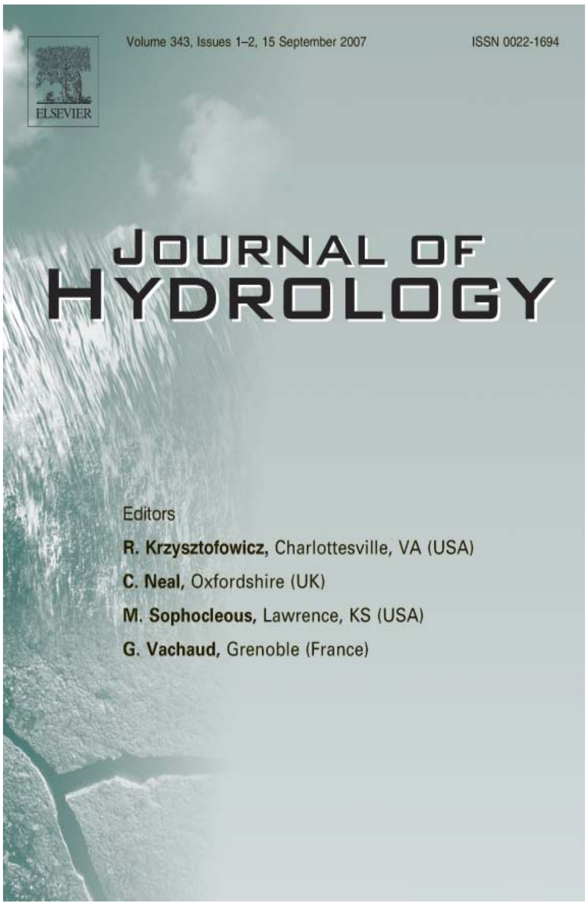

Volume 343, Issues 1–2, 15 September 2007
ISSN 0022-1694
ELSEVIER
JOURNAL OF HYDROLOGY
Editors
R. Krzysztofowicz, Charlottesville, VA (USA)
C. Neal, Oxfordshire (UK)
M. Sophocleous, Lawrence, KS (USA)
G. Vachaud, Grenoble (France)

Journal of Hydrology (2007) **343**, 12–28

ELSEVIER

available at www.sciencedirect.com

ScienceDirect

journal homepage: www.elsevier.com/locate/jhydrol

Quantification of hillslope runoff and erosion processes before and after wildfire in a wet *Eucalyptus* forest

Gary J. Sheridan *, Patrick N.J. Lane, Philip J. Noske

School of Forest and Ecosystem Science, The University of Melbourne, P.O. Box 137, Heidelberg 3084, Victoria, Australia

Received 12 October 2006; received in revised form 4 June 2007; accepted 13 June 2007

KEYWORDS
Water quality;
Fire;
Water repellence;
Sediment generation;
Erodibility;
Australia

Summary An experimental program was initiated following an intense wildfire in 2003 in NE Victoria Australia, to quantify an expected large increase in hillslope erosion by rill and interrill processes from a steep, wet, *Eucalyptus* forest. Water repellence, soil water content, infiltration capacity, infiltration-excess runoff generation, interrill erodibility, and rill erodibility were measured periodically for 3 years. Rill *erodibility* on 27° slopes increased immediately following the fire by a factor of 540 times, but decreased exponentially to very low background levels 2 years after the fire. Despite this increased erodibility, rill *erosion* was uncommon in the catchment, because soil saturated hydraulic conductivity (K_{sat}) remained very high (100–1000 mm h^{-1}) and simulated concentrated flows of 3 l s^{-1} were found to be ''transient'', infiltrating into areas with high K_{sat} a short distance downslope. Despite the very high K_{sat} values, runoff ratios from 100 mm h^{-1} simulated rainfall as high as 65% were recorded 6 months after the fire due to very high spatial variability in K_{sat}. The runoff sediment concentration under 100 mm h^{-1} simulated rainfall increased 10-fold immediately following the fire and declined exponentially to pre-fire levels within 2 years as ground cover recovered to 90–95%. Considered together, the above results suggest that most infiltration-excess overland flow reaching streams in this catchment is produced from within several metres of the stream edge, and that the transported sediment is generated by interrill processes. This conclusion contrasts strongly with common perceptions/observations of widespread overland flow dominated erosion processes following fire, and perhaps indicates substantial functional differences in erosion processes between different *Eucalyptus* forests. Measurements in unburnt areas, intended for comparative purposes only, provided unexpected insights in their own right. These areas showed a large natural seasonal oscillation in water repellence (non-repellent in winter to strongly repellent in summer) and runoff generation from rainfall simulation

* Corresponding author. Tel.: +61 3 9450 8726; fax: +61 3 9450 8644.
E-mail address: sheridan@unimelb.edu.au (G.J. Sheridan).

doi:10.1016/j.jhydrol.2007.06.005

(1% in winter to 42% in summer) as the soil gravimetric water content crossed a hysteresis-variable threshold value. These large seasonal fluctuations in runoff generation indicate that proportional increases in erosion over the unburnt condition are highly temporally variable; 18-fold immediately after the fire, increasing to 2240-fold at six months, and then reducing to only 2.5-fold by 12 months after the fire, perhaps explaining some of the diversity of reported results in the literature. The above results suggest erosion risk is greatest in the first winter following the fire. Both the runoff and erosion results show that conventional erosion modelling approaches in this catchment will fail (both in burnt and unburnt forests) because the dominant driving processes and soil properties are not represented.

Introduction

Soil erosion rates and catchment exports of sediment and nutrients are generally observed to increase considerably following forest fire (see recent review papers by Shakesby and Doerr (2006) and Shakesby et al. (2007)). Forested areas are utilized as water supply catchments in many parts of the world, and consequently understanding and predicting the magnitude of the effect of fire on sediment loads is a priority for catchment managers. However the magnitude of the increase is extremely variable. At the catchment scale, factor-increases ranging from zero (Prosser and Williams, 1998), four fold (Scott and Van Wyk, 1990), eight to ten fold (Lane et al., 2006a; Helvey, 1980), and perhaps up to 1000-fold (Brown, 1972) have been reported. At the plot/hillslope scale, factor increases in sediment delivery between 2- and 1000-fold have been reported (Morris and Moses, 1987; Scott and Van Wyk, 1992; Shakesby et al., 1993; Cerda, 1998; Prosser and Williams, 1998; Benavides-Solorio and MacDonald, 2001; Cannon et al., 2001; Moody and Martin, 2001; Pierson et al., 2001). This variability is presumably due to an interaction of factors such as climate, topography, soil properties, burn severity and vegetation recovery, altering hydrologic and erosion processes and resulting in:

1. increased input precipitation reaching ground surface due to the reduced canopy and litter interception of rainfall (Crockford and Richardson, 2000);
2. changed infiltration-excess runoff due to fire-induced water repellency (Imeson et al., 1992), burnt-out root channels providing easier flow paths for infiltrating water (Ferreira, 1996 in Shakesby et al., 2000), and possibly sealing of soil pores by ash;
3. increased rainfall energy and interrill erosion (Benavides-Solorio and MacDonald, 2001), and overland flow energy and scour erosion (Wondzell and King, 2003), due to reduced contact cover;
4. changes in soil erodibility (Giovannini and Lucchesi, 1983; Scott et al., 1998);
5. changed levels of bioturbation (Dragovich and Morris, 2002; Shakesby et al., 2007);
6. and, at larger spatial and temporal scales; increased saturation-excess runoff due to reduced evapotraspiration and subsequent increases in the water table (Langford, 1976; Kuczera, 1987); increased streampower due to increased peak-flows; increased frequency of debris-flows (Cannon et al., 2001).

The combined effect of these changes to runoff and erosion processes is a net change in erosion rates, and ultimately in sediment delivery from the catchment. Widespread hillslope erosion by concentrated overland flow is often observed after wildfire in a range of climate and forest types (Leitch et al., 1983; Shakesby et al., 2003). Modelling by Wondzell and King (2003) suggested that this is the dominant mechanism for surface erosion following fire in the USA. Despite this, changes in *interrill* erosion rates following fire have been measured far more frequently than changes in *rill* erosion rates, probably reflecting the existence of well established experimental methodologies for quantifying the former. Based on this evidence, this study explicitly focuses on the post-fire generation of runoff and subsequent erosion by rainfall and concentrated overland flow at the plot to hillslope scale (points 2, 3, and 4 above).

Fire can increase the generation of infiltration-excess runoff by increasing the spatial and temporal frequency and severity of water repellence (Robichaud, 2000; Varela et al., 2005). In addition to fire, water repellence is also strongly influenced by soil dryness (Doerr and Thomas, 2000; MacDonald and Huffman, 2004) and numerous other soil properties (Bond and Harris, 1964; Harris et al., 1964; McKissock et al., 1998; DeBano, 2000a,b). While the presence of water repellence is often reported in runoff studies, the processes responsible for water repellence are rarely investigated in these studies, limiting the ability to generalize research findings. Soil water repellence was reviewed extensively by DeBano (2000a,b). DeBano (1981) proposed a model in which fire promotes water repellence as a result of the movement and subsequent condensation of vaporized organic substances due to a strong temperature gradient in the soil during the fire. However, the contribution of fire-induced water repellence to changes in runoff generation is often unclear because research methods usually do not allow water repellence effects to be quantified independently of other fire-related changes (Shakesby et al., 2000). For example, increased runoff generation following fire has also been attributed to the washing in of ash particles, resulting in the blocking of soil pores (Shakesby and Doerr, 2006), though there is limited accessible literature to support this reasonable contention.

The presence of existing soil macropores can affect the magnitude of the runoff response to water repellency. Shakesby et al. (2000) highlight spatial contiguity as a key determinant of the magnitude of the impact of increased water repellence on soil erosion rates. This is considered both in terms of the degree of patchiness of the hydrophobic soil and the frequency of other by-pass pathways provided by cracks and macropores. Burch et al. (1989) highlighted the role of root channels when explaining the lack of a measurable impact on runoff under highly water-repellent conditions in an Australian *Eucalyptus* forest. Ferreira (1996) in Shakesby et al. (2000), reported infiltration rates over burnt-out root channels that were orders of magnitude higher than rates for the surrounding water-repellent soil.

Following fire, an increased proportion of rainfall and runoff kinetic energy; a major driver of soil erosion (Wischmeier and Smith, 1958), is expended on the soil surface due to the removal of contact cover by burning. The loss of contact cover is likely to be a major factor explaining increased soil erosion following fire in wet *Eucalyptus* forests because under unburnt conditions cover is close to 100%, resulting in extremely low erosion rates. For example, Lu et al. (2003) predicted erosion rates from *Eucalyptus* forests in SE Australia to be lower than all other landscape categories in Australia, despite the relatively steep landscapes and higher rainfall common in forested areas. Cover in close contact with the soil surface is far more effective at reducing erosion than equivalent canopy cover because falling drops are so close to the soil surface that they regain little fall velocity. In addition, cover in contact with the soil can reduce flow velocity by roughness, thereby reducing both the erosive potential and the transport capacity of the flow.

The relationship between soil cover and erosion rates is generally non-linear, with increasing cover levels resulting in decreases in erosion rates that are typically represented as either negative exponential (Renard et al., 1991; NSERL, 1995) or reciprocal functions (Loch, 2000) of the proportion of cover. The rate of erosion reduction with increasing contact cover level is highly variable, and is strongly associated with the relative contribution of rill and interrill erosion to total erosion at the site of interest. In general, cover is more effective at reducing erosion when rill erosion rates are high.

