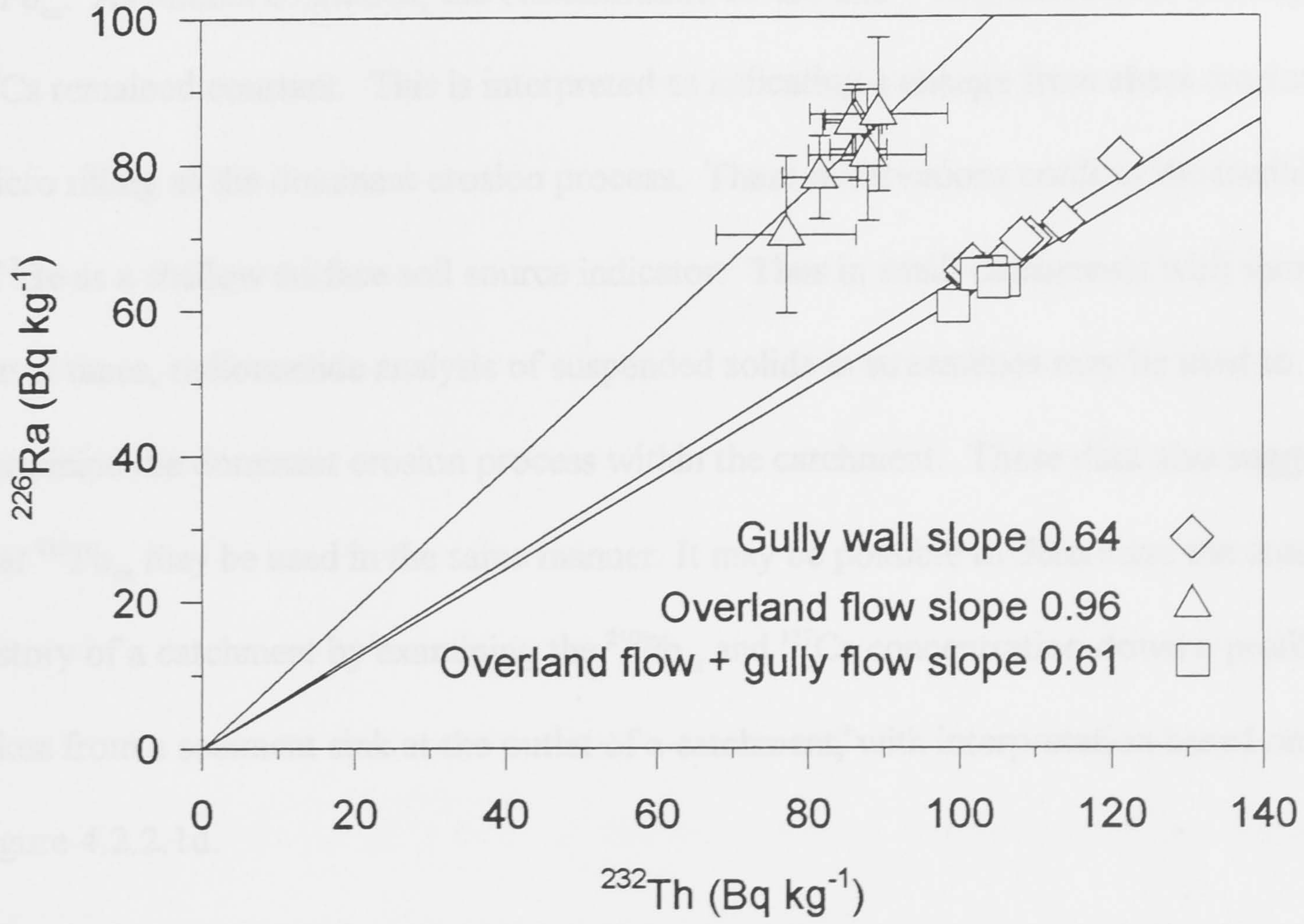

Figure 4.4.2.3 Relative mixing of surface and subsoil sources described by ratios of ^{226}Ra and ^{232}Th, at gully erosion site Whiteheads Creek, NSW.

The correlations in Figure 4.4.2.3 can therefore be used to determine the relative contribution from the two sources to the total flux leaving the system using a simple two component mixing model. Using this method, the maximum possible surface soil contribution to the total sediment flux is calculated to be ~2.5%, confirming the result based on sediment concentrations and ^{7}Be, ^{210}Pb$_{ex}$ and ^{137}Cs measurements. These findings are supported by the work of Mackenzie *et al.* (1991) and Neil and Fogarty, (1991) who used a combination of approaches to determine that gully derived material dominates channel sediment in gullied catchments in NSW, Australia.

4.4 Conclusions

At the study locations considered here, it has been shown that sediments mobilised by rainfall from undisturbed soil surfaces contain high concentrations of ^{137}Cs, ^{7}Be and ^{210}Pb$_{ex}$. As rainfall continued, the concentration of ^{7}Be and ^{210}Pb$_{ex}$ decreased, although ^{137}Cs remained constant. This is interpreted as indicating a change from sheet erosion to micro rilling as the dominant erosion process. These observations confirm the usefulness of ^{7}Be as a shallow surface soil source indicator. Thus in small catchments with short travel times, radionuclide analysis of suspended solids in streamlines may be used to determine the dominant erosion process within the catchment. These data also suggest that ^{210}Pb$_{ex}$ may be used in the same manner. It may be possible to determine the erosion history of a catchment by examining the ^{210}Pb$_{ex}$ and ^{137}Cs concentration down a profile taken from a sediment sink at the outlet of a catchment, with interpretation based on Figure 4.2.2.1d.

At the second study site a subsoil (gully wall) source was mixed with a surface soil source. The relative contributions of the two sources to the mixture were determined by three independent methods: mass balance, surface tracers (^{7}Be, ^{210}Pb$_{ex}$, ^{137}Cs) and spatial source tracers (^{226}Ra, ^{232}Th). All three methods gave consistent results, showing that sheet and rill erosion contributed a negligible fraction to the total sediment mobilised in the gully. The gully wall was the dominant source.

4.5 Summary and consideration of hypotheses

The different depth penetration characteristics of ^{137}Cs, ^{7}Be, and ^{210}Pb$_{ex}$ in undisturbed soils can be used to identify the dominant erosion processes by analysis of transported sediment. In these soils, ^{137}Cs typically decreases to half the surface value at between 30 and 50 mm; ^{7}Be has half penetration depths of between 0.7 and 10 mm, whereas ^{210}Pb$_{ex}$ has half penetration depths between 10 and 30 mm. Therefore hypothesis *(7) The tracers ^{137}Cs, ^{7}Be, and ^{210}Pb$_{ex}$ have different initial soil depth distributions in undisturbed soils,* is not rejected. Surface runoff and associated suspended sediment was artificially generated at two locations in a grazed paddock using a rainfall simulator. Suspended sediment derived from sheet flow contained initially high values of ^{137}Cs, ^{7}Be, and ^{210}Pb$_{ex}$. At the second location artificial rain was applied to an eroded gully wall. The derived suspended sediment contained no detectable ^{137}Cs, ^{7}Be or ^{210}Pb$_{ex}$, and so hypothesis *(8) Eroded material from soil surfaces will have high fallout radionuclide concentrations and subsoil derived material will have low fallout radionuclide concentrations,* is also not rejected. Overland flow from above the gully wall was then allowed to run down the gully face and mix with the water falling directly onto the gully wall. There was no detectable change in the radionuclide signature of the mixed sediment, showing that the gully wall was the predominant source of sediment. This was tested independently by mass balance and ^{226}Ra and ^{232}Th ratios, and so hypothesis *(9) The net contribution from surface erosion will exceed that from gully erosion'* must be rejected at this site. The good correlation between ^{210}Pb$_{ex}$ and ^{7}Be at this site suggests that the differential technique described here may be applicable at time and spatial scales greater than possible with ^{7}Be. It may therefore be practical to examine catchment erosion history thorough analysis of ^{210}Pb$_{ex}$ and ^{137}Cs in sediment cores.

Chapter 5: Quantifying depth sources using fallout radionuclides

Chapter 5: Quantifying depth sources using fallout radionuclides

5.1 Perspective

In Chapter 4, it was shown that a knowledge of the different depth distributions of $^{210}Pb_{ex}$, ^{137}Cs, and ^{7}Be could be used to interpret sediment sources and erosion history. However, in some circumstances a more detailed understanding of the precise depth origin of particulates is required. This chapter improves the methodology of Chapter 4, by combining the different depth distributions of $^{210}Pb_{ex}$, ^{137}Cs, and ^{7}Be to produce profile reference curves with which the radionuclide concentrations of suspended sediments can be compared. This comparison allows the depths from which the sediment material originated to be calculated with better precision than the descriptive approach of Chapter 4. The relative contributions from the different source depths can also be calculated.

Hypotheses tested in this chapter are: *(10) The combined depth profiles of* ^{137}Cs, $^{210}Pb_{ex}$, *and* ^{7}Be *provide unique concentration labels on soil material from different depths within those profiles; (11) The concentration labels of* ^{137}Cs, $^{210}Pb_{ex}$, *and* ^{7}Be *on sediments from these profiles can be used to determine their depth origin(s) in the soil profile* and; *(12) Surface material derived from shallow rill and overland flow will exceed that from sides and floors of deeper rills and gullies.*

This chapter has been reviewed and submitted as: Wallbrink, P.J., Murray, A.S. and Olley, J.M, Estimating the depth from which suspended sediment originated using fallout ^{210}Pb, ^{7}Be and ^{137}Cs, *Soil Sci. Soc. Am. J.*

5.2 Introduction

5.2.1 Background

In streams, pollutants such as heavy metals and pesticides are primarily transported in association with sediments (Kuntz, 1984; Allan, 1986; Novotny *et al.*, 1986). Many of these chemical species were originally deposited at, or close to, the soil surface. Thus, an ability to measure the depth from which stream sediments originated could aid efforts to control pollution in streams. For example, a difference in nutrient concentration between the surface and subsoil has been observed almost globally. Therefore equal volumes of sediment, derived from sheet erosion of the surface in one instance and from the base of rills in another, may transport markedly different amounts of nutrients to offsite storages and waterways. Consequently, a knowledge of the relative contribution of sheet erosion and rills to in-stream sediments and their associated pollutants could be used by land managers to design controls. Scientists also need this information in order to better understand the influence of different erosion processes on the fluxes of sediments, nutrients and pollutants.

In Chapter 4 it was shown that the origin of sediments can be determined using measurements of the fallout radionuclide tracers ^{210}Pb, ^{137}Cs, and ^{7}Be in soils. In this method it is assumed that tracer concentrations on sediment particles derived from the soil are representative of their initial depth distributions(Figure 5.2.1.1), and therefore can be used as indicators of depth source. Although this method is useful for determining the dominant sources of sediment, and describing the erosion process responsible for its production, it remains descriptive and cannot resolve the actual depths from which the sediment is derived.

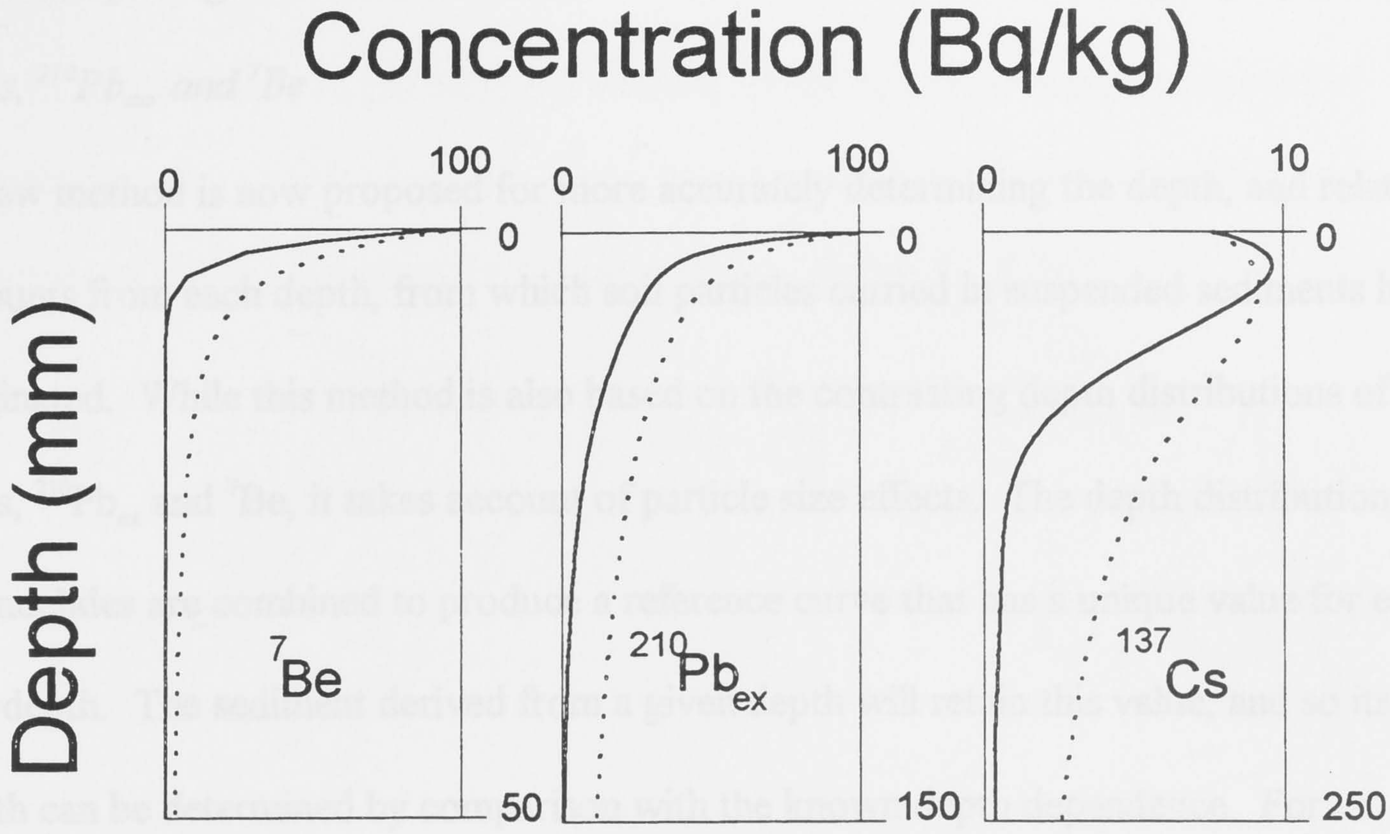

Figure 5.2.1.1 Soil distributions of ^{7}Be, ^{210}Pb$_{ex}$ and ^{137}Cs concentrations, generalised from (Fisenne, 1968; Matthews and Potipin, 1985; Walling and Woodward, 1992; Wallbrink and Murray, 1993; Basher *et al.*, 1995; Wallbrink and Murray 1996a). Note: dotted line represents integration of solid line.

In addition, the effect on radionuclide concentrations of sorting by particle size has not been formally accounted for. In general, concentrations increase as particle size decreases (Megumi *et al.*, 1982; He and Owens, 1995; Olley *et al.*, 1996; Wallbrink and Murray, 1996a). It is known that fluvial processes sort particles by size and density (Krumbein and Sloss, 1963), and thus radionuclide concentrations of sediments are likely to be different from those in the bulk soil profile from which they are derived. If this effect is not allowed for, then the sourcing of sediment can be ambiguous, eg. fine particles from depth in the soil profile may have concentrations similar to those in the bulk soil at the surface. Any method that proposes to measure the depth source of sediment material using these tracers must take this phenomenon into account.

5.2.2 Comparing sediment radionuclide concentrations with known depth profiles of ^{137}Cs, ^{210}Pb$_{ex}$, and ^{7}Be

A new method is now proposed for more accurately determining the depth, and relative amounts from each depth, from which soil particles carried in suspended sediments have originated. While this method is also based on the contrasting depth distributions of ^{137}Cs, ^{210}Pb$_{ex}$ and ^{7}Be, it takes account of particle size effects. The depth distributions of the nuclides are combined to produce a reference curve that has a unique value for each soil depth. The sediment derived from a given depth will retain this value, and so its depth can be determined by comparison with the known depth dependence. For example, the solid line in Figure 5.2.2.1a defines the ratio of the concentrations of ^{7}Be to ^{210}Pb$_{ex}$ as a function of depth (using the data of Figure 5.2.1.1). The curvature in this ratio results from their different depth distributions. Surface particles will have the highest concentration ratios; successively deeper layers have concentration ratios that move towards the origin, following the solid curve. Thus, any particle removed from the top few cm of the soil profile will have a unique ^{7}Be/^{210}Pb ratio.

However, if the sediment is derived from two distinct depths, surface and subsurface (Figure 5.2.2.1a), then the average sediment ratio (shown as point C) will plot away from the curved line. In this case, the potential depth zones which could contribute to C are defined by the intersection of the dotted mixing lines with the solid curve. These zones are limited at the surface by the maximum possible concentration values and at the origin by the minimum possible concentration values, defining the concentration ranges m^b, m^p, n^b and n^p, on the corresponding ^{7}Be and ^{210}Pb axis (Figure 5.2.2.1b). These ranges are converted to ranges of depth loss from the known concentration/depth characteristics (Figure 5.2.1.1).

The relative contributions from zones m and n can then be calculated as the inverse of their relative distance to the sediment value C. That is, to first order the fractional contribution of n to C is given by $\frac{b}{a+b}$, where a and b are the lengths of the two segments of the line bisecting the ranges m and n (Figure 5.2.2.1c).

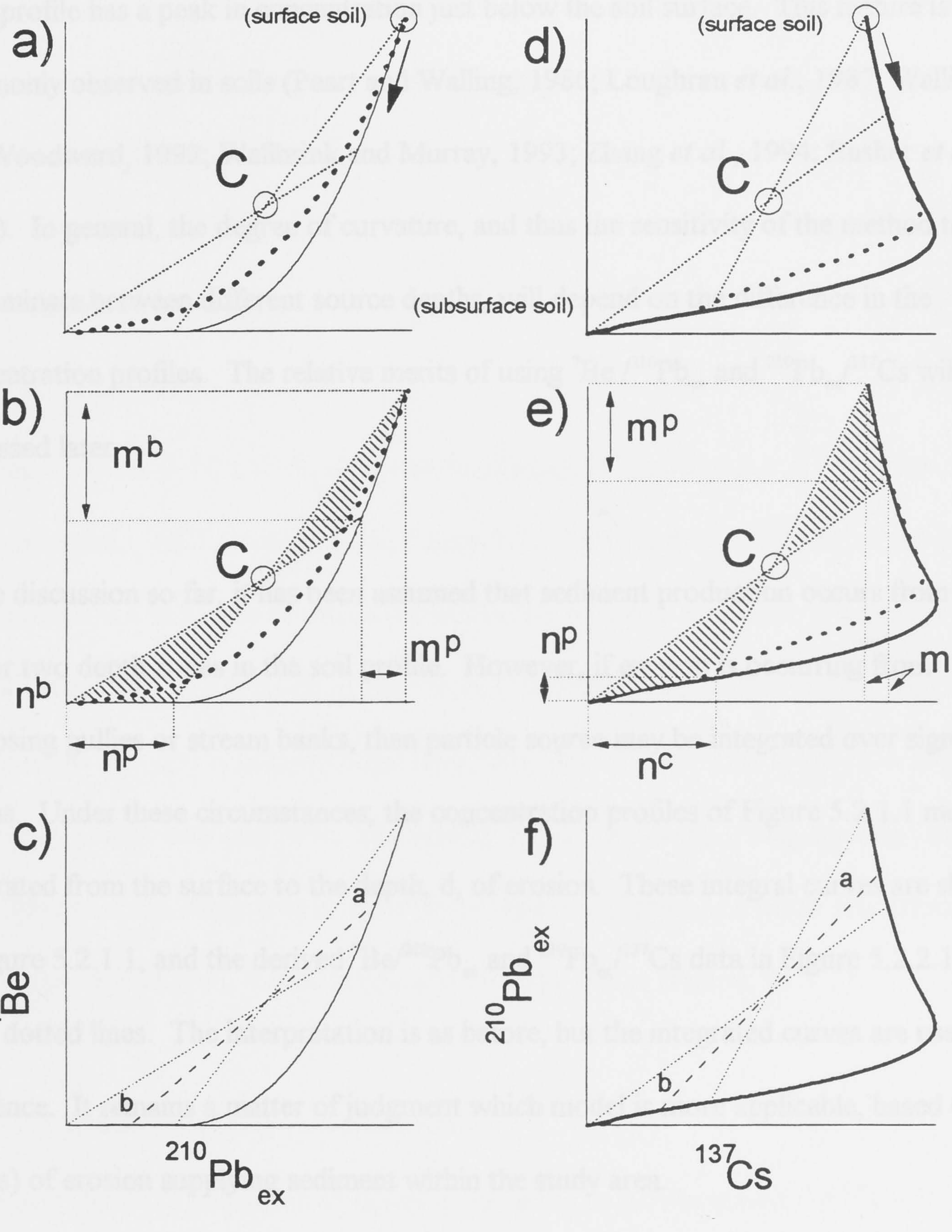

Figure 5.2.2.1 Theoretical models for quantifying sources of suspended sediment. See text for description.

The resolution of this approach is governed by the proximity of the sediment value (C) to the ^{7}Be to ^{210}Pb$_{ex}$ ratio curve. Figures 5.2.2.1d,e,f, present a similar analysis of the behaviour of ^{210}Pb$_{ex}$ and ^{137}Cs normalised concentrations. The degree of curvature is much more pronounced in Figure 5.2.21d (solid line) than in Figure 5.2.2.1a, because the ^{137}Cs profile has a peak in concentration just below the soil surface. This feature is commonly observed in soils (Peart and Walling, 1986; Loughran *et al.*, 1987; Walling and Woodward, 1992; Wallbrink and Murray, 1993; Zhang *et al.*, 1994; Basher *et al.*, 1995). In general, the degree of curvature, and thus the sensitivity of the method to discriminate between different source depths, will depend on the difference in the concentration profiles. The relative merits of using ^{7}Be /^{210}Pb$_{ex}$ and ^{210}Pb$_{ex}$/^{137}Cs will be discussed later.

In the discussion so far, it has been assumed that sediment production occurs from only one or two depth zones in the soil profile. However, if erosion is occurring from collapsing gullies or stream banks, then particle source may be integrated over significant depths. Under these circumstances, the concentration profiles of Figure 5.2.1.1 must be integrated from the surface to the depth, d, of erosion. These integral curves are shown in Figure 5.2.1.1, and the derived ^{7}Be/^{210}Pb$_{ex}$ and ^{210}Pb$_{ex}$/^{137}Cs data in Figure 5.2.2.1, as thick dotted lines. The interpretation is as before, but the integrated curves are used as a reference. It remains a matter of judgment which model is more applicable, based on the type(s) of erosion supplying sediment within the study area.

5.2.3 Objectives of this work

This chapter demonstrates a method to improve the depth resolution of descriptive sediment source models, such as those in Chapter 4 (ie., Wallbrink and Murray, 1993).

The new approach is tested by first determining the depth distributions of $^{210}Pb_{ex}$, ^{137}Cs and ^{7}Be in a study plot. The relationship between particle size and radionuclide concentration in the plot soils is then described. The reference curves of ^{7}Be and $^{210}Pb_{ex}$, and $^{210}Pb_{ex}$ and ^{137}Cs, are constructed and compared with the observed concentrations on sediment generated from the plot under different erosion conditions by simulated run-off. These data are then interpreted using the source model, and the results compared with the known depths of incision.

5.3 Materials and methods

5.3.1 Site description

A shallow hollow (~12 x 13 m, slope approximately 10°) on a colluvial footslope of Black Mountain, ACT (latitude 34^{0}S, Figure 5.3.1.1) was selected for the experiment. The bedrock is a sequence of interlayed sandstone, siltstone and shale. The soil is a Gn3.25 using the classification of Northcote (1979), that is a yellow texture contrast soil with shallow (<5 cm) A horizon made up of grey fine sandy to silty loam. The B horizon is typically yellow silty clay loam, and extends to ~50 cm depth. The dominant native grass species in the study area is *Agropyrens repens* and there is a single *Eucalyptus macrorycha* of approx. 8 m in height and 10 years in age (Figure 5.3.1.1).

5.3.2 Depth distributions of radionuclides

Three soil pits were excavated to describe the initial depth distribution of ^{137}Cs, $^{210}Pb_{ex}$, and ^{7}Be within the study plot (Figure 5.3.1.1). These each had a surface area of 0.04 m^2 (20 x 20 cm) which was defined by steel guides; the depth measurement was controlled by metal reference tabs. Given the expected shallow penetration profiles of ^{7}Be and

$^{210}Pb_{ex}$, it was important that there was fine control on the sampling depths for the surface layers. With this technique, increments of two mm were possible at the surface, although the underlying increments were larger. In addition, 15 separate surface soil scrapes (~0-20 mm) were bulked together and then separated into eight different particle size classes by wet sieving and settling. This material was then used to characterise the relationship between particle size and radionuclide concentration in this soil.

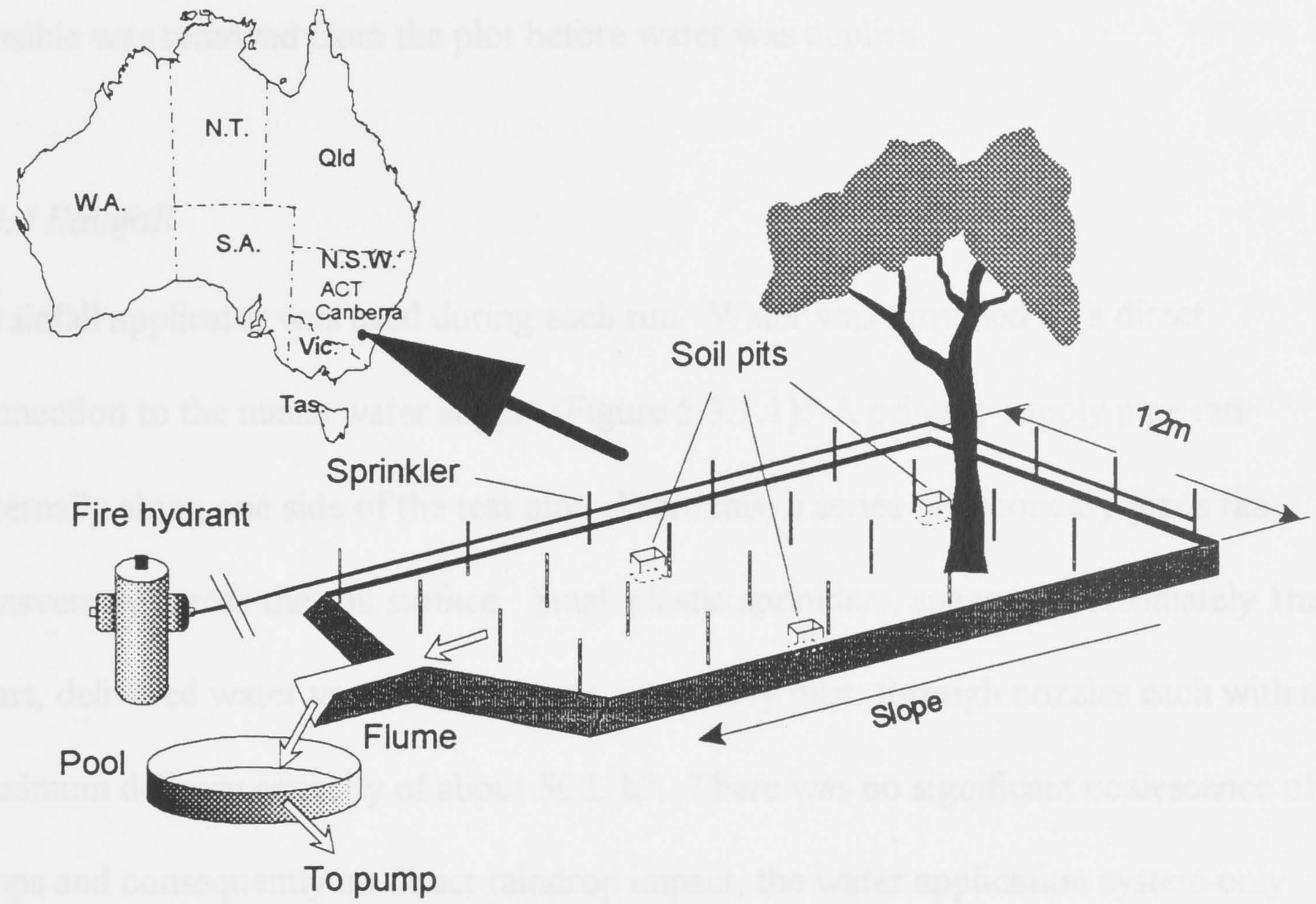

Figure 5.3.1.1 Site location for Black Mountain, ACT, and experimental design for plot experiments.

5.3.3 Site treatments

The surface of the plot was prepared in sequential stages of increasing surface disturbance to simulate four different erosion conditions. Treatment 1 involved removing the bulk of the grass to leave only a 0-5 mm stubble. In Treatment 2 the surface was scraped by a metal rake to leave a network of shallow rills of depth 5 -10 mm (widths 5-10 mm). This network drained the majority of the plot and had its exit point approximately 0.9 - 1.0 m upslope from the inlet to the flume. Treatment 3 involved deepening the rills to ~100 mm (widths 100-150 mm) with a spade. In Treatment 4, two rills that drained either side of the plot were deepened again and widened to ~250 mm depth (widths ~300 mm), to simulate sediment production from large incised rills/small gullies. In Treatment 2 the dislodged material from the shallow rills was left in place, and in Treatments 3 and 4 as much of the loosened soil material as possible was removed from the plot before water was applied.

5.3.4 Rainfall

A rainfall applicator was used during each run. Water was provided by a direct connection to the mains water supply (Figure 5.3.1.1). A primary supply pipe ran externally along one side of the test plot. From this, a series of secondary pipes ran transversely across the soil surface. Small plastic sprinklers, spaced approximately 1m apart, delivered water vertically upwards, as a heavy mist, through nozzles each with a maximum delivery capacity of about 50 L h^{-1}. There was no significant coalescence of drops and consequently no direct raindrop impact, the water application system only needed to evenly distribute water across the plot. Water was applied for periods of between 250 - 340 minutes and the application rate was kept approximately equal for the four experiments. The top and sides of the plot had trenches cut along them to prevent run-on from surrounding areas. The base of the plot was capped with Plaster of Paris to channel water into a flume. Runoff waters from the flume were directed into a partially buried container that was drained by pumps supplying a continuous flow centrifuge (Alfa Laval, Model No. MAB103b) which extracted the suspended sediment. The centrifuge was stopped and the sediment removed every 20 minutes. A further 10 minutes was then required to clean the separator bowl and get the centrifuge back to operating speed. Suspended sediment concentrations were not measured.

5.3.5 Sample analysis

All the suspended sediment and soil samples were oven dried, ashed at 400^0 C, and analysed by gamma spectrometry for ^{137}Cs, ^{7}Be ^{226}Ra, ^{232}Th, and ^{210}Pb according to Murray *et al.*, (1987), as described in previous chapters. Samples were counted for a minimum of 85 ksec. At this site the ^{210}Pb activity concentrations in the soils become a constant fraction (~88%) of ^{226}Ra at depth, ie; there was approximately a 12 % loss of ^{222}Rn activity to the atmosphere. Consequently, the values of ^{210}Pb$_{ex}$ are calculated by subtracting 88% of the observed ^{226}Ra at each depth from the appropriate total ^{210}Pb concentration. A similar phenomenon was observed in Chapter 3 at St Helens, Tasmania (Wallbrink and Murray, 1996b). Unless otherwise noted, all mean values are reported with one standard error.

5.4 Plot soil and sediment nuclide distributions

5.4.1 Soil profiles and radionuclide distributions

The concentration profiles of ^{7}Be, ^{210}Pb$_{ex}$ and ^{137}Cs (Bq kg^{-1}) have different shapes from one another (Figure 5.4.1.1a,b,c). Maximum concentrations of ^{7}Be occur at the surface, from where the values decrease approximately exponentially to detection limits at ~10 mm. The ^{210}Pb$_{ex}$ profiles also have a surface maximum; the distribution tails off to detection limits at ~100 mm. Similar distributions for both of these have been observed elsewhere (Fisenne, 1968; Matthews and Potipin, 1985; Wallbrink and Murray 1996a).

