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Effects of wildfire ash on water chemistry and biota in South-Western U.S.A. streams

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SUMMARY

1. We monitored streams within the Gila River drainage in south-western New Mexico, U.S.A., over a 5-year period, to investigate the influence of ash input on water quality and stream biota following forest wildfires.
2. Nutrients [ammonium, nitrate, soluble reactive phosphate (SRP)], potassium and alkalinity were most affected by fires; all were increased in stream water following ash input. Concentrations of each returned to prefire conditions within 4 months. Ammonium and nitrate also increased in stream water as a result of atmospheric fallout (e.g. smoke) from fires outside the catchment.
3. Periphyton biomass was not affected significantly by wildfires, although there was a shift in diatom assemblage to smaller more adnate taxa. *Cocconeis placentula* was frequently the dominant postfire species.
4. The influence of wildfires on macroinvertebrates ranged from minimal to dramatic reductions in density depending upon the duration of ash flows and the characteristics of the ash material that entered each system. Macroinvertebrate densities returned to prefire conditions within 1 year.
5. An *in-situ* ashing experiment was conducted on a first-order stream in the Gila River drainage to monitor on-site physiochemical and biotic responses during and after fire ash addition, for comparison with ash delivery from real wildfires on monitored streams. Physical–chemical parameters and algae and macroinvertebrates were monitored in an ashed and upstream reference reach for 13 months. Results generally substantiated findings from monitored streams.
6. Concentrations of major ions and nutrients, as well as turbidity, conductivity and pH, increased immediately in stream water below the point of ashing, while dissolved oxygen decreased. Changes in water chemistry were short-lived (=24 h) except for SRP. The concentration of SRP in stream water was significantly higher in the ashed reach than the control reach for at least 1 month after ash input.
7. Periphyton biomass and diatom assemblages were not significantly altered in the ashing study, whereas macroinvertebrate density was measurably lower in the ashed reach for nearly a year. Macroinvertebrate drift was over 10-fold greater in the ashed reach compared with the reference reach during ashing. Dissimilarity between macroinvertebrate communities in the reference and ashed reaches was significantly greater than variation within reaches for nearly a year.

Keywords: diatoms, macroinvertebrates, streams, water quality, wildfire

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Introduction

Information on the role of forest fires on the chemical composition of lotic systems is somewhat limited and often contradictory. For example, concentrations of nitrogen and phosphorus increased in streams in Yellowstone National Park and in Montana, U.S.A., following wildfires (Spencer & Hauer, 1991; Brass *et al*., 1996). Belillas & Rodà (1993) also reported a slight increase in nitrate concentration in the dry heathlands of north-eastern Spain and Britton (1991) reported an increase in nitrate concentration in a South African mountain stream following a prescribed burn, but not a significant increase in phosphorus. In contrast, there were no major changes in water quality following a prescribed burn in the chaparral of Arizona (Davis, 1989) nor in streams or lakes in the Superior National Forest, MN, U.S.A., following wildfires (McColl & Grigal, 1977). These contradictions presumably result from the multitude of variables that influence the impact of fire disturbance, including fire intensity and duration, geomorphology, geochemistry, soil type and distribution, amount and type of vegetation, season, recent and historic climate, and amount of time until precipitation following a fire (Britton, 1991; Townsend & Douglas, 1997).

The influence of fires is not limited to changes in water quality, and fires have been shown to impact lake and stream biota, including algae (Robinson, Rushforth & Minshall, 1994), macroinvertebrates (Minshall *et al*., 1995; 2001a; Minshall, Royer & Robinson, 2001b; Scrimgeour *et al*., 2001) and fish (Rinne, 1996). Effects of fires on stream communities depend on fire conditions (Gresswell, 1999) and are often related to the influence of fire on substratum stability (Minshall, Robinson & Lawrence, 1997).

In south-western U.S.A., the peak fire season typically occurs during the summer monsoon months of June until August. This period coincides with intense precipitation and the potential delivery of large amounts of sediment and ash into streams (Blinn, Hurley & Brokaw, 1981; Oberlin, 1995). As a result, impacts on streams in south-western U.S.A. that drain burned catchments may extend far downstream of the immediate burned area as ash material is transported through drainages.