Fire can also change the erodibility (rate of soil loss per unit of rainfall/flow energy) of the soil (Wilson, 1999), although experimentally this change is often difficult to isolate from other factors such as loss of soil cover. Immediately after fire the surface ash layer is of low density and prone to erosion, yielding an initial increase in erodibility. For the underlying soil fire can result in increased aggregate stability due to increased physical bonding between clay and silt particles (Blake et al., 2007), or reduced stability due to oxidation of organic matter. For example, Craig (1968) found that the aggregate stability of a clay-loam forest soil was strongly enhanced, and dispersion reduced, following heating above 400 °C.

The preceding discussion highlights the fact that the net increases in erosion following fire are a function of many interrelated and interacting processes and properties, and as a result the net effect of fire on erosion is both highly variable and probably difficult to predict. Compounding these difficulties are underlying issues related to the scale of measurement. Recent reviews/research on soil water repellence and erosion (Shakesby et al., 2000), wildfire and erosion (Shakesby and Doerr, 2006), and rainfall–runoff modelling (Doerr et al., 2003) have highlighted the critical role of spatial scale of measurement in the interpretation of small-plot erosion data. In particular, the reviews emphasize the need to understand how processes measured at the small-plot scale may interact with hillslope properties at larger scales, and to sediment delivery at catchment outlets.

The suite of experimental procedures described in this paper have been selected with this scaling issue in mind, and will hopefully lead to an improved conceptual understanding of erosion processes following fire, and ultimately to improved modelling capability. The presented hillslope erosion results can be interpreted in the context of the parallel catchment scale study undertaken in the East Kiewa Research Catchments after the same 2003 fire, reported by Lane et al. (2006a).

Methods

Site description

The research sites are located in wet *Eucalyptus* forest within the East Kiewa Research Catchments in NE Victoria, Australia (Fig. 1), described by Leitch (1979), Laing (1981), Hough (1983), Papworth et al. (1990) and Lane et al. (2006a). Briefly, the geology in the upper part of the research catchments (where the erosion experiments were conducted) consists of quartz diorite and Ordovician Gneiss (Laing, 1981). The elevation is 1200–1300 m and the topography is steep with slope gradients >24° on more than half the catchment (Leitch, 1979). The soils are friable brown gradational clay-loams and sandy clay-loams generally <1.5 m deep grading to coarse sand C horizons (Hough, 1983). There is decreasing pedality with depth and stones and rocks are distributed throughout the profile. The surface soils have low susceptibility to slaking, dispersion, and erosion (Hough, 1983; McKenzie et al., 2004). The soils are classified in the Australian Soil Classification (Isbell, 1996) as Acidic Eutrophic Red Dermosols.

Mean annual rainfall is around 1800–2000 mm, with a winter/spring dominant distribution. Intense summer thunderstorms are common in the area. Snow falls occur intermittently at higher elevations in cooler conditions. Mean daily maximum temperature is 17.2 °C and mean daily minimum is 5.4 °C. The vegetation is wet sclerophyll and has been described by Smith et al. (1981) and Papworth et al. (1990). Mature Alpine Ash (*Eucalyptus delegatensis* RT Baker) exists in almost pure strands in the upper catchment, often showing the effects of previous wildfires in 1919 and 1939. The East Kiewa research catchments were extensively burnt in January and February 2003 at a moderate-severe intensity, during widespread wildfires that burned more than 1.3 M ha of mostly forested land across the state of Victoria. A small (approximately 1 ha) area in the upper part of the catchment was not burnt. Severe crown scorch was apparent for most trees, and a high proportion of the Alpine Ash were killed. Details of burn severity in the research area are given by Lane et al. (2006a).

Water repellence and soil water content

The relationship between soil water content and soil water repellence was determined. Individual soil samples were collected to a depth of 5 cm from 45 locations across the

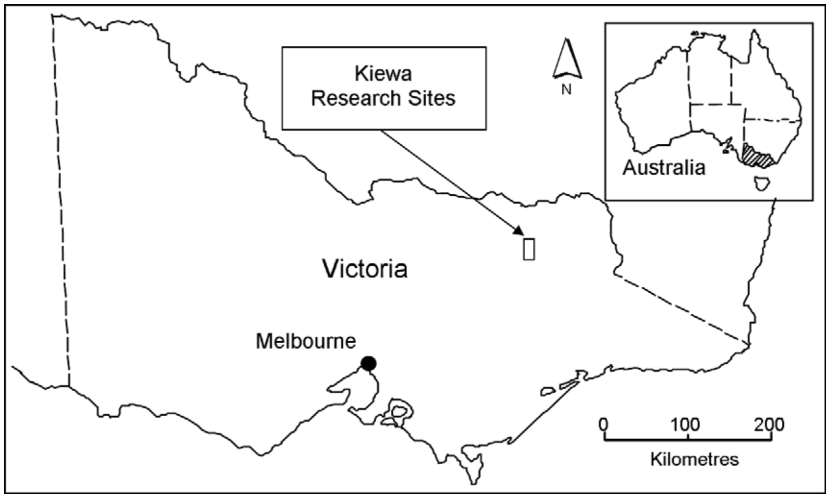


Figure 1 The location of the research area (36°48'S, 147°12'E).

burnt area in autumn 2003, and from five locations within the unburnt area in 2005. Samples were bulked as either burnt or unburnt, and a representative 2 kg sub-sample of each was taken, crushed to 2 mm and dried at 40 °C. Water repellence was determined on both the wetting and drying cycles. For the wetting cycle, a sub-sample was taken in a Petri dish for water repellence testing, and a corresponding 10 g sub-sample was taken for determination of the gravimetric water content (GWC) by oven drying overnight at 105 ° C in a soil tin. Sixty milliliter of water was then added to the remaining bulk soil and mixed thoroughly, and the above sampling procedure repeated. This approach provided a series of paired measurements of GWC and water repellence on the wetting cycle, from a GWC close to zero, up to about 40% water content. A similar procedure was adopted for the drying cycle except that the bulk soil was initially wetted close to field capacity (around 45% GWC for this clay-loam soil). The bulk soil was spread thinly on a tray and dried in an oven at 100 °C, after a time a sub sample was taken and GWC and water repellency determined, the bulk soil was then returned to the oven, procedure was repeated to yield a range of soil at different GWC. An assumption made here is that all soil particles are wetted by the addition of water, which cannot be guaranteed (Doerr and Thomas, 2000). Additionally, there is a risk that the repellency values could change during heating, which did not replicate natural drying conditions.

Water repellence was assessed using the molarity of ethanol droplet (MED), which is similar to methods described by King (1981), Doerr et al. (1998) and Leighton-Boyce et al. (2005). The MED test involves determining the lowest concentration of ethanol at which a droplet will penetrate the surface of the soil in five seconds. Ethanol–water solutions with concentrations between 0.0 and 5.0 M were prepared in 0.25 M steps. Ethanol concentrations were converted to surface tension values based on functions reported by Butler and Wightman (1932).

Rainfall simulation

Rainfall simulation (RFS) was used to quantify the infiltration-excess runoff generation and sediment concentration in the field, at approximately 6-monthly intervals over the period of the study. Simulated rainfall with an energy of 295 kJ ha^{-1} mm^{-1} was applied for 30 min at a nominal intensity of 100 mm h^{-1} to 1.5 m wide and 2 m long plots established at three representative (as determined by visual inspection) locations within each of the burnt and unburnt areas. One hundred millimeter per hour intensity is similar to the higher range of intensities employed by Wilson (1999), Benavides-Solorio and MacDonald (2001) and Lane et al. (2004) in fire erosion/infiltration studies and is a commonly used intensity in Australian rainfall simulation studies (Connolly et al., 2002).

As vegetation recovered, rainfall simulation plots were selected in order to include levels of vegetation cover that were representative of the mean surface contact cover for the entire burnt catchment as determined from vegetation and cover surveys (Lane et al., 2006a). The vegetation and cover level on the rainfall simulation plots was assessed using field cover charts (McDonald et al., 1990). The proportion of contact cover is reported including all elements that protect the soil from raindrop energy, including leaf litter, rock, wood, bark and branches, grass, and other low-growing vegetation.

Typical plot slopes were in the range 15–30°, and plots were selected to have minimal cross slope. Plots were bordered using steel sheets hammered into the ground, and overland flow was collected at the downslope end of the plot in stainless steel troughs, sealed to the soil surface with petroleum jelly. The RFS delivers raindrops from three oscillating fan-shaped sprays at 1 m centres sweeping intermittently across the plot surface and is based on an original design by Bubenzer and Meyer (1965), which was modified by Loch (1989). Water was supplied to the rainfall simulator from a 1000 L tank filled at nearby forest streams. Samples of the supply water were collected to ensure that background sediment concentration could be discounted in calculations of the sediment concentration of plot runoff.

Timed runoff samples were collected at regular time intervals in 500 mL plastic bottles for measurement of the runoff rate and sediment concentration. Typically eight samples were collected during the 30 min rainfall event.

The method for determination of suspended sediment concentration varied depending on the apparent sediment load. For low concentrations, a 0.45 μm millipore filter was used to filter 100 mL of suspension. For moderate concentrations, sub-samples were taken and dried in pre-weighed 30 mL glass vials weighed to 4 decimal places. For high concentrations, the sample was dried directly in the pre-weighed oven-proof 0.5 L container to two decimal places.

Sediment concentration and erosion rate is known to vary with slope. The selected plots for individual rainfall simulation experiments varied in slope. To allow comparison between treatments and time-steps, reported sediment concentrations are adjusted to equivalent values at 22° (the average value for all the plots) using the interrill slope adjustment equation proposed by Sheridan et al. (2003)

$$S_a = -1.50 + 6.51/[1 + \exp(0.94 - 5.30\sin\theta)] \quad (1)$$

where θ is the plot slope in the direction of overland flow. The water repellence of the soil at the time of the rainfall simulation experiments was also determined from in situ field water repellence tests at six points at the outer edge of, and immediately adjacent to, the rainfall simulation plots. The field method is a simplification of the King's (1981) molarity of ethanol droplet test, and utilizes a comparison of the ability (yes/no) for water and 2 M ethanol to penetrate the soil in 10 s. This provides a broad classification of water repellency into three classes: non-water-repellent, where a water drop infiltrates in $\leqslant$10 s; water-repellent (*R*), where water infiltrates in >10 s and 2 M ethanol infiltrates in $\leqslant$10 s, and strongly water-repellent, where 2 M ethanol infiltrates in >10 s.

Antecedent soil moisture was quantified by collecting soil samples for analysis of GWC. Samples were collected down to a depth of 50 mm from four points immediately outside each edge of the plot, samples were pre-weighed then oven-dried at 105 °C for 24 h, the re-weighed.

Rill erosion simulation

Rill erosion simulation was used to determine the relationship between concentrated overland flow and sediment delivery at the hillslope scale. Rill erosion experiments were conducted in autumn in 2003, 2004, and 2005. Water was applied to the upslope end of plots bordered with steel edges, 0.15 m wide and 3 m long. The plots had a relatively even slope and no cross slope in order to prevent water preferentially flowing down one edge of the plot. Two plot slopes were selected: 15° and 25°, with two replicates at each slope. Flow application rates were controlled in the range 0.03–1.7 L s^{-1} via a rotameter, with water pumped directly from nearby streams. Runoff water was collected at the base of the plot via a funnel-shaped trough. Four or five runoff rates were applied to each plot, varying from a low flow that did not detach sediment, to a high flow that was actively eroding the soil. Selection of appropriate runoff rates required some subjectivity, and the appropriate rates changed considerably over the course of the study.

To begin the runoff experiment, flow was introduced at the upslope end of the channel onto a plastic pit and the stopwatch was started once the pit overflowed into the channel. The time taken for the water to reach the trough at the end of the channel was recorded. Every 30 s from the time taken for overland flow to reach the trough, a sample of overland flow water was taken in a 500 mL plastic bottle, the time the sample took to collect was recorded. Three samples were taken during each flow rate, then the flow rate was increased and sampling repeated. Samples were then analysed for sediment concentration using the laboratory procedures described in the rainfall simulation section.