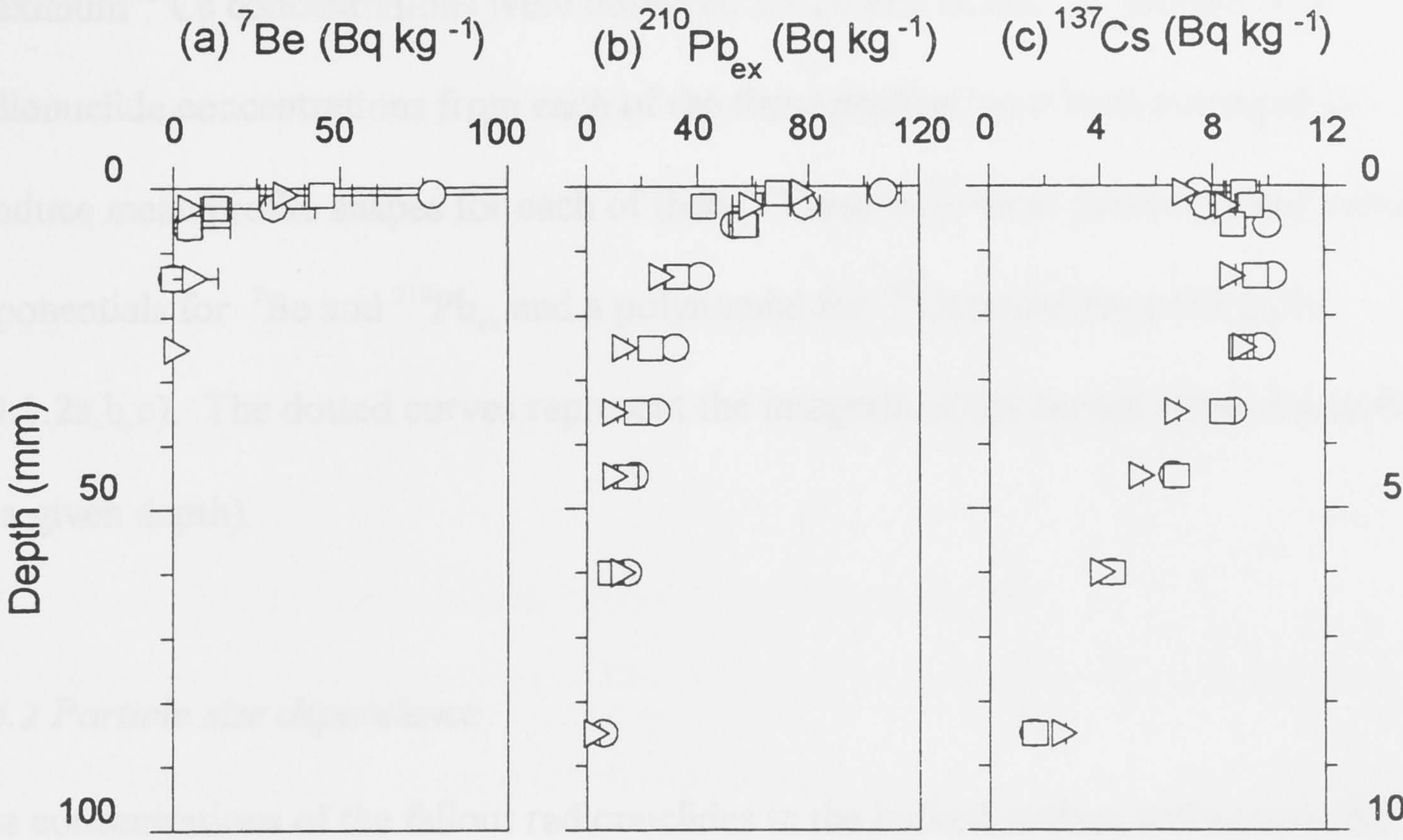

Figure 5.4.1.1: Measured bulk soil concentrations of fallout radionuclides with depth in three soil profiles at Black Mountain. Data from Profile 1 (circles), Profile 2 (squares), Profile 3 (triangles). Note depth axis of each restricted to 100 mm for clarity.

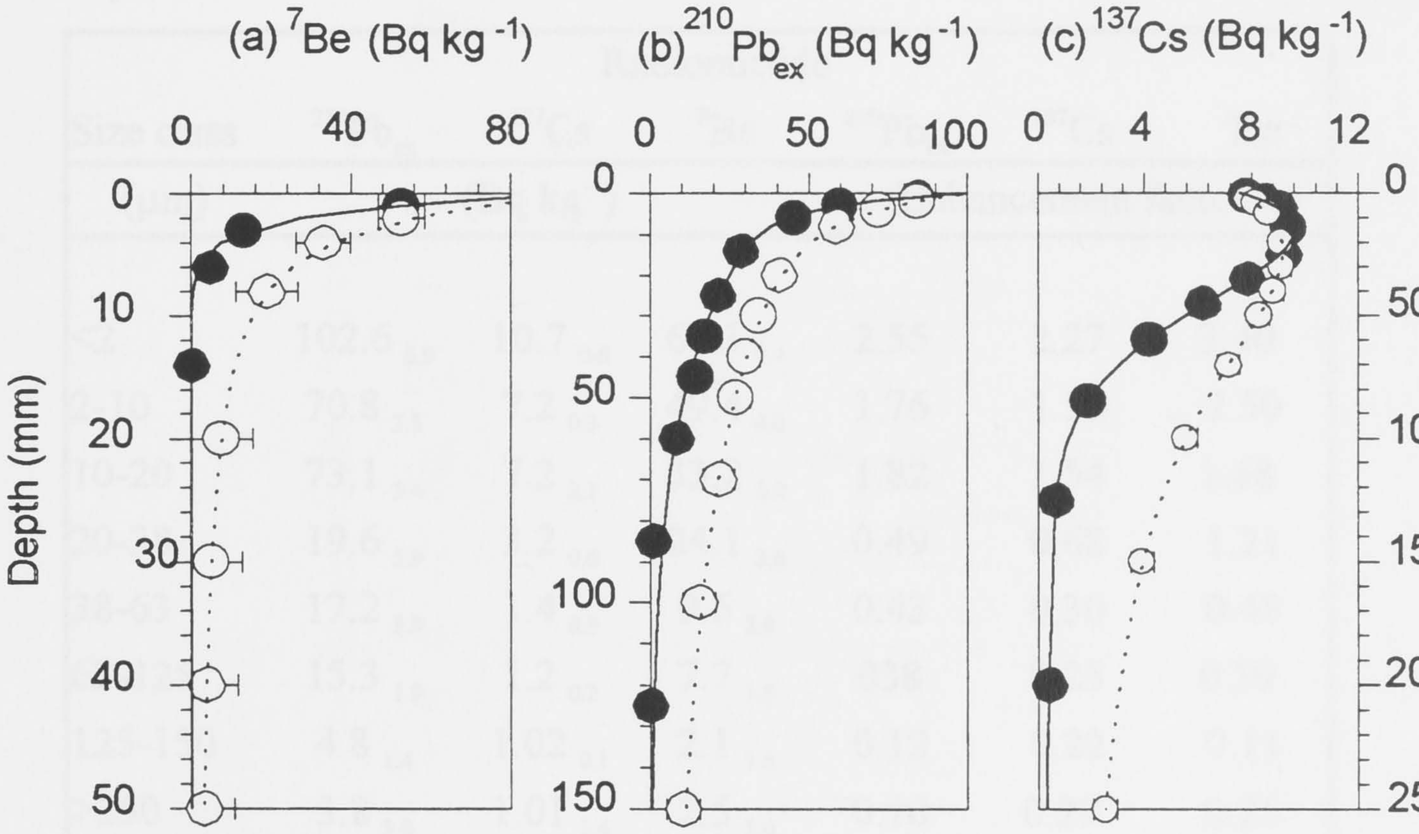

Figure 5.4.1.2 Parameterisation of the average of the bulk soil concentrations of ^{7}Be, ^{210}Pb$_{ex}$, and ^{137}Cs given in Figure 5.4.1.1. Measured data (filled circles); measured data integrated with depth (open circles), curve fitted to measured data (solid line), integral of the curve (dotted line) .

Maximum ^{137}Cs concentrations were observed 10-20 mm below the surface. The radionuclide concentrations from each of the three profiles have been averaged to produce mean profile shapes for each of them. These have been paramaterised using exponentials for ^{7}Be and ^{210}Pb$_{ex}$ and a polynomial for ^{137}Cs (solid lines in Figure 5.4.1.2a,b,c). The dotted curves represent the integrals of the curves (from the surface to a given depth).

5.4.2 Particle size dependence

The concentrations of the fallout radionuclides in the bulked surface soil scrapes for eight particle size fractions are given in Table 5.4.2.1.

Table 5.4.2.1 Concentrations and enhancement factors for ^{210}Pb$_{ex}$, ^{137}Cs, and ^{7}Be in different particle size classes at Black Mountain, ACT. Uncertainties are standard errors.

	Radionuclide					
Size class	^{210}Pb$_{ex}$	^{137}Cs	^{7}Be	^{210}Pb$_{ex}$	^{137}Cs	^{7}Be
(µm)	(Bq kg^{-1})			Enhancement factor		
<2	$102.6_{6.9}$	$10.7_{0.6}$	$67.3_{7.4}$	2.55	2.27	3.40
2-10	$70.8_{3.8}$	$7.2_{0.3}$	$49.4_{4.0}$	1.76	1.53	2.50
10-20	$73.1_{3.4}$	$7.2_{3.1}$	$33.2_{2.9}$	1.82	1.54	1.68
20-38	$19.6_{5.9}$	$3.2_{0.6}$	$24.1_{3.6}$	0.49	0.68	1.21
38-63	$17.2_{3.9}$	$1.4_{0.3}$	$9.6_{2.9}$	0.43	0.30	0.48
63-125	$15.3_{1.9}$	$1.2_{0.2}$	$7.7_{1.5}$	038	0.25	0.39
125-150	$4.8_{1.4}$	$1.02_{0.1}$	$2.1_{1.6}$	0.12	0.22	0.11
>250	$3.8_{3.0}$	$1.01_{1.4}$	$2.5_{3.0}$	0.10	0.29	0.25
Bulk soil	$40.2_{2.6}$	$4.7_{0.3}$	$19.8_{5.9}$	1.0	1.0	1.0

There is a clear dependence of radionuclide concentration on grain size; the concentrations of the three nuclides all increase as particle size decreases. Highest concentrations were observed on the <2 μm fraction of $^{210}Pb_{ex}$. The bulk soil concentration was also used to calculate enhancement factors for each nuclide for each particle size fraction (Table 5.4.2.1).

5.4.3 Tracer values in suspended sediment runoff from the plot

The fallout nuclide concentrations of the sediments from the four plot treatments are shown in Figure 5.4.3.1. Because of the low sample masses (~5g) recovered from the plot runoff, there was considerable variation between the tracer values of the individual samples obtained from each treatment run.

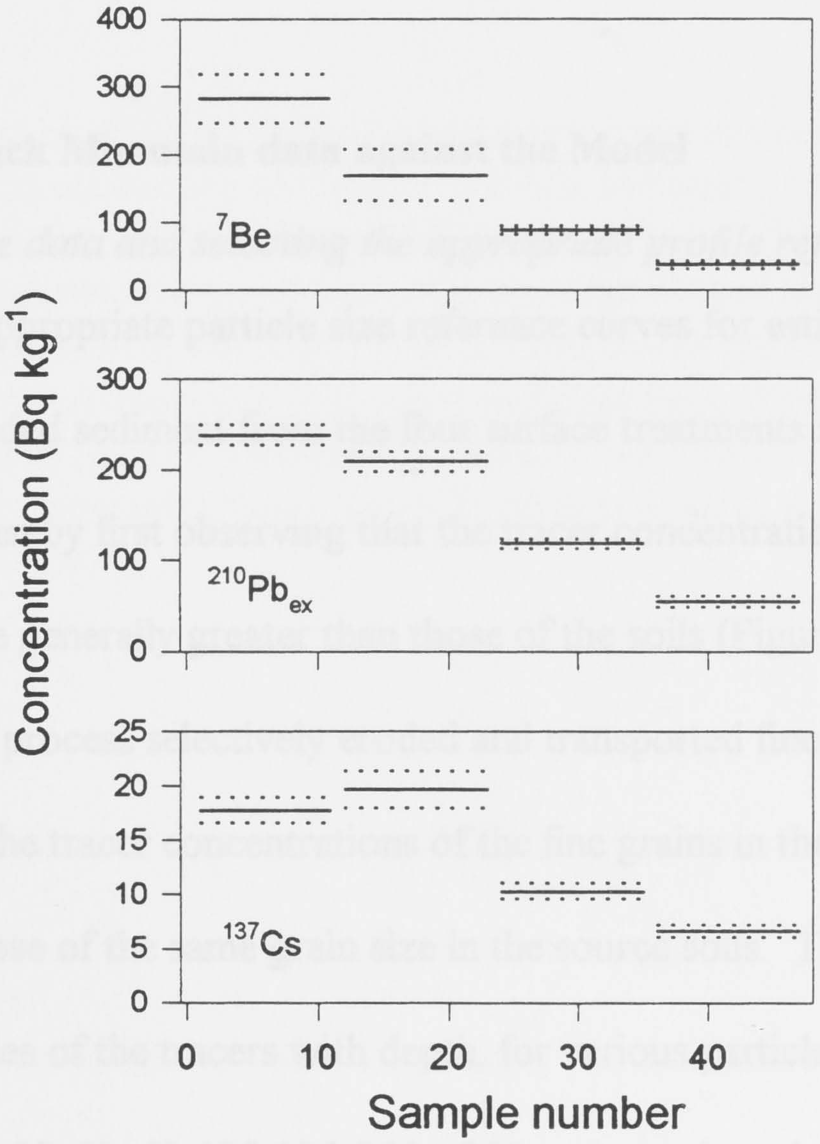

Figure 5.4.3.1: Tracer values for sediment from Treatments 1 to 4, showing the mean (solid line), and one standard error (dotted line); data from Treatment 1 (samples 1-11), Treatment 2 (samples 12-23), Treatment 3 (samples 24-35), and Treatment 4 (samples 36-47).

To improve analytical precision, all samples from each treatment were combined and then recounted as one sample. Consequently, the weighted mean and standard errors of ^{7}Be, ^{210}Pb$_{ex}$, and ^{137}Cs for these analyses have been shown plotted against sample order and plot treatment condition in this diagram. There are differences between each tracer for each treatment condition. The concentrations of ^{7}Be and ^{210}Pb$_{ex}$ are high in the suspended sediment from Treatment 1 and then progressively decrease.

The ^{7}Be concentrations decrease faster than those of ^{210}Pb$_{ex}$. The average ^{137}Cs concentrations are similar during Treatments 1 and 2 and then decrease in Treatments 3 and 4. The suspended sediment material was finer than the bulk soil, textural analysis indicated that the sediments were predominantly clay size.

5.5 Testing the Black Mountain data against the Model

5.5.1 Combining the data and selecting the appropriate profile reference curves.

In this section the appropriate particle size reference curves for estimating the depth origin of the suspended sediment from the four surface treatments are selected. The process is undertaken by first observing that the tracer concentrations of the sediments (Figure 5.4.3.1), are generally greater than those of the soils (Figure 5.4.1.2), presumably because the run-off process selectively eroded and transported finer grains. Therefore, it is appropriate that the tracer concentrations of the fine grains in the suspended sediment are compared to those of the same grain size in the source soils. The different concentration profiles of the tracers with depth, for various particle size classes (<2, 2-10, 10-20, 20-38, 38-63, 63-125,125-250 >250 μm), can be calculated using the enhancement factors given in Table 5.4.2.1. These profiles are then combined to generate the particle size reference curves (of ^{7}Be to ^{210}Pb$_{ex}$ and ^{210}Pb$_{ex}$ to ^{137}Cs) shown in

Figure 5.5.1.1a,b (note curves for grains of <38 μm, plus the bulk soil, are given only). The <2 μm particle size reference curves are the most consistent with the run-off data. As this agrees with the known particle size of the run-off sediments, these curves are used in the next section to interpret the source of the eroded sediment.

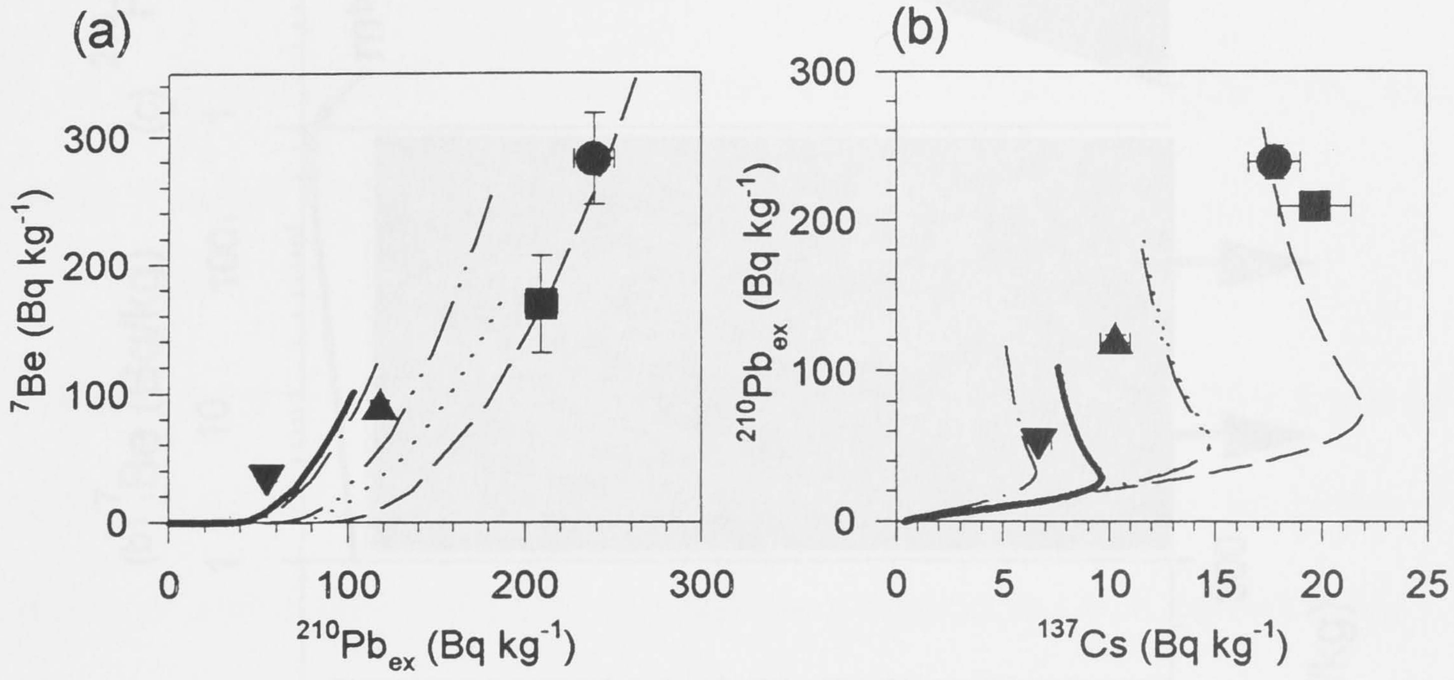

Figure 5.5.1.1 Combined profile shapes and tracer concentrations of sediment from different surface treatments at Black Mountain. (a) various profile reference curves of ^{7}Be/^{210}Pb$_{ex}$ and (b) ^{210}Pb$_{ex}$/^{137}Cs. Bulk soil concentration (solid line); <2 μm (large dashed line); 2-10 μm (dashed with 2 dots); 10-20 μm (dotted line); 20-38 μm (dashed with one dot). Symbols represent suspended sediment from: Treatment 1, surface runoff (circle); Treatment 2, shallow rills (square); Treatment 3, deep rills (upwards triangle); and Treatment 4, small gullies (downwards triangle)

5.5.2 Determining potential sources of the Treatment 3 sediment runoff using ^{7}Be and ^{210}Pb$_{ex}$

To illustrate the interpretation of sediment sources, the runoff data from Treatment 3 is first compared to the <2 μm reference curves of ^{7}Be to ^{210}Pb (Figure 5.5.2.1). The Treatment 3 tracer concentrations plot to the left of the reference curve, and if two sources are assumed, these can be defined according to Figure 5.2.2.1.

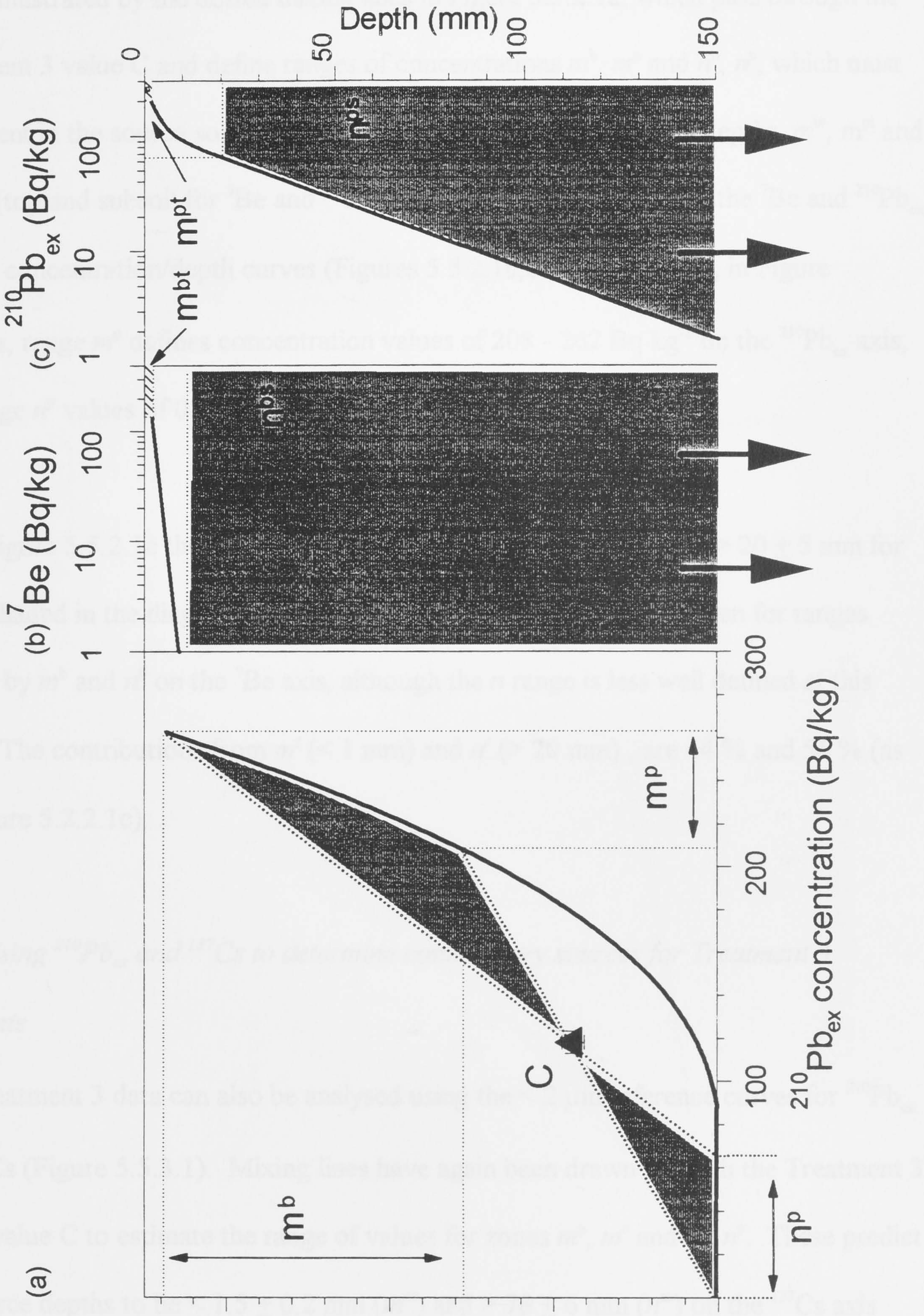

Figure 5.5.2.1 (a) Depth sources and amounts to Treatment 3 sediments at Black Mountain using ^{7}Be and ^{210}Pb$_{ex}$. (b) ^{7}Be <2 μm concentration profile with depth, and (c) ^{210}Pb$_{ex}$ <2 μm concentration profile with depth. Note: Scales for (b) and (c) given logarithmically for clarity and do not go to zero.

This is illustrated by the dotted mixing lines in Figure 5.5.2.1a, which pass through the Treatment 3 value C and define ranges of concentrations m^b, m^p and n^b, n^p, which must be present in the source soils. These ranges can be converted to soil depths, m^{bt}, m^{pt} and n^{bs}, n^{ps} (top and subsoil for ^{7}Be and $^{210}Pb_{ex}$ respectively) by reading off the ^{7}Be and $^{210}Pb_{ex}$ < 2 μm concentration/depth curves (Figures 5.5.2.1b,c). For example, in Figure 5.5.2.1a, range m^p defines concentration values of 208 - 262 Bq kg^{-1} on the $^{210}Pb_{ex}$ axis, and range n^p values of 0 - 65 Bq kg^{-1}.

From Figure 5.5.2.1c these correspond to < 1.0 ± 0.5 mm for m^{pt} and > 20 ± 5 mm for n^{ps}; as shaded in the diagrams. The same process can also be undertaken for ranges defined by m^b and n^b on the ^{7}Be axis, although the *n* range is less well defined at this depth. The contributions from m^t (< 1 mm) and n^s (> 20 mm) , are 44 % and 56 % (as per Figure 5.2.2.1c).

5.5.3 Using $^{210}Pb_{ex}$ and ^{137}Cs to determine contributory sources for Treatment 3 sediments

The Treatment 3 data can also be analysed using the < 2 μm reference curves for $^{210}Pb_{ex}$ and ^{137}Cs (Figure 5.5.3.1). Mixing lines have again been drawn through the Treatment 3 runoff value C to estimate the range of values for zones m^p, m^c and n^p, n^c. These predict the source depths to be < 1.5 ± 0.2 mm (m^{ct}) and > 78 ± 6 mm (n^{cs}) on the ^{137}Cs axis (Figure 5.5.3.1c), and < 1.2 ± 0.5 mm (m^{pt}) and > 80 ± 6 mm (n^{ps}) on the $^{210}Pb_{ex}$ axis (Figure 5.5.3.1b). This latter estimate of subsoil depth range is different from the $^{210}Pb_{ex}$ axis (n^{ps}) in the previous model (> 20 ± 5 mm).

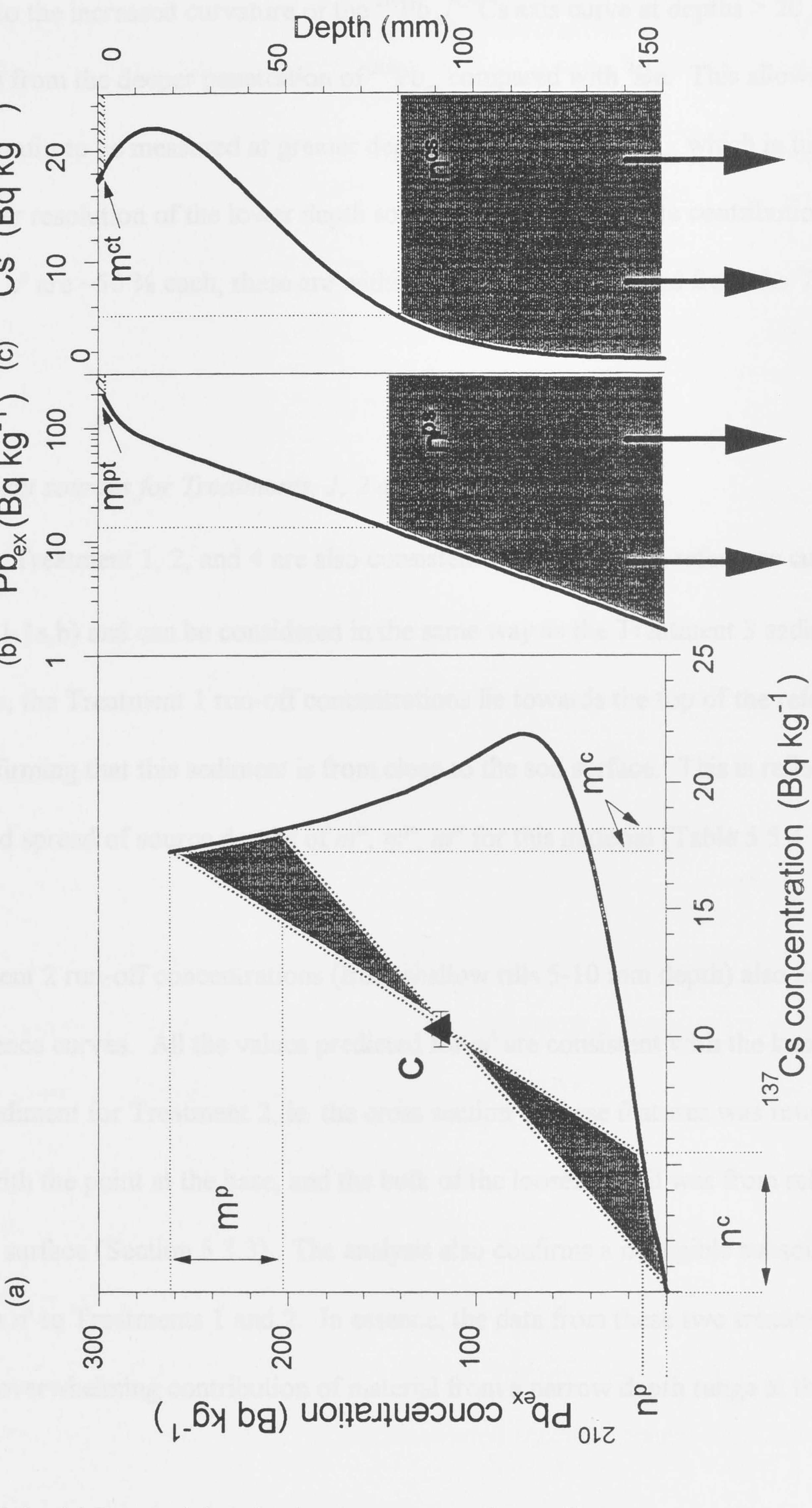

Figure 5.5.3.1: (a) Determining the depth sources and contributory amounts to Treatment 3 sediments at Black Mountain using $^{210}Pb_{ex}$ and ^{137}Cs. (b) $^{210}Pb_{ex}$ <2 μm concentration profile with depth, and (c) ^{137}Cs <2 μm concentration profile with depth. Note: $^{210}Pb_{ex}$ scale is given logarithmically for clarity and does not go to zero.

This is due to the increased curvature of the $^{210}Pb_{ex}/^{137}Cs$ axis curve at depths > 20 mm, which arises from the deeper penetration of $^{210}Pb_{ex}$ compared with ^{7}Be. This allows the $^{210}Pb_{ex}/^{137}Cs$ ratio to be measured at greater depths than for ^{7}Be /$^{210}Pb_{ex}$, which in turn permits better resolution of the lower depth source. The proportionate contributions from m^{t} and n^{s} are ~50 % each; these are within 6 % of those estimated from the ^{7}Be and $^{210}Pb_{ex}$ data.

5.5.4 Sediment sources for Treatments, 1, 2 and 4

The data for Treatment 1, 2, and 4 are also consistent with the <2 µm reference curves (Figure 5.5.1.1a,b) and can be considered in the same way as the Treatment 3 sediments. For example, the Treatment 1 run-off concentrations lie towards the top of the reference curves, confirming that this sediment is from close to the soil surface. This is reflected in the predicted spread of source depths of m^{bt}, m^{pt}, m^{ct} for this material (Table 5.5.4.1).

The Treatment 2 run-off concentrations (from shallow rills 5-10 mm depth) also lie close to the reference curves. All the values predicted for m^{t} are consistent with the known supply of sediment for Treatment 2, ie. the cross section of these features was roughly triangular with the point at the base, and the bulk of the loosened soil was from relatively close to the surface (Section 5.3.3). The analysis also confirms a negligible subsoil contribution n^{s} to Treatments 1 and 2. In essence, the data from these two treatments indicate an overwhelming contribution of material from a narrow depth range at the soil surface.

Table 5.5.4.1 Estimates of contributing depths (m^t and n^s) and their relative amounts (for m^t%) for runoff sediments generated from different surface conditions at Black Mountain. Note: Uncertainties are calculated from error bars on symbols in Figures 5.5.2.1 and 5.5.3.1.