We established a 5-year monitoring programme within and near the Gila National Forest in south-western New Mexico, U.S.A., to collect pre- and postfire chemical and biotic data from streams influenced by fire ash. The study examined the influence of fire-ash runoff on water quality and algal and macroinvertebrate assemblages in major streams in the upper Gila River drainage. We also introduced ash slurry into a first-order stream in the Gila River drainage to monitor physical–chemical and biotic responses during and after fire ash entry. Physical–chemical parameters and algal and macroinvertebrate assemblages were monitored for 13 months following ash treatment. These data were compared with information obtained from the 5-year monitoring programme.

Methods

Site description

Gila river monitoring. Five study sites on the Gila River system in New Mexico were established in 1994 as part of a monitoring programme to investigate the influence of wildfires on streams in south-western U.S.A. Upstream sites included two tributaries (East and West Forks) to the Gila River (Fig. 1). Vegetation in these two catchments was primarily mixed conifer at higher altitudes and ponderosa pine (*Pinus ponderosa* Lawson) and scrub oak (*Quercus turbinella* Greene) lower down. Sampling sites were located at 1680 m a.s.l and wetted channel width averaged 4–5 m at each site.

The main stem of the Gila River was sampled at three locations, all influenced by the Mogollon Creek catchment (Fig. 1). The upstream site, Mogollon (altitude 1400 m), was 50 river km below the East Fork site and immediately below the confluence of the ephemeral Mogollon Creek. A second site was established 26 river km downstream of Mogollon Creek at the Gila River Bird Area (GBA; 1350 m). In addition, water samples were collected at a mid-point between Mogollon and GBA near the town of Cliff, NM.

The Gila River emerges from National Forest near Mogollon and re-enters National Forest just above GBA (Fig. 1). As a result, agriculture, cattle grazing and municipal influences intensify below Mogollon Creek. Vegetation at all three sites was primarily scrub oak. Channel width at base flow averaged 5–6 m. Soil types were diverse throughout the Gila River catchment, and derived primarily from ryholite,