The experimental plots had no vegetation cover immediately following the fire. For subsequent plots, vegetation cover was selected at two levels to capture the effect of vegetation recovery on sediment generation levels from overland flow. Vegetation cover levels were assessed using visual comparison charts for percentage cover (McDonald et al., 1990). Data were analysed to determine the fire-recovery dependent relationship between the applied flow rate and the measured sediment delivery rate.

Saturated hydraulic conductivity

During the study, it became apparent that the soil saturated hydraulic conductivity (K_{sat}) could not be estimated from steady state runoff rates under simulated rainfall, and more direct methods for the measurement of K_{sat} were required. As a result, the K_{sat} measurements reported in this study do not form a temporal series of values as is the case for the rainfall simulation data. K_{sat} was measured directly using a single-ring ponded infiltrometer in September 2004 from unburnt sites (13 replicates), and in March 2005, September 2005 and February 2006, (12, 8, and 12 replicates, respectively) from burnt and unburnt sites during the same field visit when the rainfall simulation experiments were conducted.

The single-ring infiltrometer consisted of a 200 mm high and 300 mm diameter steel ring hammered 50 mm into the soil at selected locations near the rainfall simulation plots at each the burnt and unburnt sites. Locations were selected where (a) the steel ring could be inserted into the soil, i.e. not on rocks, logs, trees or in large holes, (b) in locations with least gradient on the steep mountain slopes, and (c) within operational distance of the access road. A float valve on the inside of the ring maintained the water level at between 0 and 10 mm depth at the centre of the ring. The infiltration capacity of the soil (mm h^{-1}) was determined from the straight-line portion of a plot of cumulative infiltration (mm) against time (h). Initial trials with the infiltrometer indicated very high infiltration capacities, indicating that the flow rate was dominated by gravity potential, and the use of a double-ring infiltrometer was therefore considered unnecessary for the purposes of this study. Reported values of K_{sat} are adjusted for the use of the single ring infiltrometer and for small differences in head using an adjustment equation proposed by Reynolds (1993).

Measurement of K_{sat} using the ring-infiltrometer is intended to characterize the soil in relation to infiltration and runoff generation. However, the method is limited by the small scale of the sampled area (ring infiltrometer area is 0.07 m^2) and by the difficulty in the selection of representative sample points in inaccessible, steep forests with trees, rocks, burrowing animal and tree-root holes. An alternative method for characterizing the ability of forested hillslopes to infiltrate overland flow was proposed by Hairsine et al. (2002) and applied by Lane et al. (2006b). This method, referred to as ''volume to breakthrough'' (Vbt) quantified the volume of water required to reach a specified distance downslope, and provides a quantitative assessment of the extent to which overland flow can infiltrate hillslopes.

In this study, the volume-to-breakthrough method described above was modified for the high infiltration capacity of the site so that a series of volume–distance pairs was recorded, allowing the changing rate of distance with discharge-volume to be quantified. These experiments are referred to as overland flow experiments in this paper. The downslope distance of flow D (m) as a function of volume applied V (L) was modeled as

$$D = L_{max}\left[1 - \exp\left(-\frac{V}{(S/L_{max})}\right)\right] \tag{2}$$

where L_{max} and S are fitted parameters optimized by minimizing the error sums of squares using Newton's method (Burden and Faires, 2001). If the hillslope is viewed as a series of leaky-buckets then L_{max} is a function of the infiltration rate and the flow width, while S is positively related to the storage volume of each ''bucket''. At zero volume the distance equals zero and as the applied volume approaches infinity the function asymptotes to L_{max}, so L_{max} can be interpreted as the maximum plume distance for the applied flow rate. The parameters L_{max} and S were calculated for all the overland flow experiments and mean values for each field visit are presented as a timeseries since the fire.

Water was applied at the rate of 3 L s^{-1}, pumped from creeks and metered by a rotameter. The time for the flow to reach each downslope distance was used to calculate the total volume applied. The overland flow experiments were repeated at six time-steps between 2003 and 2006.

For a subset of these experiments, the width of the flow path was measured at 1 m intervals down the slope once L_{max} was reached, allowing the approximate K_{sat} value of the soil to be estimated. The implicit assumption in this approximation is that infiltration is dominated by gravity flow (hence the steady state infiltration rate approximates the K_{sat} value) and that saturated conditions exit across the full width of the measured flow path.

In addition to the rainfall simulation, ring infiltrometer, and overland flow experiments, a subset of the rill erosion simulation experiments (aimed at quantifying *erosion* by flow) were analysed in a way that provided an independent approximation of K_{sat}. These experiments are described in detail in the rill erosion simulation section. As a modification to these experiments, after the maximum flow rate had been applied, the flow rate was gradually reduced until no runoff occurred from the narrow, bounded plots. The K_{sat} value was then calculated from the water application rate and the plot area, assuming that there were saturated conditions over the entire plot area.

Results

Water repellence and soil water content

Fig. 2 shows the laboratory-derived relationship between soil GWC and soil water repellence, as indicated by the surface tension of drops of different ethanol solution concentrations that infiltrate in 5 s. Lower surface tensions indicate greater levels of water repellence. Results are shown for unburnt and burnt soils, during both the wetting and the drying phases. The water repellence categories shown on the second vertical axis are from Leighton-Boyce et al. (2005). The results indicate clear GWC thresholds at which water repellence changes over a large spectrum from wettable to severely repellent, suggesting that GWC *alone* may explain a large proportion of the temporal variation in water repellence for these soils. Some variability in the water content-repellency relationship over the wetting and drying cycles was observed. This may be due to hysteresis or to burn severity.

Rainfall simulation and runoff generation

Table 1 lists the plot conditions (slope, vegetation cover) and total runoff from each of the 44 rainfall simulation

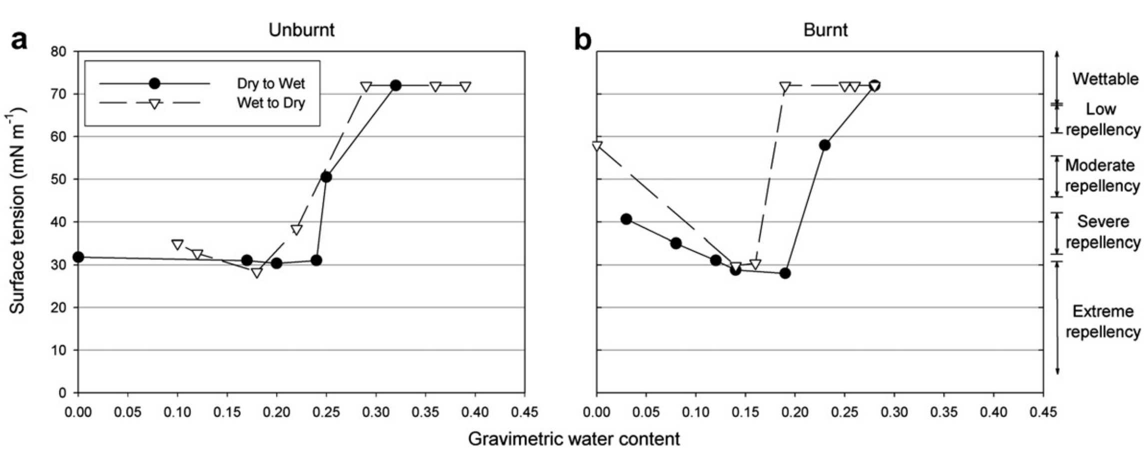


Figure 2 Variations of surface tension with soil moisture for an (a) unburnt, and (b) burnt soil from the experimental area. Water repellence categories are from Leighton-Boyce et al. (2005).

Table 1 Summary results for each of the rainfall simulation experiments at 100 mm h^{-1} for 30 min

Date	Unburnt							Burnt						
	Plot #	Cover[a] (%)	Slope (deg)	Runoff[b] (mm)	Runoff[b] ratio (%)	Sediment (g)	Slope adj.[c] sediment (g)	Plot #	Cover[a] (%)	Slope (deg)	Runoff[b] (mm)	Runoff[b] ratio (%)	Sediment (g)	Slope adj.[c] sediment (g)
2003														
Feb	U1	95	21	10	21	27	28	B1	25	27	14	28	302	267
	U2	98	16	14	29	58	77	B2	5	26	11	22	276	248
	U3	90	13	13	27	21	33	B3	10	23	27	54	1968	1916
Apr								B4	20	21	24	47	183	191
								B5	10	27	27	53	148	131
								B6	15	28	32	64	190	163
Aug	U4	100	25	0	0	0	0	B7	25	27	39	78	580	514
	U5	100	25	1	2	1	1	B8	75	25	36	71	384	352
								B9	20	27	23	46	1300	1151
2004														
Feb	U6	100	21	18	36	16	17	B10	50	27	19	38	68	59
	U7	95	16	30	60	43	57	B11	25	24	10	20	50	47
	U8	95	13	22	44	38	60	B12	55	28	32	64	122	105
	U9	93	25	20	40	5	4							
	U10	100	25	14	28	5	5							
Sept								B13	75	27	13	27	59	52
								B14	70	26	4	9	21	19
								B15	60	27	22	44	82	72
								B16	80	26	1	1	1	1
2005														
Mar	U11	100	21	7	13	1	1	B17	90	21	3	6	1	1
	U12	95	16	8	17	3	4	B18	40	25	24	47	53	48
	U13	98	13	4	9	4	6	B19	100	27	17	35	15	13
Sept	U14	100	21	7	14	1	1	B20	98	25	2	4	4	3
	U15	95	16	1	2	0	0	B21	90	27	2	4	4	3
	U16	95	13	1	1	3	5	B22	100	24	3	6	8	7
2006														
Feb	U17	100	21	17	33	2	2	B23	95	25	24	47	13	12
	U18	95	16	18	36	2	3	B24	95	26	33	67	11	10
	U19	95	13	23	47	22	34	B25	90	24	34	67	47	44

[a] Cover percentage includes all soil contact cover, including vegetation, sticks, litter, rocks.
[b] Runoff adjusted linearly for small variations in rainfall intensity.
[c] Sediment delivery adjusted to a 22° slope using Eq. (1).

experiments for the burnt and unburnt treatments over the 3 year study period. Typical burnt and unburnt plots are shown in Fig. 3. Rainfall simulation could not be conducted on April 2003 and September 2004 on the unburnt treatments due to heavy rainfall/snowfall preventing access. Total runoff volume Q_t for the 30 min duration 100 mm h^{-1} simulated rainfall event was calculated from

$$Q_t = \sum_{i=1}^{i=n} Qi\Delta t \tag{3}$$

where Q_i is the ith point-measurement of runoff rate and Δt is time period since the $i-1$th measurement of Q_i. Runoff is reported as the runoff ratio (i.e. the proportion of rainfall that becomes runoff), and was found to vary from a low mean value of 0.2% of rainfall (0.1 mm) from the unburnt plots in August 2003, to a mean of 65% (33 mm) from the burnt plots in August 2003.

