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Review

Hydrologic and erosion responses to wildfire along the rangeland–xeric forest continuum in the western US: a review and model of hydrologic vulnerability

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Abstract. The recent increase in wildfire activity across the rangeland–xeric forest continuum in the western United States has landscape-scale consequences in terms of runoff and erosion. Concomitant cheatgrass (*Bromus tectorum* L.) invasions, plant community transitions and a warming climate in recent decades along grassland–shrubland–woodland–xeric forest transitions have promoted frequent and large wildfires, and continuance of the trend appears likely if warming climate conditions prevail. These changes potentially increase overall hydrologic vulnerability by spatially and temporally increasing soil exposure to runoff and erosion processes. Plot and hillslope-scale studies demonstrate burning may increase event runoff or erosion by factors of 2–40 over small-plot scales and more than 100-fold over large-plot to hillslope scales. Reports of flooding and debris flow events from rangelands and xeric forests following burning show the potential risk to natural resources, property, infrastructure and human life. We present a conceptual model for evaluating post-fire hydrologic vulnerability and risk. We suggest that post-fire risk assessment of potential hydrologic hazards should adopt a probability-based approach that considers varying site susceptibility in conjunction with a range of potential storms and that determines the hydrologic response magnitudes likely to affect values-at-risk. Our review suggests that improved risk assessment requires better understanding in several key areas including quantification of interactions between varying storm intensities and measures of site susceptibility, the varying effects of soil water repellency, and the spatial scaling of post-fire hydrologic response across rangeland–xeric forest plant communities.

Additional keywords: cheatgrass, climate change, fire effects, grass-fire cycle, Great Basin, hydrologic risk, invasive plants, juniper, pinyon, runoff, sagebrush, wildland–urban interface, woodland encroachment.

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Introduction

Wildfire activity is increasing along the rangeland–xeric forest continuum of the interior western United States (US; Littell *et al.* 2009; Miller *et al.* 2009; Litschert *et al.* 2012; Balch *et al.* 2013). A vast expanse of the western US is dominated by an arid to semi-arid climate with less than 100-cm annual precipitation (Fig. 1*a*) and vegetation that transitions from rangelands to pinyon–juniper woodlands (*Pinus* spp., *Juniperus* spp.) or xeric ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) forests across low- to mid-elevations (Fig. 1*b*). Over the past decade, more than 1×10^6 ha of the western US were burnt by wildfire annually, and much of this was along the rangeland–xeric forest continuum (NIFC 2012). Periods of recurring high wildfire activity in the western US are not unprecedented in the paleo-record (Pierce *et al.* 2004; Heyerdahl *et al.* 2008*a*, 2008*b*; Whitlock *et al.* 2008, 2011; Marlon *et al.* 2012) but the frequency of large fires (>400 ha) and annual area burnt have increased in recent decades (Westerling *et al.* 2006; Keane *et al.* 2008; Morgan *et al.* 2008; Littell *et al.* 2009; Miller *et al.* 2011*a*).

Cheatgrass (*Bromus tectorum* L.) invasion is the primary cause of increased fire frequency and annual area burnt on sagebrush rangelands throughout the western US (Keane *et al.* 2008; Miller *et al.* 2011*a*; Balch *et al.* 2013). The species is now a major plant constituent on 4×10^6–7×10^6 ha of sagebrush rangelands in the Great Basin alone (Fig. 1*b*; Knapp 1996; Bradley and Mustard 2005; Miller *et al.* 2011*a*). Cheatgrass infill of areas between woody plants affects wildfire activity by increasing the horizontal continuity of fuels and the likelihood

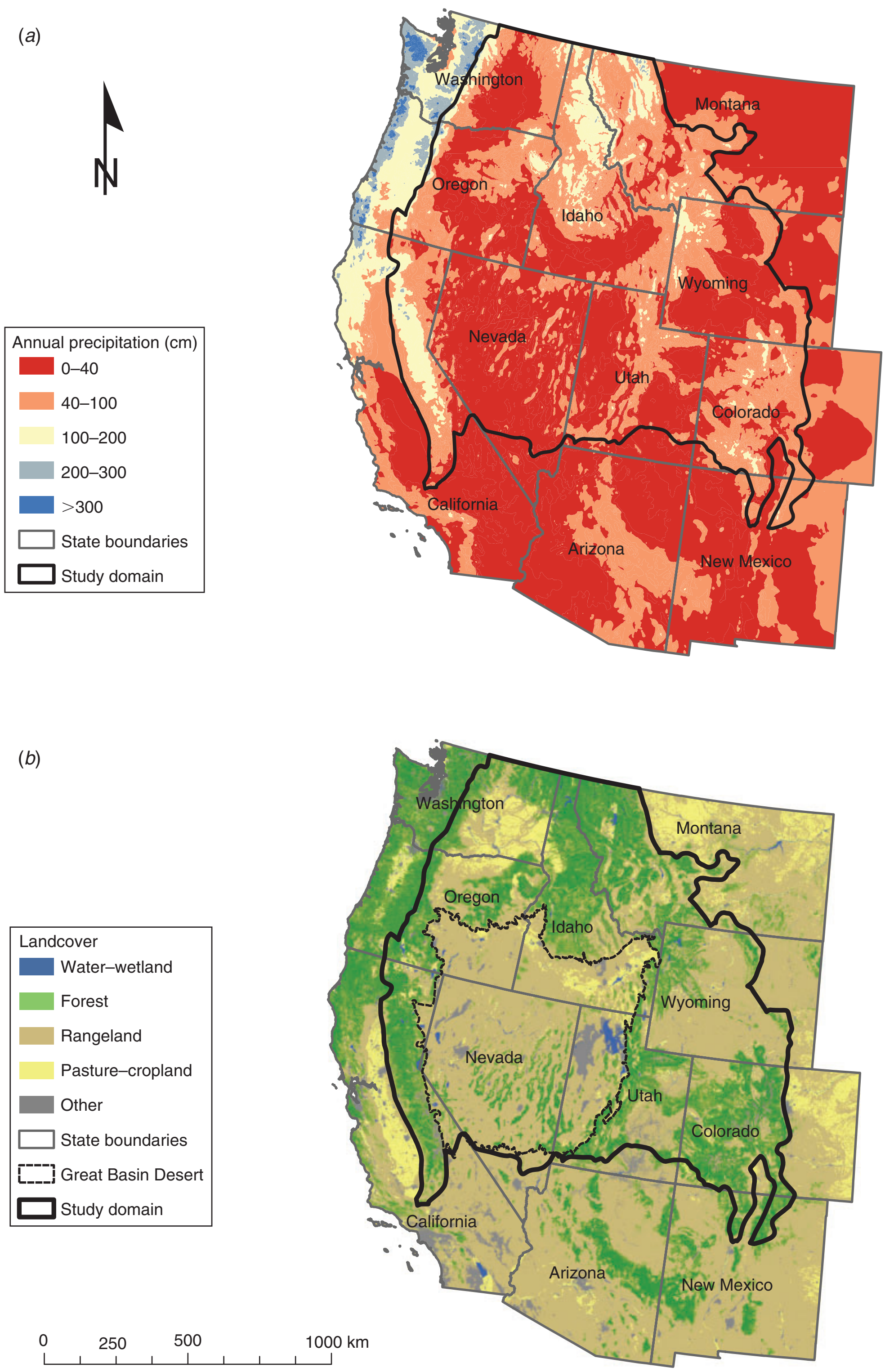

Fig. 1. Map of annual precipitation (*a*) (Prism Climate Group 2012) and landcover (*b*) (USGS 2012) across the western United States. The approximate geographic area of the study domain is delineated by the bold black line in each map. The boundary of the Great Basin Desert (rangeland and woodland region with high wildfire activity) is delineated with a dashed black line on the landcover map.