Plot condition		Radionuclide				Relative Amount
		^{7}Be (mm)	^{210}Pb$_{ex}$ (mm)	^{210}Pb$_{ex}$ (mm)	^{137}Cs (mm)	m^t (%)
1 Overland flow	m^t	$< 0.25_{0.25}$	$< 0.25_{0.25}$	$< 0.50_{0.25}$	$< 1.0_{2.0}$	96
	n^s	$< 5.0_{10.0}$	$< 6.0_{15.0}$	--	--	
2 Shallow rills (1-10 mm)	m^t	$< 1.0_{0.5}$	$< 1.0_{0.5}$	$< 1.0_{0.5}$	$< 4.0_{4.0}$	97
	n^s	$<1.5_{10.0}$	$< 1.5_{15.0}$	--	--	
3 Deep rills (100 mm)	m^t	$< 1.2_{0.3}$	$< 1.0_{0.5}$	$< 1.25_{0.5}$	$< 1.5_{0.2}$	47
	n^s	> 10	$> 20.0_{5.0}$	$> 78.0_{6.0}$	$> 78.0_{6.0}$	
4 Small gullies (250 mm)	m^t	$< 1.2_{0.2}$	$< 1.2_{0.2}$	$< 2.5_{1.5}$	$< 2.5_{1.5}$	25
	n^s	> 10	$> 50.0_{5.0}$	$> 86.0_{6.0}$	$> 84.0_{7.0}$	

The Treatment 4 runoff concentrations (small gullies to 250 mm depth) plot to the left of the reference curves (Figures 5.5.2.1 and 5.5.3.1a,b) indicating a combination of at least two distinct sources. Estimates of contributing depths for $m^{b,p,c}$ and $n^{b,p,c}$ are given in Table 5.5.4.1. The calculated contributions of ~25% for m^t and ~75 % for n^s indicate an increased subsoil input relative to Treatment 3. This is predicted to be from the base of the gully features and from a slightly greater depth on their walls than previously (Table 5.5.4.1). This is consistent with observations that particle dislodgment and entrainment occurred within the channel bottoms. It was also noted that the channel walls became more resistant over the course of the experiment.

5.6 Discussion

5.6.1 Resolution of the Models

The ability of this method to determine the depth from which sediment originates, depends on the rate at which the concentration of the nuclides change with depth. In this respect, the greatest variations in concentration occur close to the soil surface (Figure 5.4.1.2), and this permits resolution on the particle size reference curves in this region to fractions of millimetres (see estimates for m^t given in Table 5.5.4.1). At lower depths however, concentrations of these nuclides decrease less rapidly, and below ~50 mm the resolution of the subsoil contribution is measured in millimetres to centimetres. Analytical precision has a lesser effect on the accuracy of this method. For example, due to the lower concentrations, the relative measurement error for each sample increases with depth, the lowest detectable concentration of each nuclide is ~ 1.0 Bq kg^{-1}. Thus, as depth increases the relative uncertainty on the reference curves can be viewed as a steadily widening band of increasing width. However the profiles (of Figure 5.4.1.2) describe the 'average' concentration shape of the three nuclides with depth. Similarly, sediments themselves are averaged by erosion processes and so it is appropriate to compare them to the average initial profile shapes. Again, the analytical uncertainties associated with point measurements become less important.

The effective depth range is ~10 mm for ^{7}Be, ~100 mm for ^{210}Pb$_{ex}$ and ~400 mm for ^{137}Cs. In channellised catchments, however, sediment material may be predominantly derived from > 400 mm. In this case the sediment concentrations will plot at, or very close to, the origin of the respective reference curves; the depth contributions would calculate to >10, >150 and > 400 mm, respectively.

5.7 Conclusions

A technique has been presented for analysing water-borne sediments and estimating the depth in the soil from which they originated. This method is believed appropriate to any landscape in which differences in the concentration profiles of ^{7}Be, ^{210}Pb$_{ex}$ and ^{137}Cs exist. The confounding effect of particle sorting during transport was resolved by comparing runoff concentrations with those of the same particle size in the reference soil profiles. The method successfully distinguished four different surface treatments, and was able to quantify the contribution of surface and subsurface soils to the eroded sediments derived from these. The resolution of this method for estimating the depth of sediment becomes finer towards the soil surface. In conclusion, the method is a useful tool for quantifying the effect of different erosion processes and their contribution to suspended sediment runoff at a range of scales. By implication, it should help resource managers to locate the sources of sediment and their associated pollutants. Better decisions can then be made on the merit of various measures to control them.

5.8 Summary and consideration of hypotheses

A method has been developed to determine the proportions of suspended sediment derived from different depths in the surface soil profile. The concentrations of fallout nuclides ^{7}Be, ^{210}Pb, and ^{137}Cs were found to differ with depth in soils. Consequently, the ratio of one nuclide to another also varied systematically with depth, thus providing a unique label for material from these depths. Consequently hypothesis *(10) The combined depth profiles of ^{137}Cs, ^{210}Pb$_{ex}$, and ^{7}Be provide unique concentration labels on soil material from different depths within those profiles*, is not rejected. After erosion, these ratios were retained by the sediments derived from these different depths and so the origin of the sediment could be retrospectively determined. The proportion of sediment from each depth was also calculated. The method was tested using sediment generated from a natural hillslope on which the surface was prepared using four treatments of increasing surface disturbance, providing erosion from: (i) surface runoff, (ii) shallow rills, (iii) deep rills and (iv) small gullies. The confounding effect of grain size on radionuclide concentration was overcome by comparing the tracer concentrations of the suspended sediments with those of the same particle size fraction (<2 μm) in the reference soils. The radionuclide-based calculations of source were consistent with the known incision depths of these features, and therefore hypothesis *(11) The concentration labels of* ^{137}Cs, $^{210}Pb_{ex}$, *and* ^{7}Be *on sediments from these profiles can be used to determine their depth origin(s) in the soil profile*, was also not rejected. In particular, it was deduced that material was derived from 1±2 mm and from 4 ± 4 mm depth for treatments (i) and (ii) respectively. For treatments (iii) and (iv) the material was derived from a mixture of soil from the surface (0-2.5 ± 1.5 mm) and from varying depths at the base of the rill and gully features. The subsoil contribution became larger as gully depth and width increased, being ~ 50% and 75% for treatments (iii) and (iv) respectively. Thus hypothesis *(12) Material derived from shallow rill and overland flow will exceed that from sides and floors of deeper rills and gullies*, is rejected at this site.

Chapter 6: Determining the origin of suspended sediment sources at large scales

Chapter 6: Determining the origin of suspended sediment sources at large scales

6.1 Perspective

In Chapters 4 and 5, the different depth dependencies of $^{210}Pb_{ex}$, ^{7}Be, and ^{137}Cs were used explicitly to determine the depth origin of sediment in small scale, well controlled experiments. The outcomes of these tracer predictions could be tested against the independently known erosion conditions at the time. The application of such a tracer based approach needs to be examined at larger scales. In this chapter, the different labels of $^{210}Pb_{ex}$ and ^{137}Cs in material from cultivated wheatbelt lands, pastoral lands and channel banks and gully walls are used in a simple mixing model to determine their relative contributions to sediment flux in a large river basin (>10,000 km^2). The effect on these tracer labels of processes such as direct fallout of ^{210}Pb and storage and residence of particulates in channels are also considered. The results of this analysis are then used to estimate effective sediment yields from these various sources.

The hypotheses examined in this chapter are: *(13) Different land use and land form types will produce distinct fallout radionuclide concentrations in sediments eroded from them*; *(14) Surface material from the heavily cultivated lands of the mid-Murrumbidgee will dominate the flux of material in the channels draining this area*, and; *(15) The average residence times of fine particulates in this large catchment will be longer than ten years*.

This chapter has been reviewed and submitted as: Wallbrink, P.J., Murray, A.S., Olley, J.M. and Olive L.J., Determining sources and transit times of suspended sediment within the Murrumbidgee tributaries, NSW, Australia, using fallout ^{137}Cs and ^{210}Pb, *Wat. Res. Res.*

6.2 Introduction

6.2.1 General background

The Murrumbidgee River drains one of Australia's largest (~84,000 km^2) inland basins (Figure 6.2.1.1). Although average suspended sediment concentrations in the river (typically 50 mg L^{-1}) are not high by global standards (Olive and Reiger, 1986; Foster *et al.*, 1996; Ludwig and Probst, 1996), they are perceived by residents along the river to be increasing. Several communities use the river as a primary supply of water for domestic and irrigation purposes. Domestic consumption requires the water to be clarified using alum (aluminium sulphate), and so an increase in sediment concentrations in the river has both economic and health implications.

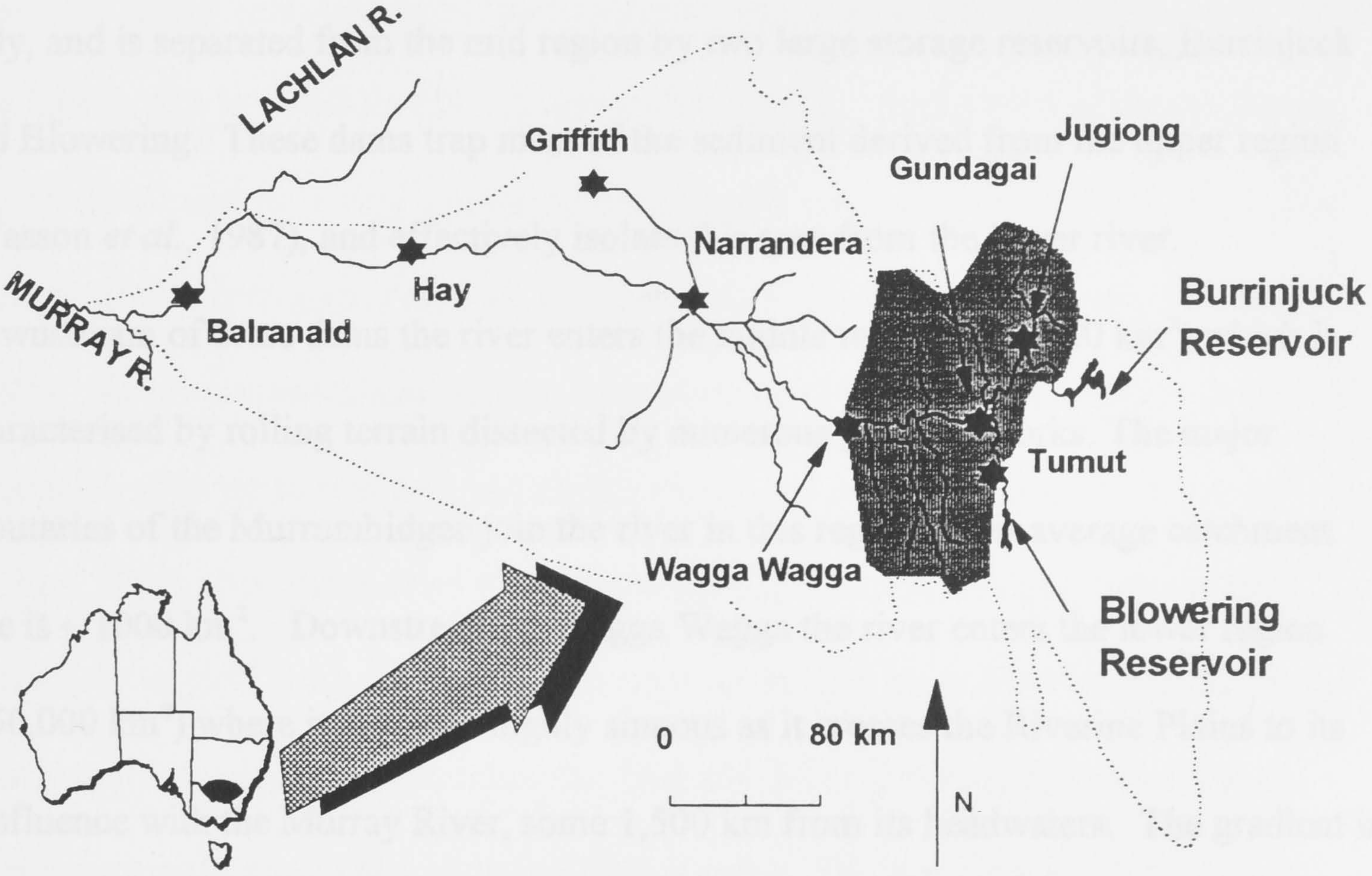

Figure 6.2.1.1 Murrumbidgee River, NSW, Australia showing location of mid-Murrumbidgee tributary catchments as shaded area.

Furthermore it is widely believed that the toxic blue-green algal blooms which occur in the river are fuelled by fertiliser phosphorus brought into the river in association with eroded surface soils. Olive *et al.* (1994) demonstrated that the suspended sediment load in the lower river is dominated by material emanating from a group of tributaries in the central part of the basin (known as the mid-Murrumbidgee, shown as the shaded area in Figure 6.2.1.1). The primary objective of this chapter is to determine which of the land use classes within the tributary catchments are the dominant source of this suspended sediment material.

6.2.2 Catchment morphology

The basin comprises three distinct geomorphic regions: the upper, mid, and lower Murrumbidgee (Page, 1994). The upper region (~20,500 km^2) is mountainous and hilly, and is separated from the mid region by two large storage reservoirs, Burrinjuck and Blowering. These dams trap most of the sediment derived from the upper region (Wasson *et al.*, 1987), and effectively isolate this area from the lower river. Downstream of these dams the river enters the middle region (~13,500 km^2) which is characterised by rolling terrain dissected by numerous gully networks. The major tributaries of the Murrumbidgee join the river in this region; their average catchment size is ~ 1000 km^2. Downstream of Wagga Wagga the river enters the lower region (~50,000 km^2) where it becomes highly sinuous as it crosses the Riverine Plains to its confluence with the Murray River, some 1,500 km from its headwaters. The gradient in this region is extremely low (<0.1%), and the climate is semi-arid.

6.2.3 Hydrology and river sediment fluxes

The Murrumbidgee River is a major supplier of irrigation water and has a series of storages and dams along its length. In low flow these break up the river below Narranderra into a series of pools. Both flow and sediment load are highly variable. Olive *et al.*, (1996) calculate that the sediment transported in a 2 week period during a one-in-12 year flood (July-August 1991) accounted for ~ 40% of the annual average sediment flux of 580,000 t at Wagga Wagga. A comparison of sedigraphs with discharge hydrographs from several stations along the river demonstrated that the tributary catchments of the mid-Murrumbidgee region (comprising ~40% of the total upstream Murrumbidgee catchment area) were the major source of this suspended sediment material.

6.2.4 Land use and erosion in the mid-Murrumbidgee catchment

The three major land uses within these tributary catchments are cultivated land for wheat and cereals (22 %), pasture (59 %), and forest (18 %). About 20% of the catchment has been assessed as having a potential for sheet/rill erosion of up to 5 t ha yr^{-1}, on the basis of aerial photography analysis and USLE modelling (Edwards *et al.*, 1989, Lucas, 1996). Significant in-channel sources have also been identified; the drainage density of channels/gullies up to 6 m deep within this region has been measured at ~2.4 m ha^{-1} (Lucas, 1996). For catchments < 10 km^2 in the upper Murrumbidgee, Sebire (1991) measured the yield from channelled landscapes as at least an order of magnitude higher than that from unchannelled catchments; Neil and Fogarty (1991) estimated that the gullies can contribute ~80% of total sediment yield. This is consistent with work in gullied catchments elsewhere in Australia (Mackenzie *et al.*, 1991); Crouch (1990) calculated that the release of material from gully sides near Bathurst, NSW, was about four times that from sheet and rill erosion. It is also consistent with measurements of up to 80% contribution from gully sources within selected basins in the United States (Glymph, 1957; Bradford and Piest, 1980).

6.2.5 Radioactive tracers ^{137}Cs and $^{210}Pb_{ex}$

The source of suspended sediments can be determined by using conservative tracers (Wall and Wilding, 1976; Oldfield *et al.*, 1979; Peart and Walling, 1986; Olley *et al.*, 1993). Burch *et al.* (1988) suggested that examination of variation in the concentrations of fallout ^{137}Cs and ^{7}Be in sediments, arising from differences in their initial soil distributions, could reveal information about the origin of that sediment. Evidence confirming the usefulness of ^{137}Cs, ^{7}Be, and ^{210}Pb$_{ex}$ to describe depth sources, and associated erosion processes, was presented in Chapters 4 and 5 (see also Wallbrink and Murray (1990; 1993) and Wallbrink *et al.* (1991) and Walling and Woodward (1992) at various basins in Devon, U.K. Alternatively, He and Owens (1995) used measurements of ^{137}Cs, ^{210}Pb$_{ex}$ and ^{226}Ra in a numerical mixing model to describe the contribution from cultivated lands, channel banks and uncultivated lands in the River Culm, U.K. Fallout ^{137}Cs and ^{210}Pb have also been used as tracers of soils and sediments in a variety of other geomorphic settings (Loughran *et al.*, 1982; Peart and Walling, 1986; Wasson *et al.*, 1987; Froelich *et al.*, 1993; Walling *et al.*, 1993; Hutchinson, 1995). A brief description of their properties is given below.

As previously elaborated in Sections 3.4.3 and 4.2.2, fallout ^{210}Pb (half-life 22 yr; also known as ^{210}Pb excess, ^{210}Pb$_{ex}$) is generated from the decay of ^{222}Rn in the atmosphere. It is continually precipitated on the soil surface by rainfall and is usually defined as the excess of ^{210}Pb activity over its parent ^{226}Ra. Anthropogenic ^{137}Cs (half-life ~30 yr) is the product of above-ground nuclear weapons testing during the 1950s-70s (Walton, 1963; Longmore *et al.*, 1983). It is also distributed approximately exponentially with depth, although often with a slight maximum below the soil surface (Walling and Bradley, 1988; Basher *et al.*, 1995; Owens *et al.*, 1996). Studies in undisturbed Australian soils have found that the majority of this nuclide (>90%) is retained within the top 10 cm of the soil (McCallan *et al.*, 1980; Campbell *et al.*, 1982; Loughran *et al.*, 1992; Wallbrink and Murray, 1993). Total soil inventories (Bq m^{-2}) in Australia are about an order of magnitude lower than those in the northern hemisphere (Longmore *et al.*, 1983). The fallout of ^{137}Cs in Australia effectively ceased by the mid 1970s, and about half of the $^{210}Pb_{ex}$ fallout currently detectable in soils has occurred in the last 20 years. Consequently, the different land forms of the tributary catchments may have concentrations of these nuclides that are distinct from one another, as a result of their differing land uses and exposure histories. The aim of this chapter is to use these differences in concentration amounts, as unique tracer properties, to determine the proportion of suspended sediment in the mid-Murrumbidgee being derived from its potential sediment sources.

6.3 Materials and methods

6.3.1 Calculating the contributions from each land use type

For simplicity the sediment sources of the mid-Murrumbidgee have been divided into three broad categories: uncultivated lands, cultivated lands, and channels/gullies. It is anticipated (from the work of Chapters 4 and 5) that the concentrations of ^{137}Cs and $^{210}Pb_{ex}$ on the suspended sediments derived from these categories will be different. In this case they can be incorporated into a simple model to determine their relative contributions to suspended sediment flux in the Murrumbidgee River. If C_u, C_c, C_b and

P_u, P_c, P_b represent the ^{137}Cs and ^{210}Pb$_{ex}$ concentrations from uncultivated, cultivated, and channel bank sources and C_s and P_s represent the respective total concentrations of ^{137}Cs and ^{210}Pb$_{ex}$ on suspended sediments, then

$$A.C_u + B.C_c + C.C_b = C_s \qquad \text{Equation (6.3.1)}$$

$$A.P_u + B.P_c + C.P_b = P_s$$

$$A + B + C = 1$$

where A, B, and C represent the relative contributions from uncultivated lands, cultivated lands and channel banks, respectively. These are the unknown quantities. The approach used to determine the radionuclide concentrations for parameters C_u, C_c, C_b, C_s, P_c, P_u, P_b, and P_s are described below.

6.3.2 Radionuclide signature of cultivated (C_c, P_c) and uncultivated lands (C_u, P_u)

The average radionuclide signature on material from contemporary sheet and shallow rill erosion from uncultivated and cultivated surfaces in the mid-Murrumbidgee is required. The minimum average depth of surface erosion across the catchment is estimated to be <0.1 mm yr^{-1}, from dividing the known annual sediment load at Wagga Wagga (~580,000 t) by the catchment area (~13,500 km^2) of the tributary catchments. However, in some locations erosion at a point will exceed this depth, and therefore a more conservative depth of <5 mm has been used to represent the average of these processes in this region. The actual material used to characterise these lands was obtained from a number of point samples which were taken at different locations and then mixed together. For example, at a given location, 5 separate samples (each of surface area ~100 cm^2, depth ~ 5 mm, mass ~ 75g) were taken randomly over an area of

~200m^2 . The process was then repeated at four additional locations, ~10 km apart, along an approximately linear transect across the land use class of interest. The samples from these five locations, (total transect length ~ 50 km, n=25) were then mixed together. The entire process was then repeated for another five transects within the same land use type, (total n = 150). The total area of each land use class represented using this sampling procedure was ~2,000 km^2. It is known that fine clays are the main particle-size fraction within the suspended sediments (Olive *et al.*, 1994) and so the <2 μm fraction was separated from the mixed soil samples by a sequential process of wet sieving and particle settling, and then analysed.

6.3.3 Radionuclide signature of channel and gully source (C_b, P_b)

Subsoil material generated from processes such as slumping, tunnelling, fluting, toppling and scouring in channels and gullies is not exposed to direct radioactive fallout before erosion occurs, and so contain no $^{210}Pb_{ex}$ or ^{137}Cs. However, the channels and gullies of the Murrumbidgee tributaries typically have near vertical sides. As these retreat, a small amount of topsoil, labelled with ^{137}Cs and $^{210}Pb_{ex}$, enters the channel, which results in measurable labels of these nuclides on the detached sediments. The net radionuclide signature on this material (C_b and P_b) was determined by dividing the tracer concentrations in the fine clays of the topsoil (defined as the region directly labelled by $^{210}Pb_{ex}$ and ^{137}Cs, i.e. the top 200 mm), by the average depth of the channel bank or gully wall beneath it. The tracer concentrations of the topsoil was calculated from the clay fraction of a series of mixed cores taken to a depth of ~200 mm at the point locations used for characterising the land use types (n=60). The average height of channel banks and gullies within the tributaries was estimated to be ~3m (Lucas, 1996).

6.3.4 Radionuclide signature of suspended sediments (C_s, P_s)

Sampling points for suspended sediments included the outlets of all the major tributary catchments (Jugiong, Tumut, Kyeamba, Tarcutta, Adelong, Billabung and Hillas) and from points along the Murrumbidgee River channel from the tributary confluences to Wagga Wagga. Sampling was undertaken during both low flow and flood conditions over a period of 3 years (1991-93). Samples were obtained using a continuous flow centrifuge (Alfa Laval, Model No. MAB103B). Water was supplied to the centrifuge at a rate of ~ 5 L min^{-1} for periods up to 2-3 hours, and masses between 5 - 300 g were recovered, depending on suspended sediment concentrations. The mean particle size of the suspended sediments was < 3 μm measured using a Sedigraph 5100™ automated particle size analyser (Jones and Bowser, 1988).

6.4 Analytical procedure

All samples were oven dried, ashed at 400° C, and analysed by gamma spectrometry for ^{137}Cs, ^{226}Ra, and ^{210}Pb as described by Murray *et al.* (1987). Small mass samples (~1-10g) were analysed in a high purity germanium well detector; and counted for a minimum of 170 ksec. Larger mass samples (up to 250g) were counted on HpGe planar or 'n' type coaxial detectors for a minimum of 84 ksec. Unless otherwise noted, all mean values are reported with the associated standard error.

6.5 Results

6.5.1 Radionuclide concentrations from different land use and land form types

The average radionuclide concentrations of particulates from the three defined source locations, and the suspended sediments, are given in Table 6.5.1.1. The highest concentrations occur in soils from uncultivated lands, $C_u = 29 \pm 3$ and $P_u = 270 \pm 26$ Bq kg^{-1} (^{137}Cs and ^{210}Pb$_{ex}$ respectively). These are comparable with similar grain size material from grazing land at Whiteheads Creek, NSW, ~30 Bq kg^{-1} for ^{137}Cs and ~300 Bq kg^{-1} for ^{210}Pb$_{ex}$, given in Chapter 4 (ie. Wallbrink and Murray, 1993), but are slightly higher than those from the undisturbed surface at Black Mountain, ACT, ~ 18 Bq kg^{-1} and 240 Bq kg^{-1} (^{137}Cs and ^{210}Pb$_{ex}$ respectively; see Chapter 5). It should be noted however, that the surface area of these two sites was only ~350 and 150 m^2 respectively; it is known from Chapter 2 that areal concentrations of ^{137}Cs, in particular, can be variable at this scale (see also Fredericks *et al.*, 1988; Sutherland, 1991; Wallbrink *et al.*, 1994; Owens and Walling, 1996).

Table 6.5.1.1 Radionuclide concentrations in <2µm material from various land use and land form types in the tributaries of the mid-Murrumbidgee catchments. Note: Subscripts are standard errors on the mean.

Location	Radionuclide	Code	Concentration (Bq kg^{-1})	(n)
Uncultivated lands	^{137}Cs	C_u	29_{3}	150
Cultivated lands		C_c	18_{2}	150
Channel/gully banks		C_b	$0.6_{0.1}$	60
Suspended sediments		C_s	$2.9_{0.2}$	71
Uncultivated lands	^{210}Pb$_{ex}$	P_u	$270_{26.0}$	150
Cultivated lands		P_c	$120_{5.9}$	150
Channel/gully banks		P_b	$2.7_{0.3}$	60
Suspended sediments		P_s	23_{4}	71

The concentrations of ^{137}Cs and ^{210}Pb$_{ex}$ on the fine grained material from the cultivated lands, C_c and P_c (Table 6.5.1.1) are significantly lower than from the uncultivated soils. However, they are higher than might be expected from surfaces that are continuously ploughed (and thus vertically mixed). This is because cultivation in this region generally involves tillage by disc harrows; these have a series of discs perpendicular to one another which break the soil to a depth of ~150 mm and move it from side to side over a horizontal distance of ~300 mm. There appears to be relatively little vertical mixing. Analysis of sectioned cores from these areas show that the initial profile shape of ^{137}Cs is still partially maintained (Section 7.4.1) confirming that vertical mixing is not complete. In contrast, the concentrations on conventionally ploughed lands tend to be uniform to the bottom of the plough layer (Ritchie and McHenry, 1973; Walling and Bradley, 1988; Owens *et al.,* 1996).

The tracer concentrations derived for channel and gully sources (C_b, P_b) are lower than those of cultivated and uncultivated lands (Table 6.5.1.1) and also lower than the suspended sediment values. The latter concentrations range from -0.5 ±1.4 to 6.6 ± 0.9 Bq kg^{-1} for ^{137}Cs, and -4 ± 5 to 109 ± 22 Bq kg^{-1} for ^{210}Pb$_{ex}$, and presumably reflect changes in contribution amounts occurring from the different sources within the catchment at different times and conditions. In particular, the lowest concentrations (of both ^{137}Cs and ^{210}Pb$_{ex}$) were generally associated with flood sediments, as is discussed later, implying that the contribution from in-channel sources increases under these conditions.

6.5.2 Initial calculation of relative contributions from the three potential sources

The concentrations of ^{137}Cs and ^{210}Pb$_{ex}$ on material from the cultivated and uncultivated lands and channel/gully subsoils C_c, P_c, C_u, P_u, C_b and P_b are different, both from one another and from the suspended sediments (C_s, P_s; Table 6.5.1.1). Using Equation 6.3.1, they provide estimates for A, B, and C of 6% (uncultivated), 4% (cultivated) and 90% (channel banks) respectively. In the above calculations we have assumed that ^{137}Cs and

$^{210}Pb_{ex}$ are acting conservatively ie. the concentrations on sediments do not change after initial mobilisation from the soil. However, it is possible that the radionuclide concentrations used to describe the various land uses are being modified by other processes operating within the fluvial system. These are examined in more detail below.

6.6 Discussion

6.6.1 Assumptions made in this analysis

In estimating A, B, and C it was assumed that no major physical or chemical alteration to particulates occurs during transport and that delivery of material from the sources to the tributary confluences is instantaneous. It was also assumed that when sediment enters a river channel it brings with it radionuclide concentrations that have been characterised by the source signature sampling method. In this respect, the desorption of ^{137}Cs from sediments has been observed in saline environments such as estuaries (Zucker *et al.*, 1984; Santschi *et al.*, 1983) or where concentrations of ^{137}Cs have been sufficient to saturate available adsorption sites on clay lattices (Stanners and Aston, 1981). However, the amount of ^{137}Cs on soils and sediments of the Murrumbidgee is relatively small, and would be easily accommodated by adsorption sites within clay lattices (Sawhney, 1970; Coman and Hockley, 1992). The waters of the Murrumbidgee also have very low salinities, and at these low concentrations the ^{137}Cs should remain strongly adsorbed onto the sediments (Evans *et al.*, 1983; Singh and Gilkes, 1990). ^{210}Pb has been shown to be more particle reactive than ^{137}Cs (Santschi *et al.*, 1983), and so should remain tightly bound under the conditions outlined here.

However, if the particles have been resident within the tributaries for a length of time that is comparable with the ^{210}Pb half life of 22 yr then the $^{210}Pb_{ex}$ sediment label may have been altered by both direct atmospheric fallout and radioactive decay. The ^{137}Cs label will be unaffected by this process (for transit times of < 25 yrs) because most was deposited > 30 yrs ago. Concentration change (through radioactive decay) is thus simultaneous and at the same rate in both soils and sediments. The effect on ^{210}Pb$_{ex}$ sediment concentrations by i) labelling by direct fallout and ii) radioactive decay, are considered separately below.

6.6.2 The direct addition of ^{210}Pb to sediments in channels from fallout

Once sediment enters the channel system it can acquire additional ^{210}Pb from fallout, but not ^{137}Cs. However, as the cumulative amount of ^{210}Pb which has fallen directly into the channel increases downstream, so too does the amount of sediment transported within the channel. In principle the net effect of these contributions can give rise to either an increase or decrease in the ^{210}Pb$_{ex}$ concentrations on transported sediments, depending on the relative size of sediment and ^{210}Pb delivery.

An estimate of the significance of this process begins by deriving a drainage density (Figure 6.6.2.1a) for the average-sized catchments of the mid-Murrumbidgee (~1000 km^2). This is best approximated by a power function of the form $d = aA^b$; where d is drainage density (m ha^{-1}); A is stream-order basin area (ha) and a and b are constants (10.3 and -0.13 respectively). These constants are derived from the relationship between the average area of the different stream-order basins in the mid-Murrumbidgee and the cumulative lengths of channels within each basin area. The average size for the fifth-order basins was measured by planimeter and then estimated for basins of decreasing order using Horton's (1945) law of basin areas in which the area ratio was assumed to be 4.5 (I.D. Prosser, 1996, pers. comm).

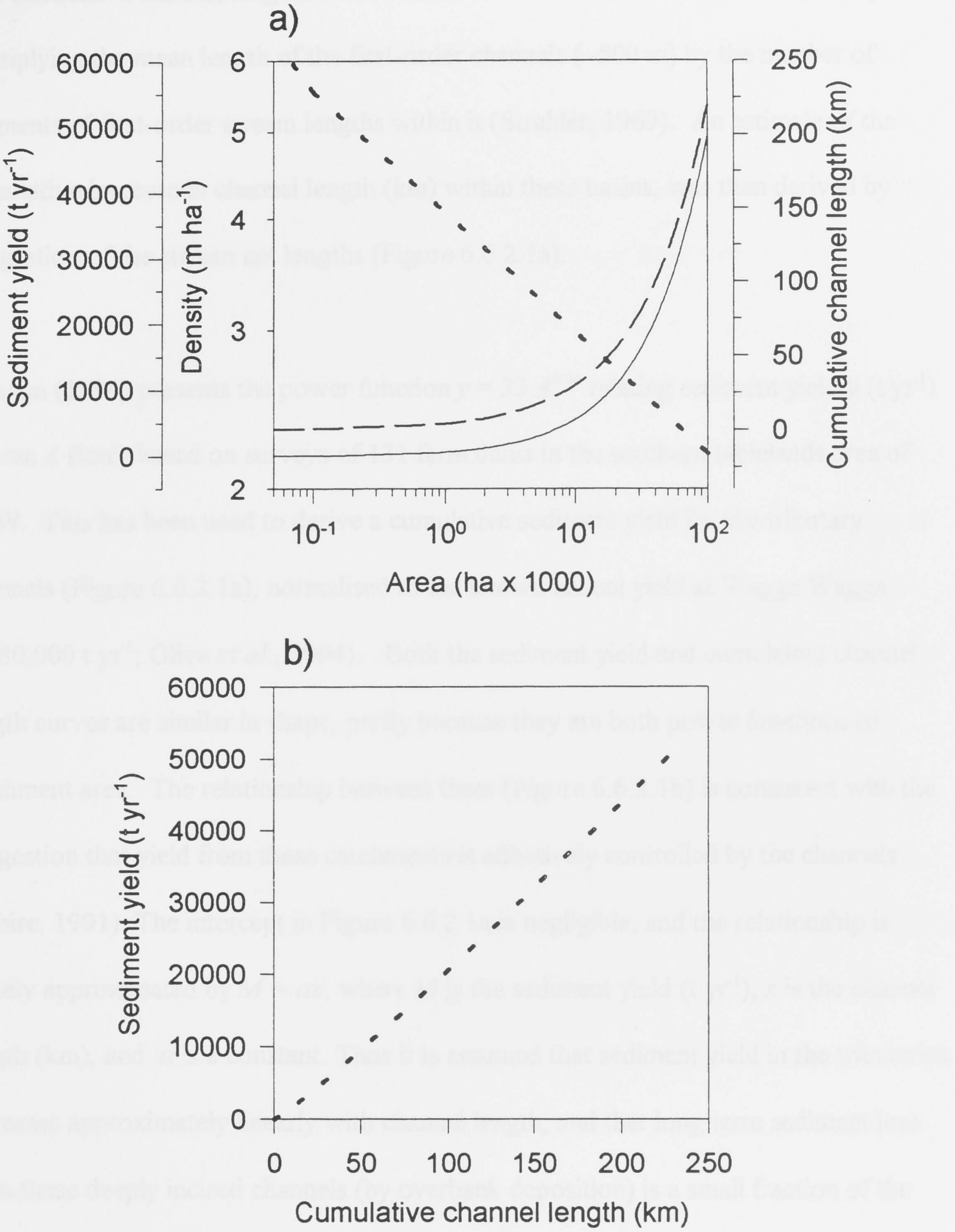

Figure 6.6.2.1 (a) Calculated drainage density (dotted line); cumulative stream length, (dashed line); and cumulative sediment yield, (solid line), for the mid-Murrumbidgee tributaries, NSW. (b) Relationship between stream length and sediment yield within mid-Murrumbidgee tributaries.

