SPECIAL ISSUE

Soil carbon and nitrogen eroded after severe wildfire and erosion mitigation treatments

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Abstract. Erosion of soil carbon (C) and nitrogen (N) following severe wildfire may have deleterious effects on downstream resources and ecosystem recovery. Although C and N losses in combustion and runoff have been studied extensively, soil C and N transported by post-fire erosion has rarely been quantified in burned landscapes. To better understand the magnitude and temporal pattern of these losses, we analysed the C and N content of sediment collected in severely burned hillslopes and catchments across the western USA over the first 4 post-fire years. We also compared soil C and N losses from areas receiving common erosion-mitigation treatments and untreated, burned areas. The concentrations of C and N in the eroded material (0.23–0.98 g C kg^{-1} and 0.01–0.04 g N kg^{-1}) were similar to those of mineral soils rather than organic soil horizons or combusted vegetation. Losses of eroded soil C and N were highly variable across sites, and were highest the first 2 years after fire. Cumulative erosional losses from untreated, burned areas ranged from 73 to 2253 kg C ha^{-1} and from 3.3 to 110 kg N ha^{-1} over 4 post-fire years. Post-fire erosion-mitigation treatments reduced C and N losses by up to 75% compared with untreated areas. Losses in post-fire erosion are estimated to be <10% of the total soil C and N combusted during severe wildfire and <10% of post-fire soil C and N stocks remaining in the upper 20 cm of mineral soil. Although loss of soil C and N in post-fire erosion is unlikely to impair the productivity of recovering vegetation, export of C and N may influence downstream water quality and aquatic ecosystems.

Additional keywords: post-wildfire recovery, sediment, watershed biogeochemistry.

Introduction

Observed and projected increases in severe wildfire frequency and extent (Westerling *et al.* 2006; Dennison *et al.* 2014) have elevated concerns about post-fire ecosystem recovery, watershed condition and long-term source-water protection (Findlay and Sinsabaugh 2003; Binkley and Fisher 2012). Fires both remove material from ecosystems and transport it downslope and downstream. For example, the post-fire transfer of nitrogen (N) and phosphorus (P) may deplete limiting soil nutrients from upland ecosystems, while enriching and potentially degrading downstream aquatic ecosystems (Durán *et al.* 2010; Smith *et al.* 2011; Silins *et al.* 2014). Losses of soil carbon (C) and N during combustion and post-wildfire transport of dissolved C and N in surface runoff and leaching have been examined in many forested regions (Bormann *et al.* 2008; North and Hurteau 2011; Santín *et al.* 2015), but the erosion of soil C and N following wildfire has rarely been quantified (Baird *et al.* 1999; Johnson

et al. 2007; Bormann *et al.* 2008). Post-fire erosion can remove up to 60 Mg ha^{-1} of soil material in the years following a severe wildfire due to fire effects on soil structure and loss of surface cover provided by organic-soil layers (Johnson *et al.* 2007; Riechers *et al.* 2008; Robichaud *et al.* 2013a). The concomitant losses of soil C and N from erosion may decrease post-fire nutrient availability and alter ecosystem productivity, while simultaneously affecting downslope and downstream resources (Gälman *et al.* 2008; Tranvik *et al.* 2009; Quinton *et al.* 2010).

High-severity wildfires combust nearly all vegetation and surface organic layers (Keeley 2009; Parsons *et al.* 2010), and have short-term effects on near surface C and N pools. Post-fire leaching and erosion of C and N, by contrast, may deplete soil pools and enrich streams over the course of many years (Rhoades *et al.* 2011; Robichaud *et al.* 2013a). Persistent soil C and N losses after severe wildfire may restrict microbially mediated nutrient-cycling processes, limit ecosystem

Table 1.   Post-wildfire erosion monitoring site information and characteristics

Superscript letters in the 'Monitoring scale' column denote: C, control; L, contour-felled logs; H, hydromulch; S, straw mulch. The 'Sediment-producing storms' listed are the total over the 4-year study period. Storm events producing <20 kg ha^{-1} of sediment not included. I_{10}, ten minute rainfall intensity

Wildfire name; fire ignition date (US State)	Location (elevation, m)	Annual precipitation (mm)	Monitoring scale (contributing area)	Sediment-producing storms	Average event $I_{10} \pm$ s.d. (maximum I_{10}, mm h^{-1})
Valley Complex; 31 Jul 2000 (Montana)	45.91°N, 114.03°W (1725)	400	Catchment (3.6[C], 2.8[L] ha)	7	25.7 ± 18 (59)
Fridley; 19 Aug 2001 (Montana)	45.51°N, 110.78°W (1940)	700	Catchment (13.3[C], 11.8[L] ha)	8	31.1 ± 17 (55)
Hayman[A]; 8 Jun 2002 (Colorado)	39.18°N, 105.36°W (2387)	400	Catchment (3.0[C], 3.1[L], 4.6[C], 5.2[H], 3.3[S] ha)	17	36.7 ± 16 (65)
			Hillslope (33.3 m^2)	23	28.5 ± 16 (72)
Cannon; 15 Jun 2002 (California)	38.45°N, 119.47°W (2230)	300	Catchment (12.6[C], 10.9[L] ha)	3	57.7 ± 67 (134)
Kraft Springs 31 Aug 2002 (Montana)	45.41°N, 104.12°W (1190)	360	Catchment (2.8[C], 3.0[S] ha)	4	36.1 ± 18 (61)
Hot Creek; 23 Jul 2003 (Idaho)	43.76°N, 115.22°W (2310)	1100	Hillslope (72 m^2)	8	24.4 ± 10 (38)
Myrtle Creek; 19 Jul 2003 (Idaho)	48.72°N, 116.46°W (1857)	790	Hillslope (285 m^2)	6	39.7 ± 19 (59)
School; 5 Aug 2005 (Washington)	46.22°N, 117.66°W (1686)	1380	Hillslope (212 m^2)	9	23.0 ± 19 (35)

[A]Two separate scale areas within the Hayman Fire perimeter were used for monitoring.

productivity (Bauhus *et al.* 1993; Certini 2005) and delay post-fire vegetation recovery. Further, these losses may contribute to long-term increases in stream water C and N observed after severe wildfire (Rhoades *et al.* 2018) and at sites across the western USA (Rust *et al.* 2018).