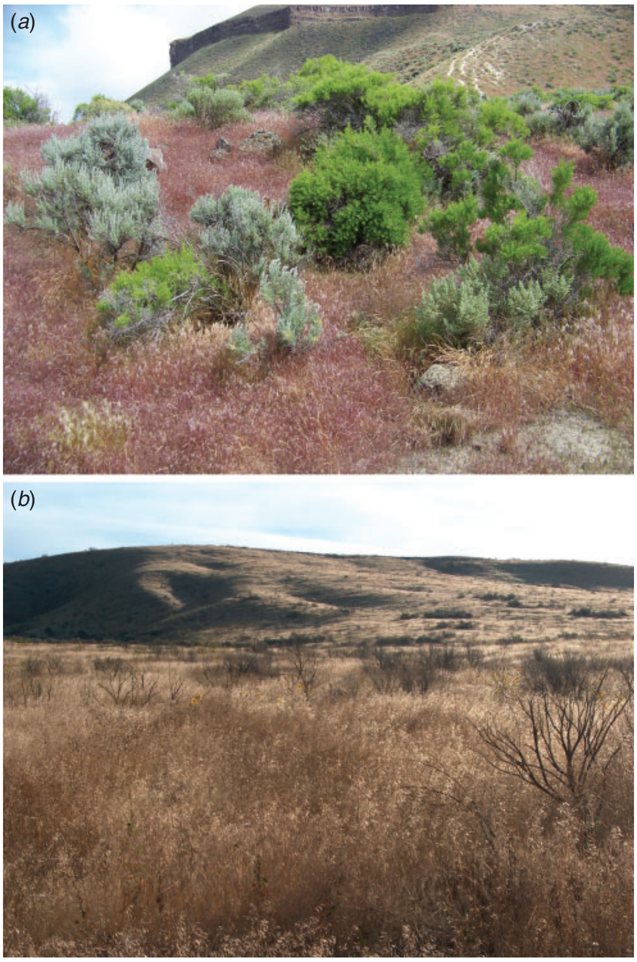

Fig. 2. Sagebrush rangeland with cheatgrass-infested interspace between shrubs (*a*) and a burnt sagebrush site with nearly 100% cover of cheatgrass 1 year post-fire (*b*).

of ignition (Fig. 2*a*; Brooks *et al.* 2004; Link *et al.* 2006; Davies *et al.* 2012). Fire return intervals in cheatgrass-infested rangelands are commonly 10-fold shorter than those for intact sagebrush–bunchgrass communities (Miller *et al.* 2011*a*). Frequent re-burning of cheatgrass-invaded rangelands promotes a grass-fire cycle that in turn perpetuates cheatgrass dominance (Knapp 1996; Brooks *et al.* 2004; Davies *et al.* 2012; Balch *et al.* 2013). Cheatgrass produces more seeds post-fire than do native species (Humphrey and Schupp 2001) and commonly outcompetes native bunchgrasses for soil nutrients and water (Harris 1967; Mack and Pyke 1983; Aguirre and Johnson 1991; Duke and Caldwell 2001). The higher seedling vigour and reproduction potential of cheatgrass relative to other species promote a decline in site species richness and evenness with increasing cheatgrass coverage (Mack 1981; Melgoza and Nowak 1991). Repeated fires over short rotations kill newly established shrubs and perennial grasses, exhaust native seed sources and propagate highly flammable cheatgrass monocultures (Fig. 2*b*; Welch 2005; Davies *et al.* 2012).

Woodland expansion and infill on rangelands have made much of the western US prone to large severe wildfires (Keane *et al.* 2008; Romme *et al.* 2009). Native pinyon and juniper species have dramatically increased their range in the past 150 years and currently occupy more than 3.0×10^7 ha of the western US (Miller and Tausch 2001; Davies *et al.* 2011; Miller *et al.* 2011*a*). Range expansion has primarily occurred through encroachment into sagebrush communities (Fig. 3*a*). Early-succession woodlands are now burning in large, high-severity wildfires due to heavy woody-fuel loading and extensive horizontal-to-vertical fuel connectivity (Fig. 3*b*; Miller and Tausch 2001). Tree infill on late-succession woodlands (Fig. 3*c*) and extreme fire weather have increased the occurrence of large, high-severity woodland fires in recent decades (Keane *et al.* 2008). Cheatgrass invasion into pinyon–juniper woodlands (Fig. 3*d*) across the western US has amplified the risk of large-scale fires associated with the annual grass-fire cycle (Young and Evans 1978; Tausch 1999; Getz and Baker 2008; Shinneman and Baker 2009). Historical wildfire regimes in pinyon and juniper woodlands consisted of high-severity fires every few hundred or more years (Baker and Shinneman 2004; Romme *et al.* 2009). Therefore, severity of modern woodland wildfires is within the historical range of variability, but the relatively high frequency of large fires and annual area burnt on woodlands in the past 20 years is likely unprecedented (Keane *et al.* 2008).

Much of the interior western US now exists in a state in which rangeland and woodland wildfires stimulated by cheatgrass and dense fuels have a greater likelihood of progressing upslope into xeric forests where fire activity is also increasing (Keane *et al.* 2008; Nelson and Pierce 2010; Balch *et al.* 2013). Wildfire activity in western xeric forests is dictated by low fuel moisture and cyclonic weather conducive to ignitions and fire spread (Heyerdahl *et al.* 2002; Gedalof *et al.* 2005; Heyerdahl *et al.* 2008*a*; Morgan *et al.* 2008; Taylor *et al.* 2008; Whitlock *et al.* 2008; Miller *et al.* 2009). In recent decades, warmer winter and spring air temperature trends at mid-elevations in the western US have resulted in decreased snowpacks (Mote *et al.* 2005; Regonda *et al.* 2005; Knowles *et al.* 2006; Trenberth *et al.* 2007; Bonfils *et al.* 2008; Nayak *et al.* 2010), earlier spring snowmelt and streamflow (McCabe and Clark 2005; Regonda *et al.* 2005; Stewart *et al.* 2005; Pederson *et al.* 2011) and drier fuels (Westerling *et al.* 2006). These shifts have lengthened fire seasons and increased fire frequency and area burnt in western forests (Pierce *et al.* 2004; Westerling *et al.* 2006; Morgan *et al.* 2008; Pierce and Meyer 2008; Littell *et al.* 2009).

Climate projections forecast geographic and elevation shifts in fuels that influence fire activity and a persistence of current fire trends along the rangeland–xeric forest continuum (Bradley *et al.* 2009; Balch *et al.* 2013). Abatzoglou and Kolden (2011) suggested cheatgrass invasibility and the length of the fire season in the Great Basin will be enhanced by a warmer climate and an increase in wet winters. Wisdom *et al.* (2003) estimated at least 35% of Great Basin shrublands remain at high risk of woodland encroachment, potentially pre-conditioning these areas to extreme fire behaviour (Keane *et al.* 2008). Miller and Tausch (2001) forecasted that land area covered by dense woodlands and the occurrence of high severity woodland fires will increase substantially in the next 40 or more years. Across the interior west, cheatgrass is migrating upslope (Keeley and McGinnis 2007; McGlone *et al.* 2009; Griffith and Loik 2010;

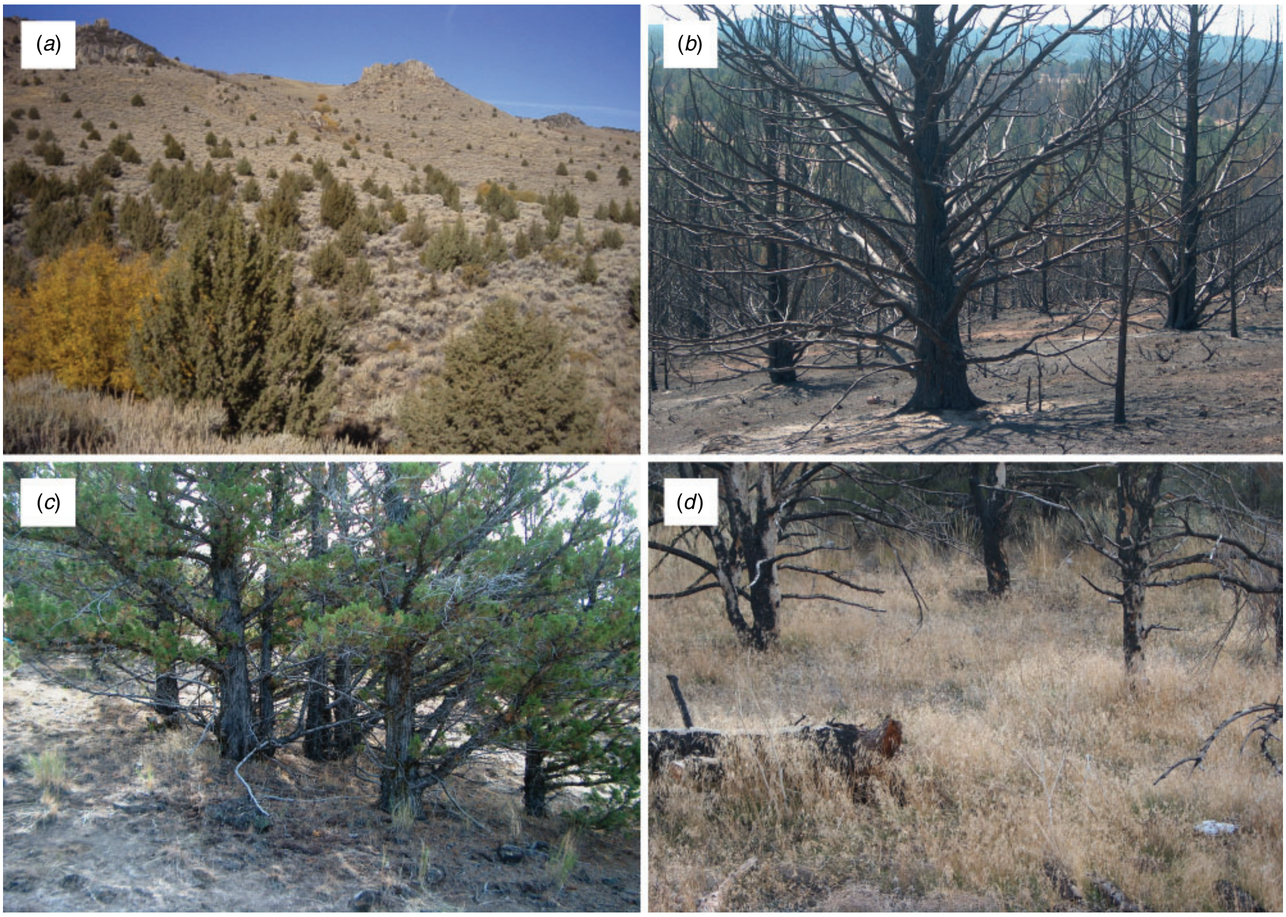

Fig. 3. Woodland encroachment on sagebrush rangeland (*a*); woodland burnt by high-severity wildfire (*b*); tree infill into persistent woodland (*c*) and cheatgrass invasion of a burnt woodland (*d*).