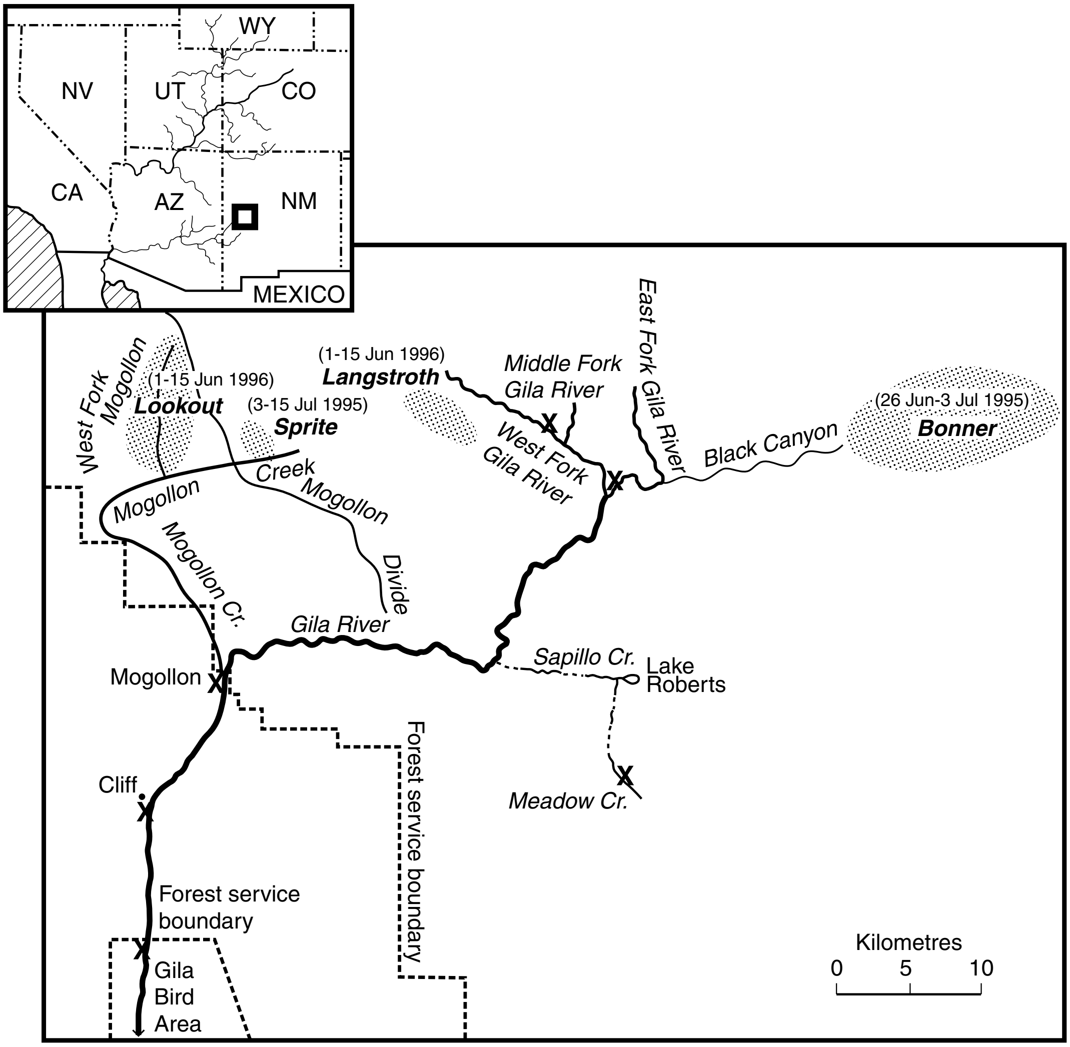

Fig. 1 Study area in the Gila National Forest, New Mexico. The experimental ash study was conducted at Meadow Creek. Monitoring locations for wildfires are denoted by an X. They include the East Fork and West Fork tributaries and the mainstream of the Gila River with sites at Mogollon Creek, Cliff, and the Gila Bird Area. The dates and approximate location and area of each wildfire with respect to the study streams are given on the figure.

Gila conglomerate, limestone, basalt and andesite (C. Souders, pers. comm., Gila National Forest, Silver City, NM).

Four moderate intensity fires occurred in our study area during 1995 and 1996 (P. Boucher, pers. comm., Gila National Forest, Silver City, NM). Such fires often produce hydrophilic soils that result in the delivery of sheets of water and particulates to stream channels during storms following fires. In late June and early July 1995, the Bonner fire burned about 10 927 ha or 28% of the East Fork catchment (Fig. 1). Rains washed fine ash debris into the East Fork of the Gila River for 2–3 days starting on 6 July 1995. Also, during 3–15 July 1995, the Sprite fire burned about 1024 ha or 2% of the Mogollon Creek catchment; however, this area received little summer rain, and ash was not washed down the ephemeral drainage that year. During 1–15 June 1996, the Lookout fire burned some 6880 ha or 5% of the Mogollon Creek catchment adjacent to the area burned during the Sprite fire. Heavy summer (15–19 July 1996) and autumn (14–20 September 1996) rains washed coarse burned and unburned material from both the Sprite and Lookout fires down Mogollon Creek into the Gila River. Also, in early June 1996, the Langstroth fire burned about 2630 ha or 5% of the West Fork catchment. Monsoon rains washed ash debris into the West Fork of the Gila River during the same summer and autumn. The location and approximate area of each burn with respect to the study streams are provided in Fig. 1.

Wildfire monitoring programme

This study was intended to investigate the role of wildfires on streams. Samples were therefore taken in anticipation of and in response to unpredictable fires, mainly around the monsoon season (Table 1). Water samples were filtered (0.45 μm filter) in the field into acid-washed bottles. One set of samples ($n = 3$) was treated with concentrated sulphuric acid to a pH = 2 for analyses of nitrate–nitrogen (NO_3–N), soluble reactive phosphate (SRP) and ammonium (NH_4) on a Technicon® Autoanalyzer II (Pulse Instrumentation, Saskatoon, Saskatchewan, Canada). Another set of water samples ($n = 3$) was treated with concentrated nitric acid to a pH = 2 for analyses of magnesium (Mg), calcium (Ca), sodium (Na) and potassium (K) on a Perkin-Elmer® flame atomic absorption spectrophotometer (Perkin Elmer Inc., Shelton, CA, USA). In addition, samples of unacidified water ($n = 3$) were collected for analyses of chloride (Cl) and sulphate (SO_4) on a Dionex® (DX-100) ion chromatograph (Dionex Corporation, Sunnyvale, CA, USA), and for specific conductance with a Radiometer CDM2 meter with a CDC114 electrode (Radiometer, Copenhagen, Denmark). The detection limit for nutrients (NO_3, NH_4, SRP) was 0.02 mg L^{-1} and 0.1 mg L^{-1} for all other ions. Samples were chilled until time of analysis within the holding times outlined by American Public Health Association (APHA) (1992). Turbidity (FTU) was measured with a Hach DR/2 spectrophotometer (Hach Company, Ames, IA, USA). Silicon dioxide (SiO_2) was measured by the silicomolybdate method (APHA, 1992). pH ($n = 2$) was measured in the field using an Oakton model WD-35615 pH meter (VWR Scientific Products, Denver, CO, USA) and alkalinity was measured by the potentiometric titration method according to APHA (1992). Dissolved oxygen (DO), percent saturation and water temperature (°C) were determined with a YSI model 55 dissolved oxygen meter (Yellow Springs Instruments, Yellow Springs, OH, USA).