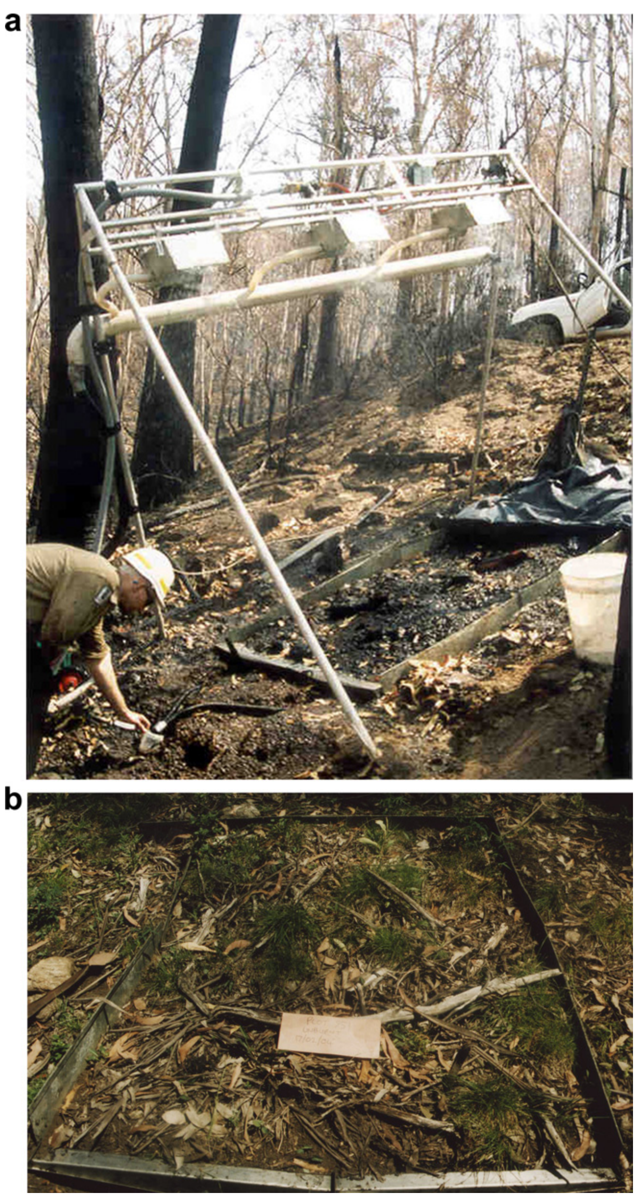


Figure 3 Rainfall simulation plots on a typical burnt plot immediately after the fire, and below, a typical unburnt plot.

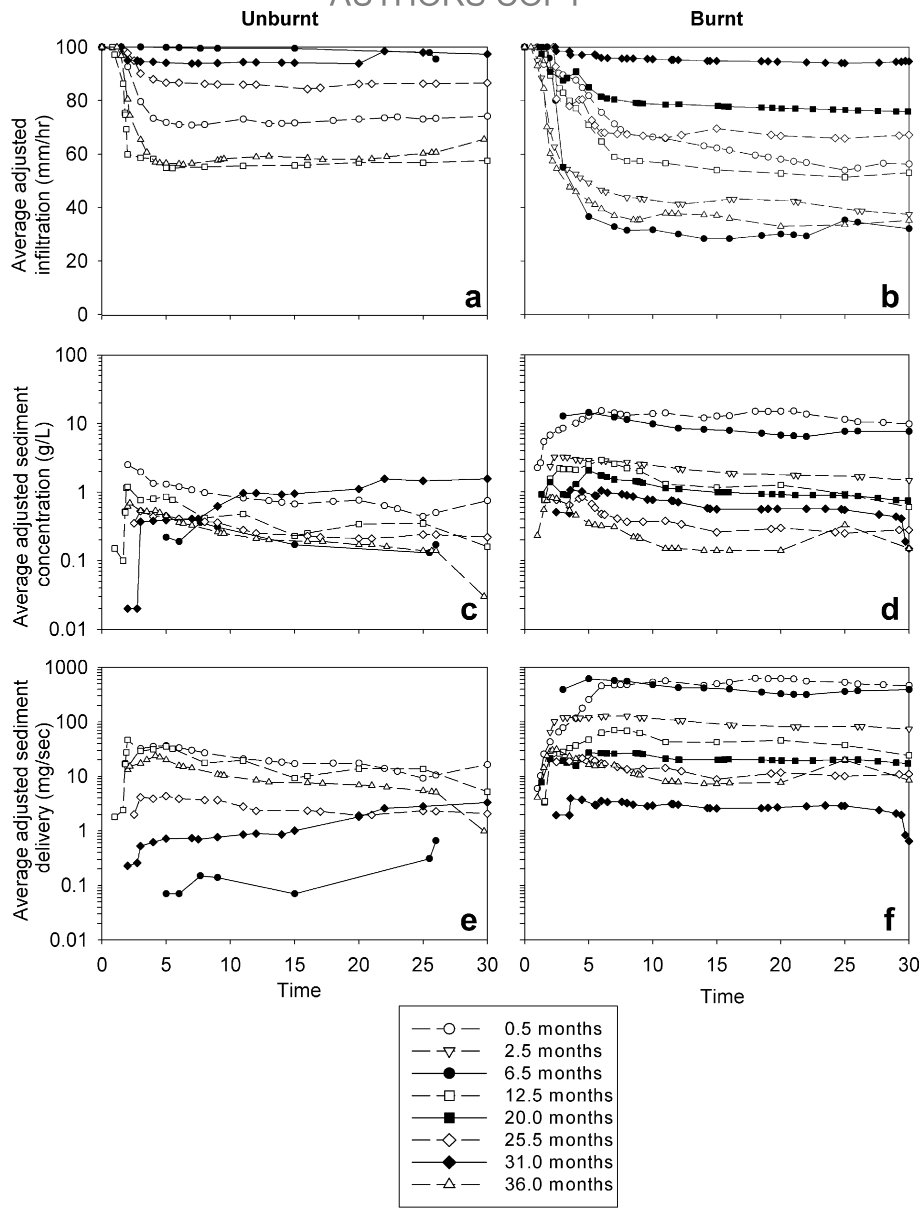
Fig. 4a and b shows the infiltration rate (mm h^{-1}) during the 100 mm h^{-1} simulated rainfall at each measurement time-step after the fire for the unburnt and burnt treatments. Each curve shows the average value for the 2–3 replicated plots listed in Table 1. Note that for clarity the variability between replicates is not shown in Fig. 4a and b. However an indication of variability is given in Table 1. The infiltration rate on the burnt plot at 0.5 months after the fire shows a steady decline over the 30 min simulation, contrasting with both the burnt plots at subsequent time-steps, and the unburnt plots at all time-steps where infiltration rates generally approach an approximately steady-state after about 10 min of rainfall.

Fig. 5a plots the temporal change in water repellence, antecedent GWC, and runoff ratio for the rainfall simulation experiments on the unburnt plots and provides a better illustration of temporal change in the runoff ratio over the 3 years of measurement. A seasonal oscillation in runoff ratio for the unburnt plots is clearly evident, with high runoff ratios of up to 42% recorded in summer/autumn, and low ratios approaching 0% in winter/spring. The seasonal oscillation in the runoff ratio is coincident with a seasonal oscillation in water repellence category, and also to a lesser extent with the antecedent GWC of the soil, also plotted in Fig. 5a.

The temporal pattern in the runoff ratio for the burnt plots (Fig. 5b) differs markedly from the unburnt plots (Fig. 5a). In particular, the runoff ratio of 65% in the first winter (i.e. when the soils are wetter) following the fire was clearly increased from the summer value of 35% immediately after the fire. In strong contrast to the unburnt sites, the strong water repellence developed in the summer prior to the fire is retained into the following winter for the burnt sites. However, from about 12 months post-fire onwards the seasonal pattern in the runoff ratio for the burnt sites appears to converge to a similar seasonal oscillation to the unburnt sites, with gradually increasing amplitude. At the end of 3 years, the seasonal oscillation in runoff ratio is of similar amplitude for both the burnt and unburnt sites.

Rainfall simulation and interrill erosion

Details of the plot slope (degrees), cover (%), total sediment delivery (unadjusted), total sediment delivery adjusted to 22° slope, and runoff ratio are given in Table 1. Soil contact cover for the 19 unburnt plots was always greater than 90%, and consisted of grass, rocks, and litter. Cover levels on the burnt treatments varied from an average of 13% immediately after the fire (consisting mostly of rocks and dead leaves) to levels >90% once contact cover vegetation recovered and forest floor litter accumulated. The average slope of the 25 burnt plots was 26° and of the 19 unburnt plots it was 18°. The average slope of all the plots was 22°, and the measured sediment delivery has been adjusted to this slope using Eq. (1) to enable direct comparison between plots and treatments (see last column in Table 1). Variation in runoff rates due to this range of slopes is assumed to be small and no slope adjustments have been made to runoff rates. The mean sediment concentration of the rainfall simulation supply water was 15 mg/L.



*Adjusted to 22% slope

Figure 4 Timeseries plots of (a, b) Infiltration rate (mm h^{-1}), (c, d) sediment concentration (g/L) and (e, f) sediment delivery rate (mg/s) under 100 mm h^{-1} simulated rainfall at each measurement time-step since the fire. Winter values are shown in solid symbols, summer and autumn values in open symbols.

Fig. 4c–f shows the change in sediment concentration (g/L) and sediment delivery rate (mg/s) *during* 100 mm h^{-1} simulated rainfall for 30 min for burnt and unburnt plots at each time-step after the fire. Each curve shows the average value for the replicated plots listed in Table 1. Sediment concentration and sediment delivery values have been adjusted for variation in slope between plots using Eq. (1) and are expressed as an equivalent rate at 22° slope.

The temporal change over the period of the study in the total event sediment concentration of runoff from the rainfall simulation plots is shown in Fig. 6a. Sediment concentration values adjusted to a 22° slope and averaged for each field visit for the rainfall simulation plots on *unburnt* plots were in the range 0.20–1.2 g L^{-1}. Sediment concentrations from the burnt plots were approximately 10 times greater than from the unburnt plots immediately after the fire. However one year later, after an increase in vegetation cover to about 43%, the burnt plots produced sediment concentrations that were only 2–3 times greater than the unburnt plots, and by 2 years the unburnt and burnt sites were producing similar low sediment concentrations, of 0.20 and 0.30 g L^{-1}, respectively. The relatively low sediment concentration of 2.0 g L^{-1} recorded in April 2003 at the burnt site may have been influenced by 80 mm of rain in the 4 days prior to the simulation run, which could have reduced available sediment levels for the simulated rainfall events.

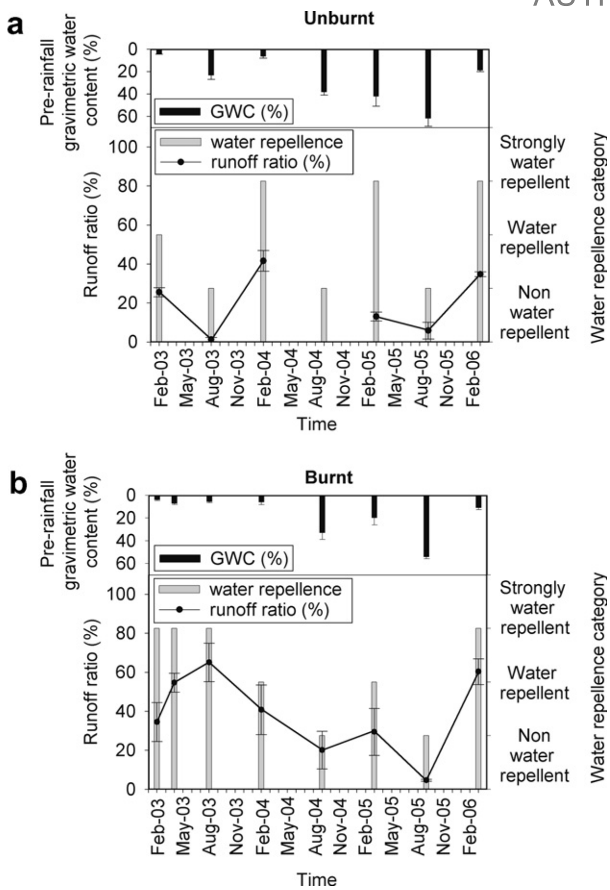


Figure 5 Temporal change in runoff ratio, water repellence category, and gravimetric water content of the soil for (a) unburnt, and (b) burnt rainfall simulation sites. Definitions of water repellence categories are given in the text. Error bars are ±1 standard error.