Bromberg *et al.* 2011), potentially introducing the grass-fire cycle at higher elevations and in xeric forests. Xeric forests adjacent to grass-dominated hillslopes will likely undergo more frequent burning than those distant from grass-dominated hillslopes (Gartner *et al.* 2012). Projections of climate and plant community transitions are highly variable (Bradley 2009), but most forecast warming, increased dry-season cyclonic storms, longer fire seasons and greater wildfire activity across the rangeland–xeric forest domain of the western US (Price and Rind 1994; Flannigan *et al.* 2000; Whitlock *et al.* 2003; Brown *et al.* 2004; Gedalof *et al.* 2005; Running 2006; Flannigan *et al.* 2009; Spracklen *et al.* 2009; Littell *et al.* 2010; Abatzoglou and Kolden 2011).

Paleo-erosion records link periods of high wildfire activity in the western US with flooding and increased erosion (Meyer *et al.* 1995, 2001; Meyer and Pierce 2003; Pierce *et al.* 2004; Pierce and Meyer 2008; Pierce *et al.* 2011). In recent decades, extensive damage to natural resources, property, and city infrastructures, and loss of human life have been well documented for post-fire flood events in the western US (Cannon *et al.* 2001*a*; Moody and Martin 2001*a*; Klade 2006; Cannon *et al.* 2011). Our ability to accurately forecast these effects and the potential hazards for values-at-risk is limited with respect to current wildfire activity (Miller *et al.* 2011*b*). Resource managers in the western US are challenged with rapidly evaluating fire effects on ecosystems, determining potential hazards to values-at-risk and conducting cost-benefit analyses of mitigation options (Calkin *et al.* 2007; Robichaud *et al.* 2010*a*). The capability of risk assessments to accurately evaluate hazards and apportion mitigation expenditures requires continued improvement in understanding fire effects, development of predictive technologies, and transfer of information and tools to resource managers (Robichaud *et al.* 2009; Robichaud and Ashmun 2013).

Current knowledge of fire effects on soils, runoff and erosion is largely based on field studies of sagebrush rangelands (*Artemisia* spp.; Pierson *et al.* 2001, 2002, 2008*a*, 2008*b*, 2009), semi-arid woodlands (Pierson *et al.* 2013; Williams *et al.* 2013), chaparral (see DeBano *et al.* 1998; Shakesby and Doerr 2006), forests (Robichaud *et al.* 2000; Benavides-Solorio and MacDonald 2001, 2005; Larsen *et al.* 2009; Robichaud *et al.* 2010*a*, 2010*b*) and Mediterranean scrublands (Cerdà 1998; Cerdà and Doerr 2005, 2008; Shakesby 2011). These studies offer valuable insight into post-fire watershed response and development of hydrologic risk assessment strategies associated with increasing wildfire activity. In this paper, we review current understanding of the hydrologic effects of increasing wildfire activity across the rangeland–xeric forest continuum in the interior western US (Fig. 1) and determine key knowledge

gaps for addressing the associated hazards to values-at-risk. Our objectives are: (1) summarise current knowledge of wildfire effects on soils, runoff and erosion; (2) frame current knowledge in a conceptual model for increasing the understanding of fire-induced hydrologic risk; and (3) identify the main knowledge gaps that limit improvement of post-fire risk assessment for increased wildfire activity.

Fire effects on soils, runoff and erosion

Water availability and surface soil conditions

The first-order effect of fire on runoff and erosion is decreased interception. Unburnt shrubs and conifers can intercept as much as 35% and 80% of rainfall during high and low intensity storms , decreasing water available for runoff and erosion (Rowe 1948; Hamilton and Rowe 1949; Skau 1964; Tromble 1983; Owens *et al.* 2006). Rainfall interception by rangeland plants can reduce erosivity of high-intensity rainfall by 50%, thereby decreasing soil detachment by rain drops (Wainwright *et al.* 1999; Martinez-Mena *et al.* 2000). Numerous studies in forested areas have found rainfall erosivity and its dissipation by cover to be primary factors controlling post-fire erosion rates (Inbar *et al.* 1998; Moody and Martin 2001*b*; Benavides-Solorio and MacDonald 2005; Spigel and Robichaud 2007; Robichaud *et al.* 2008; Moody and Martin 2009; Robichaud *et al.* 2013*a*, 2013*b*). Reduction of vegetation by fire may also result in less snow accumulation and subsequent decreases in soil water recharge and vegetation recovery. Spatial and temporal patterns of snow accumulation and melt exert significant control on soil water input, vegetation recruitment and productivity, and hydrologic processes in snow-dominated semi-arid landscapes (Flerchinger *et al.* 1998; Flerchinger and Cooley 2000; McNamara *et al.* 2005; Seyfried *et al.* 2009; Williams *et al.* 2009; Ebel *et al.* 2012*a*). Dense shrub cover (2.2 plants m^{-2}) can intercept and store 37–61% of snowfall on rangelands (Hull 1972; Hull and Klomp 1974). Reduced snow accumulation after fire may have minor influence on soil water storage where seasonal snowmelt input is substantial enough to return soils to field capacity (Ebel *et al.* 2012*a*).

Hydrologically important soil properties are strongly influenced by organic matter and soil fauna and microorganisms that are altered to varying degrees by burning (Raison 1979; Certini 2005; Shakesby and Doerr 2006; Mataix-Solera *et al.* 2009, 2011). Soil organic matter is combusted at temperatures above 200°C and is completely consumed at 450–500°C (DeBano *et al.* 1998; Neary *et al.* 1999). These temperatures are well within the range of those commonly reported for rangeland and xeric forest soils during wildfire (Wright and Bailey 1982; Neary *et al.* 1999). The combustion of organic matter in soils can alter soil structure, increase bulk density and decrease porosity and infiltration capacity (Giovannini *et al.* 1988; Giovannini and Lucchesi 1997; Hester *et al.* 1997; Pierson *et al.* 2001, 2002; Hubbert *et al.* 2006; Stoof *et al.* 2010). Aggregate stability promotes infiltration and soil resistance to erosion and may be unaffected, reduced or increased by burning. Moderate- to high-severity burning of soils stabilised by organic matter commonly reduces aggregate stability through combustion of the binding agent (Mataix-Solera *et al.* 2011). Some studies have found an increase in aggregate stability after fire associated with formation of hydrophobic soils (Mataix-Solera and Doerr 2004; Arcenegui *et al.* 2008; Jordán *et al.* 2011). Aggregate stability of soils with high clay content may be enhanced by high-severity burning due to thermal fusion of clay particles into coarser particles (Giovannini *et al.* 1988; Giovannini and Lucchesi 1997; Mataix-Solera *et al.* 2011). However, fusion of clay to silt or sand particles can increase soil erosion due to the loss of the cohesive properties inherent to clay soils (Badía and Martí 2003; Hubbert *et al.* 2006). Burning may also reduce the role of invertebrates, microorganisms, and fungal mycorrhizae in facilitating soil aggregation and infiltration (DeBano *et al.* 1998; Shakesby and Doerr 2006; Mataix-Solera *et al.* 2009). Soil temperatures of 40–210°C are fatal for most fungi and soil organisms, and organic matter combustion and nutrient volatilisation at soil temperatures above 200°C reduce the primary food source for soil fauna production (DeBano *et al.* 1998; Neary *et al.* 1999; Certini 2005; Mataix-Solera *et al.* 2011). Finally, soil moisture retention, a key component of plant and soil fauna productivity, can also be adversely affected by burning due to loss of soil organic matter, pore structure and surface insulation by litter (DeBano *et al.* 1998; Stoof *et al.* 2010; Ebel 2012, 2013).