Post-fire erosion rates vary widely, with extreme rates associated with high-intensity rainfall events (Debano *et al.* 1998; Groen and Woods 2008; Robichaud *et al.* 2013a). Similarly, rates of soil C and N removal by post-fire erosion (280–640 kg C ha^{-1} and 14–110 kg N ha^{-1}) respond to large rainfall events the initial years following wildfire (Baird *et al.* 1999; Johnson *et al.* 2007; Bormann *et al.* 2008). Erosion mitigation mulch and log barrier treatments applied shortly after severe wildfires cover bare soils and impede overland flow to reduce the erosive effects of high-intensity storm events (Robichaud and Ashmun 2012). These treatments are also likely to limit C and N losses from upland soils, although their effects have not been evaluated after severe wildfire.

The high variability in post-fire erosion rates generated by storm events, combined with additional spatial variability associated with wildfire severity and watershed characteristics (Shakesby 2011; Wagenbrenner and Robichaud 2014), require well-replicated studies conducted at multiple spatial scales in order to adequately quantify post-fire C and N losses. Such information will provide an estimate of the magnitude of soil C and N eroded from upland landscapes relative to other wildfire-related losses and will permit a comparison of the biogeochemical consequences of post-fire erosion control treatments. Here we measure C and N in sediment collected from hillslopes and catchments burned by severe wildfires at eight locations across the western USA. We investigate temporal patterns of soil C and N erosion over the first

4 post-fire years and compare losses among various erosion mitigation treatments. We expect soil C and N losses to be reduced by the mitigation treatments that reduce erosion to the greatest extent. This evaluation will help determine whether soil C and N losses represent a threat to the productivity of ecosystems recovering from wildfire, as well as if transport may impair downstream water quality.

Methods

We studied eight long-term wildfire monitoring sites (Robichaud *et al.* 2008, 2013a, 2013b) for a period of 4 years following each fire (Table 1). The contributing area of each site had $>70\%$ high soil-burn severity from wildfire. High soil burn severity is characterised by the near-complete loss of pre-fire ground cover and surface organic matter, with post-fire soil surfaces of barren mineral soil and ash cover (Parsons *et al.* 2010). Assessments of soil-burn severity were made by the US Forest Service Burned Area Emergency Response (BAER) team at each fire site. Pre-fire vegetation was a variety of conifer species with a diverse mix of grass, forb and shrub species (Robichaud *et al.* 2008, 2013a, 2013b). Soil texture varied between sites from sandy loam to silt loam. Average annual precipitation for individual sites ranged from 300 to 1380 mm year^{-1}. During the study, the majority of sites experienced three to ten sediment producing storm events, with exception of the Hayman site which experienced more than 17 sediment producing events (Table 1).