Three periphyton samples were collected with a circular open template (2 cm diam.) in each of three different riffle and pool habitats. Each periphyton sample was filtered onto a 0.45-μm glass fibre membrane. Membranes were kept on ice, dried for at least 48 h at 60 °C, ashed at 500 °C for 1 h, and recorded as gram ash-free dry mass (AFDM) of periphyton per square metre (APHA, 1992). After determining AFDM, a subsample of the periphyton was mounted onto glass slides with Hyrax® (Hyrax Company,

Table 1 Sampling dates at study sites within the Gila River drainage, NM. See text for a detailed description and location of the study sites. Letters refer to variables measured during sampling (W) water chemistry, (P) periphyton and (M) macroinvertebrates

	Gila River tributaries		Sites along the Gila River		
Sampling date	East Fork	West Fork	Mogollon	Cliff	Gila Bird Area
27 May 1994	W, P, M	W, P, M			
6 August 1994	W, P, M	W, P, M			
22 October 1994	W, P, M	W, P, M		W	
7 June 1995	W, P, M	W		W	
26 June–3 July 1995	**Bonner**				
3–15 July 1995			**Sprite**		
12 August 1995	W, P, M	W, P, M		W	
14 October 1995	W, P, M	W, P, M		W	
22 May 1996	W, P, M	W, P			W, P, M
1–15 June 1996		**Langstroth**	**Lookout**		
6 August 1996	W, P, M	W, P, M	W, P, M	W	W, P, M
24 October 1996	W, P, M	W, P, M	W, P, M	W	W, P, M
19 May 1997			W, P, M	W	W, P, M
30 June 1997		W, P, M	W, P, M	W	W, P, M
17 October 1997			W, P, M	W	W, P, M
5 June 1998			W, P, M		W, P, M
11 September 1998			W		W

Approximate time of each fire and ash entry into each respective stream is indicated in boldface print.

Auburn, CA, USA) to determine diatom composition. At least 300 valves from two slides were enumerated.

Macroinvertebrates were collected with a Surber sampler (30 × 30 cm; 600 μm mesh) in the same riffles and pools ($n = 3$ for each habitat) sampled for periphyton. Samples were preserved in the field with a mixture of formalin, acetic acid and acetone (FAA; APHA, 1992).

Macroinvertebrates were sorted to the lowest practical taxonomic category (typically genus), counted and reported as number of animals per square metre.

Treatment effects were analysed using SYSTAT version 5.1 (Wilkinson, 1989) statistical software. Water quality variables were compared using Kruskal–Wallis analyses. Periphyton biomass was compared using ANOVA and pairwise Tukey tests on square root transformed data. Macroinvertebrate densities were compared using ANOVA on $\ln(x + 1)$ transformed data. All values are reported as means (±SE).

Meadow Creek ash experiment

A before-after, control-impact type design and analysis experiment was conducted on Meadow Creek because of logistical difficulties in obtaining access to streams during natural wildfires. Meadow Creek is a small, first order, perennial stream in the Gila National Forest, NM (Fig. 1). The stream is located at an altitude of 2300 m in a predominantly ponderosa pine forest. Bank-to-bank width of the stream channel averages <1 m, and base flow during the experiment was approximately 1 L s^{-1}.