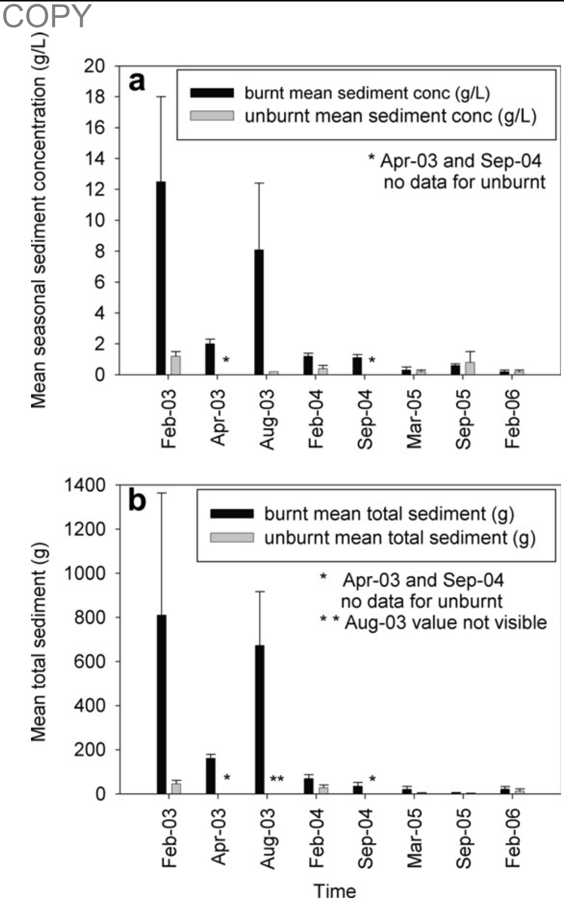


Figure 6 Temporal change in (a) mean sediment concentration and (b) mean total sediment load from 100 mm h^{-1} rainfall simulation for 30 min, adjusted to 22° slope. Error bars are 1 standard error.

The temporal change in the total sediment delivery (g) from the 30 min 100 mm h^{-1} rainfall event is shown in Fig. 6b. The total sediment delivery value integrates the effect of temporal changes sediment concentration and temporal changes in the runoff ratio during the rainfall event. Immediately following the fire the total sediment delivery from the burnt sites is 18 times greater than from the unburnt sites, due to an average 10-fold increase in sediment concentration, and an average 1.5-fold increase in the runoff ratio.

However by winter the runoff ratio from the unburnt sites decreases dramatically from 26% to only 1.3%, while for the burnt sites the runoff ratio increases from 35% to 65% over this period. At the same time the sediment concentration reduces to 0.2 g/L at the unburnt sites, while at the burnt sites increasing the cover from 13% to 40% results in sediment concentrations remaining high at 8.0 g L. In combination, these changes in hydrology and sediment concentration yield a total sediment delivery that is 2240 times higher from the burnt sites compared to the unburnt sites in the winter following the fire (Fig. 7). Note that the very low sediment load in August 2003 cannot be seen in Fig. 6b. Only six months later in the summer one year after the fire, cover levels, runoff ratios, and sediment concen-

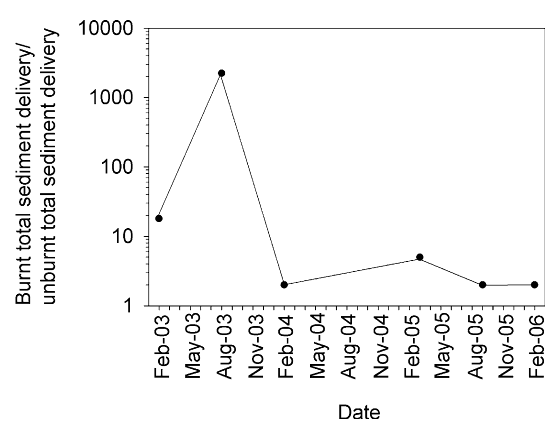


Figure 7 Change since the fire in the ratio of burnt to unburnt total sediment delivery, showing the large jump in this ratio in the first winter following the fire, and the return to a value close to 1 after 3 years (note the log scale on the vertical axis).

trations have again changed so that total sediment delivery rates are only 2.5 times greater in the burnt compared to the unburnt sites. Ratios between burnt and unburnt sites approach 1 towards the end of the monitoring period (Fig. 7).

Rill erosion

The results from the rill erosion simulation experiments in 2003, 2004, and 2005, at two plot slopes (13° and 27°), and two cover levels (<20% and 90%), are shown in Fig. 8. Each point on the graph is the mean of three sediment delivery measurements. Note that in 2003 there is only one cover level (zero) as the plots were completely burnt. Erosion rates were observed to decline rapidly during the application of a given flow rate as available sediment was removed, so the values shown in Fig. 8 should not be interpreted as absolute rates of sediment generation that would be sustained continuously at a given flow rate. Rather, the values provide relative comparisons using a consistent methodology between slopes, cover levels, and years. The rapidly declining rates of sediment production during a given flow rate (not shown) indicate that these forest soils become highly resistant to rill erosion at relatively shallow depths.

The most striking feature of Fig. 8 is the large decline in rill erodibility in the first year after the fire for both levels of land slope and cover. Rill erodibility is defined as the rate of change of sediment delivery ($g\,s^{-1}$) with discharge rate ($L\,s^{-1}$), and is therefore expressed as a concentration ($g\,L^{-1}$). One year after the fire rill erodibility is in the range 1/12th to 1/28th of the values immediately after the fire, depending on cover and plot slope. Two years after the fire the range is 1/36th to 1/540th, clearly a very large reduction.

Rill erodibility values for this undisturbed/unburnt forest are not known, though it is assumed the high-cover plots in 2005 are similar to this forest unburnt. The absolute values of rill erodibility recorded in 2005 suggest that runoff generated in unburnt forest is likely to result in very low erosion rates. It is important to note that these rill erodibility results do not indicate actual erosion rates by rill erosion in the forest. Rather, the results quantify the change in the susceptibility of the soil so rill erosion. Whether or not rill erosion actually occurs depends on the generation and concentration of overland flow during storm events.

The similarity in the flow rate vs erosion rate relationship at both levels of cover suggests the large decline in the rill erosion rates between 2003 and 2004 is largely attributable to changes in soil erodibility rather than vegetation recovery. The high erodibility immediately after the fire is probably attributable to the presence of a highly erodible ash layer on the soil surface, combined with dry, water-repellent, strongly aggregated loose soil on the soil surface.

Saturated hydraulic conductivity

Ring infiltrometer

Measured K_{sat} values from the 45 ring infiltrometer measurements are summarized in Table 2. The mean value for all the K_{sat} measurements was 886 mm h^{-1} (SD = 826 mm h^{-1}), indicating that K_{sat} is extremely high on average, and is also highly variable spatially and temporally. Mean infiltration rates were highest for unburnt sites in winter, but no significant differences (t-test, $p < 0.05$) were found between seasons (summer vs winter) and treatments (burnt vs unburnt). The mean adjustment factor (Reynolds, 1993) using this approach was 0.71 (SD = 0.07).

Overland flow experiments

A total of 41 overland flow experiments were conducted in burnt areas at six time-steps over the period of the study. The mean value at each time-step (i.e. the average value measured at each field trip) of the parameter L_{max} in Eq. (2) is plotted as a timeseries in Fig. 9. The mean L_{max} value of 18 m recorded immediately after the fire increased to about 21 m in the winter following the fire, indicating a reduction in the infiltration capacity of the soil. This pattern of runoff increase is consistent with the runoff data from the rainfall simulation on the burnt treatments shown in Fig. 5, as is the subsequent decrease in runoff also shown in Fig. 5.

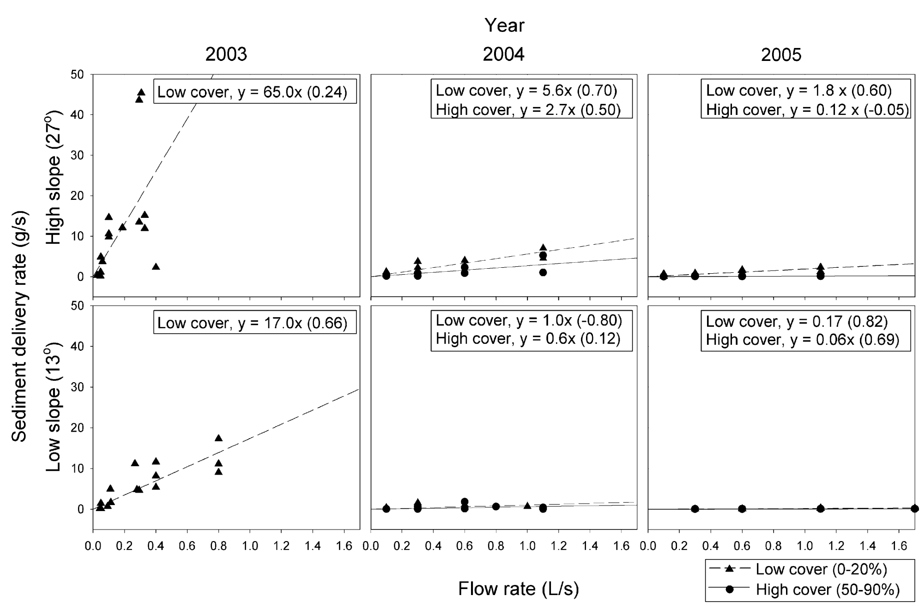


Figure 8 Sediment delivery as a function of the rate of concentrated overland flow for 3 years, 2 cover levels and 2 slopes. Regression coefficient (g/L) and coefficient of determination (in brackets) for the linear models shown in each graph. Note that the models have been forced through zero, and there is no high-cover category immediately after the fire in 2003.

Table 2 K_{sat} values measured using the single ring infiltrometer

Treatment	Summer	Winter	All
Unburnt	490 (305, 12)	1409 (1008, 15)	1001 (896, 27)
Burnt	853 (819, 12)	439 (198, 6)	715 (697, 18)
All	671 (632, 24)	1132 (960, 21)	886 (826, 45)

Values in parenthesis are the standard deviation and the number of replicates, respectively.

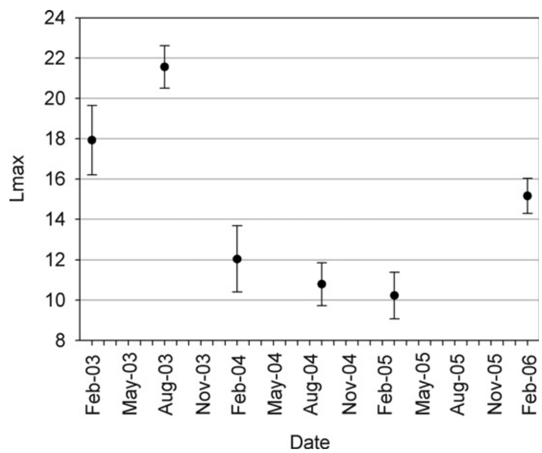


Figure 9 The temporal change in the value of the infiltration related parameter L_{max} in Eq. (2). Error bars are ±1 standard error.

The range of values for L_{max} suggest that the *infiltration capacity* varies by a factor of 2 over the period of the study, a similar seasonal factor-change to that shown for the ring infiltrometer measurements. The magnitude of this change contrasts with the 12-fold factor change in the runoff ratio after the fire for rainfall simulation shown in Fig. 5.

The model given by Eq. (2) fits the data very well in most cases, with a mean coefficient of determination value of 0.94 for the 41 overland flow experiments conducted on burnt sites. The average, minimum, and maximum values for the parameter L_{max} were 13.5, 5.5 and 25 and for S they were 3690, 206 and 19,260, respectively.