Soil heating may alter or create hydrophobic and/or hyper-dry soil conditions (Krammes and DeBano 1965; DeBano and Krammes 1966; Savage 1974; DeBano *et al.* 1998; Doerr *et al.* 2000; Hubbert *et al.* 2006; Pierson *et al.* 2008*b*; Doerr *et al.* 2009*a*; Moody *et al.* 2009; Pierson *et al.* 2009; Moody and Ebel 2012). During fires, organic matter combustion at the soil surface radiates heats downward into the soil profile and vaporises organic substances. Some of these substances are translocated downward along temperature gradients until they condense, forming a variable-thickness hydrophobic patch (DeBano *et al.* 1970; Savage *et al.* 1972; Savage 1974; DeBano *et al.* 1976; DeBano 2000; Doerr *et al.* 2004). Naturally occurring or 'background' soil water repellency has been commonly observed beneath unburnt conifers and shrubs (Lebron *et al.* 2007; Madsen *et al.* 2008; Pierson *et al.* 2008*b*; Doerr *et al.* 2009*b*; Pierson *et al.* 2009, 2010, 2013; Williams *et al.* 2013) and is typically unaffected by soil temperatures <175°C. Soil temperatures of 175–270°C may enhance 'background' water repellency or create hydrophobic soil conditions (Doerr *et al.* 2000, 2009*a*). Water repellency breaks down or is destroyed at soil temperatures of 270–400°C (Savage *et al.* 1972; DeBano *et al.* 1976; Giovannini and Lucchesi 1997; Doerr *et al.* 2004). Fire-enhanced or -induced soil water repellency is commonly found within a few centimetres of the soil surface and rapidly decreases in strength with increasing soil depth (Doerr *et al.* 2009*a*). Repellency strength and its effect on runoff pre- and post-fire is highly variable in space and time due to inherent variability in pre-fire vegetation, soil properties and conditions, and burn severity (Dekker *et al.* 2001; Huffman *et al.* 2001; MacDonald and Huffman 2004; Woods *et al.* 2007; Pierson *et al.* 2008*b*; Woods and Balfour 2008; Pierson *et al.* 2009, 2010; Stoof *et al.* 2011; Bodí *et al.* 2013; Williams *et al.* 2013). The effects of repellency on runoff generation are even more severe under hyper-dry (extremely dry) conditions immediately following high-severity fire. Extreme heating during high-severity fire can dry out small and large pores within the upper soil profile, potentially causing partial pore structure collapse

(Moody *et al.* 2009; Moody and Ebel 2012). Hyper-dry conditions require soils to be rewet before capillary and gravity-driven infiltration can occur (Moody and Ebel 2012).

Runoff and erosion at the small-plot scale

Small plot (0.7×0.7 m) rainfall simulation studies by the authors (see Table 1) on steeply sloped (35–60%) sagebrush hillslopes demonstrate the effects of vegetation cover removal, surface alteration and soil water repellency on post-fire runoff and erosion from rangelands and woodlands. For example, Pierson *et al.* (2002) investigated the hydrologic effects of wildfire on north- and south-facing sagebrush hillslopes 1 year after the Eighth Street Fire, near Boise, Idaho. Only the south-facing hillslope results are presented here. Runoff and erosion pre-fire were low from shrub coppices (areas beneath shrub canopies) and interspaces (areas between shrub canopies) due to rainfall interception by the canopy and litter and high surface roughness (Table 1). Moderate- and high-severity burning reduced vegetation and litter biomass by 75–99% and decreased surface roughness by 40%. Approximately 30–50% of applied rainfall post fire was lost to runoff over the nearly uniformly bare surface (Table 1). Fire had a greater effect on erosion than on runoff (Table 1) and severe burning increased soil erosion 10-fold from coppices and 40-fold from interspaces (Table 1). Higher runoff rates following fire were attributed to decreased interception, persistence of pre-fire soil water repellency, and reduced surface water detention following litter removal and reduced surface roughness. Increased erosion following burning was attributed to greater raindrop detachment and more efficient sediment transport, as well as increased erodibility on interspace microsites.

A 3-year investigation by Pierson *et al.* (2001, 2008*a*, 2008*b*; Table 1) measured infiltration, runoff, and erosion from rainsplash and sheetflow following the Denio Fire, in Nevada. The fire removed nearly all of the canopy and ground cover from well-vegetated, steep sagebrush hillslopes. Runoff increased by 20% immediately following burning on shrub coppices but decreased on interspaces by 40% (Table 1). The difference between runoff on burnt and unburnt coppices was attributed to the removal of canopy and ground cover by fire on strongly water repellent soils. Decreased runoff from interspace areas was associated with removal of water-shedding senescent vegetation (Pierson *et al.* 2001) and fire-reduced soil water repellency (Table 1). A decrease in soil water repellency by 50–60% on all plots 1 year after fire was concurrent with a nearly 40% increase in infiltration (Fig. 4*a*). A subsequent 40–50% increase in soil water repellency on all plots 2 years after fire coincided with a 5–15% decrease in infiltration (Fig. 4*a*; Pierson *et al.* 2008*a*, 2008*b*). Overall, canopy and ground cover removal controlled water availability whereas the strength of soil water repellency exerted greater influence on infiltration and runoff. Interestingly, burning increased erosion from coppices 3-fold but had no effect on interspace erosion (Table 1). The differing responses were attributed to a more erodible surface and greater runoff on coppices after burning. Erosion 1 year after fire was greatly reduced on all plots and similar for burnt and unburnt conditions. Two years after fire, burnt coppice plots generated 3–14 times more erosion than all other plots. Soil water repellency and infiltration were the only other variables showing the same temporal trend, implicating runoff and continued increased erodibility as causal factors (Pierson *et al.* 2008*a*).

Pierson *et al.* (2008*b*, 2009; Table 1) measured infiltration, runoff and erosion from small-plot rainfall simulations on burnt and unburnt sagebrush hillslopes the year of, and 1 year following, the Breaks Prescribed-Fire in the Reynolds Creek Experimental Watershed, Idaho. The fire reduced canopy cover to 0–10% (Table 1) and litter cover to 36 and 14% for shrub coppice and interspace plots. Runoff doubled on coppice plots immediately after fire due to canopy and ground cover reductions, decreased surface roughness and strong post-fire soil water repellency (Table 1). Burning of interspaces reduced runoff (Table 1). One year after fire, a significant decrease (by 70%) in soil water repellency on burnt and unburnt coppices and nearly uniform slight soil water repellency across all plots resulted in a 2-fold increase in infiltration (Fig. 4*b*). As in the Pierson *et al.* (2008*a*) study (Table 1), cover influenced water availability, but the strength of soil water repellency exerted a greater influence on infiltration (Fig. 4*b*) and runoff of available water. The fire had an even greater effect on erosion than on runoff (Table 1). Reductions in canopy and ground cover increased sediment yield 10-fold on coppices and 3-fold on interspaces. Fire-induced increases in erosion on coppices were attributed to greater runoff and erodibility after fire whereas significantly increased erodibility alone explained the post-fire erosion increase from interspaces (Pierson *et al.* 2009).

Benavides-Solorio and MacDonald (2001, 2002; Table 1) measured runoff from burnt and unburnt areas of a ponderosa pine forest in the Colorado Front Range, Colorado. Runoff (Table 1) from plots burnt at high severity was well correlated ($R^2 = 0.81$) with the strength of soil water repellency. Runoff was not well correlated with percentage slope or bare ground. Benavides-Solorio and MacDonald (2001) concluded that soil water repellency and soil moisture, as a controller of repellency strength, were the primary controls on runoff. Percentage bare soil explained 79% of erosion on all plots, and soil water repellency explained 43% of the variability in erosion on plots burnt with high-severity fire (Benavides-Solorio and MacDonald 2001). Erosion on moderate- and high-severity burnt plots was 2 and 16 times greater than those on unburnt or low-severity plots (Table 1).

In a forest study, Woods and Balfour (2008) evaluated the effects of ash on runoff from rainfall simulation plots 1 month following high-severity wildfire. Rainfall was applied to 0.5-m^2 plots at 75-mm h^{-1} intensity for 1 h. They found that ash provided 15 mm of water storage capacity and protected the soil surface from sealing immediate after fire. Time-to-ponding was 12 min longer and cumulative infiltration was 20 mm greater on ash- than on ash-free plots. Nine months after the fire, ash-covered and ash-free plots exhibited similar runoff behaviour. Similar ash cover and runoff relationships have been reported in studies by Cerdà and Doerr (2008), Larsen *et al.* (2009), Woods and Balfour (2010) and Ebel *et al.* (2012*b*). Bodí *et al.* (2011) found that ash may alter soil wettability, inducing surface soil water repellency when ash is hydrophobic and reducing surface soil water repellency when ash is wettable. In a laboratory rainfall simulation study, Bodí *et al.* (2012) found that a saturated ash layer promoted runoff generation from wettable soils and that an unsaturated ash layer of more than 5-mm depth protected the soil surface from rainsplash erosion

Table 1. Site characteristics, runoff coefficients and sediment yield from rainfall simulations (60 min except where noted) on unburnt (unb) and high-, moderate- (mod) and low-severity burnt semi-arid rangelands (Pierson *et al.* 2001, 2002, 2008*a*, 2008*b*, 2009), woodlands (Pierson *et al.* 2013; Williams *et al.* 2013) and forests (Benavides-Solorio and MacDonald 2001; Johansen *et al.* 2001)