On 11 August 1997, 1 140 L of ash slurry from a nearby one-acre plot that recently had been subjected to a fire of moderate intensity was manually delivered into Meadow Creek. The ash slurry was a heterogeneous mixture of material ranging in size from silt to large gravel-sized chunks and was cool at the time of delivery. The ash was made into a slurry using water from Meadow Creek and delivered at a common point into the stream over a 1.25-h period. Biotic samples were collected at locations up to 20 m below the point of ash entry; chemical parameters were measured 5 m below the point of ash entry. A reach 20 m above the point of ash entry was used as a reference site for all collections. Preliminary collections for water quality, periphyton and macroinvertebrate analyses were made in the reference and ashed reaches on 19 May 1997, and immediately prior to the introduction of ash on 11 August 1997.

Water quality measurements ($n = 3$ per period) were taken at 10- and 30-min and 1-h intervals while ash was being introduced, and at 30 min and, 3.5 and 24 h after ash input had ceased. Subsequent collections for water quality were made on 7 September and 18 October 1997, and 6 June and 12 September 1998. Alkalinity, pH, conductivity, dissolved oxygen and turbidity were also measured 1 and 2 h after ash delivery had ceased. All protocols followed those employed on the wildfire monitoring programme.

Periphyton and macroinvertebrates were collected in each of three different riffle and pool habitats above and below the point of ash entry using methods identical to those outlined for the wildfire monitoring sites in the Gila River drainage. Biotic samples were collected 24 h after the introduction of ash slurry and at subsequent intervals corresponding with collections for water quality. Filamentous Chlorophyta were sparse in the system, appearing only briefly in limited abundance late in the summer and were not included in the analyses.

Drift samples were collected simultaneously above and below the point of ash entry with three drift nets (13 × 13 cm aperture; 333 μm mesh) placed in each reach for 10 min. Care was taken to minimise disturbance of the benthos when positioning each net. Preliminary trials indicated no significant difference in drift densities above and below the net during placement. Water height was recorded and water velocity was measured with a Marsh-McBirney® Model 2000 portable water flow meter (Marsh-McBirney Inc., Fredrick, MD, USA) prior to the placement of each net. Drift collections were made prior to the experiment, and starting at 10 min and 1 h during the experiment and 30 min after ash input had ceased. Samples were preserved in the field with FAA, sorted to order, counted and converted to number of animals m^{-3} min^{-1}.

Coarse ash (=2 mm) content in the channel was determined from Surber samples taken immediately before and 24 h after the introduction of ash, and at 1, 2 and 13 months following the ash treatment in both riffle and pool habitats ($n = 3$ each). Coarse ash was manually separated under a dissecting microscope (20×) from non-ash detrital matter, dried at 60 °C for 48 h, weighed and converted to gram ash per square meter.

In addition to the statistical analyses outlined for the monitoring programme, analyses of (dis)similarity similar to those reported by Clarke (1993) and Philippi, Dixon & Taylor (1998) were used to assess the impacts of the ash treatment on diatom and macroinvertebrate communities. We calculated pair-wise distances, or dissimilarities, of samples within groups (ashed versus ashed reach; reference versus reference reach) and of samples between groups (ashed versus reference) based on the presence and abundance of taxa within those samples. If the ash treatment exerted an impact on the biota, we expected between group dissimilarity to be greater than within group dissimilarity. Pair-wise dissimilarities were computed on untransformed data using the groups function in PC-ORD (McCune & Mefford, 1995). Ecological distances between diatom assemblages (% composition) and between macroinvertebrate assemblages (density) were computed using the robust Bray-Curtis (B-C) distance metric (Faith, Minchin & Belvin, 1987; Clarke, 1993). Dissimilarity between groups was expressed as a mean between group distance minus mean within group distance, where a value >0 reflects a treatment effect and a value = 0 reflects no treatment effect. A randomisation technique was used on the distance ranks to test whether between group dissimilarity was greater than within group dissimilarity by an amount greater than expected due to chance (T. Philippi, pers. comm., Savannah River Ecology Laboratory). An alpha level of 0.05 was set in all tests.