Four sites had hillslope-scale sediment traps (20–250-m^2 contributing area), five quantified sediment output at the catchment scale (3–13-ha contributing area) and one site measured sediment at both scales. Hillslope-scale sediment traps or catchment basins were constructed in paired locations with
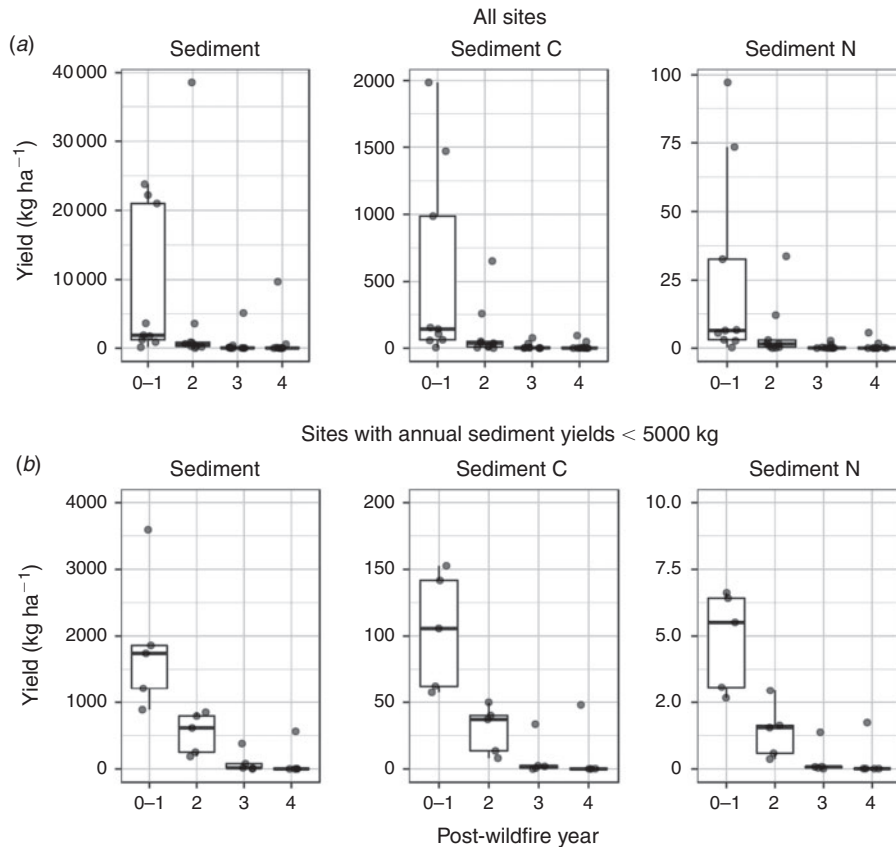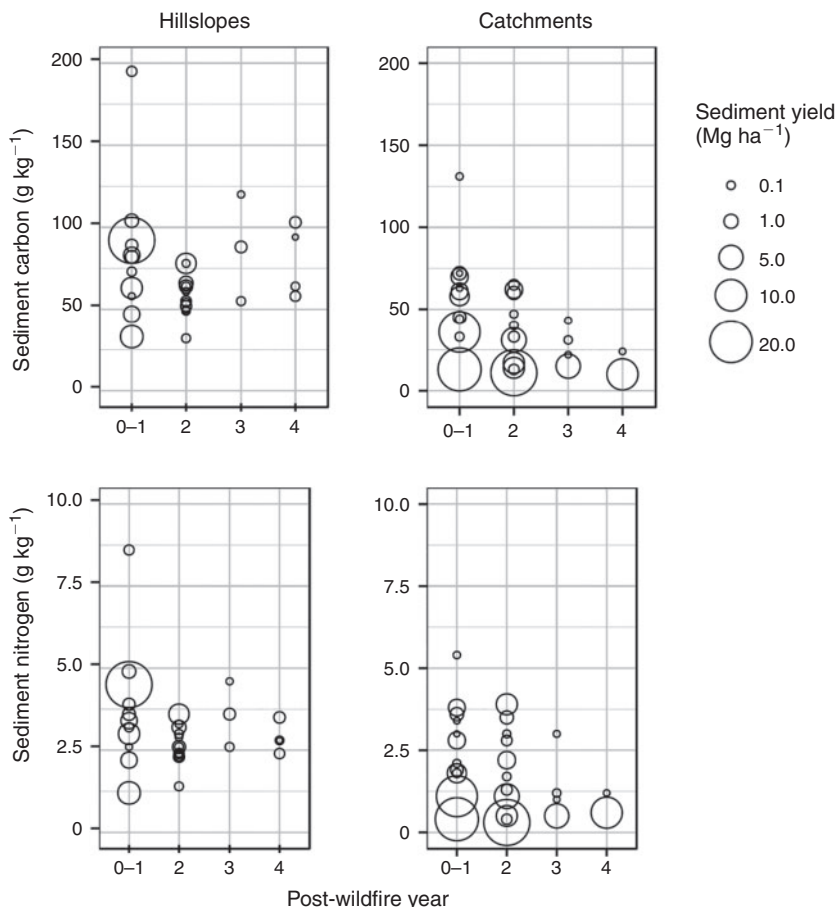
Fig. 1. Annual sediment, sediment carbon and sediment nitrogen yields from untreated areas. (*a*) Yields from all monitoring sites (includes yields from the Hayman sediment trap site and catchment site as separate points). (*b*) Excludes yields from three of the eight monitoring sites (Hayman, Fridley and Cannon) where extreme rainfall events led to cumulative sediment yields >5000 kg ha^{-1}.

similar terrain and contributing area to compare untreated, burned areas and those with erosion-mitigation treatments (Table 1; Robichaud *et al.* 2008, 2013*a*, 2013*b*). One untreated, control catchment was compared with all the treated catchments, except for the Hayman site, where two untreated catchments were monitored and sediment C and N yields were averaged. At hillslope sites, control and mitigation-treatments plots were replicated along the same hillslope contour; the number of replicates ranged from three to nine depending on the size and uniformity of the site.

Sediment traps and catchment basins were constructed as soon as logistically possible after each fire. Hillslope-sediment traps were constructed within a few weeks following the end of the fire, and the larger catchment traps were installed within 2–3 months. On-site inspection ensured that post-fire erosion had been minimal before site establishment. Eroded sediments were collected from traps for all sediment-producing storm events throughout the 4-year study period (Robichaud *et al.* 2008, 2013*a*, 2013*b*). Prior to sediment sampling, large pieces of organic material, such as straw mulch residue, newly fallen leaves and woody debris, were removed by hand. Storm events that produced less than 1000 kg of sediment were removed from sediment fences and traps with buckets and weighed on-site. For sediment deposits weighing >1000 kg, sediments were removed with mechanical equipment and an estimate of the sediment weight was based on sediment volume and bulk density. Sub-samples of sediment for C and N analysis were taken from each bucket or mechanical equipment load by collecting equivalent volumes of sediment from multiple locations throughout each load or bucket. All sub-samples were then combined for each sediment trap and storm event.

Sediment samples were dried for 24 h at 100°C to determine sediment dry weight. Samples were homogenised and passed through a 2-mm mesh sieve before sub-sampling for C and N analysis. Sediment C and N content was determined by dry combustion at 850°C (Leco TruSpec autoanalyser, St Joseph, MI, USA). Total dry-sediment mass was divided by contributing area to calculate hillslope and catchment sediment yields. For catchments, the C and N yield from each storm event cleanout was determined by the product of the sediment yield and the C and N concentration of the collected sediment. For hillslope sites, the average sediment yield from replicate sediment traps within same treatment type was multiplied by the average C and N content of the sediment samples taken during storm cleanouts. Annual yields were calculated as the dry weight sum of all sediment collected in a standard calendar year divided by the associated contributing area.