Water drop penetration time (WDPT) is an indicator of strength of soil water repellency as follows: <5 s wettable, 5–60 s slightly repellent, 60–600 strongly repellent (Bisdom *et al.* 1993). Maximum observation time for all WDPT data was 300 s with exception of study by Benavides-Solorio and MacDonald (2001, 2002) with a maximum observation time of 120 s. All data are based on average WDPTs measured over 0- to 5-cm soil depth, beginning at the mineral soil surface. Bulk density and soil water content were measured near the soil surface (<4-cm depth). Runoff coefficient is equal to cumulative runoff divided by cumulative rainfall applied. Value is multiplied by 100 to obtain percentage

Study (ecosystem)	Microsite	Burn severity	Plot size (m^2)	Slope (%)	Time post-fire (month)	Rainfall intensity ($mm\,h^{-1}$)	WDPT (s)	Bulk density ($g\,cm^{-3}$)	Soil water content (%)	Bare soil (%)	Canopy cover (%)	Ground cover (%)	Live plant biomass ($kg\,ha^{-1}$)	Litter biomass ($kg\,ha^{-1}$)	Surface roughness (mm)	Runoff coef. (%)	Sed. yield ($g\,m^{-2}$)
Pierson *et al.* (2002)[A] (sagebrush rangeland)	Coppice	Unb	0.5	35–60	12	67	–	1.21	~14	7	88	93	32 519	14372	18	11	2
		Mod	0.5	35–60	12	67	–	1.28	~5	97	11	3	341	113	12	34	30
		High	0.5	35–60	12	67	–	1.21	~5	98	13	2	744	74	12	37	22
	Interspace	Unb	0.5	35–60	12	67	–	1.35	~14	89	18	12	519	1721	18	24	4
		Mod	0.5	35–60	12	67	–	1.30	~5	95	16	5	520	212	12	26	12
		High	0.5	35–60	12	67	–	1.30	~5	99	5	1	134	61	10	49	148
Pierson *et al.* (2001, 2008*a*, 2008*b*) (sagebrush rangeland)	Coppice	Unb	0.5	30–40	1	85	200	0.93	7	1	100	99	–	–	–	30	12
		High	0.5	30–40	1	85	102	1.22	1	99	1	1	–	–	–	37	41
	Interspace	Unb	0.5	30–40	1	85	220	0.94	5	6	74	94	–	–	–	49	24
		High	0.5	30–40	1	85	97	1.21	1	99	4	1	–	–	–	30	21
Pierson *et al.* (2009) (sagebrush rangeland)	Coppice	Unb	0.5	35–50	1	85	286	1.05	7	2	84	98	–	–	34	39	17
		Mod–High	0.5	35–50	1	85	261	1.09	3	42	10	58	–	–	11	76	183
	Interspace	Unb	0.5	35–50	1	85	110	1.21	3	25	31	75	–	–	18	63	195
		Mod–High	0.5	35–50	1	85	117	1.17	4	84	0	16	–	–	11	55	705
	–	Unb	32.5	35–50	1	85	–	1.07	2	24	57	76	12 125	9517	21	4	8
	–	Mod–High	32.5	35–50	1	85	208	1.13	4	76	0	24	–	808	11	27	988
Pierson *et al.* (2013); Williams *et al.* (2013) (juniper woodland)[B]	Tree Coppice	Unb	0.5	10–25	12	102	42	–	–[B]	0[C]	17[D]	100	–	–	12	23	6
		High	0.5	10–25	12	102	54	–	–[B]	88[C]	5[D]	50	–	–	8	58	206
	Shrub Coppice	Unb	0.5	10–25	12	102	3	–	–[B]	41[C]	117	75	–	–	13	20	6
		High	0.5	10–25	12	102	11	–	–[B]	94[C]	21	43	–	–	9	23	143
	Interspace	Unb	0.5	10–25	12	102	3	–	–[B]	88[C]	20	54	–	–	9	63	36
		High	0.5	10–25	12	102	3	–	–[B]	93[C]	21	51	–	–	8	51	135
Pierson *et al.* (2013); Williams *et al.* (2013) (juniper woodland)[B]	Tree Coppice	Unb	13	10–25	12	102	–	–	–[B]	18[C]	26[D]	93	–	–	23	13	48
		High	13	10–25	12	102	–	–	–[B]	73[C]	15[D]	75	–	–	21	58	1083
	Inter–canopy	Unb	13	10–25	12	102	–	–	–[B]	89[C]	18	72	–	–	16	50	272
		High	13	10–25	12	102	–	–	–[B]	88[C]	32	61	–	–	17	50	572
Benavides-Solorio and MacDonald 2001, 2002[E]) (xeric forest)	–	Low–Unb	1.0	20–25	1–3	79	65	–	2	1	–	99	–	–	–	55	80
	–	Mod	1.0	20–35	1–3	79	50	–	2	12	–	88	–	–	–	58	179
	–	High	1.0	20–45	1–3	79	60	–	2	77	–	23	–	–	–	66	1280
Johansen *et al.* (2001)[F] (xeric forest)	–	Unb	32.5	5	3	60	–	–	~5	48		52	–	–	–	23	36
	–	High	32.5	7	3	60	–	–	~5	74		26	–	–	–	45	912

[A]Data presented from south-facing slopes solely.
[B]Simulated storm applied immediately following 45-min simulation of 64-mm h^{-1} rainfall.
[C]Includes rock cover and ash; bare areas of rock and bare soil were extensive due to woodland encroachment (Pierson *et al.* 2013).
[D]Canopy cover excludes tree cover removed to conduct the rainfall simulation experiments.
[E]Data presented for Bobcat Fire only.
[F]Rainfall applied for 60 min under dry conditions, followed by 24-h hiatus, 30 min of rainfall, 30-min hiatus and 30-min rainfall. Total rain applied was 120 mm.

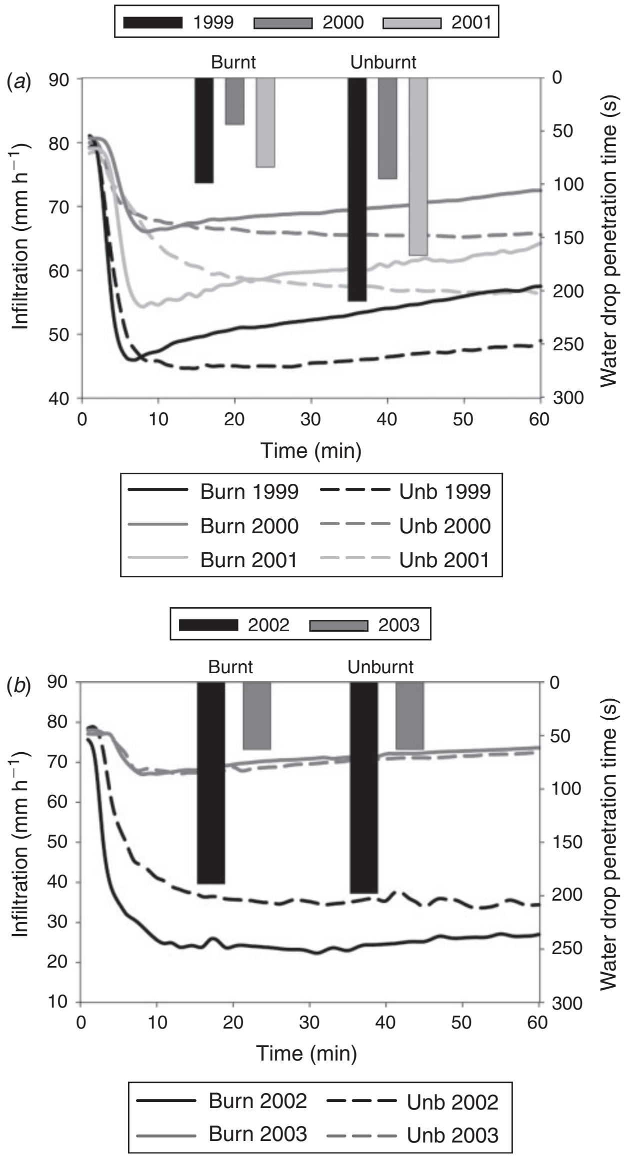

Fig. 4. Infiltration of simulated rainfall (85-mm-h^{-1} intensity) and strength of soil water repellency (measured as water drop penetration time, WDPT) on sagebrush rangeland in Nevada, USA (*a*) (Pierson *et al.* 2008*a*, 2008*b*) and Idaho, USA (*b*) (Pierson *et al.* 2009).