The cumulative channel lengths within basins of each order were then estimated by multiplying the mean length of the first-order channels (~500 m) by the number of segments of first-order stream lengths within it (Strahler, 1969). An estimate of the cumulative increase in channel length (km) within these basins, was then derived by integration of the stream net lengths (Figure 6.6.2.1a).

Wasson (1994) presents the power function $y = 33\ A^{0.94}$ relating sediment yield y (t yr^{-1}) to area A (km^2) based on surveys of 131 farm dams in the southern tablelands area of NSW. This has been used to derive a cumulative sediment yield for the tributary channels (Figure 6.6.2.1a), normalised to the known annual yield at Wagga Wagga (~580,000 t yr^{-1}; Olive *et al.*, 1994). Both the sediment yield and cumulative channel length curves are similar in shape, partly because they are both power functions of catchment area. The relationship between them (Figure 6.6.2.1b) is consistent with the suggestion that yield from these catchments is effectively controlled by the channels (Sebire, 1991). The intercept in Figure 6.6.2.1a is negligible, and the relationship is closely approximated by $M = ax$, where M is the sediment yield (t yr^{-1}), x is the channel length (km), and a is a constant. Thus it is assumed that sediment yield in the tributaries increases approximately linearly with channel length, and that long term sediment loss from these deeply incised channels (by overbank deposition) is a small fraction of the total.

Since the ^{210}Pb fallout rate (Bq m^{-2} yr^{-1}) can be assumed spatially constant, the total amount of ^{210}Pb that falls within the channels, P (Bq yr^{-1}), must be a linear function of their length, (assuming an approximately constant channel width). There can be no in-channel fallout if the channel length is zero, and so $P = bwx$, is used where P is the cumulative ^{210}Pb fallout (Bq yr^{-1}), w is the channel width (4 m, assumed constant), x is the channel distance (km) and b is a constant. Then the concentration at any point in the channel system, is $P/M = bw/a$, which is independent of time and distance. Therefore, under the assumptions given here, at steady state the concentration of ^{210}Pb added from fallout to the channel sediments, as a function of distance, is constant.

The annual amount of fallout ^{210}Pb in this region, b, is estimated to be 64 ± 8 Bq m^{-2} yr^{-1}; based on the inventories of ^{210}Pb$_{ex}$ in cores taken in undisturbed soils from the tributary catchments (2,100 ± 73 Bq m^{-2}, n=55; see Moore and Poet (1976) for a description of the methodology involved). This is consistent with a value of ~ 66 Bq m^{-2} yr^{-1} at this latitude, estimated from Turekian *et al.* (1977). An estimate of a, the average amount of sediment moving within the channel network, is ~157 kg m^{-1} yr^{-1} based on the cumulative channel length of Figure 6.6.2.1a, and the annual sediment load at Wagga Wagga. If it is then assumed that the channel system is in steady state, ie. the annual flux of sediment entering the channel system equals the amount leaving, then the addition of ^{210}Pb by direct deposition is $4b/a$, 1.6 Bq kg^{-1}. This is independent of position in the channel and residence time. However, it does not allow for radioactive decay of this added component; which will tend to reduce the concentration on this figure, depending on residence time.

It is therefore expected that the ^{210}Pb$_{ex}$ concentration of the suspended sediments will increase by 1.6 Bq kg^{-1} (neglecting radioactive decay) on average, while in transit within the tributary channels. This addition represents a small correction (~7%) to the observed concentration of ^{210}Pb$_{ex}$ in channel sediments (P$_s$) in Table 6.5.1.1.

6.6.3 Radioactive decay and residence times: their effect on the relative contributions from catchment sources.

If sediments remain in channels for a period comparable to the half-life of ^{210}Pb then radioactive decay of the catchment derived $^{210}Pb_{ex}$ signal could be important. (Although it should be noted that the actual $^{210}Pb_{ex}$ signature in the catchment itself does not decay; it is continually replenished by fallout). The $^{210}Pb_{ex}$ values for P_s have been corrected for radioactive decay as a function of residence time, by increasing their concentrations according to the half life of ^{210}Pb for a postulated number of years of 'residence'. A constant value (1.6 Bq kg^{-1} yr^{-1}) was then subtracted from P_s, to account for the addition of $^{210}Pb_{ex}$ from direct fallout, as described above. A second order correction for the decay of the added component, arising because fallout is continuous during the decay period, has been ignored. These data have been used with Equation (6.3.1) to derive Figure 6.6.3.1. The estimated contributions from uncultivated (A), cultivated (B), and channel banks (C) at time = 0 yr is now 4, 7, and 89% respectively. The total contribution from surface sources (cultivated and uncultivated) remains about 10%; the overwhelming contribution is predicted to be from in-channel sources.

It can also be seen that for a residence time of six years the predicted contribution from cultivated lands reaches 0%; residence times beyond this (for which a < 0% cultivated input is predicted) have no physical basis. This predicted short transit time for fine particulates is supported by the difference observed in ^{137}Cs concentrations on suspended sediments in the main channel carried within low flow and flood conditions. For example, the average ^{137}Cs concentration of flood sediments is 2.1 ± 0.1 Bq kg^{-1} (n=116) whilst that from low flow is 3.0 ± 0.3 Bq kg^{-1} (n=47). (Note: this also includes samples from along the main channel downstream of Wagga Wagga).

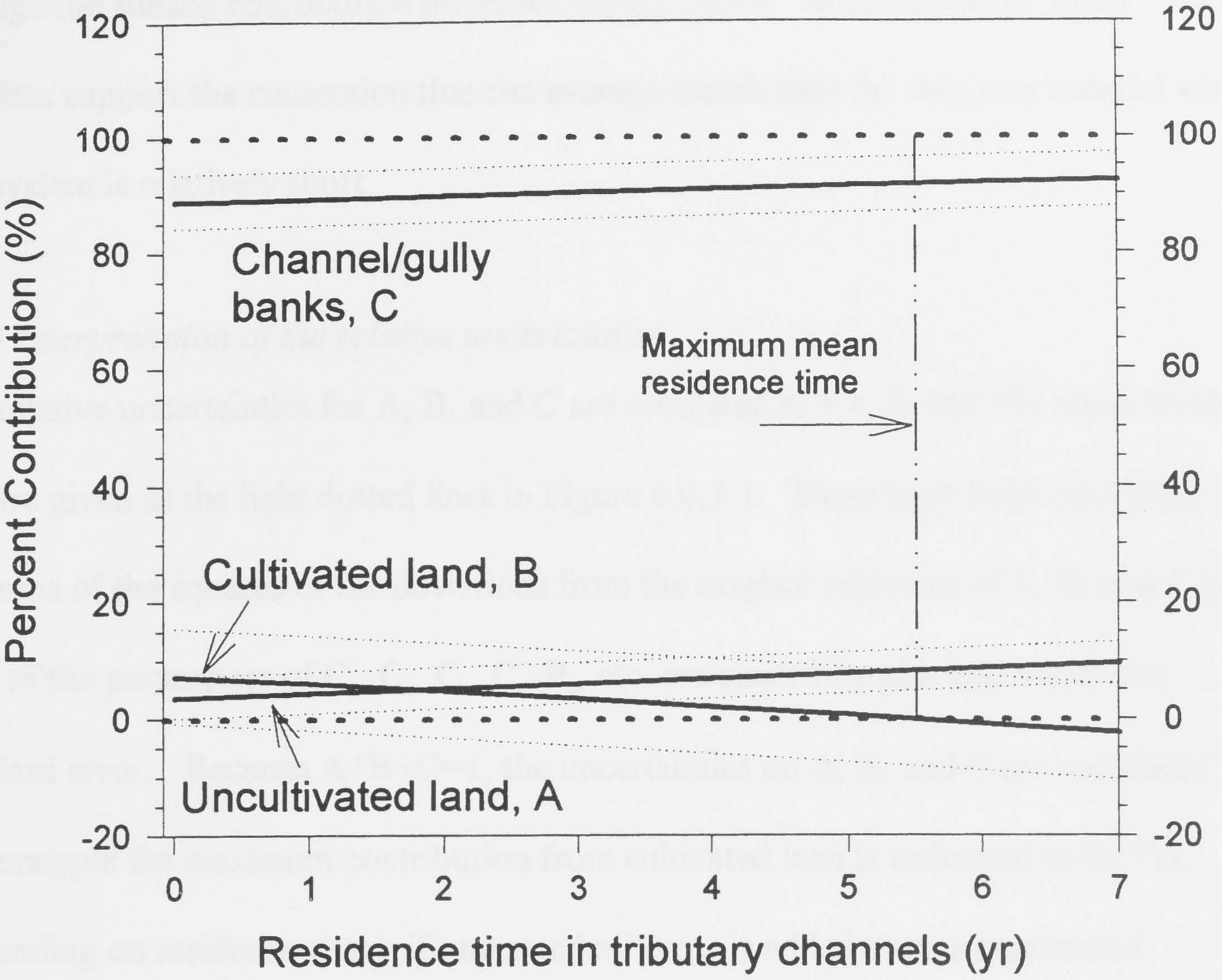

Figure 6.6.3.1 Relative contributions from channel banks, cultivated, and uncultivated land sources estimated as a function of increasing residence time.

This suggests that, on average, the contribution from subsoils increases during flood events, and more importantly that this effect is observable within the time period of the flood itself, i.e. within about a 4 week period. Furthermore, finite concentrations (up to 330 ± 40 Bq kg^{-1}) of the cosmogenic ^{7}Be were measured on some of the flood sediment samples (5 out of 116). This nuclide is also an indicator of surface soils, Chapters 4 and 5 (see also, Burch *et al.*, 1988; Wallbrink and Murray, 1990;91), but its half life is only 53 days. Typical concentrations of this nuclide on fines in Australian surface soils range from 150-350 Bq kg^{-1} , see Figure 4.4.1.3 (given in Wallbrink and Murray, 1993; 1996a). For these flood sediments to contain the high concentrations observed, their transport time must be in the order of a few weeks to months. It is also interesting to note that on occasion they must contain a significant surface soil contribution, even though, on average the subsoil contribution increases during floods. In combination, these ^{137}Cs and ^{7}Be data support the contention that the average transit time for clay size material within this system is relatively short.

6.6.4 Interpretation of the relative uncertainties

The relative uncertainties for A, B, and C are estimated at $\pm$ 4, 8, and 5% respectively, and are given as the light dotted lines in Figure 6.6.3.1. These have been calculated from the sums of the squares of the deviations from the original solutions of A, B, and C when each of the parameters of C_u, C_c, C_b ,C_s, P_u, etc. are altered by plus and minus one standard error. Because A+B+C=1, the uncertainties on A, B, and C are correlated. For example the maximum contribution from cultivated land is estimated to be 7%, (depending on residence time); if one standard error is added then the estimated contribution from this source becomes ~ 15 %. However, the contributions from the other sources must still sum to 100% and so their contributions decrease and give estimates from uncultivated and channel sources of -1 and 86% respectively. Such mixtures are not physically possible, and so set additional bounds on uncertainties.

6.6.5 Sensitivity of the model to variability in model input parameters

The sensitivity of the model to changes in C_u, C_c, C_b, C_s, P_u, etc. has been examined by adding and subtracting up to three standard errors on each parameter, while holding the other values constant (Figure 6.6.5.1). This has been undertaken using an average residence time of 3 years. In each case the changes in the estimates of A, B, and C sum to zero. The greatest changes in values of A, B, and C are associated with variations in C_s and P_s (suspended sediment), in particular the ^{210}Pb$_{ex}$ concentration The model is least sensitive to variations in ^{137}Cs and ^{210}Pb$_{ex}$ concentrations from cultivated lands and channel banks, C_c, C_b, P_c, and P_b. Generally, the estimated relative contribution of B (cultivated land) is the most affected; this is because the measured concentrations from this source lie closest to those of the suspended sediments.

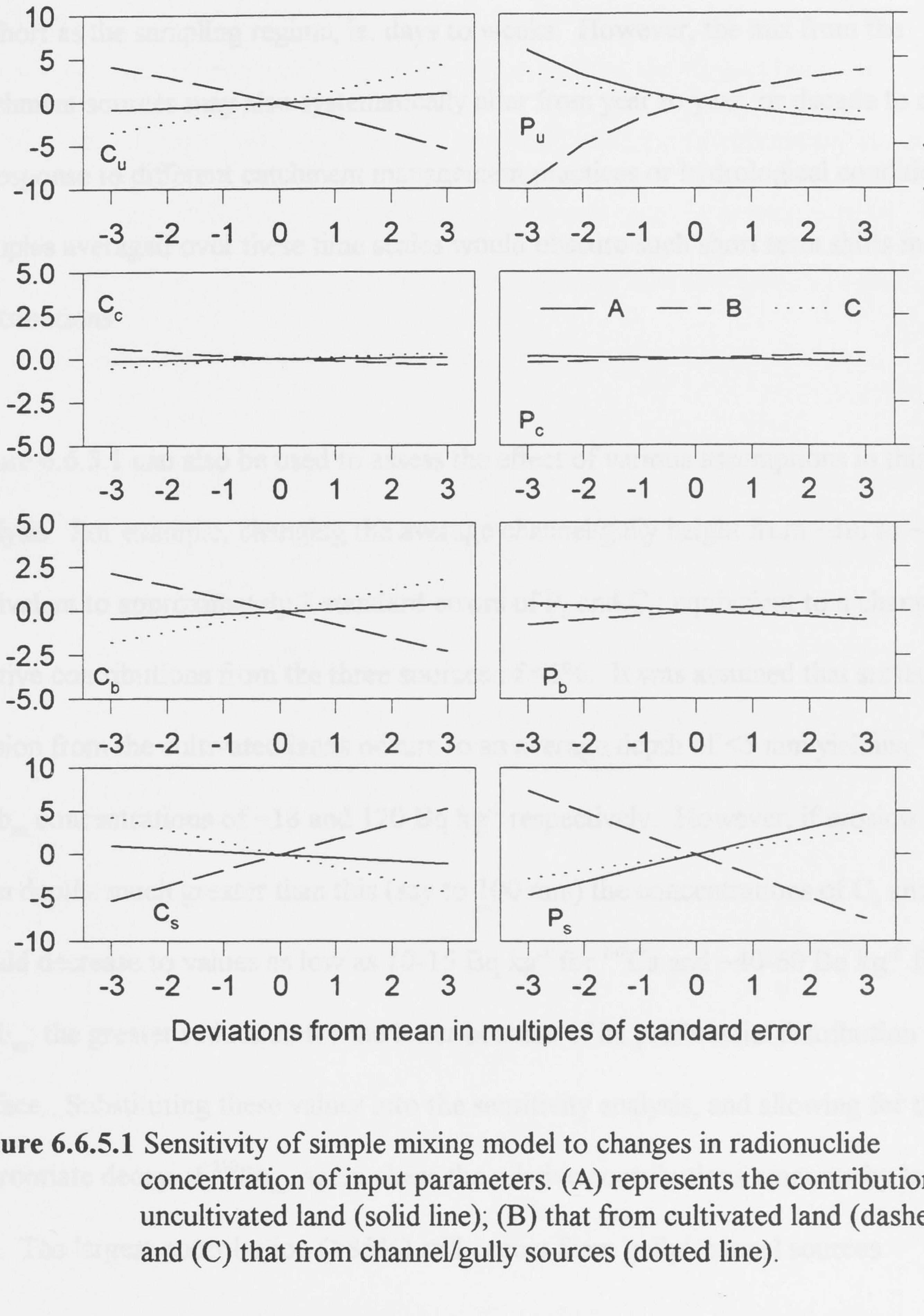

Figure 6.6.5.1 Sensitivity of simple mixing model to changes in radionuclide concentration of input parameters. (A) represents the contribution from uncultivated land (solid line); (B) that from cultivated land (dashed line); and (C) that from channel/gully sources (dotted line).

It is possible that the model predictions of source contribution could be improved by better estimating the ^{137}Cs and ^{210}Pb$_{ex}$ concentrations (C_s and P_s) on suspended sediments. However, these estimates of C_s and P_s already represent the average of many (n=71) samples over a 3 year period. It must also be remembered that sediment delivery within the catchment is a dynamic process. For example, the variations observed in P_s and C_s, imply that the mix of sediment from the various sources must vary over periods at least as short as the sampling regime, ie. days to weeks. However, the mix from the catchment sources may also systematically alter from year to year, or decade to decade, in response to different catchment management practices or hydrological conditions. Samples averaged over these time scales would obscure such short term shifts in source contributions.

Figure 6.6.5.1 can also be used to assess the effect of various assumptions in this analysis. For example, changing the average channel/gully height from ~3m to ~2m is equivalent to approximately 3 standard errors of P_b and C_b; equivalent to a change in the relative contributions from the three sources of <5%. It was assumed that surface erosion from the cultivated lands occurs to an average depth of <5 mm yielding ^{137}Cs and ^{210}Pb$_{ex}$ concentrations of ~18 and 120 Bq kg^{-1} respectively. However, if erosion occurred from depths much greater than this (say to 100 mm) the concentrations of C_c and P_c would decrease to values as low as 10-15 Bq kg^{-1} for ^{137}Cs and ~40-60 Bq kg^{-1} for ^{210}Pb$_{ex}$; the greater reduction for the latter because of its preferential distribution near the surface. Substituting these values into the sensitivity analysis, and allowing for the appropriate decay of ^{210}Pb$_{ex}$, again alters the relative contributions amounts by less than 5%. The largest contribution (>85%) still occurs from gully/channel sources.

The steady state addition of ^{210}Pb to P_s by direct fallout was calculated to be ~1.6 Bq kg^{-1}, in Section 6.6.2. It should be noted that 3 times the standard error of the P_s value (ie. ± 4; Table 6.5.1.1) is equivalent to ± 12 Bq kg^{-1}. This is at least a factor of 7 greater than that calculated from direct fallout. From Figure 6.6.5.1, this range is equivalent to a total change in contributions from the various sources of only <15%.

6.6.6 Specific sediment yield from cultivate and uncultivated lands and channel banks

The tracer based estimates of relative contribution from the various land uses and land forms of the tributary catchments can be combined with the known annual sediment yield at Wagga Wagga to provide a quantitative estimate of yield from each source. These are summarised in Table 6.6.6.1 against years of residence, and converted to specific sediment yields from the known surface area of each land use type.

Table 6.6.6.1 Estimated relative contributions and sediment yields from uncultivated, cultivated lands, and from channel/gully banks, for different sediment residence times in tributary channels of the mid-Murrumbidgee. Based on a total sediment load of ~580,000 t yr^{-1} and total channel length of ~3,300 km.

	Contribution (%) Yield (t x 1000) Specific yield [t ha^{-1} yr^{-1}]		
Residence time (yr)	Uncultivated, A	Cultivated, B	Channel/gully banks, C
0	4	7	89
	23.2 [0.02]	40.6 [0.14]	516.2 [0.38]
1	5	6	89
	29.0 [0.03]	34.8 [0.12]	516.2 [0.38]
2	5	5	90
	29.0 [0.03]	29.0 [0.10]	522.2 [0.39]
3	6	4	90
	34.8 [0.03]	23.2 [0.08]	522.2 [0.39]
4	7	2	91
	40.6 [0.04]	11.6 [0.04]	527.8 [0.40]
5	8	1	91
	46.4 [0.04]	5.8 [0.02]	527.8 [0.40]

For example, the annual yield of sediment from uncultivated sources (~23,200 t yr^{-1}) within the first year provides a specific sediment yield of ~0.02 t ha^{-1} yr, which is approximately a factor of 6 less than that from cultivated lands of ~0.14 t ha^{-1} yr^{-1}. It should be noted that these loss rates represent material actually delivered to the stream network. Presumably a substantially larger amount of soil is mobilised, but remains in storage on the slope (Walling, 1983).

The estimated loss from channel/gully sources of 0.38 t ha^{-1} yr^{-1} is consistent with values of 0.2 to 0.5 (m^3 ha^{-1} yr^{-1}) from banks/gullies in some European loam and loess soils (Poesen 1989; 1993), although they are about an order of magnitude lower than that (2.3 - 5.1 m^3 ha^{-1} yr^{-1}) from ephemeral gullying in some Mediterranean environments (Poesen *et al.*, 1996). It is also equivalent to a minimum annual linear lateral retreat of ~ 18 mm (depending on clay content of the banks), over the period of suspended sediment measurement. However, this does not take into account the significant crenulations (Crouch, 1990) observed on gully walls within the tributaries. A crenulation ratio of 2 for these features would decrease the estimated linear lateral retreat by half.

6.7 Conclusions

Olive *et al.* (1994) showed that the suspended sediment in the Murrumbidgee River was predominantly derived from the tributaries of the mid-Murrumbidgee catchments upstream of Wagga Wagga. By using the fallout tracers ^{137}Cs and ^{210}Pb$_{ex}$ it has been demonstrated that the dominant source of material (~90 %) from within these tributaries is channel banks and gully walls. This observation is consistent with Neil and Fogarty's (1991) and Wallbrink and Murray's (1993) conclusion (from Chapter 4), that channel sediments dominate flux at smaller scales in the catchments' upper region. The tracer-based estimates of contributions from this source are also consistent with results of Osborn and Simanton (1989) who describe the substantial influence on yield by subsoils in U.S. catchments where channelling occurs. Fallout ^{210}Pb data was also used to show that the mean residence times of fine grain sediments within the mid-Murrumbidgee system is unlikely to be more than a few years. Additional evidence from (i) changes in the mean ^{137}Cs concentrations between sediments from flood and low flow waters, and (ii) the presence of the short-lived tracer ^{7}Be in flood water sediments, implies that the actual transport time of fines in this river is probably weeks to months, rather than years to decades. In conclusion, analysis of fallout nuclides has substantially improved our understanding of the Murrumbidgee River system. Not only have they provided a powerful insight into the source of sediment in the mid-Murrumbidgee catchments, they have placed realistic boundaries on the transit times of material within it.

6.8 Summary and consideration of hypotheses

The tributary catchments of the mid Murrumbidgee, (~13,500 km^2) have been identified as the major contributors of fine grained sediment to the Murrumbidgee River (total catchment area ~84,000 km^2). Potential sediment sources within the mid-Murrumbidgee tributaries include cultivated lands, occupying ~22% of the surface area, uncultivated pastoral lands, occupying ~78% of the area, and the numerous channels and gullies throughout this region. The concentrations of the fallout radionuclides ^{210}Pb$_{ex}$ and ^{137}Cs were measured in these three different sources and were found to be different in each of them. Consequently, hypothesis *(13) Different land use and land form types will* *produce distinct fallout radionuclide concentrations in sediments eroded from them*, is not rejected. Comparison of these different tracer concentrations with those of the sediments in a simple mixing model suggested a dominant contribution of material from subsoil channel/gully sources. Therefore, hypothesis *(14) Surface material from the heavily cultivated lands of the mid- Murrumbidgee will dominate the flux of material in the channels draining this area*, is rejected. Alterations to the suspended sediment ^{210}Pb signature by *in situ* labelling and decay due to residence time have also been considered. The addition of ^{210}Pb from direct fallout was found to be small. Applying the model to various scenarios of decay of ^{210}Pb (as a function of residence time in channels) suggested that the fine grained material is stored, on average, for less than 6 years. On the basis of this analysis, hypothesis *(15) The average residence times of fine particulates in this large catchment will be longer than ten years*, is also rejected. Additional evidence for the short residence time of fine grained material in these systems came from variations in the mean ^{137}Cs concentrations observed between flood and low flow, and the presence of the short lived isotope ^{7}Be on flood sediments.

Chapter 7: Discussion and synthesis of the work in this thesis

Chapter 7: Discussion and synthesis of the work in this thesis

7.1 Perspective

The two main themes of this thesis are the measurement of soil erosion, and tracing the source of suspended sediments. Chapters 2 and 3 presented improved techniques to measure surface soil erosion by accounting for the initial variability of fallout radionuclides. Chapters 4, 5 and 6 presented methods for determining the amounts contributed by different sources to suspended sediments, using the different concentrations and depth dependencies of fallout nuclides. In this chapter, some of these techniques are discussed in the context of the other study areas, to examine the potential breadth of their application. This discussion is supported by additional data from these study areas where appropriate. Some general trends in the outcomes of the research are noted, and their relevance to the origin and dominant mechanisms of sediment production in parts of the Australian landscape are then discussed.

7.2 Variability of ^{137}Cs and ^{210}Pb$_{ex}$ within the study sites

7.2.1 Variability in fallout nuclides in the ACT and at St Helens.

In Chapter 2 the variability in ^{137}Cs areal concentrations in 'reference' environments was explicitly measured with a relative standard deviation (*rsd*) of ~38% at a small scale of 1 ha (Yarramundi Reach) and ~42% at a larger scale of ~ 300 km^2 (ACT region). In Chapter 3, a ^{137}Cs *rsd* of ~47% was observed within the two undisturbed forest plots (~350 m^2) near St. Helens, Tasmania. The ^{210}Pb$_{ex}$ *rsd* was also measured as ~57% in these plots, but the ratio between ^{210}Pb$_{ex}$ and ^{137}Cs was shown to be less variable (*rsd* ~28%). This ratio was subsequently used to determine soil loss with better precision than for either nuclide alone. However, the correlation between ^{210}Pb$_{ex}$ and ^{137}Cs at St

Helens was from a small area (~700 m^2). The general value of this ratio as a benchmark for soil erosion studies depends on a good correlation between $^{210}Pb_{ex}$ and ^{137}Cs in different landforms and at larger scales.

7.2.2 Regional correlations between inventories of $^{210}Pb_{ex}$ and ^{137}Cs

The tributary catchments of the mid-Murrumbidgee (~13,500 km^2) provide an appropriate region to test this correlation. Soil samples have been analysed for $^{210}Pb_{ex}$ and ^{137}Cs (Bq m^{-2}) from this larger area, and the ratios of these can be compared to those from the ACT and St Helens. The soil cores from the mid-Murrumbidgee tributary catchments were taken from apparently undisturbed lands that had good surface cover and soil loss appeared minimal; their inventories are believed to reflect fallout processes alone. Each core was taken in three sections with diameter 100 mm and depths 0-20, 20-50 and 50-200 mm (to account for the different depth distributions of $^{210}Pb_{ex}$ and ^{137}Cs). One of these sectioned cores was taken every ten km, along a transect length of ~50 km. The material from the different depth sections, from the five locations, was then mixed together to get a single representative core in three sections for that transect. The process was then repeated an additional ten times. In total, 11 sectioned cores were analysed, made up of 55 individual cores, from a total catchment area of ~1,500 km^2. The data is summarised in Table 7.2.2.1.

The ratio between the two nuclides here in the mid-Murrumbidgee is calculated to be 3.0 ± 0.1 (Table 7.2.2.1), and it can be seen that the variability (*rsd*) of this ratio (30%), is less than that of either $^{210}Pb_{ex}$ or ^{137}Cs alone (43 and 41% respectively). There is a similar reduction in the relative standard error (*rse*).

Table 7.2.2.1: Inventories of ^{137}Cs and ^{210}Pb$_{ex}$ within undisturbed soils of the mid-Murrumbidgee tributary catchments

Sample code	n	^{137}Cs Bq m^{-2}	se	^{210}Pb$_{ex}$ Bq m^{-2}	se	Pb / Cs ratio	se
ME95301	5	690	130	1840	260	2.7	0.6
ME95302	5	890	200	2630	290	2.9	0.7
ME95303	5	860	110	2830	410	3.3	0.6
ME95304	5	940	90	2660	360	2.8	0.5
ME95305	5	540	120	2140	310	4.0	1.1
ME95306	5	700	90	2060	230	2.9	0.5
ME95307	5	540	110	1680	250	3.1	0.8
ME95308	5	720	90	1770	300	2.5	0.5
ME95309	5	720	90	1890	180	2.6	0.4
ME95310	5	560	100	1600	280	2.9	0.7
ME95311	5	690	130	1930	100	2.8	0.8
Mean		710		2100		3.0	
Standard Dev. (bulk) σ		130		400		0.4	
Standard Dev.* (est) σ		290		890		0.86	
Standard Error		40		120		0.12	
rsd (%) *		41		43		30	
rse (%)		6		6		4	

Note: Uncertainties are given as subscripts, are analytical only and are equivalent to one standard error

* Because the mid-Murrumbidgee cores were bulked together the standard deviation (and *rsd*) of the individual cores is an estimate, calculated by multiplying the standard error on the mean value (derived from a sample population n = 11) by the square root of the actual number of samples (n=55).

The areal concentration of ^{210}Pb$_{ex}$ in the mid-Murrumbidgee has also been plotted against ^{137}Cs (solid circles) in Figure 7.2.2.1. The solid straight line is a least squares linear regression ($r^2 = 0.67$) of these data. The areal concentration data from St Helens (open circles) and that from the three soil pits excavated at Black Mountain (open triangles) are given for comparison. The correlation between $^{210}Pb_{ex}$ and ^{137}Cs at all these locations is encouraging, especially in the Murrumbidgee where the sampling was undertaken over a large surface area (~1,500 km^2) and each plotted value represents mixed material from five separate point locations. Mixing was shown to decrease the spread of ^{137}Cs areal concentrations in Chapter 2, (see also Owens and Walling, 1996). Therefore, it is likely that the variability in areal concentrations from the individual cores in the Murrumbidgee is significantly greater than shown in figure 7.2.2.1.

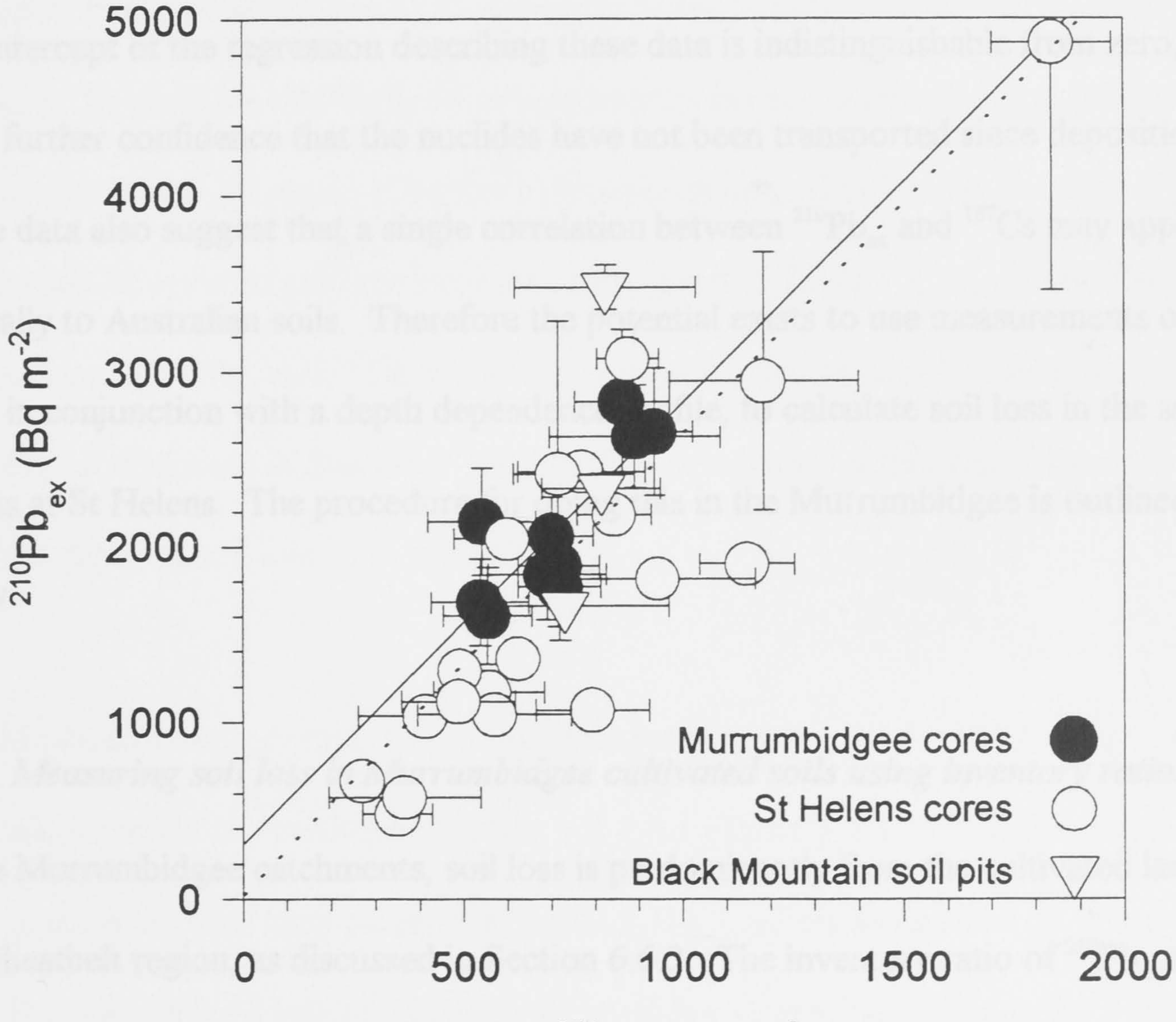

Figure 7.2.2.1 Areal concentrations of $^{210}Pb_{ex}$ and ^{137}Cs from undisturbed areas of the mid-Murrumbidgee catchment, NSW; areal concentrations from reference Plots 1 and 4, St Helens, Tasmania; areal concentrations from the Black Mountain soil pits, ACT. Data corrected to 1995. Least squares regressions of the mid-Murrumbidgee data (solid line) and combined data from all the study locations (dotted line) are also given.