Fig. 2. Carbon and nitrogen content of sediment collected from untreated hillslope and catchments after individual storm events throughout the 4-year study period.

Sediment C and N concentrations and sediment C and N yields were compared over time, across treatment types and between sampling scales (hillslope or catchment) in a generalised linear mixed-effects model (SAS Institute Inc., Cary, NC, USA). Treatment type, post-wildfire year, and either hillslope or catchment scale were used as fixed effects within the model. Site was used as the random effect. Sampling scale × treatment type within site was also used as a random effect to account for variation between plots with different scale and treatment combinations. The model incorporated a repeated-measures structure on the residuals to account for measurements taken from the same plots over time. In the model for sediment-yield comparisons, a lognormal distribution was used to transform the yield data to better approximate a normal distribution. For the same purpose, a β distribution was used to transform the confined values (e.g. 0–100%) for sediment C and N concentrations. Sediment yield and concentration differences were compared using the least-squares mean estimates with a Tukey–Kramer adjustment to account for uneven sample sizes (Kramer 1956).

Results

Across all sites, the median cumulative sediment C and N yield over the first 4 post-wildfire years was 160 kg C ha^{-1} and

8.1 kg N ha^{-1} with individual site yields ranging from 73 to 2253 kg C ha^{-1} and from 3.3 to 110 kg N ha^{-1} (Fig. 1a). No significant differences in C and N yield were found between sites, nor between hillslope and catchments scales. Annual sediment C and N yields were greater the first 2 years after fire than in years 3 and 4 ($P < 0.018$), with the greatest annual yields coming in the first year. First-year sediment C and N yields ranged from 4.4 to 1950 kg C ha^{-1} and from 0.25 to 95 kg N ha^{-1} (Fig. 1a). For the majority of the monitoring sites, C and N yields from the first 2 post-wildfire years accounted for >90% of the total sediment C and N yield over the 4-year study period. A small number of storm events were responsible for most of the sediment C and N measured during the study (Fig. 2). Single extreme-storm events at the Cannon, Fridley and Hayman (hillslope) monitoring sites generated the three largest sediment yields observed and caused >98, >95 and >76% of their respective total sediment, C and N yields. The total annual yield at sites that did not experience an extreme erosion event was <5000 kg sediment ha^{-1} (Fig. 1b).

Eroded sediment C and N concentrations from untreated areas were highly variable (4–195 g C kg^{-1}, 0.2–8.6 g N kg^{-1}) between individual sediment producing events over the 4-year study period (Fig. 2). No significant difference in sediment C and N concentration was found between individual monitoring sites.

Table 2.   Soil classification, texture, carbon content and dominant over- and understorey vegetation
Mineral soil carbon content values from adjacent unburned forest provided by the Rapid Carbon Assessment Project (2013)

| Study site	Soil classification	Soil texture class	Mineral soil (0–20 cm) carbon content (g kg^{-1})
Valley Complex	Sandy skeletal, mixed, frigid Haplustepts	Gravelly loam	35
Fridley	Loamy skeletal, mixed, typic Agriborolls; Loamy, skeletal, mixed, Mollic Eutroboralfs; Rock outcrop;	Gravelly loam	42
Hayman	Sandy skeletal, mixed, frigid shallow typic Ustorthents; Sandy skeletal, micaceous, shallow Typic Cryorthents	Gravelly coarse sand	46
Cannon	Loamy-skeletal, mixed, superactive, frigid Pachic Haploxerolls	Gravelly silt loam	38
Kraft Springs	Coarse loamy, mixed, superactive, frigid Typic Caliustepts; Coarse-loamy, mixed, superactive, frigid Typic Haplustolls	Fine sandy loam	31
Hot Creek	Loamy skeletal mixed Typic Cryorthent, Lithic Cryorthent, Entic Haploboroll, Typic Xerochrept, Typic Xerorthent, Typic Haplumbrept	Sandy loam	66
Myrtle Creek	Andic Dystrudept, Typic Udivitrands, Vitrandic Dystroxerepts	Ashy sand	33
School	Loamy skeletal, isotic, frigid Vitrandic Argixerolls; Loamy skeletal, mixed, superactive, frigid Lithic Haploxerolls	Ashy loamy sand	33

Sediment C and N concentrations rarely exceeded 1.5 g N kg^{-1} and 50 g C kg^{-1} following large sediment-producing storms ($>$5000 kg ha^{-1}) (Fig. 2). Sediments from hillslope sites contained significantly more C and N than sediments from catchment sites ($P < 0.04$, Fig. 2). The median C and N concentrations in hillslope and catchment sediments were respectively 81 and 50 g C kg^{-1} and 3.4 and 2.5 g N kg^{-1}. We saw no significant differences in C and N concentrations with time since fire.

Wheat straw and wood-strand mulch both decreased the sediment yields ($P < 0.01$), with respective $>$74 and $>$65% reductions in total C and N yield on average, relative to adjacent untreated areas (Fig. 3, Table 3). Additionally, sediment collected from mulched areas had higher C and N concentrations compared with sediment eroded from untreated areas ($P < 0.01$, Fig. 4). For the two sites that had functional contour-felled logs, sediment C and N yields were reduced by 70 and 32% on average. Hydromulch was the least effective treatment with average reduction in total sediment yield of 40 and $<$20% reductions in the yields of C and N respectively (Fig. 3).

