

Determining the effects of wildfire on sediment sources using ^{137}Cs and unsupported ^{210}Pb : the role of landscape disturbances and driving forces

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Abstract

Purpose Wildfires represent one of the major natural disturbances within forested landscapes and have potential implications for the quality and function of downstream aquatic ecosystems. This study aimed to determine if a wildfire in a mountainous, forested watershed in British Columbia, Canada, caused a change in the dominant sediment source in the immediate 1–2 years following the wildfire, and if the sediment sources changed over the medium term (3–7 years) as the landscape recovered.

Materials and methods Source materials (surface soil, subsurface soil and channel bank material) and fluvial (suspended and channel bed) sediment samples were collected over the period 2004 to 2010 from a watershed burnt by a wildfire in 2003, and from an adjacent watershed that was not impacted

by the fire. Samples were analysed for the fallout radionuclides (FRNs) caesium-137 (^{137}Cs) and unsupported lead-210 ($^{210}\text{Pb}_{\text{un}}$). An unmixing model was used to calculate the relative source contributions of the fluvial sediment samples.

Results and discussion ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ were concentrated in the upper layers of surface soils in both watersheds and were statistically different to concentrations in subsurface and channel bank material. In the burnt watershed, FRN concentrations were greatest in the ash layer. Sediment sources as determined by the unmixing model were 100% subsurface/channel bank material in the unburnt watershed, while in the burnt watershed $8.5 \pm 2.5\%$ was derived from surface soils. In both watersheds, there were no major changes in the relative contributions from surface soil and from subsurface/channel bank material over the period 2004 to 2010. Thus, while the wildfire did cause a change in sediment sources, it was fairly subtle and did not conform to the effects following wildfire described for other studies in contrasting environments, which typically document a major increase in hillslope contributions relative to channel bank sources.

Conclusions There was a limited response in terms of fine-grained sediment sources (and also sediment fluxes) in the burnt watershed. The reason for this muted response to a severe wildfire is likely to be the lack of precipitation, especially winter precipitation and the associated snowmelt, in the first year following the wildfire. Thus while the landscape was primed for erosion and sediment transport, the lack of a driving force meant that there was a limited immediate post-fire sediment response.

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1 Introduction

There is growing interest in how wildfires influence watershed processes (Shakesby and Doerr 2006). The increased delivery of sediment and associated chemicals (including nutrients) to river channels, with implications for aquatic habitats and water resources, are a major concern (Smith et al. 2011a). To date, most studies have been concerned with monitoring sediment and chemical fluxes, and have focused on either (1) soil erosion and hillslope transport, or (2) fluxes within stream channels (e.g. Cerda and Lasanta 2005; Silins et al. 2009; Ryan et al. 2011). Fewer studies have focused on documenting changes in the sources of the material following wildfires, thereby linking hillslope processes to channel impacts. Furthermore, most studies have tended to be of relatively short duration (typically 3–5 years) and may, therefore, not provide a full temporal record of both the initial responses in sediment sources and the subsequent recovery. Information on slope-channel impacts over longer timeframes is needed in order to understand the medium- to longer-term responses of landscapes to wildfires, and the subsequent impacts on aquatic systems and water resources.

Environmental fallout radionuclides (FRNs), such as caesium-137 (^{137}Cs) and unsupported lead-210 ($^{210}\text{Pb}_{\text{un}}$), have been used extensively to determine soil and sediment redistribution dynamics within contrasting landscapes (e.g. Ritchie and McHenry 1990; Walling and Quine 1991; Wallbrink and Murray 1993; Mabit et al. 2008) and determine changes in sediment sources, often as part of a composite fingerprinting approach (e.g. Collins et al. 1997; Walling et al. 1999; Owens et al. 2000). Their use within wildfire studies is relatively limited, and work to date has been almost exclusively based within Australia (e.g. Wallbrink et al. 2005; Blake et al. 2009; Wilkinson et al. 2009; Smith et al. 2011b). Those studies have shown that these FRNs offer potential to determine changes in sediment sources following wildfire events and furthermore, when coupled with watershed monitoring, to provide a comprehensive evaluation of the response of landscapes to wildfires (Wilkinson et al. 2009; Smith et al. 2011b).

The fundamental assumptions that underpin the use of ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ in this context are that: (1) they bind strongly to, and label, surface mineral and organic material; (2) their vertical distribution is limited, meaning subsurface soil and channel bank material have much lower concentrations; and importantly (3) they exhibit conservative behaviour during subsequent downstream redistribution (Walling and Quine 1991; Mabit et al. 2008). In any tracer study, data must be interpreted in the context of these assumptions, which need to be evaluated critically for every study environment (cf. Parsons and Foster 2011).

Given the anticipated increase in the number and severity of wildfires associated with future climate changes (Flannigan et al. 2009), there is a requirement to investigate landscape