Results

Water quality

Alkalinity and selected nutrients were greatly influenced by ash input in the Gila River drainage. Alkalinity, nitrate and ammonium were all high in the East Fork tributary within 6 weeks after ash input from the Bonner fire in 1995 (Table 2). Measured concentrations for nitrate and ammonium in East Fork on four dates in 1994 and 1995 prior to the fire were 0.04 mg L^{-1} compared with 0.08 and 0.1 mg L^{-1}, respectively, 6 weeks after the fire (Table 2). Concentrations of alkalinity, nitrate, and ammonium returned to prefire conditions within 4 months of the Bonner fire. Concentrations of nitrate and ammonium also increased in West Fork 6 weeks after the Bonner fire even though no major fires occurred in the West Fork catchment during 1995 (Table 3). Concentrations of each ion returned to prefire conditions in the West Fork within 4 months of the Bonner fire (Table 3). Alkalinity and nitrate concentrations also increased, while potassium and SRP increased by an order of magnitude in West Fork 1 month after ash input from the Langstroth fire in West Fork catchment in 1996 (Table 3).

Table 2 Concentration [mg L^{-1} (±SE)] of selected water chemistry variables ($CaCO_3$, NO_3–N, and NH_4) influenced by ash entry into the East Fork of the Gila River, NM. Ash from the Bonner fire (July 1995) was washed into the East Fork during heavy rains in late July and early August 1995

Sampling date	$CaCO_3$	NO_3–N	NH_4
27 May 1994	114 (±2)	bdl	0.03 (n = 1)
6 August 1994	108 (±0.1)	bdl	0.04 (n = 1)
22 October 1994	123 (±1)	bdl	bdl
7 June 1995	120 (±1)	bdl	0.03 (±0.01)
Bonner fire (26 June–3 July)			
12 August 1995	**40 (±1)**	**0.08 (n = 1)**	**0.1 (n = 1)**
14 October 1995	115 (±4)	bdl	0.06 (n = 1)
22 May 1996	123 (±1)	0.07 (n = 1)	0.07 (±0.07)
6 August 1996	120 (±1)	0.04 (±0.01)	bdl

The sampling date following ash entry is denoted in bold. All values are n = 3 unless otherwise noted. bdl: below detectable limits.

An increase in selected nutrients was also observed at sites along the Gila River following ash input. Residual ash in the Mogollon drainage from the 1995 Sprite fire, and recently formed ash from the 1996 Lookout fire, was delivered to the Gila River during heavy summer and autumn rains in 1996 (Paul Boucher, pers. comm., Fire Staff Officer, Gila National Forest, Silver City, NM). The concentration of ammonium increased from 0.04 (±0.01) mg L^{-1} on 22 May 1996 to 0.11 (±0.03) mg L^{-1} on 6 August 1996, at the Gila Bird Area site following ash input from these two fires. Although prefire water chemistry data were not available during 1996 for Cliff or Mogollon, the postash concentrations of SRP at all three sites along the Gila River exceeded prefire concentrations of SRP at Cliff and subsequent collections at all three sites. The average measured concentration of SRP at Cliff during 1994 and 1995 (n = 3 dates) was 0.2 (±0.02) mg L^{-1}. In August 1996, less than 2 months after the Lookout fire, the concentration of SRP at Mogollon was 0.63 (±0.01) mg L^{-1}, and decreased downstream to 0.53 (±0.06) mg L^{-1} at Cliff and 0.36 (±0.01) mg L^{-1} at GBA. The concentration of

Table 3 Concentration [mg L^{-1} (±SE)] of selected water chemistry variables (alkalinity, NO_3–N, NH_4 SRP, and K) influenced by ash entry into the West Fork of the Gila River, NM. Ash from the Langstroth fire (June 1996) was washed into the West Fork during heavy monsoon rains. Ash from the Bonner Fire (July 1995) in the East Fork catchment may have been aerially deposited into the West Fork tributary and catchment