Discussion

Severe wildfires reduce soil C and N stocks due to combustion of organic matter during the fire, as well as through subsequent post-fire processes, such as erosion, runoff and leaching (Certini 2005; Caon *et al.* 2014). However, the magnitude and variability for erosive losses of soil C and N have rarely been quantified directly. Our study reports soil C and N erosion over the 4 years following eight severe wildfires in the western USA. The cumulative sediment C and N yields we observed in this large regional study spanned three orders of magnitude (73–2253 kg C ha^{-1} and 3.3–110 kg N ha^{-1}), a range that is much greater than reported previously (Grier 1975; Baird *et al.* 1999). The variability in post-fire sediment losses we observed among sites was associated with differences in climate, storm-event frequency and intensity.

Total post-fire soil C and N losses from combustion, runoff and erosion are substantially higher than the contribution of erosion alone. For example, the cumulative mineral soil losses
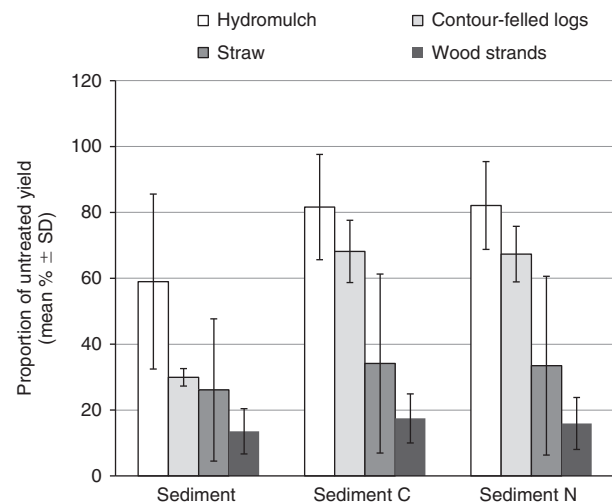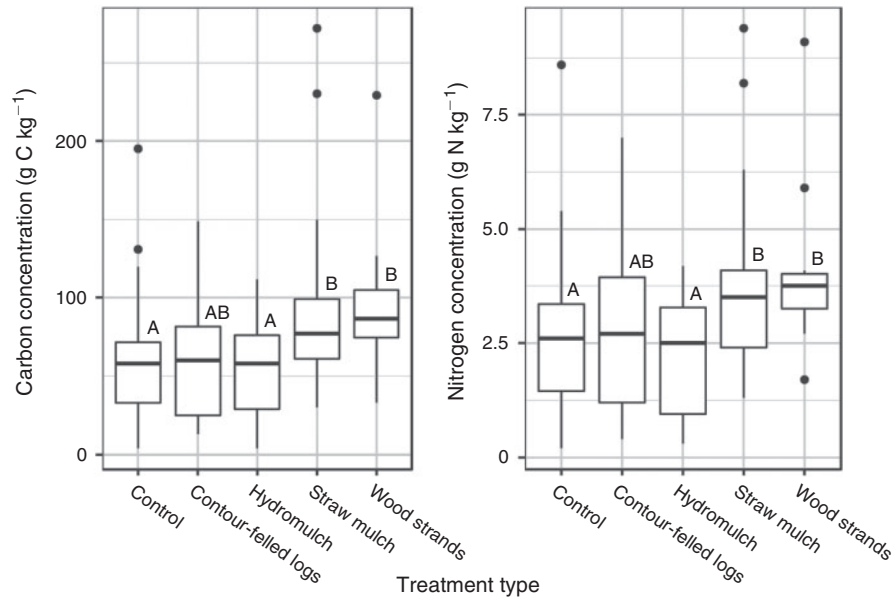

Fig. 3. Cumulative sediment, carbon and nitrogen yields over the 4-year study for areas receiving erosion mitigation treatment. Values are reported as the relative proportion of yields from adjacent untreated areas.

of C and N were 14 000 kg C ha^{-1} and 390 kg N ha^{-1} in the first year after a severe wildfire in Oregon (Bormann *et al.* 2008; Homann *et al.* 2011) and soil N losses were observed from 25 to 110 kg N ha^{-1} in the initial 3 years after the Gondola Fire in the Lake Tahoe Basin (Murphy *et al.* 2006; Johnson *et al.* 2007). Elevated post-fire N losses in leaching and runoff are also common the initial years after wildfire, with first-year losses ranging from 1.1 to 27 kg N ha^{-1} (Smith *et al.* 2011). Based on these studies, we estimate that the C and N eroded from our western USA forest study sites comprise 1–10% of total C and N losses after severe wildfires.