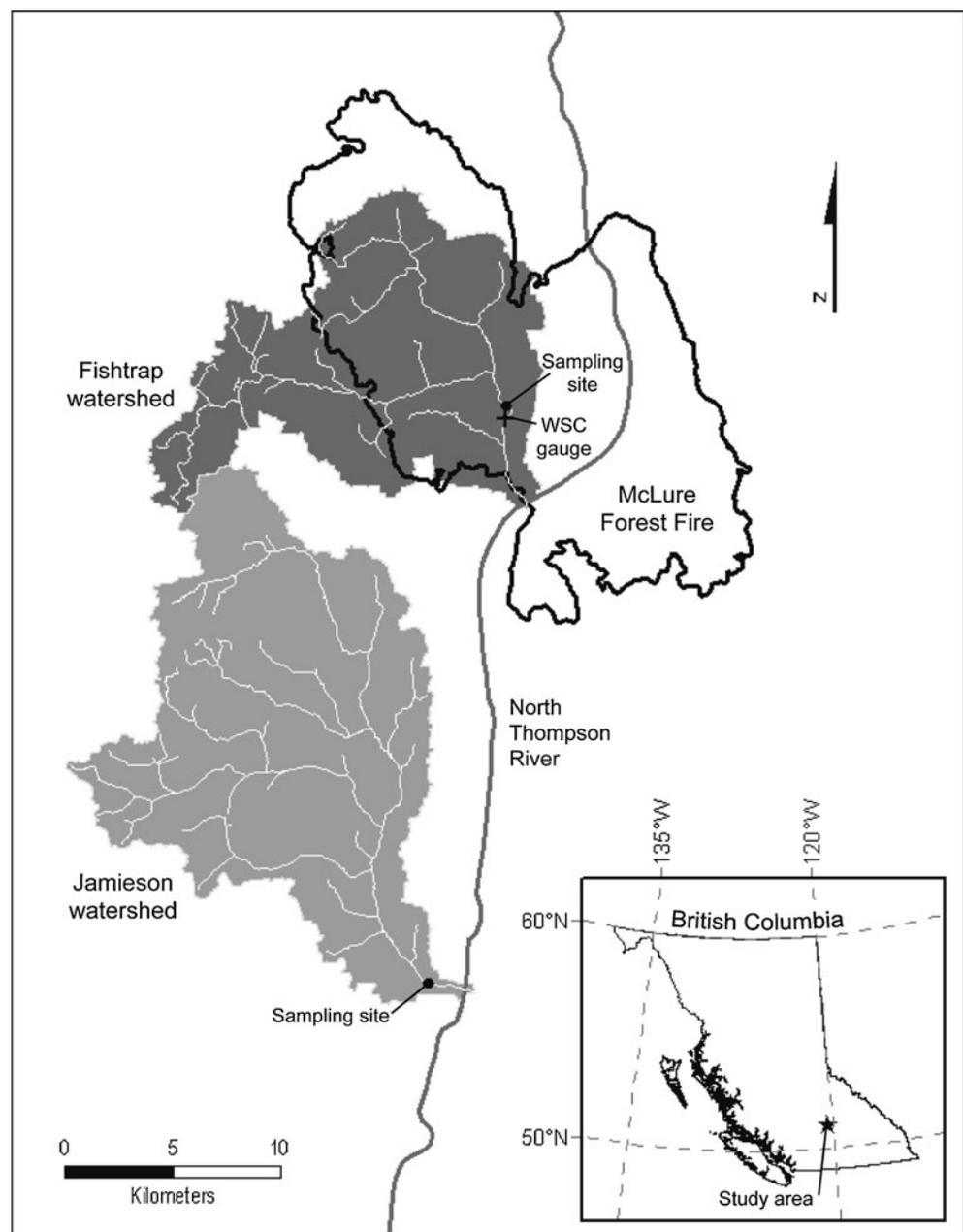
responses to wildfire in contrasting environments, and this has been identified as an important research need (Sharik et al. 2010). This study describes the use of ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ to document sediment sources over a 7-year period for a forested watershed in British Columbia (BC), Canada, in response to a wildfire event in 2003. To the best of our knowledge, this is the first study of its kind in North America, although FRNs have been used qualitatively in New Mexico, USA to investigate the effect of wildfires on ash redistribution and FRN depth profiles in soils (e.g. Johansen et al. 2003; Reneau et al. 2007). Recent Australian studies (e.g. Wilkinson et al. 2009; Smith et al. 2011b) have reported: (1) a marked increase (e.g. from ca. 20% pre-fire to ca. 86% immediately post-fire; Wilkinson et al. 2009) in the relative contribution from hillslope surface soil materials immediately after wildfire, followed by (2) a decrease in the relative contribution of surface materials after 2–3 years. Against this background, this study aims to determine if post-fire sediment sources in the forests of BC follow a similar pattern.

2 Study site

In August 2003, the McLure wildfire, in the central interior of BC, near the city of Kamloops, burnt an area of ca. 260 km². The fire was classified as a Rank 6 fire, which is the most extreme and hazardous category of fire behaviour (Eaton et al. 2010a). The Fishtrap Creek watershed (area is 158 km²; 135 km² to the Water Survey of Canada (WSC) gauging station at 51°07'24"N, 120°12'34"W, Fig. 1), is steeply sloped and mountainous (elevation range 370 to 1,620 m above mean sea level), and is forested with pines, firs, spruce and cedar trees. It is drained by several gravel-bed streams that form important habitats for salmonid species. The climate is semi-arid; mean annual precipitation (1971–2000 period) and runoff are 487 and 180 mm, respectively, and mean annual temperatures at 370 and 1,620 m elevation are 7.5 and 2.5°C, respectively (Eaton et al. 2010b). The watershed was severely burnt in the lower reaches and moderately burnt in the headwaters; total burn area in the watershed was ca. 98 km². The Jamieson Creek watershed (area is 215 km²; 50°53'N, 120°17'W, see Fig. 1) is located to the south of Fishtrap Creek watershed and was not affected by the McLure wildfire. As the Jamieson watershed has similar characteristics of vegetation cover, topography, climate and geology to the Fishtrap watershed, it serves as a good reference for comparing the effects of the 2003 fire in the Fishtrap watershed. Both creeks drain into the North Thompson River, itself a major tributary of the Fraser River which drains into the Pacific Ocean at Vancouver.

The underlying geology in both study watersheds is dominated by Palaeozoic (Pennsylvanian and Permian) volcanic and metamorphic rocks, with outcrops of Mesozoic (Triassic and Jurassic) and Cenozoic (Miocene and/or Pliocene) rocks

Fig. 1 The Fishtrap and Jamieson watersheds and the areal extent of the McLure wildfire. Also shown is the location of the sediment sampling sites; that for Fishtrap, is located just above the Water Survey of Canada gauging station. The *inset* map of British Columbia shows the location of the study area (modified from Eaton et al. 2010a)



in the headwaters of Fishtrap and Jamieson, respectively. Soils overly glacial till, glaciofluvial deposits and bedrock, and are generally well-drained (Gough 1988). The main soil associations are Alkali, Allie and Artison (soil classification is predominantly brunisolic gray luvisols and podzolic gray luvisols); valley bottoms also have Laurel and Helmcken soil associations (soil classification is predominantly orthic humo-ferric podzols), amongst others (Gough 1988).

The watersheds have a distinct snowmelt hydrological regime, with melting typically starting in late March and the main flood discharges (pre-wildfire) occurring from mid April to late May (Petticrew et al. 2006; Eaton et al. 2010a; WSC 2012), with annual discharge peaks of approximately 5 to

$8 \text{ m}^3 \text{ s}^{-1}$ (Leach and Moore 2010; WSC 2012). River discharge data are available for Fishtrap Creek at the WSC gauging station (WSC (2012), station ID 08LB024, see Fig. 1), although this was damaged by the 2003 wildfire and rebuilt. Precipitation data were obtained for the Environment Canada gauge at McLure (Environment Canada (2012), station ID 1165030; $51^{\circ}05'N$, $120^{\circ}22'W$; elevation 381 m).

3 Field and laboratory methods

Samples of source materials and river sediments were collected from both watersheds over the period 2004 to 2010. A first set

of source materials ($n=27$) was collected from Fishtrap and Jamieson watersheds soon after the 2003 wildfire in spring/summer 2004 (for details see Owens et al. 2006). A second set of source materials ($n=96$) was collected in summer 2007. In both watersheds, targeted samples of surface (0–2 cm) and subsurface (>5 cm depth) soil material were collected from active slopes (i.e. those likely to erode and supply sediment to the channels). In addition, samples were collected from actively eroding channel banks; typically, these ranged in height from ca. 0.3 to 1.0 m and had vertical faces, with a post-fire riparian vegetative cover of standing dead defoliated trees, forbaceous shrubs and fireweed (Leach and Moore 2010). Each source sample represented a composite of five to ten subsamples collected within a ca. 10 m radius, to encompass local spatial variability in soil properties. In the case of the surface soil samples collected from Fishtrap watershed, material was collected from the upper soil layers likely to have been modified by the fire (i.e. exposed, charred soil material in areas devoid of vegetation). In places, the upper layer of material consisted of ash (>1 cm thick in places), and in these situations the surface ash layer was carefully removed for analysis, in addition to the scorched mineral soil beneath, collected as above. At one site in Fishtrap watershed, samples were collected at various depths, including the ash layer, to determine the depth distribution of the FRNs.