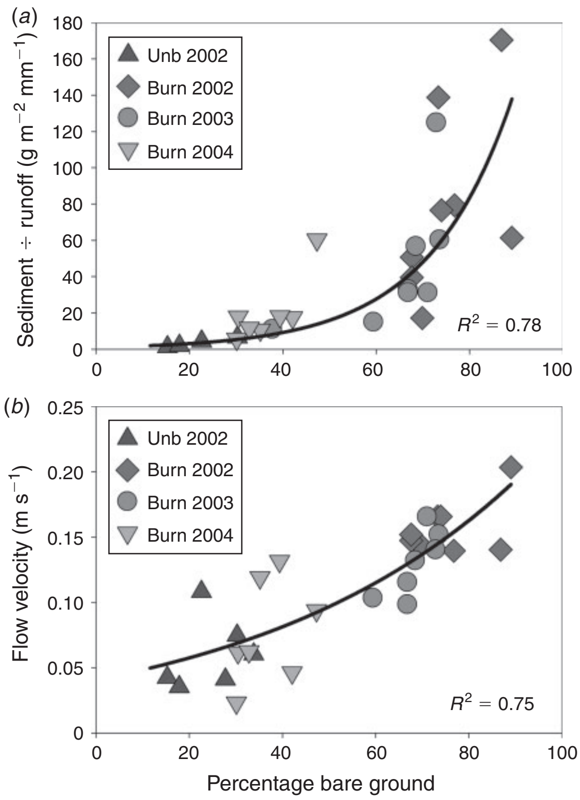

Fig. 5. Sediment yield per unit of runoff (*a*) and velocity of concentrated flow (*b*) compared with bare ground measured for rainfall simulation plots (32.5 m^2, 85 mm h^{-1}, 60 min) and overland flow experiments (12 L min^{-1}) respectively on burnt (Burn) and unburnt (Unb) areas of sagebrush rangeland immediately (2002), and 1 (2003) and 2 (2004) years after fire. Data from Pierson *et al.* (2009).

and improved infiltration into water repellent soils by fingered sub-surface flow. The study also found multiple rain events altered physical and hydraulic properties of the ash layer and reduced its effectiveness to buffer runoff generation and soil erosion. Likewise, Larsen *et al.* (2009) indicated that the positive effect of ash on infiltration is likely short-lived, and that soil sealing following winnowing of ash particles may promote runoff, especially on water repellent soils (e.g. Onda *et al.* 2008).

Runoff and erosion processes at large-plot to hillslope scales

Large-plot scale effects of burning are generally greater for erosion than for runoff due to a change from rainsplash–sheetflow to concentrated flow as the dominant process. Steep slope angles on burnt hillslopes promote concentration of runoff (Pietraszek 2006; Spigel and Robichaud 2007; Pierson *et al.* 2009; Al-Hamdan *et al.* 2012*a*, 2013). Concentrated flow has a higher velocity than sheetflow and is therefore capable of eroding and transporting more sediment. Pierson *et al.* (2009) measured a 7-fold increase in runoff from 32.5 m^2 rainfall simulation plots immediately following burning of steeply sloped sagebrush rangeland (Table 1). Greater runoff under burnt than unburnt conditions was attributed to a 3-fold ground cover reduction, canopy removal, decreased surface roughness, persistent soil water repellency and formation of high-velocity concentrated flowpaths. Runoff returned to pre-fire levels within one growing season due to a 3-fold reduction in strength of soil water repellency and ground cover recovery to 40%. Burning increased erosion more than 120-fold (Table 1, Fig. 5*a*) as a result of high velocity concentrated flow and greater runoff after fire. Cumulative runoff from consecutive 12-min releases of 7, 12, 15 and 21 L min^{-1} of concentrated flow was 406 L on burnt plots immediately following fire and 144 L on unburnt plots. Mean erosion from concentrated flow experiments was 14 363 g on the burnt plots and 2420 g on unburnt plots (Pierson *et al.* 2009). Concentrated flow velocities were 1.5–2.6 times higher on burnt than on unburnt plots the year of the fire and increased exponentially with increasing bare ground (Fig. 5*b*). Erosion from artificial rainfall and simulated concentrated flow

on burnt hillslopes approached that of unburnt hillslopes once ground cover recovered to near 60% (40% bare ground) two growing seasons after fire (Fig. 5*a*).

Limited data are available for large-plot-scale runoff and erosion from pinyon–juniper communities. Pierson *et al.* (2013) and Williams *et al.* (2013) measured runoff and erosion from 13-m^2 rainfall simulations in burnt and unburnt areas of a western juniper (*J. occidentalis* Hook.) site 1 year after fire (Table 1). Runoff from unburnt areas beneath junipers and from the intercanopy area between trees was negligible (2–6 mm) for a 64 mm h^{-1}, 45-min duration storm on dry antecedent moisture conditions. Runoff from the same storm applied to burnt tree and intercanopy plots generated 17 and 4 mm of runoff. The study applied a higher intensity (102 mm h^{-1}, 45 min) simulated storm to all plots within ~30 min of the simulation under dry conditions (Table 1). Runoff was greater for the high intensity storm, but the effects of burning on runoff were significant only for tree plots. Runoff from tree plots was four times higher for burnt than unburnt conditions and was equivalent to that of the intercanopy (Table 1). Approximately 50% of rainfall applied to burnt and unburnt intercanopy plots was converted to runoff. Erosion was high from unburnt intercanopy plots and increased 2-fold in the intercanopy after fire. Erosion increased more than 20-fold on tree plots after fire (Table 1). Williams *et al.* (2013) attributed the lack of fire effects on runoff from intercanopy plots to the already high runoff rates. Increased runoff and erosion following burning of tree plots was attributed to fire removal of dense litter cover on water repellent soils and formation of concentrated flow (Pierson *et al.* 2013; Williams *et al.* 2013).

The effects of burning and storm intensity on large-plot-scale runoff and erosion from semi-arid forests are well documented (see Robichaud *et al.* 2000; Cerdà and Robichaud 2009; Moody and Martin 2009). Johansen *et al.* (2001; Table 1) found that runoff from rainfall simulations on burnt and unburnt areas of a ponderosa pine site was positively correlated ($r = 0.76$) with percentage bare soil, and that time to runoff was negatively correlated ($r = 0.67$) with percentage bare soil. Burning increased runoff and erosion 2- and 25-fold (Table 1). Soil water repellency was highly variable spatially and had minimal effect on runoff. Erosion was strongly correlated with percentage bare soil ($r = 0.84$). Wagenbrenner *et al.* (2006) found that hillslope soil erosion (1900 m^2 plots) from burnt forests of the Colorado Front Range returned to pre-fire levels once ground cover increased to 60%. Benavides-Solorio and MacDonald (2005) used silt fences (190–6600 m^2) to measure post-fire erosion from forested slopes (25–45%) in the Colorado Front Range over varying fire severities. Over the 2-year study, percentage bare soil explained ~64% of the variability in soil erosion ($n = 48$). Approximately 90% of the sediment collected was delivered by high-intensity convective storms. Bare soil and rainfall erosivity together explained 65% of sediment production variability. Sediment yield decreased exponentially with time after fire and was highest where bare soil approached and exceeded 60%. Soil water repellency was weakly correlated with sediment production from all plots ($R^2 \approx 0.30$), but was more strongly correlated for the high-severity plots ($R^2 \approx 0.40$). Concentrated flow played an important role in post-fire erosion rates on converging topography (Benavides-Solorio and MacDonald 2005). Spigel and Robichaud (2007) used silt fences (~100-m^2 contributing area) to measure erosion responses from severely burnt, sloping (50–60% gradient) forest sites in Montana. They concluded that rainfall intensity was the dominant control on erosion from individual storms. More than 2000 g m^{-2} of soil was eroded during short-duration, high-intensity storms (75-mm h^{-1} intensity, at least 10-min duration) on sites with 60–90% bare soil and water-repellent soils. Ground cover and soil conditions influenced responses for low-intensity storms, but storms exceeding ~70-mm h^{-1} intensity over 10-min intervals led to substantial erosion regardless of site conditions. Spigel and Robichaud (2007) observed prominent, dense rill or concentrated flow networks during high intensity storms.