The C and N concentrations measured in eroded sediment (4–195 g C kg^{-1} and 0.2–8.6 g N kg^{-1}) were similar to mineral-soil layers in nearby forests (Fig. 2, Table 2) rather than ash, partially charred or unburned organic-soil or vegetation. Elevated sediment C and N concentrations expected from the

Table 3. Total sediment, carbon and nitrogen yields by mitigation treatment in the 4 years following wildfire
Sediment yields previously published in Robichaud *et al.* (2008) and Robichaud *et al.* (2013*a*, 2013*b*)

Wildfire study site	Treatment	Plot replicates (n)	Sediment (kg ha^{-1})	Carbon (kg C ha^{-1})	Nitrogen (kg N ha^{-1})
Cannon	Control	1	9831	98	5.8
Fridley Peak	Control	1	21 317	286	8.1
Hayman –	Control	2	70 921	1731	70
Catchment	Hydromulch	1	68 386	1764	68
	Contour-felled logs	1	19 344	1343	53
	Wheat straw	1	22 378	1076	39
Hayman –	Control	8	25 947	2253	110
Hillslope	Wheat straw	8	23 455	2289	114
	Wood strands	8	5301	562	26
Hot Creek	Control	6	3678	293	11
	Wheat straw	6	1098	94	3.9
Kraft Springs	Control	1	2604	156	9.6
	Wheat straw	1	0	0	0
Myrtle Creek	Control	6	3781	150	5.8
	Hydromulch	6	1530	94	4.9
	Wheat straw	6	54	2.2	<0.1
School	Control	7	1481	73	3.3
	Hydromulch	7	587	58	2.2
	Wheat straw	7	50	5.4	0.2
	Wood strands	7	99	7.3	0.3
Valley Complex	Control	1	1883	106	4.7
	Contour-felled logs	1	614	62	2.9

Fig. 4. Carbon and nitrogen concentration of eroded sediments collected from areas with differing treatment types. Data include sediment carbon and nitrogen concentrations from both hillslope plots and watershed catchments measured over the duration of the study period at all study sites. Letters above each box plot denote significant differences between mean concentrations.

incorporation of such organic residues (>10% C and >0.4% N, Fig. 2) were measured in few (<8%) samples. This is not surprising because severe wildfires consume nearly all vegetation and surface organic material (Keeley 2009).

Wind erosion of post-fire ash in the months before the onset of our sampling (Cerdà and Doerr 2008) may explain the low C and N concentrations we measured in sediments eroded shortly after each fire. The lower C and N content of sediment eroded from

catchments compared with hillslope sites suggests a greater contribution of C- and N-poor material at the larger scale. Such material may originate from deeper soil horizons exposed by erosion processes or the mobilisation of coarser sediments previously deposited along drainage channels.

We estimate that the mass of soil C and N lost in post-fire erosion is equivalent to the amount contained in the top 1.1 cm of mineral soil in forests of this region. This is consistent with soil-depth losses (0.2 to 1.4 cm) measured after wildfires in conifer forests elsewhere (Baird *et al.* 1999; Johnson *et al.* 2007). Mineral soils in nearby unburned forests contain ~15 000–90 000 kg C ha^{-1} and 800–5000 kg N ha^{-1} within the upper 20 cm (Rapid Carbon Assessment Project 2013; Soil Survey Staff 2016). Thus, the observed post-fire erosion losses are equivalent to <10% of those stocks.

Although wheat straw and wood-strand mulch were most effective, all erosion-mitigation treatments reduced soil C and N losses compared with untreated, burned areas (Fig. 3). The C and N concentration of sediments collected from areas treated with wheat straw and wood-strand mulch were slightly elevated compared with untreated areas (Fig. 4), potentially from reduced rill formation and channel erosion of deeper mineral soils with lower C and N contents. Post-fire mulch treatments also increase organic-substrate availability for soil microbial communities and promote short-term N immobilisation and nitrate retention (Berryman *et al.* 2014; Rhoades *et al.* 2015, 2017). In conjunction with improving physical soil stability, post-fire erosion mitigation treatments are likely to benefit the biogeochemical status of recovering post-fire soils, enhancing soil productivity and overall watershed recovery.

The C and N eroded from burned landscapes may have important consequences for downstream ecosystems, and the mineralisation of organic C and N deposited in eroded sediments may stimulate both terrestrial and aquatic productivity (den Heyer and Kalff 1998; Gudasz *et al.* 2010). For example, eroded sediments deposited in a lake bottom released respectively 20 and 30% of their C and N content within 5 years (Gälman *et al.* 2008). A portion of C and N in eroded sediments also form stable organo-mineral complexes or are buried by subsequent sediment layers that resist mineralisation (Sollins *et al.* 2006; Sobek *et al.* 2009; Gälman *et al.* 2008). Owing to expected increases in severe wildfire occurrence, transfer of terrestrial C and N in post-fire erosion is likely to have increasing relevance both to aquatic C and N cycling and long-term storage.

Conclusions

Post-fire losses of soil C and N were related to total erosion rates, with the greatest losses the first 2 post-fire years. The concentrations of C and N in post-fire sediments were fairly uniform across western US wildfires and were consistent with the erosion of mineral-soil horizons. Cumulative losses of soil C and N were similar at hillslope and catchment scales, although their concentrations in eroded sediments were higher at the hillslope scale. Erosive losses of soil C and N are small by comparison to losses from combustion and are not expected to substantially deplete mineral-soil stocks. Contour-felled logs, wheat straw, wood strand and hydromulch erosion-mitigation treatments reduced total sediment yields and concomitant soil C and N losses. The effects of post-wildfire sediment C and N deposition

on downstream riparian and aquatic environments are unknown, but their significance will increase with the frequency of high-severity wildfires.