Composite suspended sediment samples were collected from the outlet channel of each watershed (see Fig. 1) using two types of time-integrated, composite sediment samplers (for details of design, see Phillips et al. 2000 and Peticrew et al. 2006). In brief, both samplers work on a similar principle whereby water and entrained sediment flowing through a narrow orifice into the sampler is slowed in the relatively wide sample collection area. Tests (e.g. Phillips et al. 2000) have shown that most of the sediment (including a large fraction of the material <63 μm) is retained within the body of the sampler by settlement due to the reduced flow velocities in the sample retention area. Samplers were fixed to the bed of the channel using reinforced upright supports. For the Peticrew sampler, two pairs (i.e. $n=4$) of samplers were placed at each sampling site, while for the Phillips sampler only one pair (i.e. $n=2$) of samplers was placed at each sampling site. In both watersheds, samplers were installed at the start of the snowmelt period (freshet) in March/April and emptied at the end of the freshet (typically early to mid summer), subsequently reset and emptied at the start of the snow period (typically in fall), thereby providing composite samples of the suspended sediment transported during the period March/April to June/July (representing the period dominated by freshet and melt) and June/July to September/October (representing the period dominated by precipitation events). At certain times, samples of the fine-grained sediment stored on, and in, the upper layers of the channel bed were also collected using a resuspension approach (Owens et al. 1999; Peticrew et al. 2006). In brief, a cylinder, which

was open at both ends, was placed on the bed of the channel. A stainless steel trowel was then used to disturb the upper layers (top 5–10 cm) of the channel bed, thereby remobilizing the fine-grained sediment, which was subsequently sampled into 20-litre containers. The sediment was separated from the water by settling and centrifugation. Samples of suspended and channel bed sediment were collected in 2004, 2007, 2008, 2009 and 2010. In some years, only a limited number of sediment samples were collected as higher river flows and/or large woody debris (i.e. fallen trees) destroyed samplers or buried them under river gravels.

Source materials and fluvial sediment samples were air-dried at 30°C and disaggregated prior to screening through a 63- μm sieve. Radionuclide (^{137}Cs and $^{210}\text{Pb}_{\text{un}}$) activity concentrations (subsequently referred to as concentrations) of source materials and sediment samples were determined at the Consolidated Radioisotope Facility at Plymouth University, UK, using an EG&G Ortec (Oak Ridge, TN, USA) well (N-type) HPGe gamma spectrometer. The instrument was calibrated using soil spiked with certified, traceable mixed radioactive standards supplied by AEA Technology Plc (AEA, Harwell, UK). Calibration relationships were derived using EG&G GammaVision software and verified with inter-laboratory comparison tests with materials supplied by the International Atomic Energy Agency (IAEA, Vienna, Austria). Total ^{210}Pb was measured by its gamma emissions at 46.5 keV and its supported component calculated by subtractions of radium-226 (^{226}Ra) activity, which in turn was measured by the gamma emissions of lead-214 (^{214}Pb) at 295 and 352 keV (with correction for bismuth-214 (^{214}Bi) emissions). Samples were sealed for 21 days to allow for equilibrium between ^{226}Ra and ^{214}Pb (Joshi 1987). ^{137}Cs was determined from gamma emissions at 662 keV. Samples were counted for between 24 and 48 h; uncertainties for individual samples were derived from counting statistics and represent ± 2 sigma (where sigma represents total uncertainty associated with the measurement process at the 95% confidence level).

For ^{137}Cs concentrations, values for all source materials and sediment samples were decay corrected to the time of the wildfire in 2003. In the case of $^{210}\text{Pb}_{\text{un}}$, because of the ongoing fallout of $^{210}\text{Pb}_{\text{un}}$, such an approach would result in overcorrection of the sediment samples collected in later years (Smith et al. 2011b). Consequently, samples were decay corrected to the time that samples were collected, so as to provide an approximation of when the samples ceased to be exposed to new $^{210}\text{Pb}_{\text{un}}$ fallout; Smith et al. (2011b) adopted a similar approach. Samples with concentrations below the minimum detectable activity (MDA) were given a value of 0.5 MDA in statistical analyses; this assumes a normal distribution for those samples with values below the MDA. Given the implications of the use of 0.5MDA values for $^{210}\text{Pb}_{\text{un}}$ concentrations (which are calculated as the

difference between two values: total ^{210}Pb and ^{226}Ra), we have assumed that $^{210}\text{Pb}_{\text{un}}$ concentrations are ≥ 0 .

The relative proportions of source contributions (i.e. from surface soil and subsurface/channel bank material) to the fluvial sediment samples collected from the two study watersheds were determined using a two-source unmixing model based on that described by Collins et al. (1997) and Walling et al. (1999), and used in other similar studies in Australia (e.g. Wilkinson et al. 2009; Smith et al. 2011b). The goodness-of-fit provided by the unmixing model was determined by comparing the actual FRN concentrations for the sediment samples with corresponding values predicted by the unmixing model (cf. Collins et al. 1997; Walling et al. 1999). This approach to estimate uncertainty differs from that used by Wilkinson et al. (2009) and Smith et al. (2011b) who used a Monte Carlo-based approach.

Previous analysis of the source samples collected in 2004 (Owens et al. 2006), showed that there were no significant differences in the particle size composition between the $<63\text{-}\mu\text{m}$ fractions of surface soil and subsurface/channel bank material. Consequently, source samples were not corrected for particle size composition for comparisons between source material types. Fluvial sediment samples were, however, statistically different (i.e. finer, $p < 0.05$, Student's t test) than the $<63\text{-}\mu\text{m}$ fractions of source samples and thus corrections (for details see He and Owens 1995; Walling et al. 1999; Owens et al. 2000) were made for differences in particle size composition using the specific surface area of representative source and sediment samples. Particle size composition was determined using a Malvern Mastersizer after standard pretreatment with hydrogen peroxide and sodium hexametaphosphate.

4 Results and discussion

4.1 Radionuclide concentrations in source materials

For both watersheds, FRN concentrations were statistically ($p < 0.05$) greater in soil surface material compared to subsurface and/or channel bank material (Table 1) for samples collected in 2004 and 2007. This is in agreement with the known depth distributions of ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ in undisturbed soils, where concentrations are typically greater in surface layers and decrease with depth (Wallbrink and Murray 1993; Owens et al. 1996). Values were elevated in ash material samples collected from Fishtrap watershed in spring/summer 2004, soon after the wildfire, with mean ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ concentrations of 263 ± 18 and $387 \pm 33 \text{ Bq kg}^{-1}$, respectively (see Table 1). Figure 2 shows the depth distribution of ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ for one site in Fishtrap watershed, demonstrating the enhanced surface concentrations of these FRNs, especially in the ash layer. These findings are in keeping with other studies (e.g. Paliouris et al.