Converting L_{max} to a soil K_{sat} value was not generally possible as the width of the flow path was not known. However, for two of the latter field visits the widths of the flow paths during the overland flow experiments were measured, yielding a mean K_{sat} value from ten measurements in February 2005 of 1200 mm h^{-1} (SD 500) and from five measurements in February 2006 of 1100 mm h^{-1} (SD 200), respectively. These approximate measurements suggest that the infiltration capacity is in the order of 1000 mm h^{-1}. This infiltration value is 10–20 times higher than the *apparent infiltration rate* calculated from the difference between rainfall and runoff during the rainfall simulation experiments.

Discussion

Post-fire erosion processes

A general reading of the literature suggests widespread overland flow erosion is a common feature of the post-fire landscape (Leitch et al., 1983; Shakesby et al., 2003; Wondzell and King, 2003). This process is often attributed to an increase in runoff generation due to water repellence, an increase in erodibility, and the loss of soil cover (DeBano, 2000b). These changes were all also measured in this study – rill erodibility increased 540-fold (Fig. 8), water repellence changed from repellent to strongly repellent (Fig. 5), and cover reduced from >90% to <15% (Table 1). Despite these changes, widespread rill erosion was not observed in this catchment after the fire, for the following reasons. The ring infiltrometer measurements showed that the soil K_{sat} remained extremely high, in the order of 100s to 1000s of mm h^{-1}. The overland flow experiments showed that as a result of this high K_{sat}, high concentrated flow volumes (3 L s^{-1}) could have infiltrated within relatively short distances, suggesting that under most rainfall conditions overland flows are likely to be transient, infiltrating into areas of high K_{sat} downlsope.

Despite the very high K_{sat} values measured using the ring infiltrometer, runoff ratios from 100 mm h^{-1} simulated rainfall as high as 65% were recorded. These seemingly contradictory results between K_{sat} and the apparent infiltration rate can only be reconciled by adopting a conceptual model involving K_{sat} values that are extremely spatially variable (Sharma et al., 1980). The value in considering K_{sat} as a variable spatial field has become increasingly clear (Hawkins, 1982; Hawkins and Cundy, 1987). However in the case of this study this conceptual approach is essential.

The important water quality conclusions from the above results are that following fire, most of the infiltration-excess overland flow reaching streams in this catchment are produced from within several metres of the stream edge, and that the transported sediment is generated by interrill processes. As a consequence, the area burnt near streams is the critical factor affecting water quality impacts in these catchments, rather than the total area of the catchment that is burnt. This conclusion suggests that, conceptually, vegetated buffer strips are not areas trapping sediment from upslope, but rather these areas are the primary source of sediment and pollutants to streams.

Signs of interrill erosion in the burnt catchment were obvious. Soil pedestals were evident beneath small stones, and the collection of deposited material in small depressions was observed. The signs of accelerated interrill erosion were consistent with the rainfall simulation results. The 18-fold increase in sediment delivery rate under simulated rainfall following the fire is attributed to an on average 10-fold increase sediment concentration and an average 1.5-fold increase in runoff generation. The observed increase in sediment concentration is mostly attributed to the loss of soil cover, and to a lesser extent to changes in the soil erodibility. For example, the interrill cover adjustment function in the WEPP (NSERL, 1995) hillslope erosion model predicts a 10-fold increase in the erosion rate as cover is reduced from 92% to 0%; for the USLE model (Wischmeier and Smith, 1978) a 10-fold increase is predicted as cover is reduced from 80% to 0%. If it is assumed that cover accounts for about 10-fold of the net 18-fold increase following the fire, then the erodible ash layer must account for the remaining, approximately 2-fold, increase. It follows that the rapid recovery of surface cover in these highly productive wet forests is the dominant factor in the rapid decline in interrill erosion rates following fire.

The 18-fold increase in plot-scale sediment delivery found in this study can be compared to other plot scale erosion studies where factor increases of between 2- and 1000-fold are reported. These studies include Ronan (1986) 2–3-fold increase; Williams et al. (1973) and Pierson et al. (2001) 4-fold; Johansen et al. (2001) 25-fold; Benavides-Solorio and MacDonald (2001) 10–26-fold; DeBano et al. (1979) 34-fold; Prosser and Williams (1998) 70-fold; Moody and Martin (2001) 150–240-fold. Shakesby et al. (1993) reported splash erosion rates 100 times greater and plot scale erosion rates 1000 times greater than from burnt compared to unburnt areas. Morris and Moses (1987) estimated early post-fire increases in sediment movement of up to 1000-fold. The large range in the amount of increase in sediment delivery after fire partially reflect the fact that soil erodibility, rainfall energy, and soil cover do not act independently on sediment delivery. The amount of increase may also be highly dependent on the timing of the erosion measurement. For example, if a single ''post-fire'' measurement had been made in this study immediately after the fire then the factor increase in sediment load would have been reported as 18-fold. If a single measurement had been made 6 months later (still with very little vegetation recovery) then the factor increase due to the fire would have been reported as 2240-fold.

While higher runoff ratios were recorded on the burnt sites immediately following the fires, the difference between burnt and unburnt ($14\ \mathrm{mm\ h^{-1}}$) was far less than found in many other studies. It seems that in this study the burn has only moderately increased the water repellence that was already present prior to the burn. These results are consistent with the findings of Leitch et al. (1983) in a similar forest soil. Doerr et al. (2004) also found that fire did not enhance natural water repellence in *Eucalyptus* forest, and pointed out that it is often assumed, but rarely demonstrated, that water repellence noted after a fire has actually been induced as a result of the burn. Varela et al. (2005) also found that while fire increased the water repellence of non-water-repellent soils, for soils that were already water-repellent the fire caused little change in repellence. Reasons for the continued increase in the runoff ratio into the following winter (Fig. 5) are unclear and may include: (a) a drier soil profile at the burnt site (6% GWC) than the unburnt site (23% GWC) because of increased soil exposure and subsequent evaporation; (b) a continued increase in water repellence due to the fire; and/or (c) the washing in of ash into the soil pores, reducing infiltration over time. Only the first of these possible explanations can be evaluated in this study because the field water repellence tests do not have the resolution to quantify further increases in water repellence greater than ''strongly-repellent'', while the role of ash was not investigated as a separate treatment.

Pre-fire erosion processes

Measurements in unburnt areas, intended for comparative purposes only, provided unexpected insights in their own right. Unburnt areas showed a large natural seasonal oscillation in water repellence (non-repellent in winter to strongly WR in summer) and runoff generation (2–5% in winter to 42% in summer) as the soil gravimetric water content crossed a threshold. Laboratory experiments (Fig. 2) suggest this threshold may be between 15% and 30% GWC, but these values should be treated with some caution as the laboratory conditions may not replicate natural processes (see Section ''Water repellence and soil water content''). It may be the threshold value is greater or less than the range given here. More certainty in this threshold may be required for model parameterisation. The strong natural WR found in this study has been noted previously in SE Australian *Eucalyptus* forests by Crockford et al. (1991), Burch et al. (1989), Prosser and Williams (1998) and Doerr et al. (2006). Seasonal oscillations in runoff ratio of a similar magnitude have been reported from natural rainfall plots (in the $600–1000\ \mathrm{mm\ y^{-1}}$ zone) on similar soils by Costin et al. (1960), Gilmour (1968) and Ronan (1986), suggesting that 10-fold increases in the runoff ratio in the summer months may be common in finer textured *Eucalyptus* forest soils. Considering these results alongside the larger body of literature relating to sandy *Eucalyptus* forest soils (Shakesby et al., 2007), it can be speculated that seasonal WR may be common in a wide variety of such soils.

The absolute value of the runoff ratios measured in this study (during $100\ \mathrm{mm\ h^{-1}}$ simulated rainfall) varied between 2% and 40%. Croke et al. (1999) reported runoff ratios of 12% under similar simulated rainfall conditions, while values in the range 0–3% have been previously reported for SE Australian forest soils under natural rainfall (Costin et al., 1960; Craig, 1968; Williams et al., 1973; Bren and Turner, 1979; Ronan, 1986). These low runoff ratios result from very high K_{sat} values. Values in the range $500–1400\ \mathrm{mm\ h^{-1}}$ were measured in this study using a range of techniques. K_{sat} values reported for other well structured clay-loam soils with abundant macropores in wet *Eucalyptus* forests are of a similar magnitude (Talsma and Hallam, 1980; Huang et al., 1996; Davis et al., 1999; Lane et al., 2004). Numerous other studies have reported considerably lower K_{sat} values, but results by Topp and Binns (1976) and Davis et al. (1999) suggest many of these may be underestimates due to the sensitivity of the K_{sat} measurements to the measurement method used.

The very low erosion rates (0.2–1.2 g/L) measured from the unburnt rainfall simulator plots were expected in this study because soils in wet SE Australian *Eucalyptus* forests tend to have high aggregate stability, low inherent erodibility (Sheridan et al., 2001; McKenzie et al., 2004) and high contact cover levels.

These results can be compared with measured factor increases in sediment exports from the East Kiewa Research Catchments of 8–9-fold in the first year, and 2–4-fold in the second year after the fire (Lane et al., 2006a). However direct comparison of the hillslope erosion data with catchment export data is difficult as uncertainty remains with respect to the contribution of other erosion processes to catchment scale exports of sediment. For example, *saturation-excess overland flow* may also be generated in these near stream areas, although studies in other Australian forests in steep landscapes (e.g. Lane et al., 2004) suggest that these areas are probably only narrow in width due to the steep landscape and the high lateral hydraulic conductivity of the soils. However, this assumption is untested in this catchment, and further experiments are currently being developed to quantify the degree of near-stream saturation-excess overland flow occurring in the East Kiewa catchments. Sediment may also be generated (re-mobilised) from within the stream channel due to decreased evapotranspiration and subsequent increased total flows and peak-flows. Increases in annual total flows of between 40% and 94% were recorded in the two experimental catchments in the two years following the fire (Lane et al., 2006a). Mass-movement processes such as slumps and debris-slides were observed in some other fire-affected catchments, but were not observed in these research catchments.

A conceptual model of forest erosion

The results from the suite of experimental procedures described in this paper have led to a conceptual model of post-fire erosion that differs considerably from our expectations prior to this study. There was no evidence of debris-slides or other mass-movement processes following the fire. Our expectations of flow-driven erosion processes, affecting much of the catchment have been replaced by a conceptual model where only the near-stream areas deliver sediment to streams, from interrill rather than rill processes. Given the very high K_{sat} values observed in these environments, we propose that this model would hold for the wet eucalypt forests of SE Australia under most rainfall magnitude–frequency scenarios. The response to high magnitude–low frequency events would be an expansion of the near-stream sediment delivery area. This can be conceptualised as an infiltration-excess variable source area.

Measurements of 50% runoff ratio under 100 mm h^{-1} rainfall, at the same time as measured K_{sat} values are in the order of 100s to 1000s of mm h^{-1}, indicate that most of the common modelling approaches to predicting runoff and erosion (e.g. Green and Ampt, 1911; NSERL, 1995) are inappropriate for these forests both in the burnt and unburnt conditions. This is because the dominant driving processes and soil properties are not represented. Most importantly, the prediction of runoff generation via point-scale conceptual models of infiltration (e.g. Green and Ampt, 1911) without any consideration of the spatial variability of soil hydraulic properties, will lead to non-sensical results. This is clearly seen by comparing the runoff ratios in Fig. 5 with the K_{sat} values listed in Table 2.