References

Baird M, Zabowski D, Everett RL (1999) Wildfire effects on carbon and nitrogen in inland coniferous forests. *Plant and Soil* **209**, 233–243. doi:10.1023/A:1004602408717

Bauhus J, Khanna PK, Raison RJ (1993) The effect of fire on carbon and nitrogen mineralization and nitrification in an Australian forest soil. *Soil Research* **31**, 621–639. doi:10.1071/SR9930621

Berryman EM, Morgan P, Robichaud PR, Page-Dumroese D (2014) Post-fire erosion control mulches alter belowground processes and nitrate reductase activity of a perennial forb, heartleaf arnica (*Arnica cordifolia*). Rocky Mountain Research Station, USDA Forest Service, Rocky Mountain Research Station, Research Note RMRS-RN-69. (Fort Collins, CO, USA)

Binkley D, Fisher R (2012) 'Ecology and Management of Forest Soils'. (Wiley: New York, NY, USA)

Bormann BT, Homann PS, Darbyshire RL, Morrissette BA (2008) Intense forest wildfire sharply reduces mineral soil C and N: the first direct evidence. *Canadian Journal of Forest Research* **38**, 2771–2783. doi:10.1139/X08-136

Caon L, Vallejo VR, Ritsema CJ, Geissen V (2014) Effects of wildfire on soil nutrients in Mediterranean ecosystems. *Earth-Science Reviews* **139**, 47–58. doi:10.1016/J.EARSCIREV.2014.09.001

Cerdà A, Doerr SH (2008) The effect of ash and needle cover on surface runoff and erosion in the immediate post-fire period. *Catena* **74**, 256–263. doi:10.1016/J.CATENA.2008.03.010

Certini G (2005) Effects of fire on properties of forest soils: a review. *Oecologia* **143**, 1–10. doi:10.1007/S00442-004-1788-8

Debano LF, Neary D, Folliott PE (Eds) (1998) 'Fire's Effects on Ecosystems'. (Wiley: New York, NY, USA)

den Heyer C, Kalff J (1998) Organic matter mineralization rates in sediments: a within- and among-lake study. *Limnology and Oceanography* **43**, 695–705. doi:10.4319/LO.1998.43.4.0695

Dennison PE, Brewer SC, Arnold JD, Moritz MA (2014) Large wildfire trends in the western United States, 1984–2011. *Geophysical Research Letters* **41**, 2928–2933. doi:10.1002/2014GL059576

Durán J, Rodríguez A, Fernández-Palacios JM, Gallardo A (2010) Long-term decrease of organic and inorganic nitrogen concentrations due to pine forest wildfire. *Annals of Forest Science* **67**, 207. doi:10.1051/FOREST/2009100

Findlay S, Sinsabaugh RL (2003) 'Aquatic Ecosystems: Interactivity of Dissolved Organic Matter.' (Academic Press: Cambridge, MA, USA)

Gälman V, Rydberg J, de-Luna SS, Bindler R, Renberg I (2008) Carbon and nitrogen loss rates during aging of lake sediment: changes over 27 years studied in varved lake sediment. *Limnology and Oceanography* **53**, 1076–1082. doi:10.4319/LO.2008.53.3.1076

Grier CC (1975) Wildfire effects on nutrient distribution and leaching in a coniferous ecosystem. *Canadian Journal of Forest Research* **5**, 599–607. doi:10.1139/X75-087

Groen AH, Woods SW (2008) Effectiveness of aerial seeding and straw mulch for reducing post-wildfire erosion, north-western Montana, USA. *International Journal of Wildland Fire* **17**, 559–571. doi:10.1071/WF07062

Gudasz C, Bastviken D, Steger K, Premke K, Sobek S, Tranvik LJ (2010) Temperature-controlled organic carbon mineralization in lake sediments. *Nature* **466**, 478–481. doi:10.1038/NATURE09186

Homann PS, Bormann BT, Darbyshire RL, Morrissette BA (2011) Forest soil carbon and nitrogen losses associated with wildfire and prescribed fire. *Soil Science Society of America Journal* **75**, 1926–1934. doi:10.2136/SSSAJ2010-0429

Johnson DW, Murphy JD, Walker RF, Glass D, Miller WW (2007) Wildfire effects on forest carbon and nutrient budgets. *Ecological Engineering* **31**, 183–192. doi:10.1016/J.ECOLENG.2007.03.003

Keeley JE (2009) Fire intensity, fire severity and burn severity: a brief review and suggested usage. *International Journal of Wildland Fire* **18**, 116–126. doi:10.1071/WF07049

Kramer CY (1956) Extension of multiple range tests to group means with unequal numbers of replications. *Biometrics* **12**, 307–310. doi:10.2307/3001469

Murphy JD, Johnson DW, Miller WW, Walker RF, Carroll EF, Blank RR (2006) Wildfire effects on soil nutrients and leaching in a Tahoe Basin watershed. *Journal of Environmental Quality* **35**, 479–489. doi:10.2134/JEQ2005.0144

North MP, Hurteau MD (2011) High-severity wildfire effects on carbon stocks and emissions in fuels treated and untreated forest. *Forest Ecology and Management* **261**, 1115–1120. doi:10.1016/J.FORECO.2010.12.039

Parsons A, Robichaud PR, Lewis SA, Napper C, Clark JT (2010) Field guide for mapping postfire soil burn severity. USDA Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-243. (Fort Collins, CO, USA)

Quinton JN, Govers G, Van Oost K, Bardgett RD (2010) The impact of agricultural soil erosion on biogeochemical cycling. *Nature Geoscience* **3**, 311–314. doi:10.1038/NGEO838

Rapid Carbon Assessment Project (2013) Soil Survey Staff. United States Department of Agriculture, Natural Resources Conservation Service. Available online at: https://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/?cid=nrcs142p2_054164 [verified 26 February 2019].