1995; Johansen et al. 2003; Wilkinson et al. 2009) which have documented that wildfire may result in an increase in ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ concentrations at the soil surface, reflecting concentration of FRNs in ash by combustion of organic matter such as leaf litter and duff (the combined organic F (fermentation) and H (humus) layer, Van Wagner 1972). The effect tends to be greatest for $^{210}\text{Pb}_{\text{un}}$ due to its continuous delivery, and hence labelling of surface litter, from natural sources. In contrast, ^{137}Cs tends to have a greater association with deeper soil material due to its longer residence time in the system after the main period of fallout in the 1960s (cf. Cambay et al. 1989; Owens et al. 1996). Processes such as litter decay, bioturbation and enhanced mobility in organic-rich substrates have led to some downward transfer of ^{137}Cs (for review see Parsons and Foster 2011) and $^{210}\text{Pb}_{\text{un}}$ into the soil profile (see Fig. 2), although peak concentrations still occur in the upper 3 cm of the profile.

For both watersheds, there were no statistically significant differences between the ^{137}Cs or $^{210}\text{Pb}_{\text{un}}$ concentrations for subsurface material and channel bank material. Also, there were no statistically significant differences in the FRN concentrations of channel bank material between the two watersheds, or subsurface material between watersheds. Thus the wildfire appears to have only modified the ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ content of surface material.

There were large spatial variations in FRN values for the various source materials, especially in the Fishtrap watershed (see Table 1). This reflects: (1) the spatial variation in the intensity of the wildfire, and consequently the degree of combustion of organic litter and duff; (2) variations in ash deposition depth; and (3) variations in the characteristics of the slopes following the fire. For example, in some places trees collapsed and/or were combusted (Fig. 3) which left exposed subsurface material which may have subsequently received ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ inputs associated with deposition of FRN-rich airborne ash and some fluvial redistribution of surface materials. Thus one 'subsurface' sample had a ^{137}Cs value of $61 \pm 4 \text{ Bq kg}^{-1}$; such samples have been classified as 'exposed' (see Table 1) and have not been used in calculating statistics for subsurface samples.

The surface soil samples collected from the Fishtrap watershed in 2004 (mean concentrations of 147 ± 12 and $241 \pm 30 \text{ Bq kg}^{-1}$ for ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$, respectively) were not statistically different from the surface samples collected in this watershed in 2007 (122 ± 6 and $210 \pm 42 \text{ Bq kg}^{-1}$, respectively). The mean ^{137}Cs concentration for the 2004 samples is, however, greater than that for the 2007 samples suggesting that the wildfire increased ^{137}Cs concentrations at the soil surface, and especially FRN concentrations associated with the surface ash layer (see Fig. 2, Table 1).

Even in 2007, 4 years after the wildfire, values of ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ were statistically ($p < 0.05$) higher in surface soil materials in the Fishtrap watershed (means of 122 ± 6 and

Table 1 ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ activity concentrations of source materials collected in 2004 and 2007 from the two study watersheds

Year	Watershed	Source material	N	^{137}Cs (Bq kg ⁻¹)		$^{210}\text{Pb}_{\text{un}}$ (Bq kg ⁻¹)	
				Mean ^a	Range	Mean ^a	Range
2004	Fishtrap	Ash	3	263.2±18.0	184.6–326.9	387.2±32.8	296.0–451.0
		Surface ^b	7	146.8±11.8	52.0–326.9	241.3±29.5	96.3–451.0
		Subsurface	6	22.5±4.9	MDA–61.6	46.7±25.3	MDA–74.5
		Bank	1	MDA ^c	–	MDA	–
	Jamieson	Surface	5	96.8±10.1	36.3–168.2	140.0±43.4	52.9–229.9
		Subsurface	2	2.29±26.5	MDA–4.6	MDA	MDA
2007	Fishtrap	Surface	25	122.3±5.5	40.0–270.9	209.7±41.8	50.2–541.5
		Subsurface	21	13.2±2.3	MDA–29.9	26.4±27.6	MDA–62.1
		Exposed ^d	7	22.9±2.4	MDA–60.6	63.3±25.0	8.0–124.7
		Bank	6	29.4±9.2	MDA–28.0	68.4±28.8	20.1–134.5
	Jamieson	Surface	10	80.3±4.2	46.3–113.2	125.3±44.6	62.2–173.3
		Subsurface	12	24.1±3.4	MDA–61.4	55.6±23.0	MDA–167.9
		Bank	12	15.8±3.4	MDA–35.6	54.4±21.0	MDA–74.8

^a Mean value±the average uncertainty associated with the measurement of each sample (i.e. ±2 sigma; see text for details)

^b Incorporates surface soil and surface ash samples

^c MDA=value is below minimum detectable activity (and is assumed to be 0.5 MDA in the calculation of mean values)

^d ‘Exposed’ refers to areas of hillslope where the subsurface (i.e. material >10 cm in the soil profile) was exposed to the atmosphere probably due to vaporisation of trees/vegetation and/or collapse of trees due to the wildfire. Such material may have been labelled with FRNs after the wildfire. Such samples were not included in the statistics for the subsurface samples

210±42 Bq kg⁻¹ for ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$, respectively) than in equivalent materials in the Jamieson watershed (80±4 and 125±45 Bq kg⁻¹, respectively). This suggests that the wildfire has resulted in persistent, elevated FRN concentrations in surface soil materials. Radionuclide contents of other source materials (e.g. subsurface and channel bank), are not statistically different over time, being similar for the two watersheds (i.e. mean ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ values are <30 and <70 Bq kg⁻¹, respectively).

4.2 Radionuclide concentrations in fluvial suspended sediment

There were no statistically significant differences in the FRN concentrations for the sediment collected using either sampler (i.e. Petticrew or Phillips) or the channel bed sediment for each of the two creeks. This probably reflects the fact that the sediment was collected at essentially the same location at each site and the fact that all samples were passed through a 63-µm sieve prior to analysis. Consequently, the FRN data for each site for each time period were pooled to create an average value for all sampling types. The ^{137}Cs concentrations for the sediment samples from Fishtrap Creek were statistically different ($p<0.05$) from the sediment collected from Jamieson Creek, with values being approximately double in Fishtrap Creek (Table 2). Values