Runoff and erosion at hillslope to watershed scales

Flooding and extensive soil erosion are common where high-intensity storms occur over large areas of recently burnt, sloping terrain along the rangeland–xeric forest continuum (Craddock 1946; Cannon 2001; Cannon *et al.* 1998, 2001*a*, 2001*b*; Meyer *et al.* 2001; Moody and Martin 2001*a*; Pierson *et al.* 2002; Pierce *et al.* 2004; Klade 2006; Cannon *et al.* 2008; Pierce *et al.* 2011). Large erosion events following wildfires are typically triggered by runoff and progressive sediment bulking (Cannon *et al.* 2001*a*). For example, a torrential rainstorm 2 months after the South Canyon Fire in Colorado caused nearly 90 runoff-triggered debris-flow events that inundated a 13–14-ha area with ~70 000 m^3 of soil (Cannon *et al.* 1998, 2001*a*). The fire occurred on steep (30–70%) pinyon–juniper and shrub-dominated hillslopes. Increased runoff and erosion from rainsplash and sheetflow on bare soils facilitated formation of concentrated-flow networks and gullies with high erosive energy and sediment transport capacity. Debris flows developed during the storm mainly through bulking as the flows moved downslope, entrained material and converged in drainage channels with accumulations of wind-blown sediment. Flow velocities were estimated at 3–9 m s^{-1} (Cannon et *al.* 1998). Pierson *et al.* (2002) documented a runoff-triggered response to a short-duration high-intensity storm on steep sagebrush hillslopes 1 year after the 1996 Eighth Street Fire (6070 ha) along the Boise Front Range, Idaho. A 5–10-year return-interval storm (67 mm h^{-1}) lasting 9 min generated concentrated flow networks, flash flooding and mudflows from bare (90–100% bare ground), water-repellent soils with reduced water storage capacity and low surface roughness. In an adjacent basin on the Boise Front, similar conditions immediately following multiple cheatgrass-fuelled wildfires in 1959 resulted in widespread flooding and extensive property damage (Klade 2006). Meyer *et al.* (2001) reported that a short-duration, high-intensity storm on severely burnt ponderosa pine hillslopes in Idaho generated runoff-triggered debris flows. They found incised concentrated flow paths on the steeply sloping terrain integrated into gullies more than 1 m deep. The gullies promoted high-velocity, erosive discharge that generated sediment-laden flows reaching the North Fork Boise River. Debris flows on burnt hillslopes can also be initiated by debris slides or shallow landslides of large masses of saturated sediment (Meyer *et al.* 2001; Meyer and Pierce 2003; Wondzell and King 2003; Pierce *et al.* 2004; Parise and Cannon 2012). Debris slides are most common 4 years or more following burning of forested areas due

largely to declining root strength of dead trees (Meyer *et al.* 2001; Meyer and Pierce 2003). The studies described above clearly demonstrate that plot- to hillslope-scale effects potentially influence hydrologic and erosional responses to intense rainfall over contiguous burnt terrain.

Hydrologic risks associated with altered fire regimes

Clearly, increased wildfire activity along the rangeland–xeric continuum poses significant environmental, social and economic consequences associated with flooding and erosion. More frequent and larger fires increase the likelihood and potential magnitude of onsite and offsite effects. More frequent exposure, as a result of burning surface cover, subjects the soil surface to repeated erosion from frequently occurring storms and increases the probability that the soil surface will be exposed when less-frequent, high-intensity rainfall events occur. Larger fires create more extensive surface exposure. Annual soil loss from burnt hillslopes in sloping terrain can be 60–100 Mg ha^{-1} the first year following fire and may take 4–7 years to return to background levels (Mayor *et al.* 2007; Robichaud 2009). Such losses are detrimental if repeated on 5–10-year rotations. Loss of biologically important surface soils may be particularly critical for rangelands where soil formation takes decades (Allen *et al.* 2011; Sankey *et al.* 2012), especially where large fires are followed by drought years with minimal plant recruitment. Soils transported into sideslopes and hollows onsite may serve as a source for downstream sediment pulses during subsequent high-intensity, channel-flushing events (Cannon *et al.* 2001*a*; Meyer and Pierce 2003; Pierce *et al.* 2004; Robichaud *et al.* 2013*b*) that negatively affect water resources, fisheries and channel geomorphology (Minshall *et al.* 2001; Pierce *et al.* 2011). Studies by Meyer and Pierce (2003), Pierce *et al.* (2004) and Pierce and Meyer (2008) found that large debris flow events in the interior western US are linked to warm climatic conditions (Medieval Warm Period, 1050–750 years ago) associated with large, stand-replacing fires in xeric forests. The studies further showed that recent warming trends in western xeric forests are concomitant with occurrences of large wildland fires and post-fire debris flows. Large fire-induced debris flows are capable of transporting tremendous volumes of sediment and debris into main stem rivers (Cannon *et al.* 2001*a*; Meyer *et al.* 2001; Pierce *et al.* 2011).

The recent increase in frequent, large wildfires is particularly concerning for communities in the wildland–urban interface. Flooding in these areas presents hazards to property, infrastructure and human life. In 1945, flooding following intense rainfall over a 1-year-old 300+-ha cheatgrass burn caused more than US$6 million (2013 values) in damage to property in Salt Lake City, Utah (Craddock 1946). Multiple post-fire flooding events in the 1950s and 1990s along the Boise Front Range caused damage to property and infrastructure in the Boise metropolitan area of Idaho exceeding a value of US$4 million at 2013 rates (Klade 2006). Moody and Martin (2001*b*) evaluated the hydrologic response to a 100-year rainfall event on the 4690-ha Buffalo Creek Fire in steep, forested watersheds of the Colorado Front Range near Denver, Colorado. Two months following fire, a high-intensity (90 mm h^{-1}, 1 h) rainstorm caused flash flooding that killed two people and discharged enough sediment into the Strontia Springs Reservoir to reduce storage capacity by one-third (Agnew *et al.* 1997; Moody and Martin 2001*a*). Cannon *et al.* (2001*a*) reported debris flows from a high-intensity storm on burnt rangelands in Colorado, which engulfed 30 vehicles travelling on a flow-intersected highway and forced two people into the Colorado River. In Arizona, a 24-mm h^{-1}, 10-min storm caused widespread flooding on a recently burnt ponderosa pine site (Neary *et al.* 2012). The event flooded 85 homes, caused one death and substantially damaged city infrastructure. Post-fire mitigation expenditures exceeded US$14 million (Coconino County 2011).

Post-fire hillslope hydrologic vulnerability can be conceptualised as a function of storm magnitude (i.e. rainfall intensity) and site susceptibility (Fig. 6). In this model, storm-specific hydrologic vulnerability represents potential runoff and erosion responses for different site susceptibilities. Site susceptibility is defined by the conditions of the soil surface, cover characteristics and topography, and, therefore, encompasses burn severity as well as other key inherent site characteristics (e.g. slope, rock cover, soil erodibility) that influence hydrologic and erosion responses. For a storm of uniform intensity, hydrologic response increases exponentially with increases in site susceptibility due to a shift in hydrologic process dominance from rainsplash and sheetflow to concentrated flow (Fig. 6). Overall hydrologic vulnerability or response increases with increasing storm intensity due to amplified rainfall erosivity and greater water input with higher rainfall intensity. Fire removal of cover and decreased surface roughness increase water available for runoff over point- to small-plot scales and facilitate formation of concentrated flow paths over larger spatial scales. Runoff generation is enhanced where infiltration is inhibited by water-repellent soil conditions and on steep slopes. Fire-induced increases in erodibility and decreased surface protection against rainsplash facilitate soil detachment at small scales and promote sediment delivery by sheetflow and concentrated flow paths over larger spatial scales. Increased erosion with increasing land area results from sediment bulking of the flow as it moves downslope, potentially causing mudslides or debris flows (Cannon *et al.* 1998, 2001*a*).

Our qualitative model (Fig. 6) potentially presents a framework with which future quantitative advancements in risk assessment may be made. Kaplan and Garrick (1981) suggested risk, R, be defined based on a set of triplets,

$$R = \{<s_i, p_i, x_i>\},\ i = 1,\ 2,\ \ldots n \tag{1}$$

where s_i refers to the ith scenario or set of conditions, p_i is the probability of the ith scenario occurring, and x_i is the consequence of the ith scenario. Risk is quantified under this structure by tabulating triplets for all potential scenarios and computing a cumulative probability curve. Site susceptibility and storm intensity (or return interval) in our model of hydrologic vulnerability (Fig. 6) define the ith scenario (s_i), resulting in the ith hydrologic response or consequence (x_i). Vulnerability curves shown for the respective storm intensities in Fig. 6 can be thought of as a family of risk curves (Kaplan and Garrick 1981). The probability of the ith occurrence (p_i) and hydrologic response (x_i) is the combined probability of susceptibility and storm occurrence that define the ith scenario. The potential for damages to values-at-risk is associated with the magnitude of the hydrologic response (x_i), shown by vulnerability curves in Fig. 6, and those damages can be considered as secondary