Rhoades CC, Entwistle D, Butler D (2011) The influence of wildfire extent and severity on streamwater chemistry, sediment and temperature following the Hayman Fire, Colorado. *International Journal of Wildland Fire* **20**, 430–442. doi:10.1071/WF09086

Rhoades CC, Fornwalt PJ, Paschke MW, Shanklin A, Jonas JL (2015) Recovery of small pile burn scars in conifer forests of the Colorado Front Range. *Forest Ecology and Management* **347**, 180–187. doi:10.1016/J.FORECO.2015.03.026

Rhoades CC, Minatre KL, Pierson DN, Fegel TS, Cotrufo MF, Kelly EF (2017) Examining the potential of forest residue-based amendments for post-wildfire rehabilitation in Colorado, USA. *Scientifica* **2017**, 4758316. doi:10.1155/2017/4758316

Rhoades CC, Chow AT, Covino TP, Fegel TS, Pierson DN, Rhea AE (2018) The legacy of a severe wildfire on stream nitrogen and carbon in headwater catchments. *Ecosystems* 1–15.

Riechers GH, Beyers JL, Robichuad PR, Jennings K, Kreutz E, Moll J (2008) Effects of three mulch treatments on initial postfire erosion in north

Central Arizona. In 'Proceedings of the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States', 2–5 December 2002, San Diego, CA, USA. (Ed. MG Narog) USDA Forest Service, Pacific Southwest Research Station, General Technical Report PSW-GTR-189, pp. 107–114. (Albany, CA, USA)

Robichaud PR, Ashmun LE (2012) Tools to aid post-wildfire assessment and erosion-mitigation treatment decisions. *International Journal of Wildland Fire* **22**, 95–105. doi:10.1071/WF11162

Robichaud PR, Wagenbrenner JW, Brown RE, Wohlgemuth PM, Beyers JL (2008) Evaluating the effectiveness of contour-felled log erosion barriers as a post-fire runoff and erosion mitigation treatment in the western United States. *International Journal of Wildland Fire* **17**, 255–273. doi:10.1071/WF07032

Robichaud PR, Lewis SA, Wagenbrenner JW, Ashmun LE, Brown RE (2013*a*) Post-fire mulching for runoff and erosion mitigation; Part I: effectiveness at reducing hillslope erosion rates. *Catena* **105**, 75–92. doi:10.1016/J.CATENA.2012.11.015

Robichaud PR, Wagenbrenner JW, Lewis SA, Ashmun LE, Brown RE, Wohlgemuth PM (2013*b*) Post-fire mulching for runoff and erosion mitigation; Part II: effectiveness in reducing runoff and sediment yields from small catchments. *Catena* **105**, 93–111. doi:10.1016/J.CATENA.2012.11.016

Rust AJ, Hogue TS, Saxe S, McCray J (2018) Post-fire water-quality response in the western United States. *International Journal of Wildland Fire* **27**, 203–216. doi:10.1071/WF17115

Santín C, Doerr SH, Preston CM, González-Rodríguez G (2015) Pyrogenic organic matter production from wildfires: a missing sink in the global carbon cycle. *Global Change Biology* **21**, 1621–1633. doi:10.1111/GCB.12800

Shakesby RA (2011) Post-wildfire soil erosion in the Mediterranean: review and future research directions. *Earth-Science Reviews* **105**, 71–100. doi:10.1016/J.EARSCIREV.2011.01.001

Silins U, Bladon KD, Kelly EN, Esch E, Spence JR, Stone M, Emelko MB, Boon S, Wagner MJ, Williams CH, Tichkowsky I (2014) Five-year legacy of wildfire and salvage logging impacts on nutrient runoff and aquatic plant, invertebrate, and fish productivity. *Ecohydrology* **7**, 1508–1523. doi:10.1002/ECO.1474

Smith HG, Sheridan GJ, Lane PN, Nyman P, Haydon S (2011) Wildfire effects on water quality in forest catchments: a review with implications for water supply. *Journal of Hydrology* **396**, 170–192. doi:10.1016/J.JHYDROL.2010.10.043

Sobek S, Durisch-Kaiser E, Zurbrügg R, Wongfun N, Wessels M, Pasche N, Wehrli B (2009) Organic carbon burial efficiency in lake sediments controlled by oxygen exposure time and sediment source. *Limnology and Oceanography* **54**, 2243–2254. doi:10.4319/LO.2009.54.6.2243

Soil Survey Staff (2016) Natural Resources Conservation Service, United States Department of Agriculture. Web Soil Survey. Available online at: https://websoilsurvey.sc.egov.usda.gov/. [verified 1 December 2016].

Sollins P, Swanston C, Kleber M, Filley T, Kramer M, Crow S, Caldwell BA, Lajtha K, Bowden R (2006) Organic C and N stabilization in a forest soil: evidence from sequential density fractionation. *Soil Biology & Biochemistry* **38**, 3313–3324. doi:10.1016/J.SOILBIO.2006.04.014

Tranvik LJ, Downing JA, Cotner JB, Loiselle SA, Striegl RG, Ballatore TJ, Kortelainen PL (2009) Lakes and reservoirs as regulators of carbon cycling and climate. *Limnology and Oceanography* **54**, 2298–2314. doi:10.4319/LO.2009.54.6_PART_2.2298

Wagenbrenner JW, Robichaud PR (2014) Post-fire bedload sediment delivery across spatial scales in the interior western United States. *Earth Surface Processes and Landforms* **39**, 865–876. doi:10.1002/ESP.3488

Westerling AL, Hidalgo HG, Cayan DR, Swetnam TW (2006) Warming and earlier spring increase western US forest wildfire activity. *Science* **313**, 940–943. doi:10.1126/SCIENCE.1128834