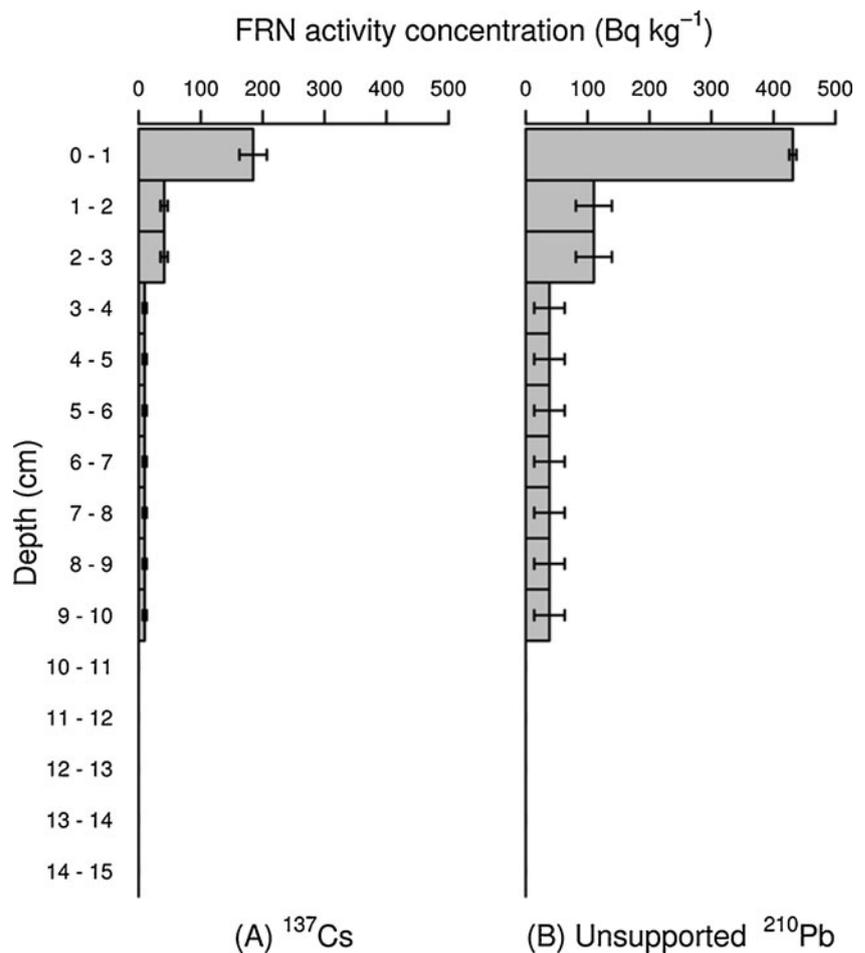
of $^{210}\text{Pb}_{\text{un}}$ were, however, not statistically different between the two watersheds.

In the case of ^{137}Cs , there was no significant trend over time, with values being constant at ca. 20 Bq kg⁻¹ in Fishtrap Creek and ca. 12 Bq kg⁻¹ in Jamieson Creek. The highest ^{137}Cs concentration (mean of 24±5 Bq kg⁻¹) was recorded in 2008. Values of $^{210}\text{Pb}_{\text{un}}$ were more variable over time; the highest value (225±42 Bq kg⁻¹) was for Jamieson Creek in 2008, although this was based on a single sample. The reason for these peak FRN values in 2008 is unknown at present, as precipitation and discharge (described later) for this year were not noticeably different (high or low) than other years with similar FRN concentrations.

4.3 Changes in sediment sources due to the wildfire

As there were no statistically significant differences between subsurface soil and channel bank material for both watersheds, these samples were combined into one source group representing subsurface/channel bank material from both catchments; thereby creating a more robust sample population. A bi-plot of the ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ concentrations for the source materials and fluvial sediment samples for both watersheds (Fig. 4) shows that there are reasonably distinct differences between surface soil and subsurface/channel bank material. Those subsurface/channel bank samples that lie within the cluster

Fig. 2 Depth distributions of ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ from 0 to 15 cm for a site in Fishtrap watershed. The surface sample represents the ash layer. Error bars represent the 95% confidence associated with the measurement process. FRN activity concentrations were zero below 15 cm depth



represented by the surface soil samples from the Fishtrap and Jamieson watersheds (i.e. ^{137}Cs concentrations above ca.



Fig. 3 Subsurface material exposed to the atmosphere, due to the combustion of roots and below-ground vegetation during the 2003 wildfire in Fishtrap watershed. Such exposed subsurface material may have received additional fallout inputs of $^{210}\text{Pb}_{\text{un}}$, in addition to inputs of redistributed surface soil material and ash enriched with ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$. The vertical black structure is the base of a burnt tree and exposed roots (photo taken in July 2004)

50 Bq kg $^{-1}$; see Fig. 4) were those samples collected from the Fishtrap watershed in areas of recently exposed subsoil caused by fallen and/or combusted trees (see Fig. 3), and thus may have received some FRN material from wind and/or fluvial redistribution following the wildfire, as described above.

There are also reasonably distinct differences between surface soil influenced by the wildfire (i.e. Fishtrap watershed) and surface material that was not burnt (i.e. Jamieson watershed). Furthermore, the three surface ash samples collected from Fishtrap watershed in 2004 are distinct (i.e. ^{137}Cs concentrations >180 Bq kg $^{-1}$; Fig. 4) from the other surface soil samples collected at the same time (i.e. 2004) from Fishtrap watershed. The non-ash 2004 surface samples have much lower FRN concentrations (i.e. ^{137}Cs concentrations <90 Bq kg $^{-1}$ and $^{210}\text{Pb}_{\text{un}}$ <190 Bq kg $^{-1}$; Fig. 4). These samples also lie within the cluster formed by the 2007 surface soil samples from Jamieson watershed, which have lower FRN concentrations (i.e. ^{137}Cs concentrations between ca. 45 and 115 Bq kg $^{-1}$; Fig. 4). Therefore, the surface samples collected from Fishtrap watershed in 2007 with the higher concentrations of FRNs (i.e. ^{137}Cs >150 Bq kg $^{-1}$ and $^{210}\text{Pb}_{\text{un}}$ >250 Bq kg $^{-1}$) may represent a combination of ash and surface soil that has become partly mixed and homogenized by bioturbation since

Table 2 ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ activity concentrations of fluvial sediment samples collected between 2004 and 2010 from the two study watersheds

Year	Watershed	N	^{137}Cs (Bq kg^{-1})		$^{210}\text{Pb}_{\text{un}}$ (Bq kg^{-1})	
			Mean ^a	Range	Mean ^a	Range
2004	Fishtrap	4	19.2±3.6	12.1–28.3	83.0±33.1	44.9–140.2
	Jamieson	4	8.4±4.1	5.7–12.2	71.2±35.9	53.5–100.3
2007	Fishtrap	1	18.2±4.5	–	125.7±35.0	–
	Jamieson	1	12.9±4.2	–	133.0±32.6	–
2008	Fishtrap	5	24.2±5.3	20.2–26.0	124.5±33.3	97.3–151.0
	Jamieson	1	11.2±5.6	–	224.5±41.7	–
2009	Fishtrap	8	19.5±3.9	13.3–24.0	94.1±24.7	49.2–142.5
	Jamieson	9	12.6±3.8	9.1–24.5	119.1±26.4	64.3–160.3
2010	Fishtrap	8	20.1±4.9	13.5–24.3	103.5±26.9	59.7–226.2
	Jamieson	10	11.8±3.1	8.6–15.2	101.6±22.3	45.3–158.6