The correlation between the nuclides in the Murrumbidgee is also similar to that observed in Tasmania, and the ACT; the relationship between $^{210}Pb_{ex}$ and ^{137}Cs at all these sites is indistinguishable. A least squares regression of all the data (given as the dotted line in Figure 7.2.2.1) yields an r^2 of 0.7, implying that some 70 percent of their current distribution can be accounted for by similarities in their initial fallout behaviour, and subsequent lateral transport before incorporation into the soil matrix. (It should be remembered that all these samples were taken from undisturbed areas in which no significant soil movement was believed to have occurred).

The intercept of the regression describing these data is indistinguishable from zero, which gives further confidence that the nuclides have not been transported since deposition. These data also suggest that a single correlation between $^{210}Pb_{ex}$ and ^{137}Cs may apply generally to Australian soils. Therefore the potential exists to use measurements of this ratio, in conjunction with a depth dependence profile, to calculate soil loss in the same way as at St Helens. The procedure for doing this in the Murrumbidgee is outlined below.

7.2.3 Measuring soil loss in Murrumbidgee cultivated soils using inventory ratios

In the Murrumbidgee catchments, soil loss is predominantly from the cultivated lands of the wheatbelt region, as discussed in Section 6.6.3. The inventory ratio of $^{210}Pb_{ex}$ to ^{137}Cs (Bq m^{-2}) has been measured here from 30 sectioned cores taken at 0-20, 20-50 and 50-200 mm depth increments. The average areal concentrations of $^{210}Pb_{ex}$ and ^{137}Cs, and the ratio between them in the cultivated and undisturbed regions are summarised in Table 7.2.3.1. These are all statistically different (using $p < 0.05$, one tailed test).

Table 7.2.3.1 Average inventories of $^{210}Pb_{ex}$, ^{137}Cs, and the ratio between them, in soils from cultivated lands and undisturbed lands (see Table 7.2.2.1), of the mid-Murrumbidgee catchment, NSW.

	n	$^{210}Pb_{ex}$ (Bq m^{-2})	se	^{137}Cs (Bq m^{-2})	se	$^{210}Pb_{ex}/^{137}Cs$ -	se
Undisturbed	55	2100	120	710	40	3.0	0.1
Cultivated	30	960	50	550	60	1.8	0.1
Percent depletion		54		23		-	
Note: n = total number of cores taken							

The inventory ratio can be converted to a depth of soil loss (using the approach outlined in Chapter 3) if an appropriate depth dependence relationship can be derived as a reference. To illustrate this the areal concentrations in each segment of the soil pits excavated at Black Mountain (Chapter 5) have been normalised to the average of the undisturbed Murrumbidgee inventory amounts, given in Table 7.2.3.1. The different distributions, of $^{210}Pb_{ex}$ and ^{137}Cs, were then summed from the base to the surface (Figure 7.2.3.1a), as discussed in Section 3.5.1, and the resulting inventory ratios used to calculate depth of soil loss as in Section 3.5.4.

The calculated inventory ratio for undisturbed lands is 3.0 ± 0.1 (Table 7.2.3.1). The ratio calculated for the cultivated lands is 1.8 ± 0.1 (Table 7.2.3.1). Comparing the latter value to the inventory ratio curve (Figure 7.2.3.1b), predicts a total loss of about 37 ± 6 mm. This represents a loss of ~1.2 mm yr^{-1} over the 31 year period since the peak ^{137}Cs fallout (~1964, Longmore, 1983), and is equivalent to an annual yield of about 14 ± 2 t ha^{-1} yr^{-1} given the measured bulk density in these soils of ~1.2 g cm^{3}. This estimate can be compared to losses of ~ 20 ± 8 mm (8 ± 3 t ha^{-1} yr^{-1}) from ^{137}Cs and 25 ± 6 mm (10 ± 2 t ha^{-1} yr^{-1}) from $^{210}Pb_{ex}$ using the depth dependence of these nuclides given in Figure

7.2.3.1a. These are consistent with estimates of loss of about 8 t ha^{-1} yr^{-1}, from erosion plot studies of summer cropping areas in NSW, Australia (Edwards, 1987) and with the range of values given for agricultural lands elsewhere (Evans and Cook, 1986; Basher *et al.*, 1995). Note: the value of Edwards (1987) may be out by a factor of 3, Lang (1992).

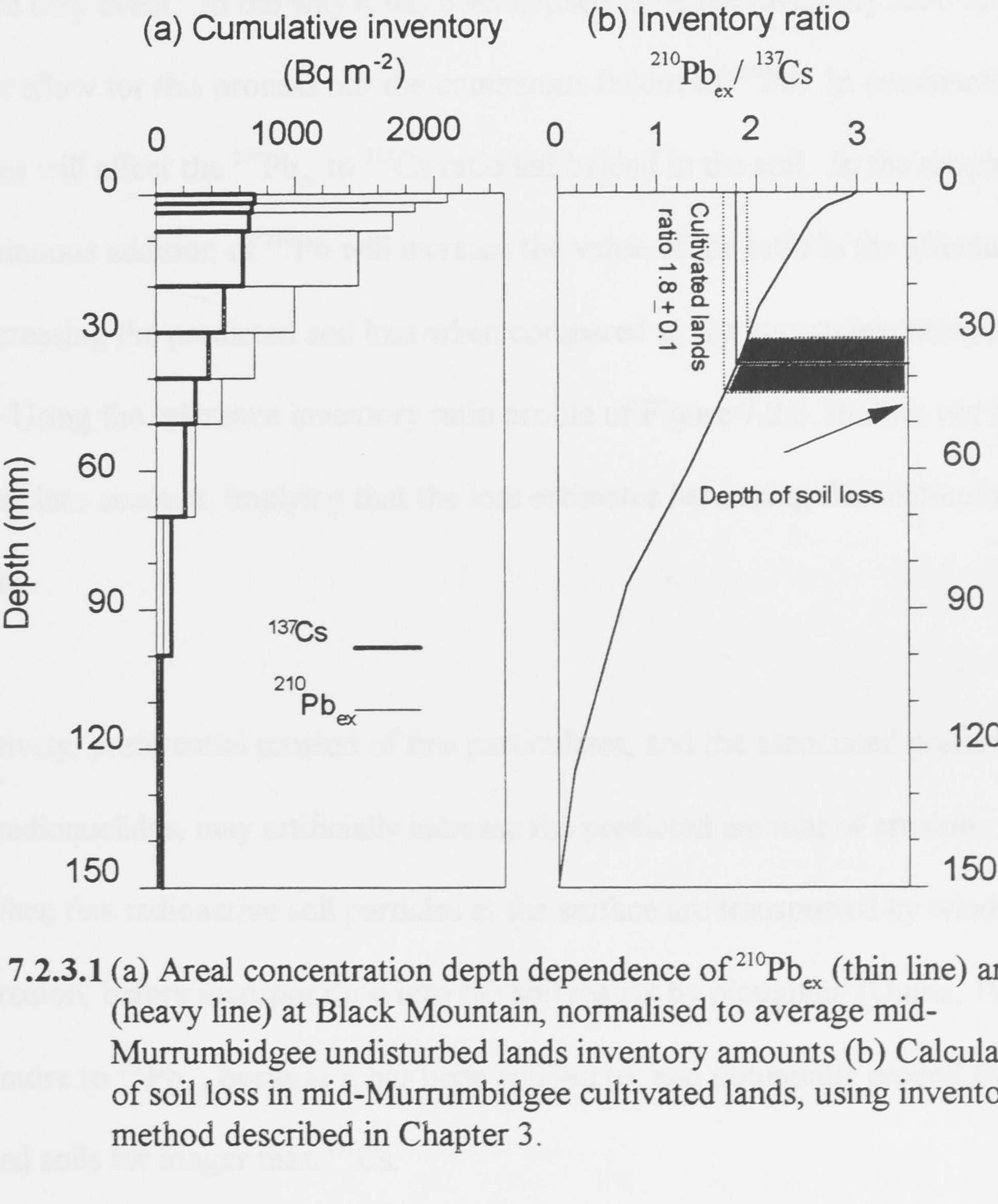

Figure 7.2.3.1 (a) Areal concentration depth dependence of $^{210}Pb_{ex}$ (thin line) and ^{137}Cs (heavy line) at Black Mountain, normalised to average mid-Murrumbidgee undisturbed lands inventory amounts (b) Calculated depth of soil loss in mid-Murrumbidgee cultivated lands, using inventory ratio method described in Chapter 3.

It is interesting that the inventory ratio method estimated higher rates of soil loss than either $^{210}Pb_{ex}$ or ^{137}Cs in the mid-Murrumbidgee. The ratio method also predicted the highest rates of soil loss at St Helens, which was attributed to potential differences between reference site and erosion plot inventories. This is unlikely to be true in the mid-Murrumbidgee however, as the undisturbed reference inventory values were calculated from the average of many (n=55) individual cores. The average values from cultivated lands were similarly calculated from a large data set (n=30). One process that will affect the interpretation presented here is that soil loss from the mid-Murrumbidgee cultivated lands is presumably continuous, unlike St Helens where it essentially occurred as a once only event. In the way it has been applied here, the inventory ratio approach does not allow for this process nor the continuous fallout of ^{210}Pb. In combination, these processes will affect the ^{210}Pb$_{ex}$ to ^{137}Cs ratio left behind in the soil. In the simplest case, the continuous addition of ^{210}Pb will increase the value of the ratio in the affected soil, thus decreasing the predicted soil loss when compared to the known inventory ratio profile. Using the reference inventory ratio profile of Figure 7.2.3.1b does not take these processes into account, implying that the loss estimates here using this method may be a minimum.

Alternatively, preferential erosion of fine particulates, and the associated preferential loss of radionuclides, may artificially increase the predicted amount of erosion. This can occur when fine radioactive soil particles at the surface are transported by wind and water erosion, before incorporation into the soil matrix by ploughing (Quine, 1995). This applies more to ^{210}Pb$_{ex}$, because it has been applied to, and potentially eroded from, these cultivated soils for longer than ^{137}Cs.

As a result of these uncertainties it is concluded that soil loss in this region cannot be determined more precisely using inventory ratios of ^{210}Pb$_{ex}$ to ^{137}Cs until a more accurate erosion model can be constructed. At the least, this will need to take into account the continuous fallout of ^{210}Pb, the (annual) loss of particulates due to cultivation, and the redistribution of particles resulting from cultivation.

7.2.4 Comparing variability of ^{137}Cs and $^{210}Pb_{ex}$ inventories in the different study areas

In this section the average areal concentrations (Bq m^{-2}) of ^{137}Cs and ^{210}Pb$_{ex}$, and their variability (*rsd*), from the different study areas, are contrasted with one another. The variability of ^{137}Cs here may be different from overseas; some tentative explanations for this are explored.

The mean ^{137}Cs concentration of the Murrumbidgee data, 710 ± 40 Bq m^{-2}, can be compared with that of the ACT, 660 ± 50 Bq m^{-2}, and St Helens, 760 ± 80 Bq m^{-2} (Table 7.2.4.1). They agree surprisingly well given the latitudinal and rainfall difference between them; both of these have been shown to influence fallout of ^{137}Cs due to the presence of trends in zonal meteorological conditions (Trefry, 1975). The *rsd* of ^{137}Cs in the Murrumbidgee (Table 7.2.4.1) is consistent with that in both the ACT region and St Helens, although it is larger than that for the three detailed soil profiles taken at Black Mountain. However this latter figure represents the average of only 3 samples, (see Section 5.3.2), and the *rsd* may be atypically small as an artefact of this small data set.

Table 7.2.4.1 Comparison of ^{137}Cs and ^{210}Pb$_{ex}$ areal concentration amounts and variability from different scales and locations, Australia.

Location	n	^{137}Cs (Bq m^{-2})	*rsd* (%)	*rse* (%)	^{210}Pb$_{ex}$ (Bq m^{-2})	*rsd* (%)	*rse* (%)	Pb/Cs -	*rsd* (%)	*rse* (%)
ACT, Black Mt.#	3	790	5	3	2490	30	17	3.1	26	15
ACT, All sites #	82	660	41	5	-	-	-	-	-	-
St Helens #	18	760	47	11	1820	57	13	2.4	28	7
Murrumbidgee #	55	710	40	6	2100	41	6	3.0	30	4

Note: # Undisturbed locations from this study, all values decay corrected to 1995

The fallout inventories of $^{210}Pb_{ex}$ in the study areas can also be compared with one another. The highest inventory, at Black Mountain (2,490 ± 420 Bq m^{-2}) is within uncertainty of that in the Murrumbidgee (2100 ± 120 Bq m^{-2}). The lowest value is at St Helens, Tasmania. This is most likely because Tasmania is an island of only ~50,000 km^2, whereas the Australian mainland is ~7,700,000 km^2. The production of ^{210}Pb is known to be dependent on the size of continental land mass upwind from the deposition site (Turekian *et al.*, 1977). Tasmania's prevailing wind comes from the west; there is no significant land mass in this direction. Alternatively, the same prevailing winds must cross a substantial fraction of the Australian continent before reaching the mainland study sites used here. The estimated *rsd* of the St Helens $^{210}Pb_{ex}$ inventories (particularly Plot 4, Table 3.5.1.1) is higher than that from the Murrumbidgee tributaries. This is attributed to the forest canopy increasing spatial heterogeneity from leaf drip and stem flow effects at St Helens (Section 3.6.6). This phenomenon has been observed elsewhere for ^{137}Cs and ^{7}Be (Franklin *et al.*, 1967; Wallbrink and Murray, 1996a).

The ratio of $^{210}Pb_{ex}$ to ^{137}Cs is highest at Black Mountain and lowest at St Helens, due to the smaller ^{210}Pb inventory at the latter site. The *rsd* of this ratio is lower than that of both $^{210}Pb_{ex}$ and ^{137}Cs at St Helens and the mid-Murrumbidgee, but not at Black Mountain, where the variability of ^{137}Cs was atypically small, as discussed. The lower overall *rsd* of the $^{210}Pb_{ex}/^{137}Cs$ ratio in the Murrumbidgee samples supports the concept of using this nuclide ratio to compensate for the initial spatial heterogeneity of these tracers at this scale.

7.2.5 Global comparison of spatial variability in ^{137}Cs inventory amounts

The variability of ^{137}Cs measured in undisturbed 'reference' areas of this thesis can be compared with that measured in 'reference' areas elsewhere (Table 7.2.5.1). The values have been broadly grouped into those from the northern hemisphere and those from the southern hemisphere; the latter group being dominated by measurements from Australia. It should be noted that the values of Table 7.2.5.1 represent the results from a number of different researchers using a variety of different sampling techniques. In Section 3.6.5, soil sampling area was shown to influence measured spatial variability, and so it is possible that the different sampling strategies involved in this comparison may cause systematic differences in the reported values. For example, the values for the ACT region in Section 2.4.1, were obtained from soil pits with a surface area of ~0.04 m^2; whereas those from St Helens, in Section 3.4.2, were obtained by both soil cores with surface area ~0.008m^2 and soil pits with combined surface area of 0.16m^2 and 0.04m^2. It was proposed (Section 3.6.5) that the larger surface area of the soil pits at the latter site may have reduced some of the variability occurring at the spatial frequency of the smaller cores. Other researchers have used a variety of different methods, from cores with surface area as small as 0.003 m^2 (Basher and Matthews, 1993) to scraper plates covering as much as 0.1 m^2 (Loughran *et al.*, 1982).

In addition to this, the *rsd* has been calculated for, from between 4 to 54 samples. Both Fredericks *et al.* (1988) and Sutherland (1991) report on the effect of sample number on properly characterising both sample variability and mean inventory amount. Nonetheless, if all these considerations are taken into account, a clear difference remains between the mean of the measured *rsd* values from the different hemispheres (Table 7.2.5.1).

Table 7.2.5.1 Comparison of spatial variability of ^{137}Cs inventories from Black Mountain, St Helens, and the mid-Murrumbidgee with measurements from elsewhere (adapted in part from Owens and Walling, 1996)

Location	(n)	*rsd* (%)	Reference
Southern hemisphere			
Inverell	11	34	Fredericks *et al.*, (1988)
"	15	33	"
"	7	45	"
Maluna Creek.	5	23	Loughran *et al.*, (1982)
Darling Downs	6	32	McCallan *et al.*, (1980)
Yarramundi (Chapter 2)	50	38	Wallbrink *et al.*, (1994)
ACT region *	32 (68)	42	"
St Helens (Chapter 3)	18	47	Wallbrink and Murray, (1996b)
Murrumbidgee *	11 (55)	40	This work
Msana, Africa	24	67	Owens and Walling (1996)
New Zealand	8	30	Basher and Matthews, (1993)
Mean		**39**	
Standard Deviation		**11**	
Standard Error		**3**	
Northern Hemisphere			
Oregon, (USA)	9	29	Brown *et al.*, (1981)
Bavaria	25	29	Bunzl and Kracke (1988)
New Brunswick	4	23	de Jong *et al.*, (1986)
English Midlands	4	24	Foster *et al.*, (1994)
"	4	5	"
U.S.	54	19	Lance *et al.*, (1986)
Devon	12	26	Loughran *et al.*, (1987)
Korea	7	30	Menzel *et al.*, (1987)
Spain	9	19	Navas and Machin (1991)
Hawaii	20	20	Sutherland (1991)
"	36	18	"
"	20	23	"
Yendacott	57	18	Owens and Walling (1996)
Mean		**22**	
Standard deviation		**7**	
Standard error		**2**	

Note: * samples mixed together

The *rsd* value for the southern hemisphere locations (39 ± 3; n=11) is nearly a factor of two greater than the average of all the northern hemisphere locations (22 ± 2; n=13). With the exception of Loughran *et al.*, (1982) who observed an *rsd* of 23 % (possibly due to the large surface area of the scraper plate used), the *rsd* of all of the southern hemisphere locations is greater than 30%. Conversely, only one location in the northern hemisphere has an *rsd* greater than 30%; (30.4%, Menzel *et al.*, 1987).

The high variability of ^{137}Cs areal concentrations in Australian, New Zealand and African 'reference' sites is intriguing, and may possibly be related to the lower overall fallout of ^{137}Cs in the southern hemisphere, combined with the more variable distribution of rainfall delivering it (Fleming, 1974). High intensity, short duration rainfall events are common in Australian and African summers, which is more likely to cause overland flow, and therefore loss of nuclides with runoff waters. A similar phenomenon was observed for ^{7}Be in bare soils (Wallbrink and Murray, 1996a). New Zealand also has highly variable rainfall with rainfall amount closely following topography (Basher and Matthews, 1993). In contrast, the northern hemisphere had up to an order of magnitude higher total fallout, (Longmore, 1982) and, in the U.K. at least, a more even distribution of rainfall.

Although it is clear that the *rsd* of ^{137}Cs in the northern hemisphere is generally lower than that observed in the Australian study sites, it is not known how much this has been affected by the different methods of measurement. Further investigation of global trends in the variability of fallout of ^{137}Cs probably requires the adoption of a standardised approach to measuring ^{137}Cs soil concentrations.

7.3 Describing sediment sources and erosion processes using fallout nuclides

In Section 7.2 correlations between areal concentrations (Bq m^{-2}) of $^{210}Pb_{ex}$ and ^{137}Cs were explored as an alternative means of measuring soil erosion. The next part of this chapter is concerned with the use of absolute concentrations (Bq kg^{-1}) of fallout ^{7}Be, $^{210}Pb_{ex}$, and ^{137}Cs to determine sediment sources. The discussion on this topic begins by applying the methods developed at the study locations to other sites. For example, the crossplot method described in Chapter 4 is very simple, yet in the absence of prior knowledge of catchment condition can provide a large amount of information on sediment sources and erosion processes. In Section 4.2.3, concentrations of ^{7}Be, $^{210}Pb_{ex}$, and ^{137}Cs were plotted against one another to show that sediment yield was dominated by gully wall erosion at Whiteheads Creek, NSW. The usefulness of this approach can be examined further, by considering the runoff data from the controlled runoff experiments at Black Mountain, ACT.

7.3.1 Describing erosion processes at Black Mountain using crossplots of ^{7}Be, $^{210}Pb_{ex}$ and ^{137}Cs

The radionuclide concentrations of the sediments from the four surface treatments at this site, can be plotted against one another to compare their location on the graph with known erosion condition (as in Figure 4.2.2.1). For the purposes of this discussion, it is assumed that the only catchment information available is in the form of particle size analysis and radionuclide concentrations of the surface scrapes from this area (Table 5.4.2.1). This surface soil data provides a reference point for purely surface derived sediment. There is a clear separation between the values when ^{7}Be is plotted against ^{137}Cs (Figure 7.3.1.1a).

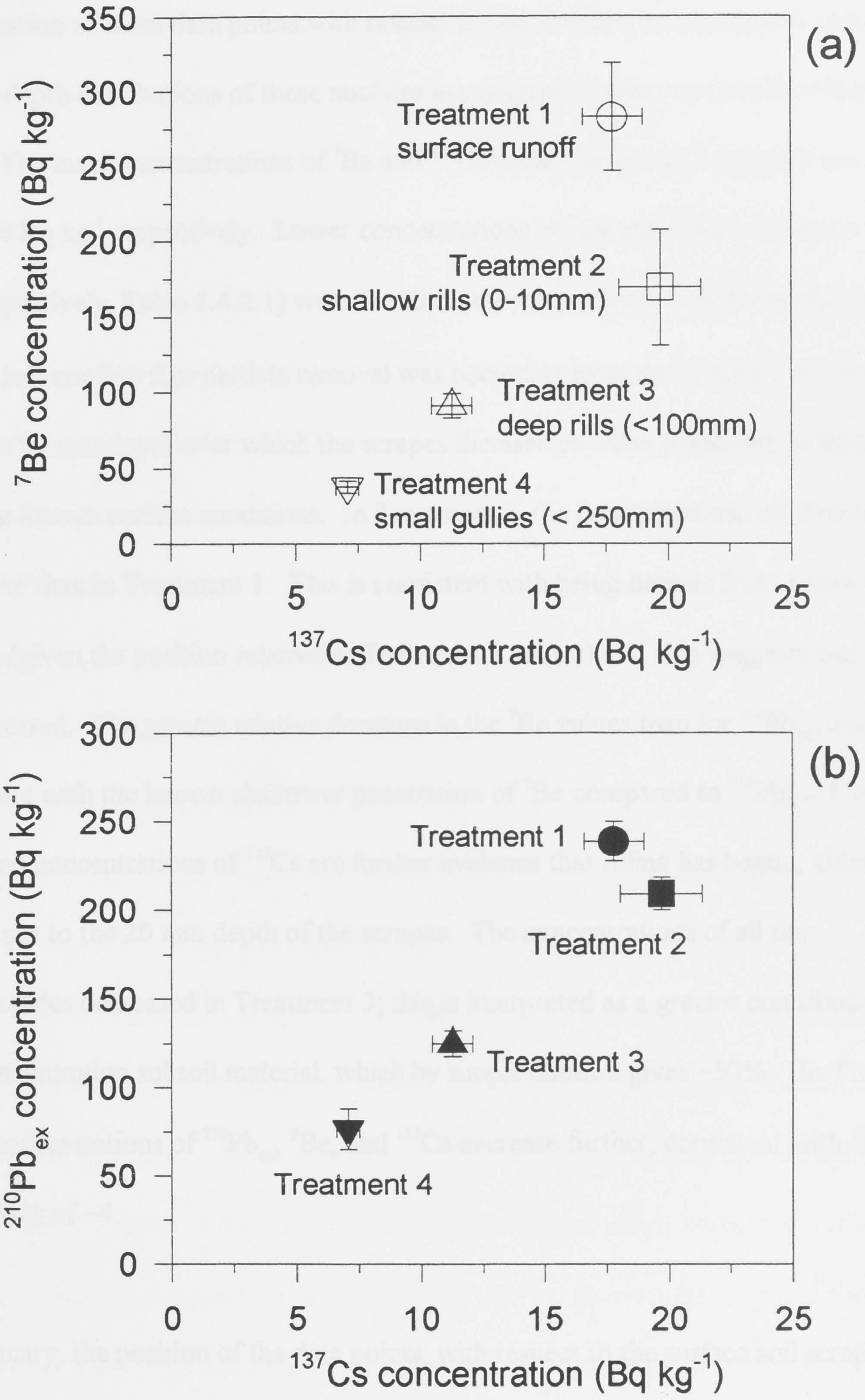

Figure 7.3.1.1 Radionuclide concentrations of suspended sediment runoff from four different surface erosion treatments at Black Mountain, ACT. (a) ^{7}Be against ^{137}Cs, and (b) ^{210}Pb$_{ex}$ against ^{137}Cs.

The same pattern is observed when $^{210}Pb_{ex}$ is plotted against ^{137}Cs (Figure 7.3.1.1b). The location of these data points with respect to one another, are consistent with the known depth distributions of these nuclides in undisturbed soils, as described in Section 4.2.2. The mean concentrations of ^{7}Be and ^{137}Cs in the Treatment 1 samples are ~280 and ~18 Bq kg^{-1} respectively. Lower concentrations of ^{7}Be and ^{137}Cs (~70 and ~11 Bq kg^{-1} respectively, Table 5.4.2.1) were found on the <2μm fraction of the surface scrapes. These data confirm that particle removal was occurring from closer to the soil surface than the 20 mm depth over which the scrapes themselves were integrated; in agreement with the known surface conditions. In Treatment 2, the concentrations of ^{7}Be and $^{210}Pb_{ex}$ are lower than in Treatment 1. This is consistent with being derived from below the surface (given the position relative to Treatment 1 values) and also suggests that rilling has occurred. The greater relative decrease in the ^{7}Be values than for $^{210}Pb_{ex}$, is also consistent with the known shallower penetration of ^{7}Be compared to $^{210}Pb_{ex}$. The increased concentrations of ^{137}Cs are further evidence that rilling has begun, although clearly not to the 20 mm depth of the scrapes. The concentrations of all the radionuclides decreased in Treatment 3; this is interpreted as a greater contribution from low concentration subsoil material, which by simple dilution gives ~50%. In Treatment 4, the concentrations of $^{210}Pb_{ex}$, ^{7}Be, and ^{137}Cs decrease further, consistent with dilution by a factor of ~4.

In summary, the position of the data points, with respect to the surface soil scrapes and one another, implies an initial contribution from shallow surface erosion (Treatment 1), followed by a shift to rilling processes (Treatment 2); after that, material is derived from a combination of Treatment 2 material diluted by varying amounts of subsoil material

(Treatment 3 and 4). These interpretations are consistent with the known erosion conditions on the plot at the time, confirming the reliability of this simple approach.

However, the data from Black Mountain and Whiteheads Creek are from relatively small surface areas (<400 m^2) in which the transport paths are short, and the erosion conditions are well known. A more rigorous test of the crossplot method involves using the data from the tributary catchments of the mid-Murrumbidgee region. The scale of this site is much larger, and the influence of the various erosion processes is not intuitively obvious.

7.3.2 The large scale: the mid-Murrumbidgee tributary catchments

The $^{210}Pb_{ex}$ and ^{137}Cs signatures on the catchment sources and sediments of the mid-Murrumbidgee are all clearly separated (Figure 7.3.2.1). The concentrations of $^{210}Pb_{ex}$ and ^{137}Cs from the surface scrapes of uncultivated lands were the highest, and these plotted in the upper right hand corner of this diagram, as expected. This material also had the greatest range of values, which is not surprising given the diversity of soils and landforms within this category. The radionuclide concentration of material from the cultivated lands plotted between the uncultivated values and the origin; the value estimated for subsoil erosion from channel/gully sources lies closest to the origin. The radionuclide signature of the suspended sediments (with a minor correction for labelling by direct fallout, as discussed in Section 6.6.2) also lies towards the origin of the crossplot, and adjacent to the subsoil erosion value. This implies a dominant contribution from subsoil sources to the sediments of the Murrumbidgee River, with the material being derived from erosion of channel bank and gully walls.

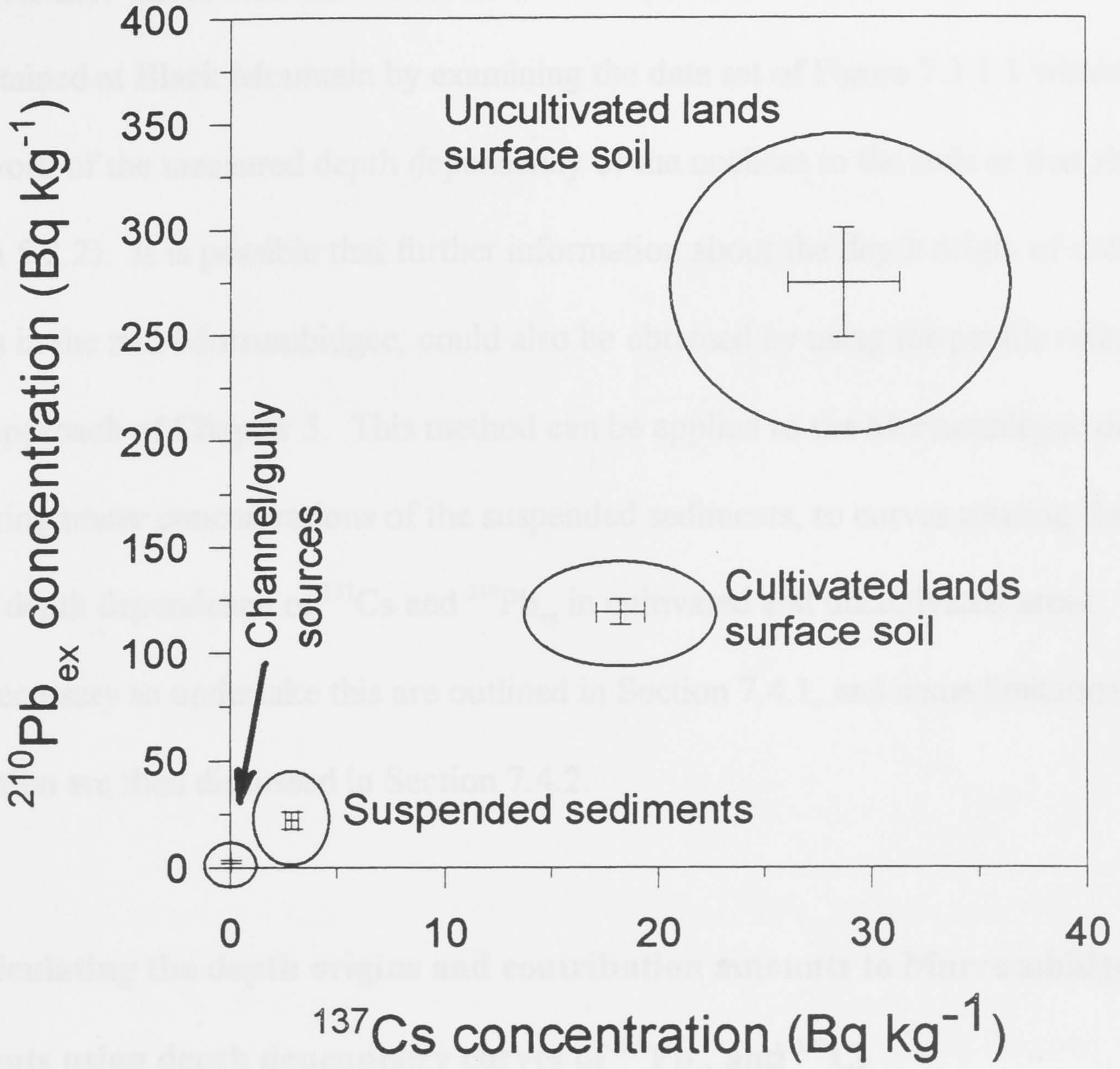

Figure 7.3.2.1 Radionuclide concentrations of suspended sediment and catchment sources in the mid-Murrumbidgee region, NSW. The standard error of the mean of each of these parameters is given by the error bars, the standard deviation of the sample population is given by the ellipse surrounding the data error bars.