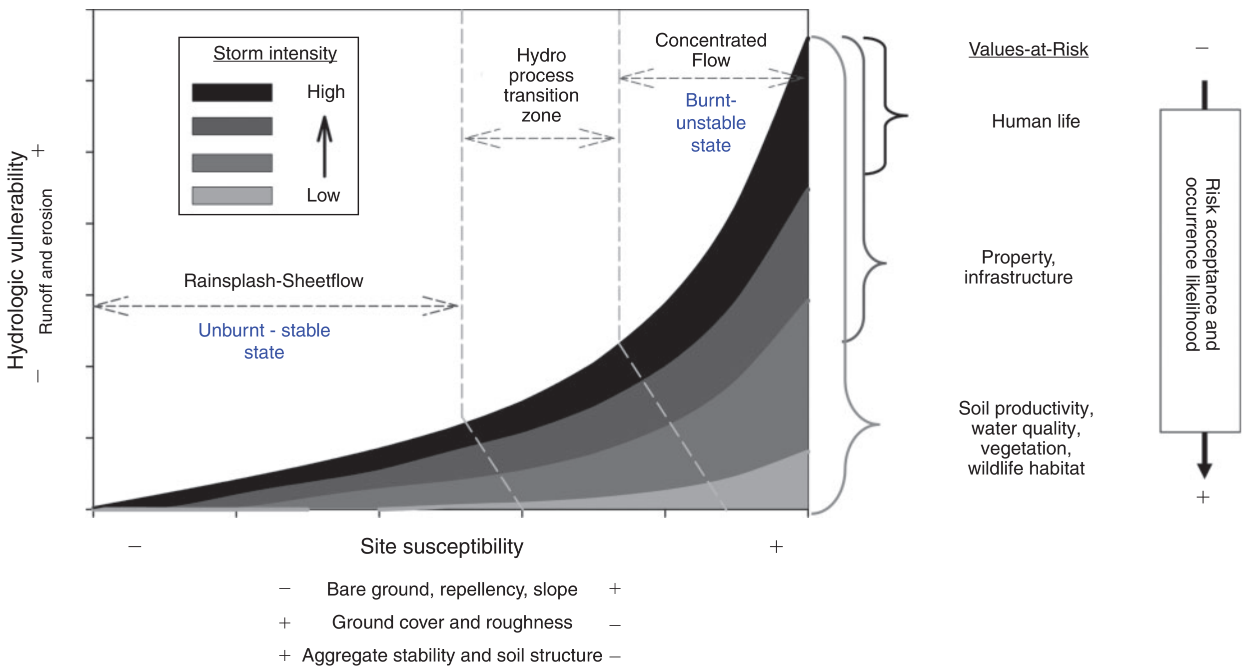

Fig. 6. Conceptual model of hillslope-scale hydrologic vulnerability (runoff and erosion response, *y*-axis) for varying site susceptibility (*x*-axis) and rainfall intensity. Site susceptibility is defined by the surface soil, ground cover and topographic conditions that affect runoff and erosion responses. Symbols indicate directional increase (+) or decrease (–) in respective variable. Hydrologic vulnerability and response increases exponentially as ground cover, roughness, aggregate stability and soil structure decrease, and bare ground and soil water repellency increase. Responses are amplified with increasing hillslope angle. Rainsplash and sheetflow processes dominate on gentle portions of the vulnerability curves, where conditions are hydrologically stable (unburnt state); concentrated flow dominates where curves steepen and conditions become hydrologically unstable (burnt state). The transition zone occurs where decreased surface protection or increased water availability facilitate concentrated flow initiation. Hydrologic responses are generally greater with increasing rainfall intensity. Potential values-at-risk for varying magnitudes of hydrologic response are shown to illustrate potential consequences of respective runoff and erosion events. Acceptance and likelihood of damage to values-at-risk are illustrated by the arrow on the right side of the figure. Figure is modified from Pierson *et al.* (2011).

consequences resulting from the x_i hydrologic response. Of course, assessing potential harm to values-at-risk requires knowledge of the storm and/or runoff magnitudes necessary to cause the respective damage (for example, see Cannon *et al.* 2008, 2011). Clearly, such damages occur on western US landscapes (Cannon *et al.* 1998, 2001*a*; Meyer *et al.* 2001; Moody and Martin 2001*a*; Pierson *et al.* 2002; Klade 2006; Cannon *et al.* 2008) and their occurrences will likely be amplified by ongoing increases in wildfire activity. We propose that recent advances in understanding and quantification of fire effects from small-plot to hillslope scales provide an initial point for populating fire effects models in a probabilistic framework that incorporates probabilities of site susceptibility, storm occurrence and magnitude of hydrologic response (e.g. Robichaud *et al.* 2007; Cannon *et al.* 2010). The Erosion Risk Management Tool (Robichaud *et al.* 2007) is one model that, in part, utilises the above conceptual framework to predict hillslope-scale soil erosion in probabilistic terms based on site-specific climate, vegetation, soil texture, burn severity and topography.

Knowledge gaps in the assessment of post-fire hydrologic risk

This review of post-fire hydrology and erosion studies offers insight into potential confounding issues in the field interpretation of post-fire hydrologic vulnerability. The studies reviewed from burnt rangelands and forested sites (Table 1) demonstrate that field assessments may be challenged by spatial and temporal variability in fire effects and post-fire site conditions, and inherent differences in recovery rates for runoff *v.* erosion. For example, Pierson *et al.* (2002) found that runoff and erosion on burnt sagebrush rangeland was significantly greater on south-facing than on north-facing slopes 1 year following wildfire. Overland flow generated on south-facing slopes during a convective thunderstorm caused intense flash flooding. Assessment of north-facing slopes alone would not have detected the potential storm response. Runoff and erosional responses may also exhibit temporal variation that masks fire effects. Annual variation in climate influences vegetative recovery, litter recruitment, soil erodibility and soil water repellency. Pierson *et al.* (2008*a*, 2008*b*, 2009) reported that temporal variability in naturally occurring (not fire-induced) soil water repellency on burnt sagebrush sites exerted greater influence on runoff than did direct fire effects. Pierson *et al.* (2009) observed that soil erosion from burnt sagebrush coppices exhibited significant temporal variability, but it was not determined whether this resulted from differences in infiltration, runoff or erodibility. Finally, the conditions required for hydrologic stability differ for runoff *v.* erosion, and for rainsplash–sheetflow processes *v.* concentrated flow (Pierson *et al.* 2008*a*; 2009). Pierson *et al.* (2008*a*, 2009) and Benavides-Solorio

and MacDonald (2001, 2002) have shown that fire effects are greater with respect to erosion than to runoff. Our review of field studies on fire effects indicates that post-fire assessments focusing on one aspect of hydrologic vulnerability (e.g. runoff) or on one process (e.g. rainsplash) may not accurately reflect fire effects. Meaningful field studies of landscape-scale effects may require multiple year assessments, annual control treatments and field evaluation of runoff and erosion at different scales, and should include assessment of rainsplash, sheetflow and concentrated flow processes. However, such comprehensive studies are seldom possible or practical. Investigations that focus on a single hydrologic parameter or process at only one scale should therefore acknowledge the potential errors associated with broad-scale inferences on overall hydrologic vulnerability.

The qualitative model presented in this study (Fig. 6) illustrates the general hydrologic and erosional relationships affected by ongoing plant community transitions and increased fire activity in the western US, but our current ability to populate the model relating to this problem is confounded by several key issues. First, we are still learning how the variables that define site susceptibility at different spatial scales interact to influence hydrologic and erosion responses. Second, current understanding is inadequate with regards to quantifying effects of within-storm varying rainfall intensity and site conditions. Third, knowledge of how to incorporate soil water repellency and its inherent variability in space and time into predictive models is particularly limiting. Fourth, runoff and erosion data are extremely scant for many plant communities. Finally, advancements in predictive erosion models have been made (e.g. Robichaud *et al.* 2007; Nearing *et al.* 2011; Wagenbrenner *et al.* 2010; Al-Hamdan *et al.* 2012*b*), but most models remain focused at the hillslope scale given the lack of watershed-scale data sources. Spatial scaling of hydrologic and erosion processes has long been difficult for scientists, which remains a problem for landscape-scale modelling. Scaling limitations further inhibit linkages of plot- and hillslope-scale responses to offsite effects on values-at-risk (Cawson *et al.* 2012). Nevertheless, current models based on plot-to-hillslope scale knowledge provide a means of predicting post-fire hillslope responses and evaluating mitigation efforts.

Summary and conclusions

Increased wildfire activity associated with cheatgrass invasions, plant community transitions and a warming climate along the rangeland-xeric forest continuum in the western US poses hydrologic risks to natural resources, property and human life. Large and frequent fires promote loss of biologically important soils and increase the likelihood of damaging flood and mass erosion events. Projections of climate warming suggest that current trends towards an increase in wildfire activity are likely to continue. Future climate scenarios also predict large-scale shifts in plant communities that may further enhance wildfire activity in the rangeland–xeric forest continuum. Field studies of post-fire runoff and erosion have advanced our understanding of key physical processes and have contributed to hydrologic and erosion model development. In our review, we present a conceptual model of post-fire hydrologic vulnerability and risk based on current understanding, and we identify remaining knowledge gaps that limit post-fire risk assessment. We found that current understanding is lacking in several key areas with regard to quantitative modelling of post-fire hydrologic responses and effects on values-at-risk. Current knowledge is particularly deficient regarding the interacting effects of hydrologic variables (i.e. varying rainfall intensity, infiltration, runoff generation) and spatially variable post-burn conditions and topography. Knowledge of how to incorporate soil water repellency and its variability into hydrologic models is critically limited. Finally, most physically based models are designed to simulate hillslope-scale responses and are not directly applicable to current landscape-scale fires extending across diverse watersheds with steeply sloping xeric forest and rangeland plant communities. Our review suggests that future post-fire risk research should focus on advancing understanding in the key areas noted above and on probability-based modelling of the interacting controls on post-fire responses across relevant spatial scales and for changing climate conditions.

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