The samples represent a mix of suspended sediment (collected by both the Petticrew and Phillips samplers) and channel bed sediment

^a Mean value±the average uncertainty associated with the measurement of each sample (i.e. ±2 sigma: see text for details)

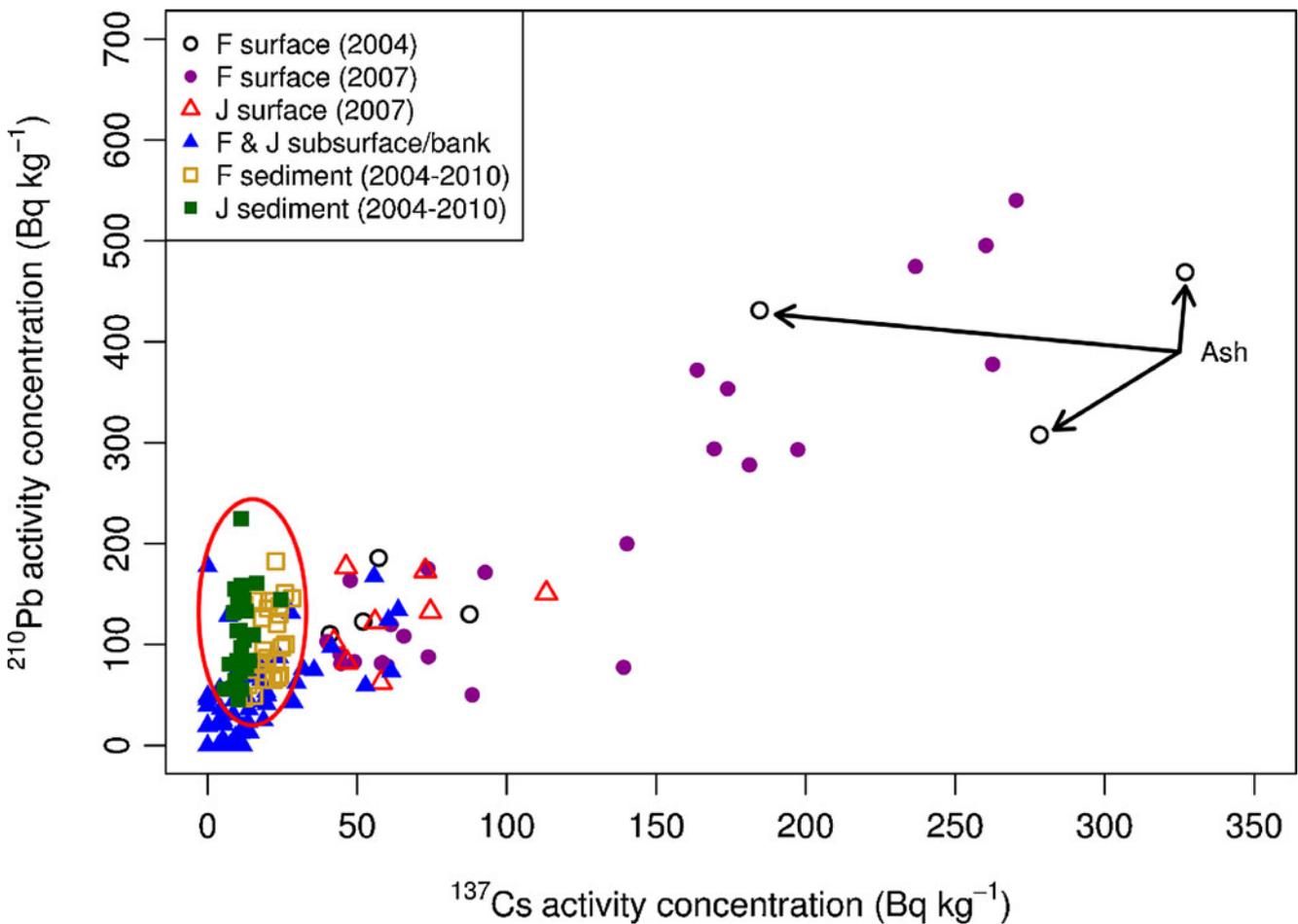


Fig. 4 ^{137}Cs and $^{210}\text{Pb}_{\text{un}}$ activity concentrations of source and sediment samples for Fishtrap (F) and Jamieson (J) watersheds. Subsurface/channel bank material represents a combined source group of subsurface soil and channel bank material for both catchments. The source materials are those collected in 2007, although the surface soil

from Fishtrap collected in 2004 (which includes three ash samples) are also presented as these were influenced by the wildfire. The sediment samples collected between 2004 and 2010 are enclosed in the envelope (bottom left)

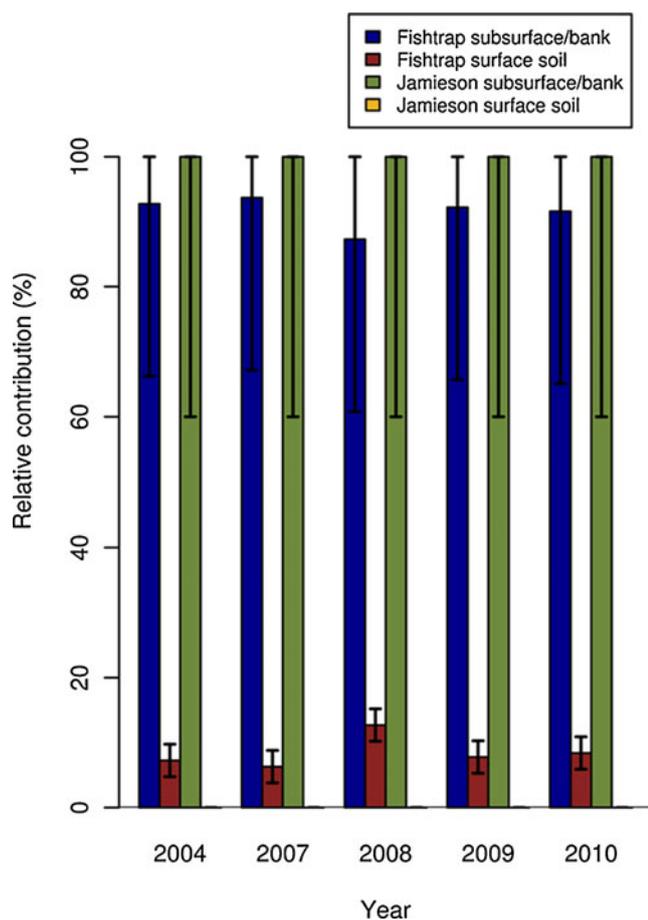


Fig. 5 Relative contributions from surface soil and subsurface/channel bank material for the fluvial sediment samples collected from Jamieson and Fishtrap Creeks for the period 2004–2010. See text for explanation of estimates of source contribution uncertainty; it is assumed that the relative contributions are ≥ 0 and $\leq 100\%$ and thus the errors cannot be outside this range

the wildfire event in 2003, and/or may represent the in-wash of ash into pore spaces.