It is concluded that this crossplot approach can reveal much about erosion occurring within landscapes. In particular, the method has been shown to qualitatively determine the origin, and dominant erosion mechanism, of suspended sediment generated under different erosion conditions in three distinct locations of widely varying size (Whiteheads Creek, Black Mountain, and the mid-Murrumbidgee).

However, in reality this approach is a simplification of the mixing model of Chapter 6, which was used to quantify the relative contributions from the three potential sources of

Figure 7.3.2.1. More information on the actual depth from which sediment was derived, was obtained at Black Mountain by examining the data set of Figure 7.3.1.1 within the framework of the measured depth dependency of the nuclides in the soils at that site (see Section 5.2.2). It is possible that further information about the depth origin of sediment sources in the mid-Murrumbidgee, could also be obtained by using the profile reference curve approach of Chapter 5. This method can be applied to the Murrumbidgee data by comparing tracer concentrations of the suspended sediments, to curves relating the known depth dependence of ^{137}Cs and ^{210}Pb$_{ex}$ in cultivated and uncultivated areas. The steps necessary to undertake this are outlined in Section 7.4.1, and some limitations to its application are then discussed in Section 7.4.2.

7.4 Calculating the depth origins and contribution amounts to Murrumbidgee sediments using depth dependency curves of ^{210}Pb$_{ex}$ and ^{137}Cs

7.4.1 Constructing the reference curves for uncultivated and cultivated lands

The interpretation of the suspended sediment data from the mid-Murrumbidgee requires two sets of reference curves to be constructed: (i) the depth dependence for uncultivated lands, and (ii) that for cultivated lands. The appropriate curves for the uncultivated lands can be estimated from the sectioned cores used to determine the regional ^{210}Pb$_{ex}$ and ^{137}Cs inventories (Bq m^{-2}) in the undisturbed lands (Section 7.2.2). However, these cores (given as histograms in Figure 7.4.1.1) do not provide the resolution necessary for this analysis. The soil pits excavated at Black Mountain (Figure 5.4.1.2) provide improved resolution; these have been normalised to the uncultivated lands surface concentration value (Figure 7.4.1.1; dotted lines). It can be seen that the normalised curves are consistent with the observed average concentrations in the low resolution data (histograms). Therefore it is reasonable to use these normalised curves as reference profiles for the uncultivated lands.

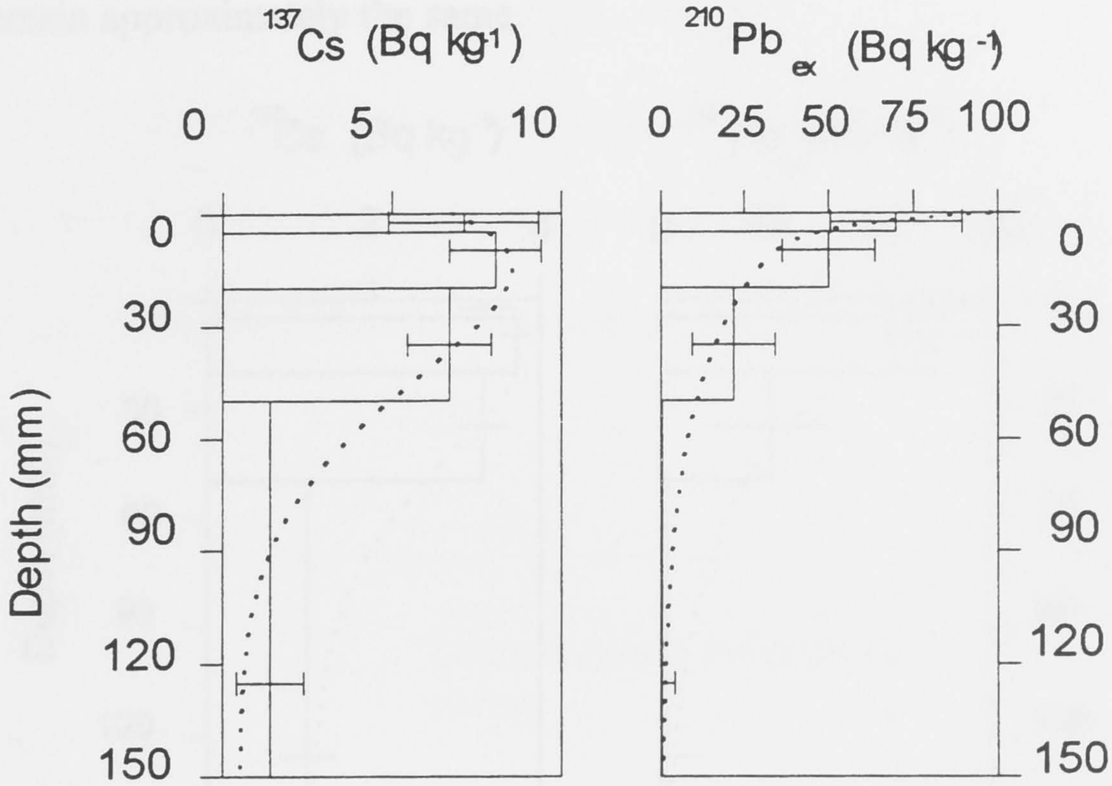

Figure 7.4.1.1 Measured depth distribution of ^{137}Cs and ^{210}Pb$_{ex}$ in bulk soils from undisturbed lands of the mid-Murrumbidgee. The dotted lines represent the depth dependence of ^{137}Cs and ^{210}Pb$_{ex}$ from Black Mountain (Figure 5.4.1.2), normalised to undisturbed lands' surface scrapes concentration value.

The erosion curves from the cultivated lands however are more difficult to construct, because the only data on which to base the shape of the curve come from cores (n=30) taken within them at 0-20, 20-50 and 50-200 mm depth increments. A thin scrape of the soil (5 mm) was also taken to estimate the surface concentrations. The average of these 30 cores is given in Figure 7.4.1.2 for ^{210}Pb$_{ex}$ and ^{137}Cs. The distribution of these nuclides have been approximated by a polynomial for ^{137}Cs and a first order exponential for ^{210}Pb$_{ex}$. It is clear that a memory of the initial soil distribution of both these nuclides is preserved in these cultivated soils. The ^{210}Pb$_{ex}$ distribution has a less pronounced surface maximum than it does in the undisturbed soils, although the fact that the general shape persists at all is surprising given the history of cultivation. However, as discussed in Section 6.5.1, the cultivation process used in this area does not turn the soil over, rather the soil is pushed from side to side; the original depth positions of particulates appear to remain approximately the same.

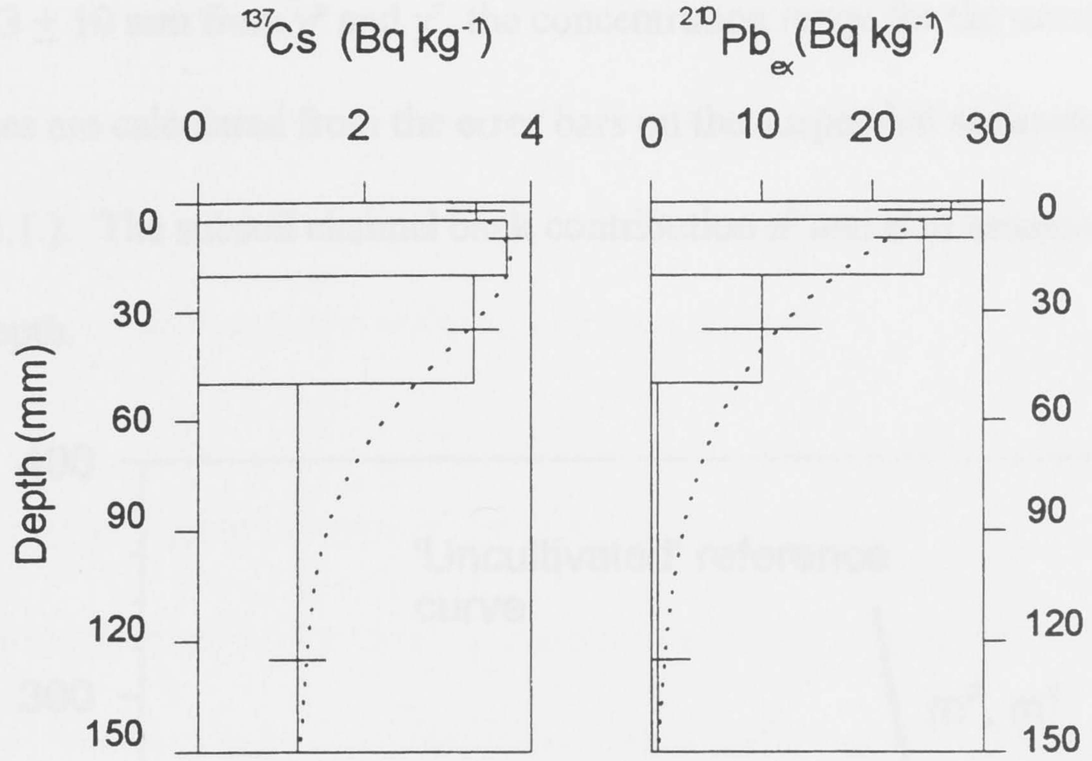

Figure 7.4.1.2 Measured depth distribution, and fitted curves, for ^{137}Cs and ^{210}Pb$_{ex}$ in bulk soils of cultivated lands in the mid-Murrumbidgee

The overall concentrations of the radionuclides in the cultivated cores are lower than in the uncultivated soils. This could be due to a combination of i) particle removal by erosion, and ii) the gradual downwards movement of fine particulates in the soil profile due to the cultivation process. Nonetheless, these cultivated profiles can also be normalised to their measured average <2 μm concentrations, and then plotted against one another to create the appropriate ^{210}Pb$_{ex}$ to ^{137}Cs reference curve for this land use type (Figure 7.4.1.3). Mixing lines are then drawn to bound the potential depth ranges, and relative contributions, from the cultivated and uncultivated reference curves. These have been converted to eroded depths in a similar manner to that described in Section 5.5.3, by comparing the measured concentration ranges on the respective ^{210}Pb$_{ex}$ and

^{137}Cs axis (Figure 7.4.1.3; m^p and m^c; uncultivated; y^p and y^c; cultivated; the superscripts *p* and *c* represent $^{210}Pb_{ex}$ and ^{137}Cs respectively) with the known depth dependencies. These give <2 ± 6 mm from m^p and m^c, the concentration range for the uncultivated lands, and <3 ± 10 mm from y^p and y^c, the concentration range for the uncultivated lands. (Uncertainties are calculated from the error bars on the suspended sediment sample in Figure 7.3.2.1.). The subsoil channel bank contribution n^p and n^c is assumed to be from >400 mm depth.

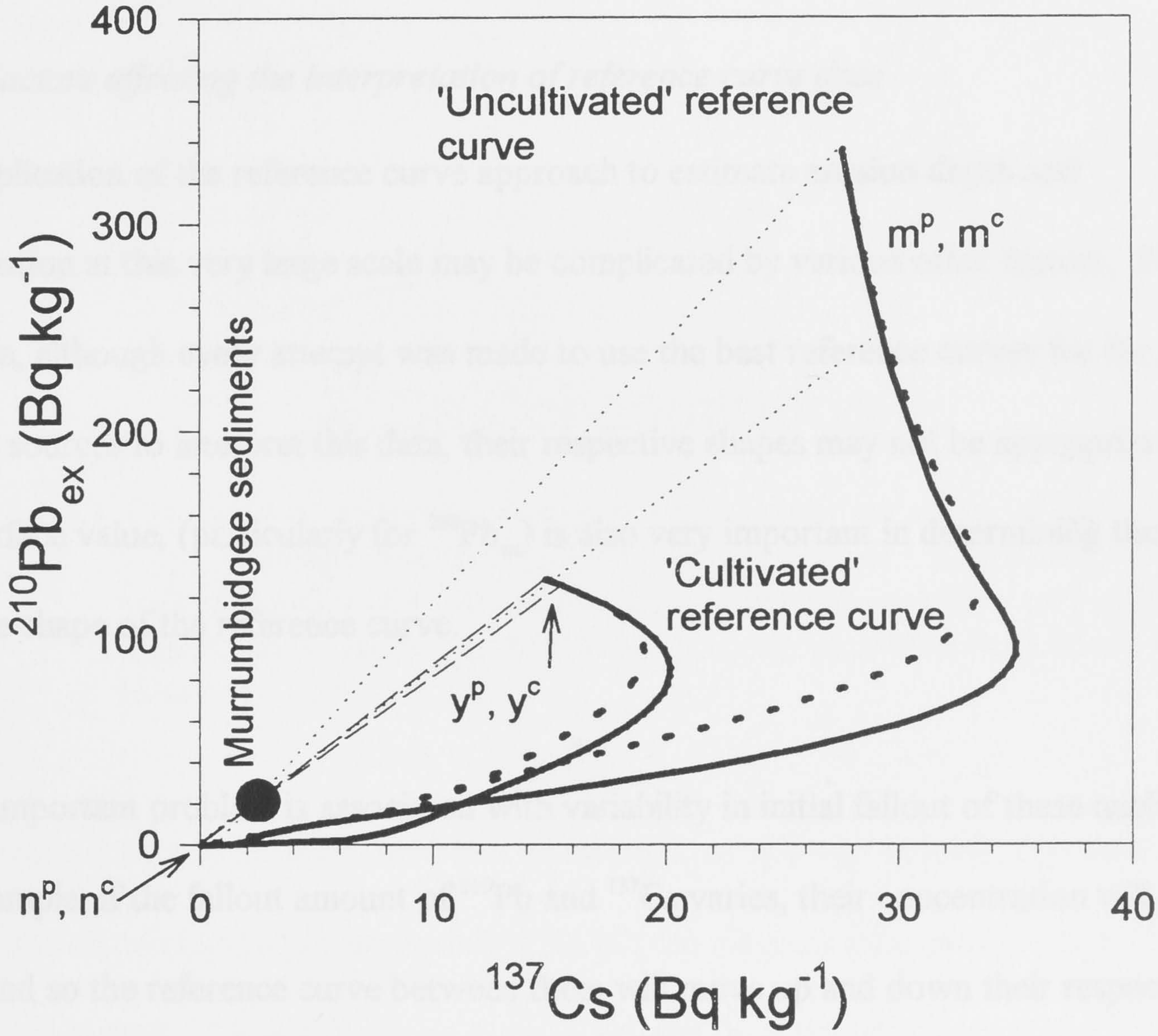

Figure 7.4.1.3 Suspended sediments of the Murrumbidgee River with respect to different $^{210}Pb_{ex}/^{137}Cs$ erosion reference curves for uncultivated lands and cultivated lands. Interpretation of solid and dotted curves as for Figure 5.5.3.1.

However, the spatial relationship between the suspended sediment value and the two profile reference curves is not independent. For example, a large 'uncultivated lands' surface contribution could increase the radionuclide concentrations on the suspended sediment. This would move it closer to the cultivated curve, thereby increasing the apparent contribution from this source. Thus, the surface contribution from cultivated soils, estimated at ~16% (y^{pc}) is most likely an upper limit because of the possible influence from uncultivated surface material. Conversely, the subsoil contribution (estimated at 84%) is most probably a minimum, because it must include at least some of the 93% from subsoils associated with uncultivated surfaces.

7.4.2 Factors affecting the interpretation of reference curve data

The application of the reference curve approach to estimate erosion depth and contribution at this very large scale may be complicated by various other factors. For example, although every attempt was made to use the best reference curves for the two surface sources to interpret this data, their respective shapes may not be appropriate. The surface value, (particularly for $^{210}Pb_{ex}$) is also very important in determining the ultimate shape of the reference curve.

A less important problem is associated with variability in initial fallout of these nuclides. For example, if the fallout amount of ^{210}Pb and ^{137}Cs varies, their concentration will also differ and so the reference curve between them will move up and down their respective concentration axis. However, if the fallout patterns of $^{210}Pb_{ex}$ and ^{137}Cs are correlated (as seen in Figure 3.5.4.1 and 7.2.2.1) then even if their absolute concentrations change, the shape of their respective depth profiles, and the relationship between them, should not. Therefore when profiles of different $^{210}Pb_{ex}$ and ^{137}Cs concentrations are plotted against one another, the shape of their combined reference curves should not be different, rather they will be altered with respect to the average condition. This has been illustrated in

Figure 7.4.2.1, in which the concentrations of $^{210}Pb_{ex}$ and ^{137}Cs have been altered by factors of 0.1, 0.2 and 0.4, to simulate various scenarios of initial variability.

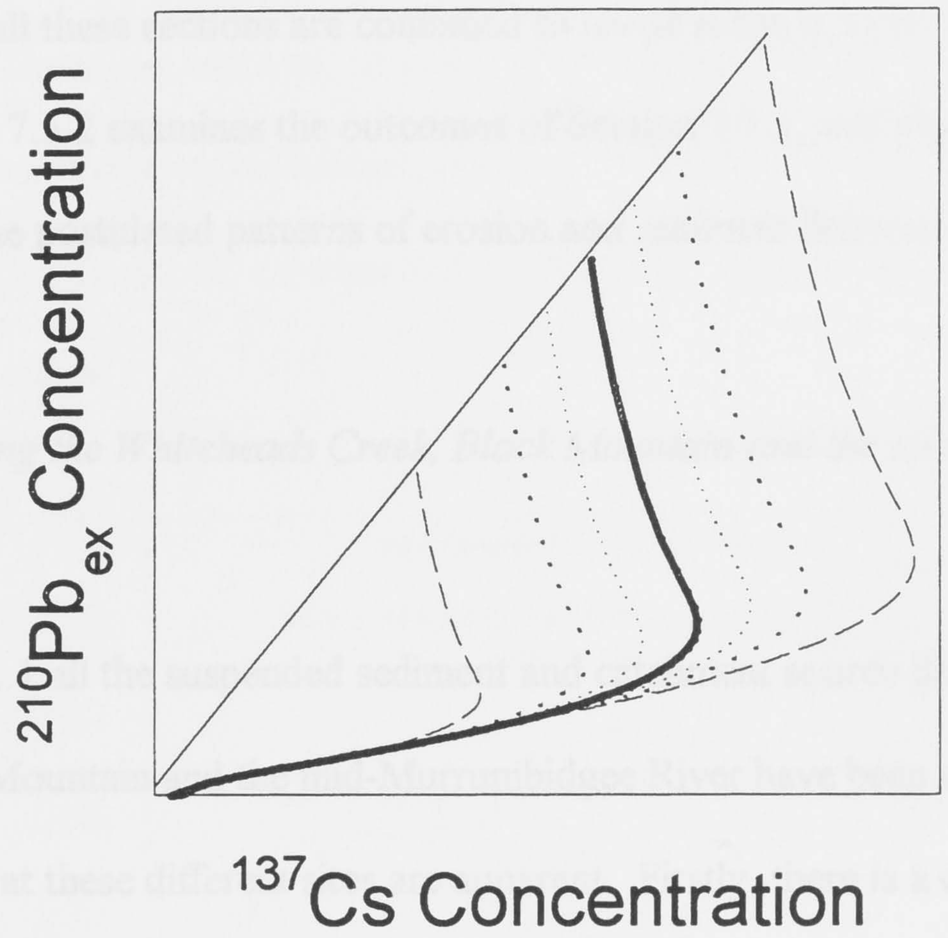

Figure 7.4.2.1 The effect on reference profiles of initial fallout variability of $^{210}Pb_{ex}$ and ^{137}Cs. Average condition (solid line); variability of ±10% (small dotted line); variability of ± 20% (thick dotted line) and variability of ± 40% (dashed line). Based on the detailed soil pits excavated from Black Mountain, Section 5.3.2.

Nonetheless, it should be considered that sediment is an 'average' in itself, representing the average of erosion processes occurring within the range of ratio profile shapes from which it is derived. The width of this range is defined by their initial variability. Thus it is appropriate to compare the 'average' of the Murrumbidgee sediment radionuclide concentrations, with the 'average' shape of the reference curves within the catchment. However despite this, the results of this analysis in the mid-Murrumbidgee must remain at least partly ambiguous, because the appropriate reference curves are poorly known.

7.5 Erosion trends within the study areas and related Australian catchments

In Sections 7.2, 7.3 and 7.4, various methods developed in the thesis were applied in different study areas, to test the breadth of their application. In the next Section (7.5.1) the data from all these sections are combined to reveal some common elements between them. Section 7.5.2 examines the outcomes of Section 7.5.1, and provides supporting evidence for the postulated patterns of erosion and sediment delivery in this region.

7.5.1 Combining the Whiteheads Creek, Black Mountain and the mid-Murrumbidgee River data

In Figure 7.5.1.1 all the suspended sediment and catchment source data from Whiteheads Creek, Black Mountain and the mid-Murrumbidgee River have been combined. Some general trends at these different sites are apparent. Firstly, there is a clear overlap between the $^{210}Pb_{ex}$ and ^{137}Cs values measured from surface erosion processes at the three experimental sites. This is encouraging as the region covered by them is ~ 50,000 km^2. The differences between the values can most probably be accounted for by changes in geology and soil type. For example, the bedrock at Whiteheads Creek is principally granite (Olley *et al.*, 1993). Grain size analysis confirms that the particle size distribution here is approximately bi-modal. It can be hypothesised that a greater fraction of fallout nuclides is attached to the fine clay component in preference to the larger grains, thus giving rise to elevated $^{210}Pb_{ex}$ and ^{137}Cs concentrations in the fine suspended sediment in runoff waters. A similar phenomenon applied to ^{7}Be in granitic soils at St Helens, (Wallbrink and Murray, 1996a). At Black Mountain however, the bedrock is a sequence of interlayed sandstone, siltstone and shale; the soil is texture contrast and the A horizon is made up of grey fine sandy to silty loam (Section 5.3.1). Consequently, the soil matrix here has a large amount of intermediate sized grains competing for the available fallout radioactivity, thus reducing the concentrations on the fine clays.

The mid-Murrumbidgee region contains a diverse range of lithologies, including basalts, granites and sedimentary material. A range of soils have developed over these bedrock types. As a result the average concentrations of $^{210}Pb_{ex}$ and ^{137}Cs from all these different mid-Murrumbidgee soils, lie between and overlap with, those from Whiteheads Creek and Black Mountain.

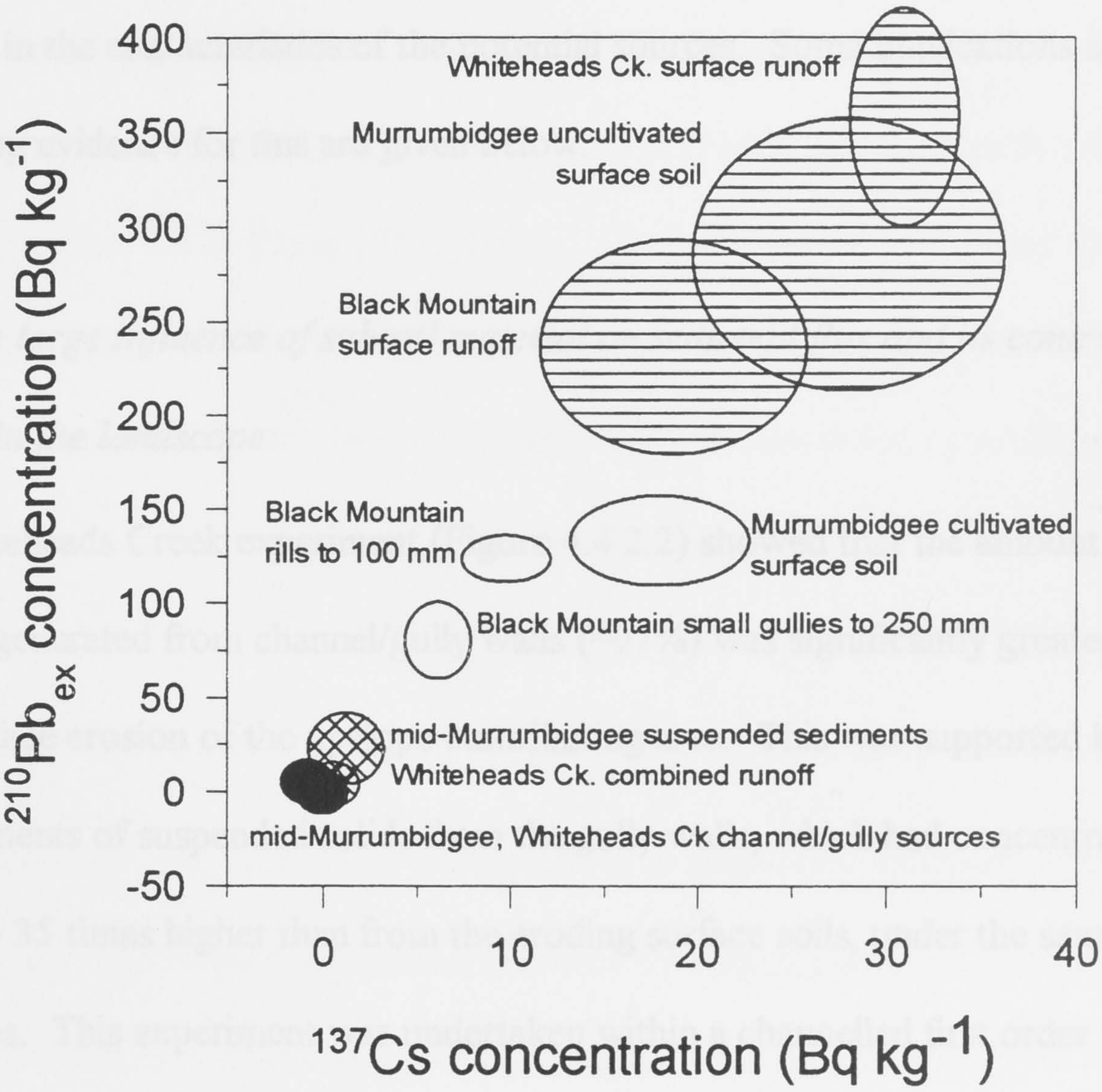

Figure 7.5.1.1 Suspended sediment, surface runoff and catchment source radionuclide concentrations from Whiteheads Creek, Black Mountain and the mid - Murrumbidgee region. Note: The values for the different parameters are given as a circle defined by the mean at the centre and the variability to one standard deviation surrounding it. The Treatment 1 and 2 values from the Black Mountain experiment have been combined to give one surface runoff value. All concentrations are for fine clay size material

The values from streambank/gully sources are also very similar and are well separated from the surface erosion values. The material from the Black Mountain deep rill (100 mm) and shallow gully (250 mm) experiments have concentrations between the surface and subsurface extremes. The radionuclide concentrations on suspended sediment material from the small scale and large scale are very similar and their location adjacent to the subsoil values suggests domination from this latter source at these locations. It appears that this method consistently predicts a large contribution from subsoils at the different scales and landform types presented here, and that there is relatively little variation in the characteristics of the potential sources. Some implications and supporting evidence for this are given below.

7.5.2 The large influence of subsoil material on sediment flux and its control by channel position in the landscape

The Whiteheads Creek experiment (Figure 4.4.2.2) showed that the amount of subsoil material generated from channel/gully walls (~97%) was significantly greater than that from surface erosion of the upslope contributing area. This was supported by measurements of suspended solids from the gully walls, which had concentrations (mg L^{-1}) up to 35 times higher than from the eroding surface soils, under the same rainfall conditions. This experiment was undertaken within a channelled first order stream and implies that the substantial contribution by subsoils to suspended sediment flux begins at the point in the landscape where gullies and channels first occur.

The results of Figure 7.3.2.1 and the mixing model analysis of Section 6.6.3, showed that the material from channel/gully sources (~90%) also exceeds that from the surface (~10%) at the much larger scale of the mid-Murrumbidgee. Within this system, channels and gullies are found right up to the headwaters and it can be assumed that these exert a similar control on sediment yield in this landscape as at Whiteheads Creek. Whiteheads Creek is also a first-order channel of the Wollondilly catchment (~5,400 km^2), which ultimately drains into Lake Burragorang (catchment area, ~9000 km^2), the main water supply for Sydney. Sediment flux in this catchment was similarly found to be dominated by subsoils (~70%) derived from channel/gully sources (Fredericks, 1994). The results from Whiteheads Creek, mid-Murrumbidgee and Lake Burragorang are also consistent with the research of Neil and Fogarty, (1991) and Sebire (1991) who found, in a systematic survey of farm dam sediments in the southern tablelands region of NSW, that sediment yield from channelled catchments exceeded that from native forest and improved pasture by an order of magnitude. The latter author concluded that gully erosion dominates sediment sources of small catchments, regardless of use. The Murrumbidgee and Wollondilly data suggest that this conclusion applies at much larger scales.

7.6 The contribution of subsoils to suspended sediments elsewhere in Australia

Comparisons of $^{210}Pb_{ex}$ and ^{137}Cs concentrations in suspended sediments with those of their sources, has revealed a common influence by subsoil material (Sections 7.3, 7.4 and 7.5). Additional data from Lake Burragorang and farm dam surveys (Neil and Fogarty, 1991; Sebire, 1991) implied that the result may be widespread. In this final section, these findings are extrapolated to other channelled catchments in Australia. Unfortunately, $^{210}Pb_{ex}$ measurements are not available in all these catchments. However, estimates of the subsoil contribution can still be made using concentrations of ^{137}Cs alone (Ritchie *et al.*, 1974; Campbell *et al.*, 1982; Peart and Walling, 1986; 1988; Froelich *et* *al.*, 1993; Wallbrink *et al.*, 1996a; 1996b). In this approach a simple two component mixing model, Equation 7.6.1 below, is used to compare the concentrations of ^{137}Cs on sediments with those from subsoils and topsoils and thus estimate the contribution from each of them.

$$Ax + By = z$$

$$A + B = 1 \qquad \text{Equation (7.6.1)}$$

Where A and B are the unknown amounts (%) from the subsoil and surface soil, and x, y and z are the known ^{137}Cs concentrations in the subsoil, surface soil and suspended sediments respectively. Ideally, the average concentration on surface soil material from all the catchment sources would be measured, as there may be differences between them. Similarly, the average channel bank and gully height in each catchment would also be estimated to calculate the appropriate ^{137}Cs concentration for subsoil material likely to reach the channel bed. This data is not available for all sites considered below, although the surface soil and subsoil data collected at the sites investigated in this thesis can be used in their absence. (Although it is acknowledged that systematic differences between these values derived here and elsewhere may exist because of differences in fallout rates and patterns). The average of all the surface soil measurements in this thesis, including that from the mid-Murrumbidgee cultivated lands and Black Mountain surface runoff, is 26.0 ± 1.0 Bq kg^{-1} (n=320). The average of all the subsoil concentrations from the mid-Murrumbidgee and Whiteheads Creek is 0.6 ± 0.1 Bq kg^{-1} (n=90) assuming a gully/bank height of 3 metres. The concentrations of ^{137}Cs measured on sediments from other channelled catchments are presented in Table 7.6.1. The proportionate contribution from subsoil to each of them using Equation 7.6.1, is given in a separate column.

The concentrations of ^{137}Cs in these other catchments are generally very low, implying a similar control on sediment flux by channel/gully sources. The estimated contribution by subsoil material in them is > 70%. The ^{137}Cs concentrations from the Werriberri and Goodradigbee catchments are also given for comparison, (in parenthesis) below the larger catchments of which they form a part.

Table 7.6.1 ^{137}Cs concentrations, and calculated subsoil contributions to, fine suspended and/or deposited sediment, in some channelled catchments in Australia.

Location	(n)	Grain size (µm)	^{137}Cs (Bq kg^{-1})	Area (km^2)	Percent of continent (%)	Subsoil amount (%)
Molonglo River, NSW. *[3]	75	<63	1.6	870	0.01	96
Murrumbidgee, NSW. *[!1]	164	<3	2.4	84000	1.1	93
(Goodradigbee) *[3]	5	<63	15.5		-	(41)
Ord River, W.A. #[3]	53	<63	3.8	46100	0.6	87
Swan River, W.A. *[3]	2	<63	3.2	135000	1.75	90
Wollondilly River, NSW. #[2]	51	<5	7	9000	0.1	74
(Werriberri) *[2]	8	<5	27.5	210	-	(0)
Murray River, VIC. *[3]	5	<5	2.3	421000	5.5	93
Darling River, NSW. *[#!3]	117	<10	1.7	641000	8.3	95
Total	378		2.9	1400000	~18	91

Note:

(i) * recently deposited core material, # grab samples of deposited material in stream channels, ! suspended sediments sampled by continuous flow centrifuge

(ii) [1] Values from total length of river channel, Wallbrink *et al.* (1996b)

(iii) [2] Fredericks (1994)

(iv) [3] Unpublished data derived from collaborative research with Olley, J.M. Murray, A.S., Olive, L.J., Hancock, G., Caitcheon, G.G., Wasson, R.J. and Crockford, H.