Analytical modelling solutions to this problem proposed by Hawkins (1982) and Hawkins and Cundy (1987) have been shown to work effectively in agricultural areas at the plot scale by Yu et al. (1997) and Yu (1999). Yu et al. (1998) has shown that, given some reasonable assumptions, common empirical catchment scale runoff estimation procedures such as the SCS curve number method (SCS, 1985) implicitly represent K_{sat} as a spatially variable parameter. Recent improvements (Morbidelli et al., 2007) to earlier numerical methods at the plot scale (Woolhiser et al., 1996; Smith and Goodrich, 2000; Govindaraju et al., 2006) also show considerable promise. The common conceptual theme behind all of these approaches is the representation of, at least, the *effects* of spatial variability of soil properties on runoff generation. These approaches are also likely to lead to improved scaling of hydrological results, a critical issue in hydrology (Grayson and Bloschl, 2000; Doerr and Moody, 2004).

Failure to represent the temporal variability of K_{sat} due to water repellence will also lead to results that are incorrect, not only in terms of absolute *magnitude*, but also *direction* of change on a seasonal basis. The results also suggest that the washing-in of ash may play an important role in the temporal variation in K_{sat} and further work is required to quantify the effect of this process.

The prediction of rill erosion as an increasing function of either flow rate, shear stress, or stream power, will grossly overestimate erosion rates in this environment because the soil erodibility reduces dramatically with depth. In the study catchment after the fire the soil can be conceptually considered as consisting of two layers; a highly erodible shallow layer of loose soil aggregates and ash, overlying a soil matrix of extremely low erodibility. This arrangement is not commonly represented in erosion models which have been predominantly developed for tilled agricultural soils with homogeneous erodibility over the depth of interest. In contrast with the above, current approaches to the prediction of interrill erosion as a function of rainfall energy and runoff volume (Kinnell, 1997) appear to be reasonable in the post-fire environment.

Conclusion

The suite of experimental methods utilized in this study enabled the dominant post-fire erosion processes to be identified and quantified for a wet *Eucalyptus* forest. The first key finding from this study is that following fire, hillslope erosion in these catchments is generally localized, and dominated by interrill processes. This finding will have significant implications for those attempting to estimate the likely impact of fire on water quality because the source areas for sediment are restricted to the streamside zone. These results imply that the condition (i.e. unburnt, severely burnt, etc.) of most of the catchment is probably of little consequence with respect to catchment exports of sediment, because these areas remain disconnected from the stream system during most rainfall events.

Secondly, the study clearly shows that the erosion risk (relative to the background level) in the first winter following the fire is orders of magnitude greater than both before and after this season. One year after the fire, the erosion risk is only 2–3 times higher than the background level. These results provide some guidance to water authorities managing risks to water quality following wildfire in water supply catchments.

Lastly, the results indicate that many of the current conceptual models underlying runoff and erosion prediction are inappropriate for both the burnt and unburnt forest because the dominant driving processes and soil properties are not represented. Common point-scale infiltration modelling approaches (e.g. Green and Ampt, 1911) will lead to results that are incorrect in magnitude, but also in direction on a seasonal basis. Realistic conceptual models of runoff and erosion are required, in particular that will incorporate the effects of spatial and temporal variability of K_{sat}.

Future work will attempt develop better conceptual models of runoff generation and erosion that will link the processes and properties quantified in this paper, with the catchment scale exports reported by Lane et al. (2006a).

Acknowledgments

This study was jointly funded by Land and Water Australia project DSE1, the Victorian Department of Sustainability and Environment Bushfire Recovery Program and the Victorian Department of Sustainability and Environment Land Stewardship and Biodiversity Division. The authors thank John Costenaro, Gabi Szegedy, Phill McKenna and Chris Sherwin for their tireless efforts in data collection and analysis over many years. Thanks to David Fox for assistance with water repellence assessment of soils. Thanks also to the anonymous referees whose comments improved the paper.

References

Benavides-Solorio, J., MacDonald, L.H., 2001. Post-fire runoff and erosion from simulated rainfall on small plots, Colorado Front Range. Hydrological Processes 15, 2931–2952.

Blake, W.H., Droppo, I.G., Humphreys, G.S., Doerr, S.H., Shakesby, R.A., Wallbrink, P.J., 2007. Structural characteristics and behavior of fire-modified soil aggregates. Journal of Geophysical Research 112, F02020. doi:10.1029/2006JF000660.

Bond, R.D., Harris, J.R., 1964. The influence of microflora on physical properties of soils I effects associated with filamentous algae and fungi. Australian Journal of Soil Research 2, 111–122.

Bren, L.J., Turner, A.K., 1979. Overland flow on a steep, forested infiltrating slope. Australian Journal of Soil Science 30, 43–52.

Brown, J.A.H., 1972. Hydrologic effects of a bushfire in a catchment in South-Eastern New South Wales. Journal of Hydrology 15, 77–96.

Bubenzer, G.D., Meyer, L.D., 1965. Simulation of rainfall and soils for laboratory research. Transactions of the ASAE 17, 1033–1037.

Burch, G.J., Moore, I.D., Burns, J., 1989. Some water repellent effects on infiltration and catchment runoff. Hydrological Processes 3, 211–222.

Burden, R.L., Faires, D.J., 2001. Numerical analysis. Wadsworth Group, Pacific Grove, California, United States of America.

Butler, J.A.V., Wightman, A., 1932. Adsorption at the surface of solutions. Part I. The surface composition of water–alcohol solutions. Journal of the Chemical Society 293, 2089–2097.

Cannon, S.H., Bigio, E.R., Mine, E., 2001. A process for fire-related debris flow initiation, Cerro Grande fire, New Mexico. Hydrological Processes 15, 3011–3023.

Cerda, A., 1998. Changes in overland flow and infiltration after a rangeland fire in a Mediterranean scrubland. Hydrological Processes 12, 1031–1042.

Connolly, R.D., Silburn, D.M., Glanville, S.G., Bridge, B.J., 2002. Using rainfall simulators to derive soil hydraulic parameters. In: McKenzie, N.J., Coughlan, K.J., Cresswell, H.P. (Eds.), Soil Physical Measurement and Interpretation for Land Evaluation. CSIRO Pub, Collingwood, Victoria, Australia, pp. 163–176.

Costin, A.B., Wimbush, D.J.,Kerr, D., 1960. Studies in catchment hydrology in the Australian Alps. II Surface run-off and soil loss. Division of Plant Industry Technical Paper No. 14. Commonwealth Scientific and Industrial Research Organisation, Melbourne, Australia.

Craig, F.G., 1968. Effects of fire and heat on the physical properties of soils with particular reference to clay minerals. PhD Thesis, School of Forestry, The University of Melbourne, Australia.

Crockford, R.H., Richardson, D.P., 2000. Partitioning of rainfall into throughfall, stemflow and interception: effect of forest type, ground cover and climate. Hydrological Processes 14, 2903–2920.

Crockford, H., Topalidis, S., Richardson, D.P., 1991. Water repellency in a dry sclerophyll eucalypt forest – measurements and processes. Hydrological Processes 5, 405–420.

Croke, J., Hairsine, P., Fogarty, P., 1999. Runoff generation and redistribution in logged *Eucalyptus* forests, south-eastern Australia. Journal of Hydrology 216, 56–77.

Davis, S.H., Vertessy, R.A., Silberstein, R.P., 1999. The sensitivity of a catchment model to soil hydraulic properties obtained by using different measurement techniques. Hydrological Processes 13, 677–688.

DeBano, L.F., 1981. Water repellent soils: a state-of-the-art. United States Department of Agriculture Forestry Service General Technical Report, PSW-46. Berkeley CA, 21pp.

DeBano, L.F., 2000a. Water repellency in soils: a historical overview. Journal of Hydrology 231–232, 4–32.

DeBano, L.F., 2000b. The role of fire and soil heating on water repellency in wildland environments: a review. Journal of Hydrology 231–232, 195–206.

DeBano, L.F., Rice, R.M., Conrad, C.E., 1979. Soil heating in chaparral fires: effects on soil properties, plant nutrients, erosion, and runoff. Research Papers PSW vol. 145. United States Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experimental Station.

Doerr, S.H., Moody, J.A., 2004. Hydrological effects of soil water repellency: on spatial and temporal uncertainties. Hydrological Processes 18, 829–832.

Doerr, S.H., Thomas, A.D., 2000. The role of soil moisture in controlling water repellency: new evidence from forest soils in Portugal. Journal of Hydrology 231–232, 134–147.

Doerr, S.H., Shakesby, R.A., Walsh, R.P.D., 1998. Spatial variation of soil hydrophobicity in fire-prone *Eucalyptus* and pine forests, Portugal. Soil Science 163, 313–324.

Doerr, S.H., Ferreira, A.J.D., Walsh, R.P.D., Shakesby, R.A., Leighton-Boyce, G., Coelho, C.O.A., 2003. Soil water repellency as a potential parameter in rainfall–runoff modelling: experimental evidence at point to catchment scales from Portugal. Hydrological Processes 17, 363–377.

Doerr, S.H., Blake, W.H., Shakesby, R.A., Stagnitti, F., Vuurnes, S.H., Humphreys, G.S., Wallbrink, P.J., 2004. Heating effects on water repellency in Australian eucalypt forest soils and their value in estimating wildfire soil temperatures. International Journal of Wildland Fire 13, 157–163.

Doerr, S.H., Shakesby, R.A., Blake, W.H., Chafer, C.J., Humphreys, G.S., Wallbrink, P.J., 2006. Effects of different wildfire severities on soil wettability and implications for hydrologic response. Journal of Hydrology 319, 295–311.

Dragovich, D., Morris, R., 2002. Fire intensity, slopewash and biotransfer of sediment in eucalypt forest, Australia. Earth Surface Processes and Landforms 27, 1309–1319.

Ferreira, A.J.D., 1996. Processors hidrologicos e hidroquimicos em povoamentos de *Eucalyptus* globulus Labill. e Pinus pinaster Aiton. PhD thesis, Universidade de Aveiro, Portugal.

Gilmour, D.A., 1968. Water repellence of soils related to surface dryness. Australian Forestry 32, 143—148.

Giovannini, G., Lucchesi, S., 1983. Effect of fire on hydrophobic and cementing substances of soil aggregates. Soil Science 136, 231—236.

Govindaraju, R.S., Corradini, C., Morbidelli, R., 2006. A semi-analytical model of expected areal-average infiltration under spatial heterogeneity of rainfall and soil saturated hydraulic conductivity. Journal of Hydrology 316, 184—194.

Grayson, R., Bloschl, G., 2000. Spatial patterns in catchment hydrology: observations and modelling. Cambridge University Press, Cambridge, United Kingdom.

Green, W.H., Ampt, G.A., 1911. Studies on soil physics Part 1 — The flow of air and water through soils. Journal of Agricultural Science 4, 1—24.

Hairsine, P.B., Croke, J.C., Mathews, H., Fogarty, P., Mockler, S.P., 2002. Modelling plumes of overland flow from roads and tracks in forestry environments. Hydrological Processes 16, 2311—2327.

Harris, R.F., Chester, G., Allen, O.N., 1964. Mechanisms involved in soil aggregate stabilization by fungi and bacteria. Soil Science Society of America Proceedings 28, 529—532.

Hawkins, R.H., 1982. Interpretation of source area variability in rainfall—runoff relationships. In: Rainfall Runoff Relationships. V.P. Singh Water Resources Publications, Fort Collins, Colorado, pp. 303—324.

Hawkins, R.H., Cundy, T.W., 1987. Steady-state analysis of infiltration and overland flow for spatially varied hillslopes. Water Resources Bulletin 23 (2), 251—256.

Helvey, J.D., 1980. Effects of a north central Washington wildfire on runoff and sediment production. Water Resources Bulletin 16, 627—634.

Hough, D.J., 1983. Effects of Alpine Ash logging on stream sediment levels in the East Kiewa River catchment. 3. Soils of the hydrological project area. Forest Commission Victoria, Research. Branch Report 220: 14 (unpublished).