The sediment samples from both watersheds occur exclusively within the cluster formed by the subsurface/channel bank material, suggesting that this is the primary source of the actively transported (and stored in the channel bed) fine-grained sediment in Fishtrap and Jamieson Creeks. The FRN (especially ^{137}Cs) concentrations of the sediment samples for Fishtrap Creek tend to be greater than those for Jamieson Creek, suggesting that there is an increase in the relative contribution of surface soil material (possibly including ash) in Fishtrap Creek. As there are no marked changes in ^{137}Cs concentrations of sediment samples over time (see Table 2), then the relative contribution of surface vs subsurface/channel bank material does not change markedly over the period 2004 to 2010. The elevated values of $^{210}\text{Pb}_{\text{un}}$ associated with the sediment samples, when compared to the main ^{137}Cs - $^{210}\text{Pb}_{\text{un}}$ axis for source materials in Fig. 4 (i.e. the $^{210}\text{Pb}_{\text{un}}$ concentrations of the sediment samples exceed the main cluster for the

subsurface/channel bank material) may reflect direct fallout of $^{210}\text{Pb}_{\text{un}}$ onto sediment during transit (Wallbrink and Murray 1993; Wilkinson et al. 2009).

Figure 5 shows the relative contributions of surface soil and subsurface/channel bank material for the fluvial sediment samples collected from Fishtrap and Jamieson Creeks, as determined by the unmixing model. For Jamieson Creek, the unmixing model indicates that 100% (95% confidence range is 60 to 100%; i.e. the contribution cannot be $>100\%$) of the fine-grained sediment transported at the collection site was derived from subsurface/channel bank sources. This finding is in keeping with the location of the sediment samples within the cluster of subsurface/channel bank samples in Fig. 4. For Fishtrap Creek, the mean contributions from surface soil and subsurface/channel bank material were 8.5% (95% confidence range is 6.0 to 11.0%) and 91.5% (95% confidence range is 64.5 to 100%), respectively. The relatively high uncertainties associated with the unmixing model results partly reflect the larger measurement errors associated with the $^{210}\text{Pb}_{\text{un}}$ measurements, and thus the variations in $^{210}\text{Pb}_{\text{un}}$ values compared to relatively constant ^{137}Cs values. It also reflects the generally high spatial variability in FRN values for surface soil and subsurface/channel bank sources due to the effects of the wildfire, as described above. The implication of this is that emphasis should be placed on the magnitude of the contributions. The relative contributions from surface soil sources were approximately constant over the period 2004–2010. The sediment fingerprinting results suggest that while subsurface/channel bank material was the dominant sediment source in these forested watersheds, wildfires may increase relative contributions from surface soil sources.

There is, however, one important caveat to consider. While there is the potential for mobilized source material stored in upper reaches to have been labelled with additional fallout $^{210}\text{Pb}_{\text{un}}$ during transportation, as indicated above, the elevation in concentration does not appear to influence the outcome, i.e. subsurface and channel bank sources are still shown to dominate.

5 Sediment response of Fishtrap watershed to the wildfire

Concentrations of FRNs in source materials and fluvial sediment samples show that subsurface/channel bank material was the dominant sediment source in both watersheds, including that affected by the wildfire, i.e. Fishtrap watershed. Surface soil was, however, a measurable sediment source in Fishtrap watershed, contributing on average $8.5 \pm 2.5\%$, suggesting that the wildfire did increase the relative contribution from surface soil sources, and thus had some effect on surface erosion processes on hillslopes. While this finding of the dominance of subsurface/channel bank material is in contrast to work (using FRNs) in Australia (e.g. Wilkinson et al. 2009; Smith

et al. 2011b), it does agree with other studies in the USA that have used other techniques to determine sediment responses following wildfire (e.g. Moody and Martin 2001, 2009). In the USA studies, the increase in importance of channel bank sources reflected a large increase in channel erosion (due to increased flows) following wildfire. The results presented here are also in agreement with other non-wildfire-impacted forested environments, where gully and river bank erosion of subsoil are the prime sediment sources (Wallbrink and Croke 2002; Nagle et al. 2007). In Fishtrap watershed, there was limited evidence of mass movement events and gully erosion, and it is concluded that erosion of channel banks represents the main process contributing fine (and coarse) sediment to the creek. This finding is corroborated by the work of Eaton et al. (2010a) on the channel bank dynamics of Fishtrap Creek, which has documented significant channel bank erosion and channel migration due to root decay and a lack of vegetative root strength 3–5 years following the wildfire.

The fairly constant nature of the relative contributions of surface and subsurface/channel bank materials over time is contrary to many other studies in watersheds affected by wildfires (e.g. Wilkinson et al. 2009; Smith et al. 2011b). Such studies have documented an increased contribution from surface soil material in the first few years following the wildfire, often up to contributions of 80–100% in the first year following the wildfire (Wilkinson et al. 2009; Smith et al. 2011b), with a subsequent reduction in surface sources and, thereby, relative increase in subsurface/channel bank sources in subsequent years. In the case of the Fishtrap watershed, this lack of distinct changes in sediment sources (and, in particular, the apparent minimal increase in hillslope erosion) is matched by the lack of

a major increase in fine sediment fluxes immediately following the wildfire (Petticrew et al. 2006; Eaton et al. 2010a; unpublished data). Thus, Petticrew et al. (2006) estimated that seasonal (April–October) fine-grained sediment fluxes in Fishtrap watershed in 2004 were low (ca. 855 kg km⁻²) and only 66% greater than those in Jamieson watershed. In subsequent years, preliminary estimates of channel sediment fluxes (unpublished data) appear to be higher, but were still similar for the two creeks; i.e. they were not dramatically increased in Fishtrap Creek.