(v) Total surface area of Australia ~7,700,000 km^2

(vi) All ^{137}Cs concentration values decay corrected to 1995

The Werriberri catchment is known to be surface erosion dominated (Fredericks, 1994); the Goodradigbee is largely contained within a national park that is extensively forested and contains minimal gullying. It can be seen that their ^{137}Cs concentrations are much higher than elsewhere, and consequently their subsoil contribution estimates are much smaller than in the catchments with extensive channel /gully networks. These two catchments provide an important internal check on one of the assumptions of this analysis, ie that surface soil material retains its characteristic 'surface' signature irrespective of erosion and transport over large distances.

The results presented for the subsoil contribution in Table 7.6.1, assume that the grain size of the sediments from each catchment is similar to the <2 μm fraction used for the derivation of the surface soil component. Although the grain size cutoff is reported as <63 μm in some of these samples, the majority of this material was actually clay sized. If the mean grain size of the sediments was much greater than 2μm, this would cause the subsoil contribution amount to be overestimated. However, the ^{137}Cs concentrations of the sediments in these different regions are similar (Table 7.6.1), suggesting that the correlation between grain size and concentration is probably of second-order importance. A more likely explanation is that catchment processes exercise the primary control on the nuclide concentration of these sediments.

Another source of uncertainty in the calculation of source contributions, is the estimate of an appropriate average bank height. In this analysis an average height of 3m has been assumed. Although different channel/gully bank heights to this obviously occur, on average they are unlikely to be systematically different. Nonetheless, a change in this value of up to ± 2m results in only a ~5% difference in subsoil contribution amount using this model; this is within the uncertainties of the measurements themselves. Using this average value of ~3 m implies that the average contribution from subsoils to fine grained sediment yield in all of these channelled catchments is 90 ± 10%.

This is an intriguing finding for such a large percentage of the total land mass (~18%), especially given that rates of surface sheet and rill erosion in Australia are reported to be high (Department of Arts, Heritage and Environment, 1986; Department of Environment, Sport and Territories, 1996). The data here implies that this surface material must be inefficiently delivered to streamlines. Such a conclusion has been reported previously in Australia (Wasson, 1987); the primary causes are believed to be the characteristically low relief of the continent, sediment exhaustion from surface sources, and the generally low rainfall and runoff available to deliver eroded material (Olive and Reiger, 1986). Conversely, material from channels and gullies is rarely supply limited, and is very efficiently delivered due to their position within streamlines. The ~90% estimated subsoil contribution to suspended sediment flux in channelled catchments in Australia, is consistent with the upper estimates of channel bank/gully contribution to sediment yields elsewhere (Glymph, 1957; Poessen, 1989;1993; Osborn and Simanton, 1989; Crouch, 1990). This has clear implications for the management of sediments, nutrients and pollutants within the affected waterways.

7.7 Summary

Areal concentrations of $^{210}Pb_{ex}$ and ^{137}Cs have been shown to be similarly correlated at the large scale of the Murrumbidgee, and at the small scale of St. Helens. The variability in the ratio between them was shown to be smaller than for either nuclide alone. This implies that the ratio approach could be used to measure soil erosion in the Murrumbidgee landscape, with a consequent improvement in precision.

The method of using crossplots of ^{7}Be, $^{210}Pb_{ex}$ and ^{137}Cs (Bq kg^{-1}) to determine sediment source, and thus infer erosion mechanism, was successfully applied at the Black Mountain study site and was able to distinguish surface, shallow rill, deep rill and small gully surface treatments. Suspended sediment from the Murrumbidgee River was also examined in this way, leading to the inference that subsoils are the major source of this material. The Murrumbidgee suspended sediments were then examined using profile reference curves of $^{210}Pb_{ex}$ and ^{137}Cs, which estimated the cultivated material to be from <3 ± 10 mm at the surface, the uncultivated material from <2 ± 6 mm, and the channel bank material from below 400 mm. The relative amounts from these were consistent with those of the mixing model in Section 6.6.3, although problems with characterising the depth dependence of $^{210}Pb_{ex}$ and ^{137}Cs at this large scale were identified.

The suspended sediment data from the study sites were then combined and it is apparent that subsoil material dominated sediment flux at both small and large scales. Evidence was presented that subsoil dominance begins at the point in the landscape where channels first appear. Some ^{137}Cs analyses from other channelled catchments allowed the extrapolation of these results to a larger portion of the Australian landscape. A simple two component mixing model suggests that up to 100% (ie. 90 ± 10) of the material at the very large scale integrated by these river systems could be dominated by subsoil material, depending on various assumptions made about the ^{137}Cs concentrations on surface and subsoil material.

Chapter 8: Conclusions and suggestions for further research

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8.1 Perspective

The major aim of this thesis was to improve the way in which methods based on fallout radionuclides may enhance studies of erosion and sedimentation processes. Techniques have been developed and tested at a variety of study locations, and at a range of different scales. The major conclusions of this work are outlined in Section 8.2 below, in the context of the two major themes. Section 8.3 outlines suggestions for additional research that would improve the generality and reliability of the methods developed in this thesis.

8.2 Conclusions

8.2.1 Measuring soil erosion

An important part of assessing environmental degradation is measuring soil loss. In this way, the effect of different land management practices on soil erosion can be quantified, and the benefit of a (primary) industry can be balanced against its environmental cost. One method of measuring soil loss involves comparing the amount of radioactive fallout ^{137}Cs (Bq m^{-2}) in a reference area, with that of a disturbed area. The difference is taken to be a function of soil loss (or soil gain by sedimentation). A comparison on this basis assumes that the initial distribution of ^{137}Cs was spatially uniform. However, if the initial distribution of this nuclide was spatially variable then such comparisons could be in error. Consequently, the initial variability of this nuclide in reference areas was investigated, both as a function of different cover conditions (pasture or forest) and as a function of scale, and a method of allowing for this source of uncertainty developed.

The variability (relative standard deviation, *rsd*) of areal concentrations of ^{137}Cs was first measured in reference forest and grassland environments, and was found to be ~40%. This was independent of location and scale, and thus hypothesis *(1) ^{137}Cs is uniformly distributed within reference areas*, was rejected. The variability (~42%) of a small hillslope 1 ha plot was also found to be similar to that of the surrounding references areas. It was not possible to attribute a significant part of the variability of the nuclide to redistribution of the soil, and thus hypothesis *(2) The distribution of ^{137}Cs at the small (1 ha) scale is associated with soil movement alone*, was rejected. The *rsd* of ^{137}Cs and ^{210}Pb$_{ex}$ were also found to be independent of the presence of trees. For example, the highest and lowest *rsd* of both of these nuclides was found within forests (although not at the same location), and consequently hypothesis *(3) The variability of ^{137}Cs within tree covered areas will be higher than in pasture*, was also rejected.

The distribution of ^{210}Pb$_{ex}$ and ^{137}Cs (Bq m^{-2}) in the study sites appeared to be correlated, and it was shown that ~70% of the variability of both these nuclides in reference areas could be accounted for by the same factors, presumably related to their initial distribution. Consequently, the effects of initial fallout variability were partly compensated for by taking the ratio between them. At the two different locations studied, the *rsd* of this ratio was lower than that for either nuclide alone. Therefore hypothesis *(4) The ratio of the soil inventories of ^{210}Pb$_{ex}$ and ^{137}Cs has a lower overall variability in 'reference' areas than the areal concentrations of either nuclide alone*, was not rejected. Differences between this ratio in 'reference' areas and those in disturbed areas of a logged forest were observed and then quantified to soil loss using a measured depth dependence relationship; it was found that soil loss under 'normal' impact harvesting was greater than under 'minimal' impact harvesting. Consequently, hypotheses *(5) The ratio of $^{210}Pb_{ex}$ to ^{137}Cs in forested reference areas is different from those in harvested areas*, and *(6) Soil loss from a 'normal' impact logged plot will exceed that from a 'minimal' impact logged plot*, were not rejected. In cultivated soils however, interpretation of this ratio requires a better understanding of the effect of continuous soil erosion and fallout of ^{210}Pb.

The study areas in which the correlation between these nuclides was observed represents a large proportion of southeastern Australia. This region supports many valuable primary industries, yet there is a paucity of information regarding their impact on the soil store. This thesis suggests that although these impacts can be quantified using ^{137}Cs alone, potentially better estimates of loss can be made by incorporating measurements of fallout ^{210}Pb. This is because the effects of initial nuclide variability will be reduced, and thus differences between reference area and disturbed area values are more likely to represent genuine soil losses.

8.2.2 Methods for tracing and quantifying the sources of suspended sediments

The impact of soil erosion is rarely limited to the slopes from which it has occurred. The greatest effect may be on aquatic environments downstream; where water quality declines, habitats are destroyed, or water storages are filled and/or polluted. Effective catchment (or reservoir) management should begin from a knowledge of sediment and pollutant sources. In this way, the problem can be tackled at its source. Consequently, techniques are required to accurately and reliably predict the predominant source of eroded material. A method is proposed in this thesis which involves the fallout radionuclides ^{7}Be, ^{137}Cs, and ^{210}Pb and takes advantage of their different depth distributions in soils. For example, the depth distributions of ^{7}Be, $^{210}Pb_{ex}$, and ^{137}Cs in soils were found to be distinctly different, and so hypothesis *(7) The tracers* ^{137}Cs*,* $^{210}Pb_{ex}$*, and* ^{7}Be *have different initial soil depth distributions in undisturbed soils*, was not rejected. These different distributions were found to give rise to different concentrations (and nuclide ratios) on sediments derived from them. In general, material derived from surface erosion sources had high concentrations of these tracers and that from subsoil sources had low concentrations, thus hypothesis *(8) Eroded material from soil surfaces will have high fallout radionuclide concentrations and derived subsoil material will have low fallout concentrations*, was not rejected. Sediments from erosion processes that cut into the solum, i.e. deep rills or shallow gullies, were shown to have radionuclide concentrations somewhere between these two extremes. Tracer concentrations on suspended sediments were also shown to approximate the depth of origin of those sediments, and hence infer the most likely erosion process for their production. This approach was used to demonstrate that the contribution of subsoil material from gully erosion exceeded that from surface erosion, by at least an order of magnitude, at Whiteheads Creek, NSW. Therefore hypothesis *(9) The net contribution from surface erosion will exceed that from gully erosion*, was rejected in this particular case.

The known depth distributions of these nuclides were combined to produce reference curves of ^{7}Be to ^{210}Pb$_{ex}$ and ^{210}Pb$_{ex}$ to ^{137}Cs, at Black Mountain, ACT. The ratio of one nuclide to another was shown to vary systematically with depth, and so hypothesis *(10) The combined depth profiles of* ^{137}Cs*,* $^{210}Pb_{ex}$*, and* ^{7}Be *provide unique concentration labels on soil material from different depths within those profiles*, was not rejected. Concentrations of the tracers in suspended sediment were then compared to these curves and used to calculate sediment source depths and the relative contribution from each depth layer. Therefore, hypothesis *(11) The concentration labels of* ^{137}Cs*,* $^{210}Pb_{ex}$*, and* ^{7}Be *on sediments from these profiles can be used to determine their depth origin(s) in the soil profile*, was also not rejected. This method apparently resolved the depth contribution from four different erosion processes to within millimetres. Resolution of the subsurface component was about an order of magnitude less precise. For example, in overland flow and shallow rill experiments, the method predicted the material to be from surface depths of 1 ± 2 mm and 4 ± 4 mm, respectively. In later experiments, deep rills and gullies were incised into the plot surface, and subsoil contribution amount was shown to increase from ~50% to ~75% as their depths and widths were enlarged. Thus, hypothesis *(12) Surface material derived from shallow rill and overland flow will exceed that from sides and floors of deeper rills and gullies*, was rejected.

The concentrations of ^{137}Cs and ^{210}Pb$_{ex}$ were found to be different on material from the three potential sources of fine grained sediment in the mid-Murrumbidgee catchment: uncultivated lands, cultivated lands, and channel banks/gully walls. Thus hypothesis *(13) Different land use and land form types will produce distinct radionuclide concentrations in sediments eroded from them*, was not rejected. A simple mixing model estimated that the contribution, from cultivated lands to fine grained sediment in the Murrumbidgee River was about twice that from uncultivated lands. However, as this land use occupies only ~22% of the catchment surface area, its specific yield (t ha^{-1}) is ~6 times that from uncultivated lands. The greatest contribution (~89%) was from subsoil sources, presumably from erosion of stream channels and gully walls. Consequently, hypothesis *(14) Surface material from the heavily cultivated lands of the mid-Murrumbidgee will dominate the flux of material in the channels draining this area*, was rejected. The mixing model estimates of contribution from these sources was dependent on residence time of the fine grained sediments, which on average, was estimated to be no more than ~6 years, and so hypothesis *(15) The average residence times of fine particulates in this large catchment will be longer than 10 years*, was also rejected. Additional evidence from changes in ^{137}Cs concentrations on flood and non flood sediments, and the presence of the short lived cosmogenic ^{7}Be, suggest probable residence times of this fine grained material are in the order of weeks to months, rather than years to decades. The fate of other particle size classes are unknown.

Analysis of suspended sediments from the overland flow and gully wall experiments at Whiteheads Creek suggests that subsoil begins to make a significant contribution to the total sediment flux at the point in the landscape where channels or gullies first appear. An analysis of sediments from the much larger mid-Murrumbidgee catchment revealed a similar domination by subsoil sources, also presumably controlled by material emanating from channel/gully sources. It appears that the large subsoil contribution that occurs at small scales applies at much larger scales, most likely resulting from the postulated short transit time and likely conservative transport of fine grained material in these deeply incised systems.

Measurements of ^{137}Cs on sediments in other large channelled catchments in Australia suggest that some ~18% of the continent, at least, may be similarly controlled by subsoil erosion. Although surface soil erosion is shown to dominate in specific undisturbed catchments, it is concluded that subsoils dominate the sources of suspended sediments in those catchments where gullying and channelling occurs. Consequently, any work that aims to reduce the supply and delivery of sediment in these catchments should focus on the yield from these subsoil sources. This may involve measures to prevent stock access such as fencing and the provision of off-channel watering points for domestic stock. It may also be necessary to undertake minor engineering works to stabilise streambanks, by installing grade control structures, revegetation of gully floors and walls, and contour banks to exclude water from gully headwater regions. In the final analysis however, it must be remembered that gullying and channelling remain natural phenomena, that their extent is considerable, and the resources to effectively treat them may be better used.

8.2.3 Review of hypotheses

A review of the hypotheses tested in this thesis are given in Table 8.2.3.1. The result of the testing is noted, as well as a comment about the region over which the hypothesis outcome applies. For example hypothesis *(9) The net contribution from surface erosion will exceed that from gully erosion,* was initially rejected on the basis of the Whiteheads Creek experiment alone. However, the regional analysis of sediment sources in Sections 7.5 and 7.6, combined with the farm dam work of Neil and Fogarty (1991) and Sebire (1991), allow this result to be tentatively extrapolated to other channelled/gullied regions in Australia. The related hypothesis; *(12) Surface material derived from shallow rill and overland flow will exceed that from sides and floors of deeper rills and gullies* was similarly rejected. However, in this case the outcome applies only to Black Mountain, because of the specific nature of the experimental work at that site. In some cases, the generalisation of a hypothesis outcome to larger regions, i.e. *(13) Different land use and land form types will produce distinct radionuclide concentration in eroded sediments,* is provided on the basis of supporting evidence from other locations, cited in the literature.

Table 8.2.3.1 Summary of hypotheses and consideration of region to which hypothesis outcome applies

Hypothesis	Outcome	Region for which hypothesis outcome applies
(1) ^{137}Cs is uniformly distributed within reference areas	**rejected**	Global, from this work, and others in the literature, (eg. Sutherland, 1991)
(2) The distribution of ^{137}Cs at the small (1 ha) scale is associated with soil movement alone	**rejected**	Global, on the basis of the rejection of (1)
(3) The variability of ^{137}Cs within tree covered areas will be higher than in pasture	**rejected**	ACT region only
(4) The ratio of the soil inventories of $^{210}Pb_{ex}$ and ^{137}Cs has a lower overall variability in 'reference' areas than the areal concentrations of either nuclide alone	**not rejected**	Southeastern Australia, from this work
(5) The ratio of $^{210}Pb_{ex}$ to ^{137}Cs in forested reference areas is different from those in harvested areas	**not rejected**	Possibly southeastern Australia
(6) Soil loss from a 'normal' impact logged plot will exceed that from a 'minimal' impact logged plot	**not rejected**	St Helens study area
(7) The tracers ^{137}Cs, $^{210}Pb_{ex}$, and ^{7}Be have different initial soil depth distributions in undisturbed soils	**not rejected**	Global, from this work and others in the literature (eg. Walling and Woodward, 1992)
(8) Eroded material from soil surfaces will have high fallout radionuclide concentrations and subsoil derived material will have low fallout concentrations	**not rejected**	Global, providing soil is initially undisturbed
(9) The net contribution from surface erosion will exceed that from gully erosion	**rejected**	Gullied catchments in Australia
(10) The combined depth profiles of ^{137}Cs, $^{210}Pb_{ex}$, and ^{7}Be provide unique concentration labels on soil material from different depths within those profiles	**not rejected**	Undisturbed soils of southeastern Australia
(11) The concentration labels of ^{137}Cs, $^{210}Pb_{ex}$, and ^{7}Be on sediments from these profiles can be used to determine their depth origin(s) in the soil profile	**not rejected**	Undisturbed soils of southeastern Australia
(12) Surface material derived from shallow rill and overland flow will exceed that from sides and floors of deeper rills and gullies	**rejected**	Black Mountain, the outcome elsewhere is dependent on erosion conditions at each location
(13) Different land use and land form types will produce distinct radionuclide concentrations in sediments eroded from them	**not rejected**	Possibly global, from this work and others (eg. He and Owens, 1995)
(14) Surface material from the heavily cultivated lands of the mid-Murrumbidgee will dominate the flux of material in the channels draining this area	**rejected**	Mid-Murrumbidgee region, NSW
(15) The average residence times of fine particulates in this large catchment will be longer than 10 years	**rejected**	Mid-Murrumbidgee region, NSW

8.3 Suggestions for further research

8.3.1 Soil erosion

In Sections 7.2.2 , 7.2.3, and 7.2.4, inventories of $^{210}Pb_{ex}$ and ^{137}Cs (Bq m^{-2}) were shown to be correlated in the mid-Murrumbidgee, and the *rsd* of the activity ratio was smaller than for either nuclide alone. However, quantifying this ratio in terms of soil loss was complicated by the poorly described effects of continuous cultivation and erosion. Additional modelling is needed to explicitly take into account the different initial soil stores of these nuclides, their contrasting fallout histories, the effect of continual losses, and the mixing of soil particulates due to cultivation and lateral transport. Further work is also required to investigate the correlation between $^{210}Pb_{ex}$ and ^{137}Cs inventories elsewhere in Australia, and the way in which they vary with depth.

8.3.2 Sediment source tracing

It is assumed in this thesis that the arguments for the concentrations of fallout ^{137}Cs, ^{210}Pb, and ^{7}Be being conservative tracers of sediments are sound, and therefore that no major physical or chemical alteration occurs to them during transport. There is ample evidence that the primary initial control on radionuclide concentration is particle size and soil depth, and that these concentrations are not markedly altered by transport. However, further examination of tracer behaviour under various conditions needs to be undertaken to support these empirical observations. This could be undertaken by simulating the possible alterations to sediments that occur during transport, eg. by mechanically abrading sediments while subjecting them to a range of different redox, pH, and varying salinity conditions. Such experiments would identify the environmental conditions where the tracers are most likely not to reflect the source characteristics.

This thesis also concluded that fine grained sediment in rivers draining ~18% of the Australian continent may be dominated by material from subsoil sources. Further measurements need to be made in the soils and sediments of these systems, and in other large basins, to generalise this finding. Measurements of ^{137}Cs should be complemented by similar measurements of ^{210}Pb$_{ex}$. However, interpretation of the latter nuclide will require careful consideration; the potential for alteration to ^{210}Pb's signature by direct fallout increases with scale, deposition-resuspension behaviour of suspended sediment material within the channel system, and residence time.

The residence time of fine grained material was shown to be quite short in the mid-Murrumbidgee catchments. Clarification of the in-channel behaviour of ^{210}Pb$_{ex}$, as well as a more detailed understanding of the sediment yield relationship M (Section 6.6.2), would provide better estimates of the residence time of the fine grained material within these catchments. This would benefit our understanding of the nature of sediment delivery, not only in the mid-Murrumbidgee, but in drainage systems elsewhere. This information could also be incorporated into a tentative sediment budget for the mid-Murrumbidgee region by combining (i) the known total sediment flux with (ii) tracer-based estimates of soil loss (when available) and (iii) the estimated contributions of these to stream sediment flux. Sediment budgets such as these are essential for scientists and river regulators to understand the potential effects of various land management practices on sediment fluxes downstream, and the time scales over which these effects may occur.

8.4 Concluding remarks

Soil and water are commodities of great value. In Australia, where the rainfall is low and the environment is fragile, the health of the two are inextricably linked. Thus, if we hope to inhabit this land for any ongoing period we must respect these life giving factors. I contend that indicators which allow us to gauge soil degradation are a key to protecting these resources. This thesis has provided indicators for measuring soil erosion and consequent impacts on downstream environments, tools by which we can seek to protect our soil and water. Failure to use these, and other indicators, could turn neglect by default into conscious injury.

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Appendix

A: Analytical methods

This section describes how the radionuclide analyses of the soils and sediments in this thesis were undertaken. The procedures for pretreating soils and sediments prior to analysis are presented first, followed by a discussion of the gamma spectromic methods used to measure their radionuclide content.

A.1 Soil and sediment processing

A.1.1 Soil samples

Upon return to the laboratory the soil samples were weighed, oven dried and then weighed again, to determine moisture content. A small amount ~10g was then placed in a labelled, and sealed, container for future reference. The residual soil was then ashed in a muffle furnace, at 400^0 Celsius, to remove any organic material and then weighed again. The loss on ignition (LOI) from combustion of organic material was then calculated. The ashed mineral material was then ground to a fine powder in a Rocklabs™ ring mill, (to a diameter of ~3μm). In this form the soil material was then ready for casting in polyester resin, prior to radionuclide analysis.

A.1.2 Suspended sediment samples

As outlined previously, suspended sediments were obtained by continuous flow centrifuging of sediment from stream waters and plot runoff. The centrifuged material was generally returned to the laboratory as a slurry (with water). It was first oven dried and then weighed, to determine the sample mass. A subsample was put aside for future reference as above. The sample was weighed again and ashed in a muffle furnace to determine the LOI. The ashed sediment material was then ground in the ringmill, ready for casting in polyester resin.

A.1.3 Standard sample geometries

Before the ashed and ground samples were analysed for their radionuclide content, they were first distributed in one of three distinct geometric shapes. Polyester resin was used as the retaining matrix. The specific geometric shape used was determined by the sample mass. For example, large masses were placed in the 'inverted Marinelli beaker' geometry; the total volume of which was 201 cm^3, containing roughly 220-250g of soil/sediment, depending on sample density. Smaller mass samples, of between 20 and 40 g were cast as a 'planar disk' with volume of 33 cm^3. Samples with masses of below 10 g were cast as a rod, with internal volume of 7 cm^3.

A.1.4 The resin casting process

The different geometric shapes were produced by aluminium moulds. The moulds themselves were constructed from a series of components, which if assembled in the correct order would leave the appropriate geometric shape exposed as a void inside them. The soil/sediment powder (following ashing and grinding in the ringmill) was then mixed with polyester resin and a small amount of hardener was added, ~1% by volume of resin. This mixture was then poured into the moulds while still viscous. The weight of the soi/sediment sample, the polyester resin and the hardener were all carefully noted during this process, in order to calculate the net amount of soil/sediment retained within each sample. After 24 hours of curing the moulds were then dismantled to reveal the sample in solid geometric form.

A.1.5 Benefits of using a polyester resin matrix

There are advantages and disadvantages to producing soil/sediment samples within a polyester resin matrix. Some of the benefits are listed below.

i) The determination of ^{210}Pb in excess of its parent ^{226}Ra (ie ^{210}Pb$_{ex}$) requires analysis of both ^{210}Pb and the ^{226}Ra components. Unfortunately, although ^{210}Pb is readily identifiable at 46 keV, the ^{226}Ra line is obscured at 186 keV by emissions from ^{235}U and so must be measured through its daughters, ^{214}Bi and ^{214}Pb at 295, 352 and 609 keV. These are the product of a very short lived parent, and gaseous ^{222}Rn (half life 3.5 days). Thus in order to measure ^{226}Ra through ^{214}Bi and ^{214}Pb, the whole sample must be gas tight to ensure equilibrium in the lower part of the chain. The polyester resin effectively retains all ^{222}Rn and thus its daughters.

ii) The radioactive standards used to calibrate the detectors are prepared in the same three geometries as the soil/sediment samples, using the same polyester resin. This ensures that the sample matrix and density are constant for both the calibration standards and the 'unknown' soil/sediment samples. This reduces the potential for systematic errors due to density differences between the calibration standards, and the 'unknowns' with which they are compared.

iii) If all the unknown samples are presented to the detectors with the same matrix and density, the potential for systematic and random errors between analyses are reduced.

iv) The polyester resin matrix is very robust, this reduces the chances of breakage and physical damage, in addition they can be stored for long periods for future access.

Disadvantages of this approach include the effort and cost of preparing samples with polyester resin. Alternatives to this involve pressing samples into gas tight tins, or plastic containers with a hydraulic ram. In these methods the sample is accessible, whereas with polyester resin it can only be retrieved by ashing in a muffle furnace.

A2 Procedures for analysis of radionuclide content

A.2.1 Detectors used in this thesis

Five detectors were used to generate analyses for this thesis; two 'Ortec' HPGe closed Ended 'n' type Coaxials, one 'Canberra' HPGe closed Ended 'n' type Coaxial, one 'Ortec Poptop' HPGE 'n' type and one 'Ortec' HpGe well detector 'p' type. The advantage of the 'n' type coaxial detectors is that they have relatively good efficiencies at energies below 50 keV, this is very important given that ^{210}Pb is an emitter at 46 keV. The loss of efficiency at low energies with the 'p' type well detector is compensated for by the greater resolution afforded by the re-entrant geometry. (Note: HPGe stands for Hyperpure Germanium). Samples prepared in the 'inverted marinelli' shape fitted over the end of the detector can, effectively surrounding the germanium crystal assembly. Samples in the 'planar disk' form were placed directly on top of the detector can and held in place by a locating ring. The 'rod' samples were used only in the re-entrant of the well detector. A description of the different efficiency curves for these geometries is given in Murray *et al*, (1987).

A.2.2 Calibration of HPGe Gamma spectrometers

The five gamma spectrometers were directly calibrated using standard sources of known activity. This overcomes problems of different attenuation characteristics of gamma emitters in the energy range of interest. ie 46 (^{210}Pb) to 662 keV (^{137}Cs). The different procedures for ^{137}Cs, ^{7}Be and ^{210}Pb are outlined below (after Murray *et al.*, 1987).

The ^{137}Cs and ^{7}Be standards were both based on standardised solutions provided by Amersham International. A quantity of these solutions was evaporated to dryness on an appropriate ground matrix (such as an inert sand) and then homogenised by ringmill grinding before casting. Three standards of each nuclide in each geometry were prepared. These replicates provided an estimate of the errors associated with the mechanical preparation of the standards. Each standard was then counted on each of the detectors for a period of ~84 ksecs. The total peak area, less the background, was then converted to a normalised count rate (cksec^{-1} per Bq kg^{-1}), to which the count rate of 'unknown' soil/sediment samples could then be compared to calculate their activity (Bq kg^{-1}). The 'n' type detectors used in this work were able to resolve the 662 keV line of ^{137}Cs from that of an interfering Bismuth-214 line at 665 keV. Consequently no corrections for the presence of this nuclide was necessary. There is no interference near the ^{7}Be peak at 478 keV.

Excess lead-210 is usually determined by subtracting the concentration of ^{210}Pb from that of its parent, ^{226}Ra. However analysing for ^{226}Ra and ^{210}Pb requires calibrating the detectors for the U-238 series, of which both are daughters. The process was begun by creating U-series standards from Uranium ore of known activity (BL-5) provided by CANMET. (These suppliers produce evidence that ^{210}Pb is in equilibrium with ^{226}Ra). The ore is diluted by weight into the same inactive matrix as the ^{137}Cs and ^{7}Be standards, and then ground in the ringmill again. Three replicates of each geometry were then made and set aside for at least five half lives of ^{222}Rn (23 days - to allow build up of the radon daughters to equilibrium with the parent ^{226}Ra). These standards were then counted in the same way as described for ^{137}Cs, generating normalised count rates for ^{235}U (186 keV), ^{234}Th (63 keV), ^{230}Th (63keV), ^{226}Ra (186 keV), ^{214}Pb (295 keV), ^{214}Pb (352 keV) ^{214}Bi (609 keV) and ^{210}Pb (46 keV). In order to separate the ^{226}Ra contribution to the 186 keV peak from ^{235}U interference however, a separate measurement of ^{226}Ra and its daughters is also needed. This is undertaken using a pure

^{226}Ra solution provided by ANSTO, again evaporated to dryness and cast into replicate standards using the method outlined for ^{137}Cs above. These were also stored for a period of at least 23 days before counting. The ratio of the intensity of the 186 keV line (from ^{226}Ra alone), to that of each of the radon daughters at 295, 352 and 609 keV can then be used to determine the concentration of ^{226}Ra in samples containing ^{235}U. The intensity of the 186 keV line now gives the ^{235}U concentration after subtraction of the ^{226}Ra contribution. At this stage both absolute ^{210}Pb and ^{226}Ra are known, and so the difference between them, where applicable (ie the excess) can be calculated by subtracting the ^{210}Pb from the ^{226}Ra. The overall uncertainties using this calibration approach (including systematic effects) are estimated to be less than 3%, at the two sigma level.

A.2.3 Background interference to HPGe detectors

The background interference to a detector is generally the product of internal and external factors. The internal component is usually a combination of the materials used in the detector itself and the method of construction, once present it is difficult to remove. The external component consists of all the interfering radiation present from cosmic and terrestrial sources. For the gamma spectrometers used in this thesis, the external background was reduced by surrounding each detector with a lead shield of 10 cm thickness. Inside this was a steel shield of 10 mm thickness, to attenuate high energy X-rays generated from the external lead shielding. Residual interference to the detector itself, was then directly measured in the same way as for the 'unknown' or standard samples. In this case no active sample is present on the detector, only an inert resin disk. The residual background count rates from this process, for the energy lines of interest, are then subtracted from those of the 'unknown' or standard samples to produce the net concentration (Bq kg^{-1}) of each radionuclide.