Huang, J., Lacey, S.T., Ryan, P.J., 1996. Changes in soil hydraulic properties due to forest harvesting in two Pinus radiata plantations and three *Eucalyptus* forests. Soil Science 161, 79—86.

Imeson, A.C., Verstraten, J.M., van Mulligen, E.J., Sevink, J., 1992. The effects of fire and water repellency on infiltration and runoff under Mediterranean type forest. Catena 19, 345—361.

Isbell, R.F., 1996. The Australian Soil Classification. CSIRO Publishing, Victoria, Australia, p. 143.

Johansen, M.P., Hakonson, T.E., Breshears, D.D., 2001. Post-fire runoff and erosion from rainfall simulation: contrasting forests with shrublands and grasslands. Hydrological Processes 15, 2953—2965.

King, P.M., 1981. Comparison of methods for measuring severity of water repellence of sandy soils and assessment of some factors that affect its measurement. Australian Journal of Soil Research 19, 275—285.

Kinnell, P.I.A., 1997. Runoff ratio as a factor in the empirical modelling of soil erosion by individual rainstorms. Australian Journal of Soil Research 35, 1—13.

Kuczera, G.A., 1987. Prediction of water yield reduction following a bushfire in ash-mixed species eucalypt forest. Journal of Hydrology 94, 215—236.

Laing, A.C.M., 1981. Geology of East Kiewa hydrological project area. Geological Survey of Victoria, unpublished report.

Lane, P.N.J., Croke, J.C., Dignan, P., 2004. Runoff generation from logged and burnt convergent hillslopes: rainfall simulation and modelling. Hydrological Processes 18, 879—982.

Lane, P.N.J., Sheridan, G.J., Noske, P.J., 2006a. Changes in sediment loads and discharge from small mountain catchments following wildfire in south eastern Australia. Journal of Hydrology 331, 495—510.

Lane, P.N.J., Hairsine, P.B., Croke, J.C., Takken, I., 2006b. Quantifying diffuse pathways for overland flow between the roads and streams of the Mountain Ash forests of Central Victoria Australia. Hydrological Processes 20, 1875—1884.

Langford, K.J., 1976. Change in yield of water following a bushfire in a forest of *Eucalyptus* regnans. Journal of Hydrology 29, 87—114.

Leighton-Boyce, G., Doerr, S.H., Shakesby, R.A., Walsh, R.P.D., Ferreira, A.J.D., Boulet, A., Coelho, C.O.A., 2005. Temporal dynamics of water repellency and soil moisture in eucalypt plantations, Portugal. Australian Journal of Soil Research 43, 269—280.

Leitch, C.J., 1979. Effects of Alpine Ash logging on stream sediment levels in the East Kiewa River catchment. 1. First Progress Report. For. Comm. Victoria, Res. Branch Rep. 148, 19 (unpubl).

Leitch, C.J., Flinn, D.W., van de Graaff, R.H.M., 1983. Erosion and nutrient loss resulting from Ash Wednesday (February 1983) wildfires: a case study. Australian Forestry 46, 173—180.

Loch, R.J., 1989. Aggregate breakdown under rain: its measurement and interpretation. PhD Thesis, University of New England, Armidale, NSW.

Loch, R.J., 2000. Effects of vegetation cover on runoff and erosion under simulated rain and overland flow on a rehabilitated site on the Meandu Mine, Tarong, Queensland. Australian Journal of Soil Research 38, 299—312.

Lu, H., Prosser, I.P., Moran, C.J., Gallant, J.C., Priestley, G., Stevenson, J.G., 2003. Predicting sheetwash and rill erosion over the Australian continent. Australian Journal of Soil Research 41, 1037—1062.

MacDonald, L.H., Huffman, E.L., 2004. Post-fire soil water repellency: persistence and soil moisture thresholds. Journal of Soil Science Society of America 68, 1729—1734.

McDonald, R.C., Isbell, R.F., Speight, J.G., Walker, J., Hopkins, M.S., 1990. Australian Soil and Land Survey Field Handbook. Inkata, Melbourne.

McKenzie, J., Jacquier, D., Isbell, R., Brown, K., 2004. Australian Soils and Landscapes. CSIRO Publishing, Collingwood, Australia.

McKissock, I., Gilkes, R.J., Harper, R.J., Carter, D.J., 1998. Relationships of water repellency to soil properties for different spatial scales of study. Australian Journal of Soil Research 36, 495—507.

Moody, J.A., Martin, D.A., 2001. Initial hydrologic and geomorphic response following a wildfire in the Colorado front range. Earth Surface Processes and Landforms 26, 1049—1070.

Morbidelli, R., Corradini, C., Govindaraju, R.S., 2007. A field-scale infiltration model accounting for spatial heterogeneity of rainfall and soil saturated hydraulic conductivity. Hydrological Processes 20, 1465—1481.

Morris, S.E., Moses, T.A., 1987. Forest fire and the natural soil erosion regime in the Colorado front range. Annals of the Association of American Geography 77, 245—254.

NSERL, 1995. USDA-Water Erosion Prediction Project (WEPP) Technical Documentation. National Soil Erosion Reseat Laboratory (NSERL) Report No 10, West Lafayette.

Papworth, M.P., Hartland, R., Lucas, A., 1990. Logging Alpine Ash in the East Kiewa River Catchment. Part I: Effects on Stream Sediment Levels. Dept. of Cons. and Environ, Land Protection Division, Res. Rep. 4.

Pierson, F.B., Robichaud, P.R., Spaeth, K.E., 2001. Spatial and temporal effects of wildfire on hydrology of a steep rangeland watershed. Hydrological Processes 15, 2905—2916.

Prosser, I.P., Williams, L., 1998. Effect of wildfire on runoff and erosion in native *Eucalyptus* forest. Hydrological Processes 12, 251—264.

Renard, K.G., Foster, G.R., Weesies, G.A., Porter, J.P., 1991. Revised universal soil loss equation. Journal of Soil and Water Conservation 46, 30—33.

Reynolds, W.D., 1993. Saturated hydraulic conductivity: laboratory measurement. In: Carter, M.E. (Ed.), Soil Sampling and Methods of Analysis. Canadian Society of Soil Science. Lewis Publishers.

Robichaud, P.R., 2000. Fire effects on infiltration rates after prescribed fire in Northern Rocky Mountain forests, USA. Journal of Hydrology 231–232, 220–229.

Ronan, N.M., 1986. The hydrological effects of fuel reduction burning and wildfire at Wallaby Creek. Report for the Melbourne Metropolitan Board of Works Report Number MMBW-W-0015, pp. 229.

Scott, D.F., Van Wyk, D.B., 1990. The effects of wildfire on soil wettability and hydrological behaviour of an afforested catchment. Journal of Hydrology 121, 239–256.

Scott, D.F., Van Wyk, D.B., 1992. The effects of fire on soil water repellency, catchment sediment yields and streamflow. In: Van Wilgen, B.W., Richardson, D.M., hKruger, F.J., Van Hensbergen (Eds.), Fire in South African mountain Fynbos. Springer-Verlag, Berlin, pp. 216–239.

Scott, D.F., Versfeld, D.B., Lesch, W., 1998. Erosion and Sediment yield in relation to afforestation and fire in the mountains of the Western Cape Province, South Africa. South African Geographical Journal 80, 52–59.

SCS National Engineering Handbook, 1985. Sec. 4, Hydrology, US Department of Agriculture, Soil Conservation Service, Washington, DC, United States of America.

Shakesby, R.A., Doerr, S.H., 2006. Wildfire as a hydrological and geomorphological agent. Earth-Science Reviews 74, 269–307.

Shakesby, R.A., Coelho, C.O.A., Ferreira, A.D., Terry, J.P., Walsh, R.P.D., 1993. Wildfire impacts on soil erosion and hydrology in wet Mediterranean forest, Portugal. International Journal Wildland Fire 3, 95–110.

Shakesby, R.A., Doerr, S.H., Walsh, R.P.D., 2000. The erosional impact of soil hydrophobicity: current problems and future research directions. Journal of Hydrology 231–232, 178–191.

Shakesby, R.A., Chafer, C.J., Doerr, S.H., Blake, W.H., Wallbrink, P.J., Humphreys, G.S., Harrington, B.A., 2003. Fire severity, water repellency characteristics and hydrogeomorphological changes following the Christmas 2001 Sydney forest fires. Australian Geographer 34, 147–175.

Shakesby, R.A., Wallbrink, P.J., Doerr, S.H., English, P.M., Chafer, C., Humphreys, G.S., Blake, W.H., Tomkins, K.M., 2007. Distinctiveness of wildfires effects on soil erosion in south-east Australian eucalypt forests assessed in a global context. Forest Ecology and Management 238, 347–364.

Sharma, M.L., Gander, G.A., Hunt, C.G., 1980. Spatial variability of infiltration in a watershed. Journal of Hydrology 45, 101–122.

Sheridan, G.J., Lane, P.N.J., Murphy, S.C., 2001. Testing and development of a simple field based erosion hazard assessment scheme. In: Proceedings of the symposium ''Soil erosion research for the 21st century''. Ala Moana Hotel, Honolulu, Hawaii January 3–5th, 2001.

Sheridan, G.J., So, H.B., Loch, R.J., 2003. Improved slope adjustment functions for soil erosion prediction. Australian Journal of Soil Research 41, 1–20.

Smith, R.E., Goodrich, D.C., 2000. Model for rainfall excess patterns on randomly heterogeneous areas. Journal of Hydrologic Engineering ASCE 5, 355–362.

Smith, R.B., Wehner, B., Black, P., 1981. Survey of vegetation and timber resources of the East Kiewa experimental catchments, 1981. Forest Inventory Report No 17. Forests Commission, Victoria.

Talsma, T., Hallam, P.M., 1980. Hydraulic conductivity measurement of forest catchments. Australian Journal of Soil Research 30, 739–748.

Topp, G.C., Binns, M.R., 1976. Field measurement of hydraulic conductivity with a modified air-entry permeameter. Canadian Journal of Soil Science 56, 139–147.

Varela, M.E., Benito, E., de Blas, E., 2005. Impact of wildfires on surface water repellency in soils of northwest Spain. Hydrological Processes 19, 3649–3657.

Williams, D.F., Bren, L.J., Craig, F.G., 1973. Effects of wildfire on overland flow and erosion in the Macalister River watershed. Forest Commission of Victoria, Research Branch Report. 29, (unpublished).

Wilson, C.J., 1999. Effects of logging and fire on runoff and erosion on highly erodible granitic soils in Tasmania. Water Resources Research 35, 3531–3546.

Wischmeier, W.H., Smith, D.D., 1958. Rainfall energy and its relationship to soil loss. Transactions/American Geophysical Union 39, 285–291.

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall erosion losses – a guide to conservation planning. US Department of Agriculture, Ag. Handbook No. 537.

Wondzell, S.M., King, J.G., 2003. Postfire erosional processes in the Pacific Northwest and Rock Mountain regions. Forest Ecology and Management 178, 75–87.

Woolhiser, D.A., Smith, R.E., Giraldez, J.V., 1996. Effects of spatial variability of saturated hydraulic conductivity on Hortonian overland flow. Water Resources Research 32, 671–678.

Yu, B., 1999. A comparison of the Green-Ampt and a spatially variable infiltration model for natural storm events. Transactions of the ASAE 42, 89–97.

Yu, B., Rose, C.W., Coughlan, K.J., Fentie, B., 1997. Plot-scale rainfall–runoff characteristics and modelling at six sites in Australia and Southeast Asia. Transactions of the ASAE 40, 1295–1303.

Yu, B., Cakurs, U., Rose, C.W., 1998. An assessment of methods for estimating runoff rates at the plot scale. Transactions of the ASAE 41, 653–661.