The situation in Fishtrap watershed may reflect the fact that while the system (especially the hillslopes) in the watershed was primed for erosion and sediment delivery from hillslopes to rivers, there was a lack of precipitation (especially in winter) and subsequent snowmelt in the immediate years following the wildfire. For example, total annual precipitation in 2003 (336.2 mm; Table 3) was low compared to the longer-term (1980–2002) average (469.5 mm). While 2004 was considerably wetter (annual total=572.2 mm; Table 3), most of the ‘excess’ precipitation (relative to the long-term average) occurred during summer months, when vegetation had started to recover. In these snowmelt-driven systems, most water and sediment transport occurs during the April to June freshet period, thus a better measure of available water for erosion and sediment transport is provided by the winter (November to March) precipitation total for the period preceding the collection of sediment samples (i.e. between April to October). Winter precipitation for the first three years following the wildfire (i.e. 2004, 2005 and 2006) was less than the longer-term average (180.5 mm; Table 3). This is corroborated by flood frequency analysis

Table 3 Annual mean and maximum daily mean discharge at the Water Survey of Canada gauging station at Fishtrap Creek (see Fig. 1 for location) and total annual (January to December calendar year) and

winter (November to March) precipitation for the Environment Canada precipitation gauge at McLure

Year	Annual mean discharge ^a (m ³ s ⁻¹)	Maximum daily mean discharge (and day) ^a (m ³ s ⁻¹)	Total annual precipitation (mm)	Winter precipitation ^b (mm)
1980–2002/2003 (mean)	0.76	14.9 (May 15 1997)	469.5	180.5
2003	0.52	4.09 (May 2)	336.2	162.2
2004	0.67 ^c	4.57 (April 27)	572.2	159.8
2005	1.05	9.74 (April 24)	408.9	165.2
2006	1.08	8.61 (April 30)	469.4	118.7
2007	0.93	7.67 (May 9)	397.6	247.5
2008	0.74	7.57 (May 17)	343.6	156.0
2009	0.56	6.72 (May 3)	264.3	170.0
2010	0.83	7.58 (April 21)	N/A	N/A

^a Discharge data have been reanalyzed and represent hourly averages based on 15-min data, and therefore differ slightly from values given at WSC (2012)

^b Represents precipitation (probably as snow) prior to the freshet for that calendar year (i.e. value for 2003 is sum of monthly totals for November 2002 to March 2003 inclusive). Freshet period is April to June inclusive

^c Likely to be an underestimate due to damage to the WSC gauging station due to the wildfire in 2003 and lack of data in January and February 2004
N/A not available

which showed that the return periods for the peak flows during the period 2004–2007 (all during the freshet) were between 1.5 and 2.5 years, compared to the long-term return period for the pre-fire years 1971–2002 (range ca. 1 to 35 years; Eaton et al. 2010a). Metrics describing discharge (annual mean and maximum daily mean) were low immediately after the wildfire, with maximum post-wildfire river discharges recorded in 2005 and 2006 (see Table 3). Two to three years after the wildfire, there was sufficient vegetation recovery to limit hillslope erosion, even though winter precipitation was significantly higher in 2007 (247.5 mm), and flows were greater in 2005 and 2006 (see Table 3). From 2008, rivers flows have decreased to values consistent with the long-term average. Unfortunately, sediment samples were not collected in 2005 and 2006; the years with maximum river flows. However, the annual mean discharge for 2007 (during which sediment samples were collected) was similar to that in 2005 and 2006 (see Table 3), and relative contributions from surface soil sources in 2007 were not significantly different from years before or after (see Fig. 5).

It has also been suggested (e.g. Phillips and Eaton 2008) that there was limited evidence of fire-induced soil hydrophobicity in the Fishtrap watershed. This contrasts with other wildfire events, where an increase in soil hydrophobicity due to wildfire was partly responsible for increases in surface runoff and hillslope erosion rates (Shakesby and Doerr 2006).

The limited changes in sediment sources (e.g. Fig. 5) and limited increases in sediment fluxes (e.g. Petticrew et al. 2006; Eaton et al. 2010a) following the 2003 wildfire could reflect intermediate storage effects between sediment sources and the channel system, which could delay or attenuate the downstream response. Several studies (e.g. Trimble 1981) have shown that

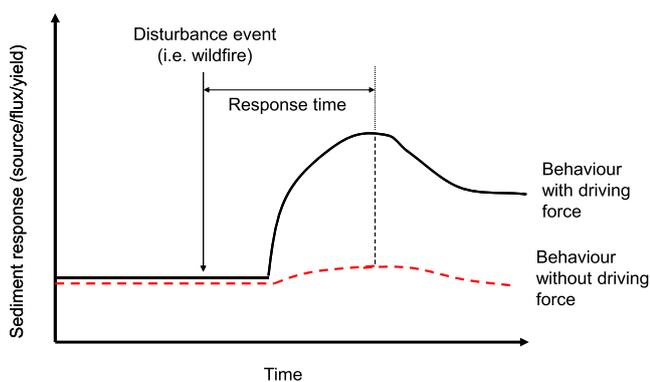


Fig. 6 Different types of sediment response (i.e. changes in sediment sources, increases in sediment fluxes) over time to disturbances such as wildfires, showing the effect of the presence or absence of driving forces (e.g. precipitation and snowmelt) on the magnitude of the response. The response time represents the combined reaction time (i.e. there is often a time lag between the disturbance event and the time at which the system starts to respond) and relaxation time (based on Prosser and Williams 1998; Viles et al. 2008; Owens et al. 2010)

changes in intermediate storage (and thus sediment delivery ratios) can mask increased erosion on hillslopes, such that there are no major changes in sediment fluxes in downstream channels; i.e. increased soil erosion on hillslopes is matched by increased sediment storage at toeslope and other similar locations. However, major areas of fine sediment storage following the wildfire were not observed during field surveys of Fishtrap Creek, although further work is required to verify this.

Thus in the context of disturbance-response assessment (cf. Viles et al. 2008; Owens et al. 2010), the response of landscapes to disturbances, such as wildfire, is a function of both the disturbance event and the susceptibility of the system to respond due to driving forces, such as precipitation and snowmelt, within a ‘window of disturbance’ (Prosser and Williams 1998; Fig. 6). In the case of the 2003 wildfire event at Fishtrap watershed, while there was a disturbance event, the driving force was not conducive to result in the response often recorded in other wildfire-impacted watersheds. Further measurement and monitoring are required to determine whether the response to the wildfire is muted over the medium to longer term (i.e. 10–50 years) or whether there will be a significant, but delayed, response in sediment sources and fluxes in the next few years/decades as any existing fire-impacted vegetation completely breaks down (i.e. existing standing, but dead, trees collapse and their root systems break-down).

6 Conclusions

Fine-grained sediment transported within Fishtrap (burnt) and Jamieson (unburnt) Creeks was predominantly derived from subsurface/channel bank sources. The lack of mass movements and gully erosion, and the obvious channel erosion and migration, suggest that channel banks are the prime source of fine-grained sediment in both of these watersheds, despite the influence of wildfire. Assuming that Jamieson watershed is representative of the situation in Fishtrap watershed prior to the wildfire, then the 2003 wildfire in Fishtrap watershed slightly increased (from 0% to $8.5 \pm 2.5\%$) the relative contribution from surface soil on the hillslopes adjacent to the channel network. There was also a limited response—at least in terms of that which can be ascribed to the wildfire—in terms of fine-grained sediment fluxes in the burnt watershed. The reason for this muted response to a severe wildfire is likely to be the lack of winter precipitation in the first year following the wildfire. Thus while the landscape was primed for erosion and sediment transport, the lack of a driving force (i.e. precipitation and snowmelt) in the immediate post-fire period meant that there was limited sediment response.

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