Table B1

St Helens, radionuclide concentrations from soil pit excavated at reference Plot 1

Code	Depth Mid Point (mm)	Pb-210 (ex) (Bq/kg)	se	Cs-137 (Bq/kg)	se	Pb-210 (ex) Cumulative inventory (Bq/m2)	se	Cs-137 Cumulative inventory (Bq/m2)	se
SS92001/1,	0.5	53.7	4.6	12.2	0.8	1543.5	15.7	638.1	29.6
SS92001/2,	1.5	72.5	4.2	15.1	0.5	1519.2	15.6	632.5	30.0
SS92001/3,	2.5	87.5	7.2	16.6	0.8	1476.1	15.5	623.6	30.0
SS92001/4,	3.5	122.9	6.3	27.7	1.0	1420.1	15.4	612.9	30.0
SS92001/5,	4.5	144.9	5.8	31.1	0.7	1353.2	15.3	597.8	30.0
SS92001/6,	5.5	118.6	6.6	34.3	1.3	1271.9	15.2	580.4	30.0
SS92001/7,	6.5	132.2	9.3	31.9	1.4	1224.7	15.1	566.7	30.0
SS92001/8,	7.5	113.7	9.7	30.9	1.3	1164.2	14.9	552.1	30.0
SS92001/9,	8.5	123.8	6.7	36.0	0.9	1077.7	14.7	528.6	30.0
SS92001/10	9.5	84.0	10.7	25.6	1.6	959.5	14.5	494.3	30.0
SS92001/11	11	74.9	3.1	26.6	0.4	884.0	14.1	471.3	30.0
SS92001/12	13	58.8	4.4	19.1	0.5	730.7	13.9	416.8	30.0
SS92001/13	15	45.6	3.4	15.8	0.5	592.6	13.5	372.0	29.9
SS92001/14	17	30.6	3.6	10.4	0.4	500.7	13.3	340.1	29.9
SS92001/15	19	30.3	2.5	9.6	0.2	428.1	12.9	315.6	29.9
SS92001/16	22.5	16.9	2.6	6.7	0.2	365.0	12.7	295.6	29.9
SS92001/17	27.5	10.8	2.4	4.2	0.2	278.4	12.2	261.5	29.9
SS92001/18	35	7.3	2.3	3.2	0.2	219.3	11.7	238.8	29.9
SS92001/19	45	4.3	3.2	2.2	0.3	106.9	10.0	189.3	29.7
SS92001/20	55	2.5	3.5	1.4	0.3	62.6	8.2	167.3	29.6
SS92001/21	65	1.5	2.1	1.1	0.2	27.7	7.6	148.4	29.3
SS92001/22	75	0.8	2.1	0.9	0.3	14.4	6.2	138.8	29.3
SS92001/23	85	-0.7	2.1	0.7	0.2	-2.0	6.1	121.8	28.8
SS92001/24	95	1.4	2.1	0.5	0.2	3.0	5.2	114.1	28.8
SS92001/25	105	-	-	0.6	0.2	-	-	107.1	28.7
SS92001/26	115	-	-	0.5	0.1	-	-	96.9	28.6
SS92001/27	125	-	-	0.4	0.1	-	-	89.8	28.5
SS92001/28	135	-	-	0.3	0.1	-	-	83.6	28.4
SS92001/29	145	-	-	0.5	0.3	-	-	79.2	28.4
SS92001/30	155	-	-	0.4	0.1	-	-	69.7	27.9
SS92001/31	165	-	-	0.1	0.3	-	-	62.5	27.8
SS92001/32	175	-	-	0.2	0.3	-	-	61.8	27.7
SS92001/33	185	-	-	0.3	0.3	-	-	57.9	27.3
SS92001/34	195	-	-	0.3	0.3	-	-	58.4	26.5
SS92001/35	210	-	-	0.2	0.3	-	-	22.1	26.0
SS92001/36	235	-	-	0.1	0.3	-	-	10.6	23.5

Table B2

St Helens, radionuclide concentrations from soil pit excavated at reference Plot 4

Code	Depth Mid Point (mm)	Pb-210 (ex) (Bq/kg)	se	Cs-137 (Bq/kg)	se	Pb-210 (ex) Cumulative inventory (Bq/m2)	se	Cs-137 Cumulative inventory (Bq/m2)	se
SS92009/1	1	11.0	2.2	2.5	0.2	671.9	18.5	270.9	8.8
SS92009/2	3	16.0	5.8	5.0	0.7	651.1	18.4	266.1	9.2
SS92009/3	5	29.9	2.7	7.3	0.3	634.8	18.2	261.0	9.9
SS92009/4	7	28.5	2.4	8.2	0.4	581.3	18.1	248.0	10.2
SS92009/5	9	23.5	4.1	8.1	0.8	523.5	18.0	231.5	10.6
SS92009/6	11	21.5	3.1	6.5	0.6	446.6	17.6	204.8	10.6
SS92009/7	13	14.0	3.1	5.1	0.5	395.2	17.4	189.3	10.6
SS92009/8	15	17.8	3.3	4.4	0.7	359.5	17.1	176.2	10.6
SS92009/9	17	12.4	2.2	3.5	0.2	304.9	16.9	162.8	10.5
SS92009/10	19	10.6	4.2	3.7	0.8	271.7	16.7	153.4	10.4
SS92009/11	21	6.7	2.5	2.7	0.5	243.0	16.3	143.4	10.3
SS92009/11	23	7.4	3.0	2.3	0.6	224.8	16.1	136.0	10.3
SS92009/12	25	6.5	4.0	2.0	0.5	204.6	15.9	129.9	10.2
SS92009/13	27	5.1	2.9	2.1	0.3	187.2	15.5	124.6	10.1
SS92009/14	29	3.8	5.8	1.8	1.2	173.6	15.3	118.9	10.1
SS92009/15	35	3.0	2.5	1.6	0.4	162.8	14.7	113.9	9.9
SS92009/16	45	2.2	1.4	0.6	1.3	106.8	12.3	84.1	9.5
SS92009/17	55	2.4	2.0	0.7	0.4	69.0	8.8	73.4	8.3
SS92009/18	65	2.9	1.5	0.8	0.5	37.3	6.1	64.2	8.1
SS92009/19	75	2.0	3.6	0.4	0.6	8.6	12.5	54.0	7.6
SS92009/20	85	-	-	0.4	0.4	-	-	47.2	7.0
SS92009/21	95	-	-	0.7	0.3	-	-	39.2	6.4
SS92009/22	105	-	-	0.2	0.8	-	-	24.1	5.9
SS92009/23	115	-	-	0.3	0.3	-	-	16.9	4.4
SS92009/24	125	-	-	0.1	0.4	-	-	9.9	3.6
SS92009/25	135	-	-	0.5	0.3	-	-	7.9	2.3
SS92009/26	145	-	-	0.1	0.1	-	-	1.3	1.0
SS92009/27	155	-	-	0.1	1.3	-	-		

Table B3

Whiteheads Creek, suspended sediment and radionuclide values from First experiment

ampl umbe	Time (mins)	Suspended sediments (mg/L)	Pb-210 (ex) (Bq/kg)	se	Cs-137 (Bq/kg)	se	Be-7 (Bq/kg)	se	Th-232 (Bq/kg)	se	Ra-226 (Bq/kg)	se
1	00:15	1013	289.2	19.9	29.4	1.6	319.9	29.9	104.4	2.0	86.9	2.0
2	00:43	399	316.4	28.9	32.1	2.2	262.0	36.6	105.9	2.6	82.0	2.8
3	01:08	263	300.0	34.1	27.7	2.6	289.9	43.1	100.9	3.0	79.2	3.3
4	01:33	411	238.4	18.7	29.8	1.5	223.9	25.7	101.7	1.8	78.3	1.9
5	01:57	326	262.3	22.4	27.5	1.7	214.2	29.3	95.0	2.5	77.3	2.2
6	02:12	543	228.9	24.3	30.6	2.0	205.2	20.6	100.7	2.3	79.3	2.5
7	02:41	308	251.7	25.7	32.7	2.0	208.9	21.5	105.1	2.4	83.5	2.6
8	03:07	303	262.4	27.8	31.1	2.2	229.7	23.5	103.4	2.6	78.2	2.7
9	03:32	362	220.6	8.5	30.3	0.9	203.5	18.3	99.1	1.6	79.7	1.3
10	03:58	417	226.7	21.7	30.6	1.8	180.2	18.5	99.8	2.1	78.5	2.2
11	04:22	349	211.0	15.2	32.1	1.2	177.4	14.5	103.8	1.5	78.4	1.6
12	04:27	299	165.3	6.0	28.5	3.4	136.9	36.4	97.2	3.9	74.9	4.3

Whiteheads Creek, radionuclide values from Second experiment

Radionuclide concentrations on material from overland sheet/rill flow

	Start	Finish										
1	10:18 AM	10:33 AM	400.5	46.4	34.0	3.4	247.0	47.6	86.9	3.8	81.8	4.3
2	10:42 AM	10:57 AM	241.0	62.2	25.0	4.5	292.8	63.4	81.8	5.0	78.1	5.7
3	11:07 AM	11:22 AM	374.4	50.8	33.8	3.7	336.9	55.2	86.6	4.1	85.9	4.7
4	11:31 AM	11:46 AM	286.3	47.7	35.4	3.7	141.0	48.4	86.2	4.0	85.4	4.5
5	11:55 AM	12:11 PM	421.9	117.0	34.4	8.5	131.2	121.1	77.3	9.3	70.2	10.7
6	12:20 PM	12:35 PM	411.9	97.7	35.4	7.1	116.5	102.2	88.2	7.8	81.1	9.0
7	12:44 PM	12:52 PM	356.9	129.2	36.7	9.0	634.1	143.4	97.2	10.2	77.6	11.7
8	01:07 PM	01:22 PM	380.2	115.3	33.6	8.2	249.3	124.9	89.6	9.1	86.7	10.5
9	01:31 PM	01:46 PM	483.7	164.6	29.4	11.9	50.8	183.0	-	-	-	-

Radionuclide concentrations on material from main gully wall

1	10:00 AM	10:00 AM	-21.4	11.8	1.0	0.8	18.7	14.0	121.4	1.6	80.5	1.4
2	10:15 AM	10:30 AM	1.6	12.5	0.3	0.9	-4.8	15.3	101.2	1.5	64.6	1.4
3	10:30 AM	11:00 AM	-4.2	13.4	0.8	0.9	21.3	16.7	106.8	1.6	67.2	1.5
4	11:00 AM	11:30 AM	5.3	15.3	0.4	1.0	-15.9	19.0	109.6	1.8	69.8	1.8
5	11:30 AM	12:00 PM	-1.9	14.0	0.3	1.0	-22.0	23.0	108.4	1.7	69.7	1.6
6	12:00 PM	12:30 PM	-25.9	13.7	1.3	0.9	31.9	17.6	101.1	1.6	65.4	1.5
7	12:30 PM	01:00 PM	-15.5	13.3	-0.2	0.9	-4.3	17.0	101.9	1.6	66.2	1.5
8	01:00 PM	01:30 PM	-11.8	15.4	-0.3	1.0	5.8	19.8	113.9	1.8	72.1	1.7

Radionuclide concentrations on material from combined overland sheet/rill flow and main gully wall

1	03:00 PM	03:30 PM	-0.6	19.7	3.1	1.3	-38.3	40.0	104.4	2.1	65.0	2.1
2	03:30 PM	04:00 PM	-14.7	13.9	1.5	0.9	-6.3	13.7	99.4	1.6	60.4	1.5
3	04:00 PM	04:30 PM	20.0	14.5	-0.5	1.0	2.8	14.5	102.4	1.7	64.7	1.6
4	04:30 PM	05:00 PM	16.1	14.6	0.4	1.0	16.0	15.2	106.0	1.7	63.8	1.6
5	05:00 PM	05:30 PM	-6.4	14.7	-0.4	1.0	-0.1	15.9	104.7	1.7	63.7	1.6

Table B4

Black Mountain, radionuclide concentrations on suspended sediments

	Pb-210 (ex) (Bq/kg)	se	Cs-137 (Bq/kg)	se	Be-7 (Bq/kg)	se
Treatment 1, overland flow	238.8	11.2	17.8	1.2	282.4	36.0
Treatment 2, shallow rills	209.2	9.2	19.7	1.7	169.0	38.0
Treatment 3, deep rills	118.1	5.2	10.3	0.7	87.9	6.8
Treatment 4, small gullies	54.6	5.6	6.7	0.6	37.0	5.6

Radionuclide concetrations with depth from soil pits

		Pb-210 (ex) (Bq/kg)	se	Cs-137 (Bq/kg)	se	Be-7 (Bq/kg)	se
Excavated soil Pit number # 1							
0-2mm	1	107.3	6.1	7.5	0.9	77.8	24.3
2-4mm	3	78.8	5.3	7.7	0.4	30.0	12.5
4-8mm	6	53.6	2.2	10.0	0.2	4.5	4.5
8-20mm	14	41.1	3.1	10.2	0.3	-2.0	4.4
20-30mm	25	32.2	3.0	9.7	0.3	-	-
30-40mm	35	25.3	1.6	8.7	0.2	-	-
40-50mm	45	17.6	3.0	6.6	0.3	-	-
50-70mm	60	15.0	3.0	4.4	0.2	-	-
70-100mm	85	6.3	3.0	1.6	0.2	-	-
100-150mm	125	0.5	3.0	0.5	0.2	-	-
150-250mm	200	-0.5	3.0	0.3	0.2	-	-
Excavated soil Pit number # 2							
0-2mm	1	68.7	7.6	9.1	0.6	44.3	16.7
2-4mm	3	42.2	2.1	9.3	0.2	12.9	4.5
4-8mm	6	56.8	5.4	8.7	0.4	5.0	12.2
8-20mm	14	32.5	2.9	9.5	0.3	-1.0	2.3
20-30mm	25	23.1	3.1	9.0	0.3	-	-
30-40mm	35	17.9	3.0	8.3	0.2	-	-
40-50mm	45	14.0	2.5	6.7	0.2	-	-
50-70mm	60	8.3	3.0	4.4	0.2	-	-
70-100mm	85	-1.8	3.0	1.6	0.2	-	-
100-150mm	125	-2.2	2.7	0.7	0.2	-	-
150-250mm	200	2.3	4.0	0.5	0.2	-	-
Excavated soil Pit number # 3							
0-2mm	1	76.7	3.0	6.9	0.2	36.3	6.6
2-4mm	3	58.1	5.4	8.4	0.4	4.1	12.4
4-8mm	6	25.8	3.0	8.5	0.3	5.6	8.0
8-20mm	14	12.3	2.7	8.5	0.2	0.1	3.2
20-30mm	25	8.8	2.4	9.0	0.2	-	-
30-40mm	35	8.8	2.4	6.6	0.2	-	-
40-50mm	45	11.6	1.7	5.3	0.1	-	-
50-70mm	60	1.9	2.8	3.9	0.2	-	-
70-100mm	85	-1.2	1.2	2.5	0.1	-	-
100-150mm	125	-0.7	3.3	0.8	0.2	-	-
150-250mm	200	0.5	1.0	0.5	0.2	-	-

Table B5

Mid-Murrumbidgee suspended sediment radionuclide values

Location	Cs-137	se	Pb-210 (ex)	se	Location	Cs-137	se	Pb-210 (ex)	se
JUGIONG CK	10.9	7.5	-21.9	65.4	KYEAMBA CK	1.0	0.3	-6.7	5.4
JUGIONG CK	2.8	1.0	38.0	9.0	TARCUTTA CK	1.3	1.2	8.8	8.8
TUMUT @ BRUNGLE	2.0	7.0	-3.9	61.7	ADELONG CK	1.7	0.5	23.9	4.7
TUMUT @ BRUNGLE	4.9	3.0	93.7	27.7	HILLAS CK	2.7	1.2	28.2	8.8
ADELONG	2.5	0.9	26.4	8.0	ADELONG CK	2.0	0.9	18.8	5.9
HILLAS	3.6	1.7	94.6	13.8	HILLAS CK	0.7	0.7	5.2	6.2
KYEAMBA	0.7	1.4	11.4	9.1	BRUNGLE @ TUMUT	2.9	1.2	20.5	12.5
BILLABUNG	2.2	1.9	54.2	17.5	BILLABUNG CK	3.0	0.5	15.0	5.9
TUMUT @ BRUNGLE	4.6	1.4	24.2	12.6	ADELONG CK	1.5	1.3	5.9	12.0
TUMUT @ BRUNGLE	4.3	1.1	36.4	10.5	HILLAS CK	0.9	1.1	9.6	10.0
JUGIONG CK	3.9	0.9	41.9	8.8	TARCUTTA CK	1.2	0.3	-73.0	86.4
ADELONG CK	2.3	1.7	34.8	16.4	KYEAMBA CK	2.2	16.0	-1.8	4.8
TUMUT @ BRUNGLE	6.5	2.6	80.1	24.2	BILLABUNG CK	2.5	0.5	16.0	4.0
HILLAS CK	2.7	1.2	3.4	10.3	JUGIONG CK	3.6	3.1	-20.0	47.0
HILLAS CK	1.7	0.3	65.3	4.5	BIDGEE @ JUGIONG	5.9	1.4	18.6	12.8
BILLABUNG CK	1.9	0.7	2.4	5.6	BIDGEE@JUGIONG	4.5	1.3	48.8	12.4
TARCUTTA CK	0.9	0.2	10.2	2.9	BIDGEE @ JUGIONG	6.6	0.9	61.1	7.9
KYEAMBA CK	2.0	0.3	20.1	4.0	BIDGEE @ JUGIONG	1.1	2.6	109.0	21.7
JUGIONG CK	2.1	1.0	26.6	7.0	COOLAC	2.7	1.4	34.7	9.2
TUMUT @ BRUNGLE	2.4	1.6	13.6	9.7	JUGIONG(WTW)	4.2	1.0	29.9	15.2
ADELONG CK	3.1	0.8	25.5	6.8	GUNDAGAI	9.0	8.4	-27.2	74.8
HILLAS CK	2.4	0.7	11.3	6.8	GUNDAGAI	3.7	1.6	80.8	14.8
TARCUTTA CK	1.6	0.5	-11.9	9.6	GUNDAGAI	4.7	1.1	53.8	10.1
TUMUT@BRUNGLE	5.6	2.0	33.3	14.8	GUNDAGAI	5.5	3.4	78.2	26.8
TARCUTTA CK	1.9	0.4	35.9	5.3	GUNDAGAI	2.2	0.9	39.0	6.1
BILLABUNG CK	3.3	0.5	20.3	4.6	GUNDAGAI	3.6	3.4	37.4	17.8
HILLAS CK	0.7	0.4	2.0	6.8	GUNDAGAI	3.4	1.6	15.6	7.6
KYEAMBA CK	-0.1	0.8	-2.8	8.5	MUNDARLO	2.6	9.8	50.2	85.2
ADELONG CK	1.6	0.5	17.8	8.3	MUNDARLO	3.3	1.4	43.7	9.3
KYEAMBA CK	1.0	0.3	-4.2	4.8	MUNDARLO	2.1	3.3	50.3	25.9
TARCUTTA CK	1.3	0.5	11.0	5.0	MUNDARLO	4.5	1.2	61.2	11.4
JUGIONG CK	1.7	0.5	23.5	8.3	MUNDARLO	4.3	1.4	2.3	2.4
BRUNGLE @ TUMUT	2.6	0.7	37.7	6.9	WAGGA (EUNONY)	1.8	6.5	-40.4	56.9
EUNONY	2.5	1.0	-19.0	25.0	WAGGA WAGGA	3.7	1.5	24.6	11.9
EUNONY	3.8	1.0	-35.6	48.0	WAGGA WAGGA	1.2	0.9	23.5	6.1
EUNONY	3.5	1.4	25.8	9.5	WAGGA WAGGA	2.7	0.9	41.8	9.1

Table B6

Radionuclide data from mid-Murrumbidgee bulked cores

Undisturbed lands

	Core depth	Pb-210 (ex) (Bq/kg)	se	Cs-137 (Bq/kg)	se
ME95101/A	surface	72.1	5.5	10.0	0.6
ME95101/B	0-2	52.6	3.0	10.9	0.3
ME95101/C	2-5	14.6	1.9	5.6	0.2
ME95101/D	5-25	-4.7	2.8	1.0	0.2
ME95102/1	surface	65.3	3.7	8.5	0.6
ME95102/B	0-2	77.7	2.8	9.4	0.3
ME95102/C	2-5	30.6	3.2	8.9	0.4
ME95102/D	5-25	3.5	3.3	1.7	0.4
ME95103/A	surface	82.8	5.9	8.7	0.9
ME95103/B	0-2	91.7	4.1	12.1	0.3
ME95103/C	2-5	23.2	3.2	6.8	0.4
ME95103/D	5-25	2.9	2.5	1.6	0.2
ME95105/1	surface	114.9	6.3	12.5	0.4
ME95105/B	0-2	83.9	3.1	13.4	0.3
ME95105/C	2-5	26.1	3.4	9.2	0.3
ME95105/D	5-25	-1.5	1.8	1.5	0.1
ME95106/A	surface	99.2	4.8	9.1	0.4
ME95106/B	0-2	69.9	4.2	7.4	0.5
ME95106/C	2-5	29.4	3.8	6.8	0.4
ME95106/D	5-25	0.7	2.2	1.1	0.2
ME95107/A	surface	48.8	4.6	5.4	0.7
ME95107/B	0-2	52.5	2.9	6.8	0.2
ME95107/C	2-5	25.4	2.4	7.3	0.2
ME95107/D	5-25	2.8	2.5	1.5	0.2
ME95108/A	surface	58.5	5.0	6.3	0.5
ME95108/B	0-2	60.9	4.6	9.7	0.5
ME95108/C	2-5	24.3	3.1	8.0	0.3
ME95108/D	5-25	-1.7	2.6	0.9	0.2
ME95109/A	surface	30.6	2.8	4.9	0.3
ME95109/B	0-2	43.7	3.1	6.6	0.2
ME95109/C	2-5	19.3	3.1	6.3	0.3
ME95109/D	5-25	2.8	3.0	1.8	0.2
ME95110/1	surface	60.9	3.4	7.5	0.5
ME95110/B	0-2	53.8	2.9	6.9	0.2
ME95110/C	2-5	21.9	1.7	6.3	0.1
ME95110/D	5-25	-2.9	3.0	1.8	0.2
ME95111/1	surface	62.0	7.3	5.2	0.7
ME95111/B	0-2	34.4	4.7	6.5	0.5
ME95111/C	2-5	18.5	4.4	7.2	0.3
ME95111/D	5-25	2.7	3.2	1.5	0.2
ME95112/A	surface	50.5	6.1	4.7	0.5
ME95112/B	0-2	39.0	3.2	4.2	0.3
ME95112/C	2-5	18.9	2.9	4.7	0.3
ME95112/D	5-25	-1.8	3.9	1.4	0.2

Cultivated lands

	Core depth	Pb-210 (ex) (Bq/kg)	se	Cs-137 (Bq/kg)	se
ME96001/A	surface	20.5	4.0	3.3	0.4
ME96001/B	0-2	18.2	2.7	3.9	0.2
ME96001/C	2-5	10.0	1.8	3.6	0.1
ME96001/D	5-25	1.4	4.1	1.3	0.4
ME96002/A	surface	29.5	2.5	4.0	0.2
ME96002/B	0-2	27.6	2.6	4.3	0.2
ME96002/C	2-5	9.2	3.6	3.4	0.4
ME96002/D	5-25	1.5	2.4	1.4	0.2
ME95104/A	surface	31.5	3.8	2.6	0.3
ME95104/B	0-2	28.1	1.8	2.8	0.2
ME95104/C	2-5	10.8	3.5	3.1	0.4
ME95104/D	5-25	0.5	1.4	2.6	0.1

Table B7

Concentrations of Cs-137 on sediments in Australian rivers (Raw data for Table 7.6.1, note all values decay corrected to 1995)

Code	Cs-137 (Bq/kg)	se	Code	Cs-137 (Bq/kg)	se
Molonglo River, NSW					
MN90001/1, MAC1	1.2	0.5	MN89001/5a, 3	4.4	0.5
MN90001/2, MAC2	1.2	0.4	MN89001/7, 4	2.8	0.4
MN90001/3, MAC3	0.7	0.2	MN89001/9, 6	6.1	1.5
MN90001/4, MAC4	1.6	0.5	MN89001/11a, 6	2.3	0.3
MN90001/5, MAC5	1.5	1.8	MN89001/11, 6B	2.9	0.3
MN91001/6, MDC1	0.3	0.2	MN89002/1, 8-10	4.4	1.9
MN91001/7, MDC2	0.3	0.3	MN89002/1, 8-10	4.0	1.5
MN91001/8, MDC3	0.6	0.5	MN89002/3, 12-13	5.4	0.4
MN91001/9, MDC4	0.3	0.2	MN89002/5, 14-15	2.4	0.4
MN91001/10, MDC	0.0	1.1	MN89012/1, 22/23	6.1	0.3
MN90001/6, MUB1	0.3	0.8	MN89009/3, 46	1.2	0.3
MN90001/7, MUB2	-0.3	0.8	MN89009/1, 45	0.9	0.3
MN90001/8, MUB3	0.3	0.4	MN89009/6, 47	5.2	0.4
MN90001/9, MUB4	-0.1	0.7	MN89022/1, 44	1.5	0.2
MN90001/10, MUB	-0.0	0.4	MN89008/3, 37	0.2	0.3
MN90001/1, MDB1	0.7	0.3	MN89008/5, 38	0.4	0.3
MN90001/2, MDB2	1.6	0.8	MN89008/1, 36	1.5	0.2
MN90001/3, MDB3	-0.5	0.7	MN89004/3, 40	2.6	0.3
MN90001/4, MDB4	1.0	0.5	MN89004/1, 39	0.8	0.2
MN90001/5, MDB5	0.6	0.4	MN89004/3, 41	3.0	0.3
MN90002/1, MUR1	0.1	0.8	MN89004/5, 41	2.6	0.2
MN90002/2, MUR2	0.3	0.3	MN89021/3, 19	0.7	0.6
MN90002/3, MUR3	1.0	0.9	MN89021/1, 17/18	1.3	0.5
MN90002/4, MUR4	0.6	0.2	MN89021/5, 20/21	3.2	0.4
MN90002/5, MUR5	0.1	0.2	MN89020/1, 24/25	2.0	0.3
MN90002/11, MDR	1.4	0.9	MN89020/3, 26/27	-0.1	0.6
MN90002/12, MDR	0.2	0.2	MN89020/5, 28/29	0.7	0.5
MN90002/13, MDR	0.9	0.7	MN89019/3, 34	1.9	0.3
MN90002/14, MDR	0.7	0.5	MN89019/1, 33	0.5	0.2
MN90002/15, MDR	0.3	0.2	MN89019/5, 35	0.8	0.3
MN90007/1, MUP1	1.9	0.6	MN89018/1, 30	2.4	0.2
MN90007/2, MUP2	3.3	0.4	MN89018/5, 32	0.0	0.1
MN90007/3, MUP3	0.2	0.1	MN89018/3, 31	0.5	0.2
MN90007/4, MUP4	3.1	0.4	MN89015/1, 42	1.2	0.3
MN90007/5, MUP5	4.9	0.6	MN89015/3, 43	1.6	0.2
MN90007/6, MBP1	4.4	0.8			
MN90007/7, MBP2	3.2	0.4			
MN90007/8, MBP3	1.9	0.2			
MN90007/9, MBP4	4.5	0.5			
MN90007/10, MBP5	2.1	0.4			

Table B7 continued

Concentrations of Cs-137 on sediments in Australian rivers
(Raw data for Table 7.6.1, note all values decay corrected to 1995)

Code	Cs-137 (Bq/kg)	se
Ord river, W.A.		
core 3/2	5.0	0.7
core 37/3	6.9	0.5
core 18/4	0.8	0.3
core 24/5	1.3	1.1
OR91013/1, OAP1	0.4	0.2
OR91013/2, OAP2	2.0	0.1
OR91013/3, OAP3	1.3	0.7
OR91013/4, OAP4	2.8	0.6
OR91013/6, OAP6	2.1	0.2
OR91013/7, OAP7	1.7	0.5
OR91013/8, OAP8	1.7	0.5
OR91010/1, PUO1	2.1	1.3
OR91010/6, PUO6	1.2	0.2
OR91010/7, PUO7	-0.1	2.1
OR91010/8, PUO8	-0.5	2.0
OR91012/1, OBP1	3.1	0.7
OR91012/2, OBP2	3.4	0.2
OR91012/4, OBP4	3.2	0.4
OR91012/7, OBP7	1.7	5.1
OR91028/1, OANB1	13.3	12.0
OR91028/2, OANB2	0.3	4.2
OR91028/3, OANB3	102.2	123.2
OR91028/4, OANB4	4.1	3.0
OR91028/5, OANB5	-9.5	5.7
OR91028/6, OANB6	0.9	1.7
OR91028/7, OANB7	1.7	1.4
OR91028/8, OANB8	1.5	3.5
OR91028/9, OANB9	2.2	1.0
OR91029/1, LN1	1.2	0.3
OR91029/2, LN2	2.0	0.5
OR91029/3, LN3	3.7	3.6
OR91029/4, LN4	4.1	1.5
OR91029/5, LN5	1.2	1.8
OR91029/6, LN6	2.1	0.6
OR91029/7, LN7	2.0	0.4
OR91030/1, NO1	2.3	0.6
OR91030/2, NO2	2.4	0.4
OR91030/3, NO3	2.5	0.3
OR91030/4, NO4	-1.8	5.3
OR91030/5, NO5	2.6	1.2
OR91030/8, NO8	2.8	0.5
OR91032/1, OAS1	2.5	0.8
OR91032/2, OAS2	0.2	4.2
OR91032/3, OAS3	0.4	1.9
OR91032/6, OAS4	-0.0	2.5
OR91033/1, SAO1	0.2	0.2
OR91033/2, SAO2	0.6	0.3
OR91033/3, SAO3	-2.9	7.6
OR91033/4, SAO4	0.4	1.2
OR91034/1, OBS1	2.1	0.1
OR91034/2, OBS2	4.2	1.5
OR91034/4, OBS4	2.0	0.8
OR91034/5, OBS5	9.5	4.8
Swan River, W.A.		
SR93001/1	2.7	1.5
SR93001/2	3.5	0.6
Murray River, VIC.		
Barmah forest/1	2.1	1.0
Barmah forest/2	2.3	1.0
Barmah forest/3	3.4	0.5
Lock 2	3.2	1.0
Lock 3	0.5	0.5

Table B7 continued

Concentrations of Cs-137 on sediments in Australian rivers
(Raw data for Table 7.6.1, note all values decay corrected to 1995)

Code	Cs-137 (Bq/kg)	se	Code	Cs-137 (Bq/kg)	se
Darling River, NSW					
DR93001	1.1	0.6	DR94001	2.3	0.2
DR93002	1.2	0.7	DR94002	2.7	0.2
DR93003	1.7	0.3	DR94003	2.3	0.1
DR93004	0.8	0.7	DR94004	2.0	0.5
DR93005	1.4	0.5	DR94005	2.4	0.3
DR93006	0.8	0.6	DR94006	0.8	0.2
DR93007	1.1	0.5	DR94007	1.7	0.2
DR93008	1.5	0.9	DR94008	1.9	0.2
DR93009	1.2	0.6	DR94009	1.9	0.2
DR93011	0.4	0.7	DR94010	1.7	0.3
DR93013	3.3	0.7	DR94011	1.8	0.1
DR93014	4.6	0.8	DR94012	1.7	0.2
DR93020	0.1	1.4	DR94013	1.6	0.2
DR93021	2.2	0.9	DR94014	2.1	0.3
DR93022	1.1	0.3	DR94015	1.9	0.3
DR93023	1.7	0.8	DR94016	2.0	0.1
DR93024	2.1	0.3	DR94017	1.9	0.2
DR93025	1.6	0.4	DR94018	1.3	0.3
DR93026	1.5	0.8	DR94019	0.2	0.2
DR93027	0.9	1.3	DR94020	1.7	0.2
DR93028	1.6	0.8	DR94021	1.9	0.6
DR93029	2.9	1.1	DR94022	2.7	0.4
DR93030	2.2	1.4	DR94023	1.2	0.7
DR93031	1.2	1.2	DR94024	1.3	0.4
DR93032	2.5	1.7	DR94025	1.2	0.3
DR93033	1.1	1.0	DR94026	2.0	0.8
DR93034	1.0	0.4	DR94027	1.7	0.2
DR93035	2.6	0.6	DR94028	1.7	0.2
DR93036	2.5	0.3	DR94029	1.5	0.2
DR93037	1.6	0.4	DR94030	0.5	0.2
DR93038	1.3	0.6	DR94031	1.0	0.2
DR93039	2.0	0.9	DR94032	1.9	0.1
DR93040	2.0	0.6	DR94033	2.0	0.3
DR93041	3.0	0.3	DR94034	0.4	0.2
DR93042	2.0	0.4	DR94035	1.0	0.2
DR93043	2.2	0.5	DR94036	1.8	0.1
DR93044	2.3	0.3	DR94037	1.5	0.2
DR93045	2.6	0.8	DR94038	1.6	0.1
DE91001/2	1.8	0.5	DR94039	1.7	0.2
DE91006/2	2.6	0.8	DR94040	1.8	0.2

Table B7 continued

Concentrations of Cs-137 on sediments in Australian rivers (Raw data for Table 7.6.1, note all values decay corrected to 1995)

Code	Cs-137 (Bq/kg)	se
Darling River, continued		
DR94041	0.8	0.2
DR94042	1.7	0.2
DR94043	1.6	0.3
DR94044	2.4	0.2
DR94045	1.7	0.3
DR94046	1.6	0.2
DR94047	1.6	0.1
DR94048	1.4	0.1
DR94049	1.6	0.2
DR94050	1.4	0.3
DR94051	0.8	0.2
DR94052	1.1	0.1
DR94053	0.8	0.3
DR94054	0.7	0.2
DR94055	1.0	0.3
DR94056	0.6	0.1
DR94057	0.8	0.2
DR94058	0.7	0.8
DR94059	0.4	0.7
DR94060	0.8	0.2
DR04080/1	0.7	0.2
DR94080/2	0.8	0.2
DR94080/3	0.6	0.1
DR94080/4	0.9	0.2
DR94080/5	0.7	0.2
DR04084/1	7.3	0.3
DR04084/2	5.5	0.4
DR04084/3	4.0	0.2
DR04084/4	2.0	0.4
DR94084/5	6.5	0.4
DR04085/1	2.6	0.4
DR04085/2	2.5	0.4
DR04085/3	2.9	0.3
DR04085/4	3.0	0.4
DR04085/5	2.3	0.4
DR95203	0.4	0.2
DR95207	0.6	0.2