Sampling Date	Alkalinity	NO_3-N	NH_4	SRP	K
27 May 1994	62 (±2)	bdl	0.04 (n = 1)	0.12 (n = 1)	2.1 (n = 1)
6 August 1994	70 (±1)	bdl	bdl	0.09 (n = 1)	1.7 (n = 1)
22 October 1994	63 (±1)	bdl	bdl	0.04 (±0.01)	2.0 (n = 1)
7 June 1995	69 (±1)	bdl	0.02 (±0.01)	0.08 (±0.01)	1.4 (n = 1)
			Bonner fire (26 June–3 July)		
12 August 1995	70 (±1)	**0.07 (n = 1)**	**0.10 (n = 1)**	nc	1.7 (n = 1)
14 October 1995	76 (±1)	bdl	bdl	0.06 (n = 1)	0.1 (n = 1)
			Langstroth fire (1–15 June)		
6 August 1996	**101 (±1)**	**0.07 (±0.01)**	**0.02 (±0.01)**	**0.60 (±0.09)**	**5.2 (±1.8)**
30 June 1997	81 (±1)	bdl	bdl	0.09 (±0.09	0.3 (±0.01)

The time of the fire and variables influenced by ash input are denoted in boldface. All values are n = 3 unless otherwise noted. bdl: below detectable limits; nc: not collected.

SRP decreased at all three sites 2 months later to 0.23 (±0.03), 0.19 (±0.01) and 0.22 (±0.01) mg L^{-1} at Mogollon, Cliff and GBA, respectively.

The experimental introduction of ash into Meadow Creek also resulted in immediate changes in water quality. Concentrations of major ions and nutrients (nitrate, ammonium and SRP), as well as, alkalinity, pH, turbidity and specific conductivity, increased rapidly after ash entry (Table 4; Fig. 2). In contrast, dissolved oxygen concentration decreased from 6.3 to 5.7 mg L^{-1} and percent saturation of oxygen decreased from 63% to 57% within 10 min of ash input (Fig. 2). The concentration of SiO_2 was unaltered by the input of ash.

Table 4 Concentration [mg L^{-1} (±SE)] of measured ions in the treatment reach of Meadow Creek, NM on 11 August 1997 immediately prior to the introduction of ash (preash), peak concentration measured while ash was being introduced (peak), and corresponding concentration of ions in the upstream reach at the time of the peak concentration (upstream). Peak concentrations of all ions were reached within 0.5–1.0 h of ash introduction

Ion	Preash	Peak	Upstream reach
NH_4	bdl	1.35 (±0.02)	bdl
NO_3–N	bdl	0.08 (±0.01)	bdl
O-PO_4	0.12 (±0.01)	3.78 (±0.16)	0.12 (±0.12)
Na	5.9 (±0.2)	11.6 (±0.6)	5.3 (±0.1)
K	0.2 (±0.01)	42.0 (±0.3)	bdl
Mg	9.0 (±0.1)	14.2 (±0.1)	8.6 (±0.1)
Ca	16.5 (±0.1)	33.1 (±0.4)	12.7 (±0.1)
SO_4	15.8 (±0.9)	50.5 (±0.4)	15.4 (±0.9)
Cl	1.4 (±0.1)	3.0 (±0.1)	1.4 (±0.1)

All values are mean mg L^{-1} (±SE); bdl: below detectable limits.

Water quality parameters in the ashed reach returned to a concentration similar to that in the reference reach within 24 h of ash input except for SRP, which remained significantly (P = 0.04) higher (0.20 ± 0.01 mg L^{-1}) 1 month later (Fig. 3). No significant difference (P = 0.10) in SRP concentration of the ashed (0.13 ± 0.01 mg L^{-1}) and reference (0.15 ± 0.01 mg L^{-1}) reaches was found 2 months after the introduction of ash.

Coarse ash (2 mm) was transported about 95 m downstream from the point of entry at base flow (1 L s^{-1}) 1 h after ash input, whereas finer ash (silt-like) was transported up to 130 m downstream. A large quantity of ash was present in riffles (40 ± 15 g m^{-2}) and pools (35 ± 10 g m^{-2}) 24 h after ash introduction. Most of the ash had washed out of the riffles 1 month later (5.0 ± 0.1 g m^{-2}) but more had accumulated in pools (245 ± 100 g m^{-2}). The amount of ash did not decrease in pools until 2 months after the experiment at which time it was reduced to 15 (± 10) g m^{-2}. Ash mass in the reference reach was <1 g m^{-2} throughout the study.

Periphyton response

A trend for reduced periphyton biomass in riffle habitats was found in West Fork and riffles and pools at two sites along the Gila River (Mogollon and GBA) following ash entry. However, periphyton biomass was highly variable at all sites throughout the study and was not significantly different after fires or between collections from similar seasons at any of

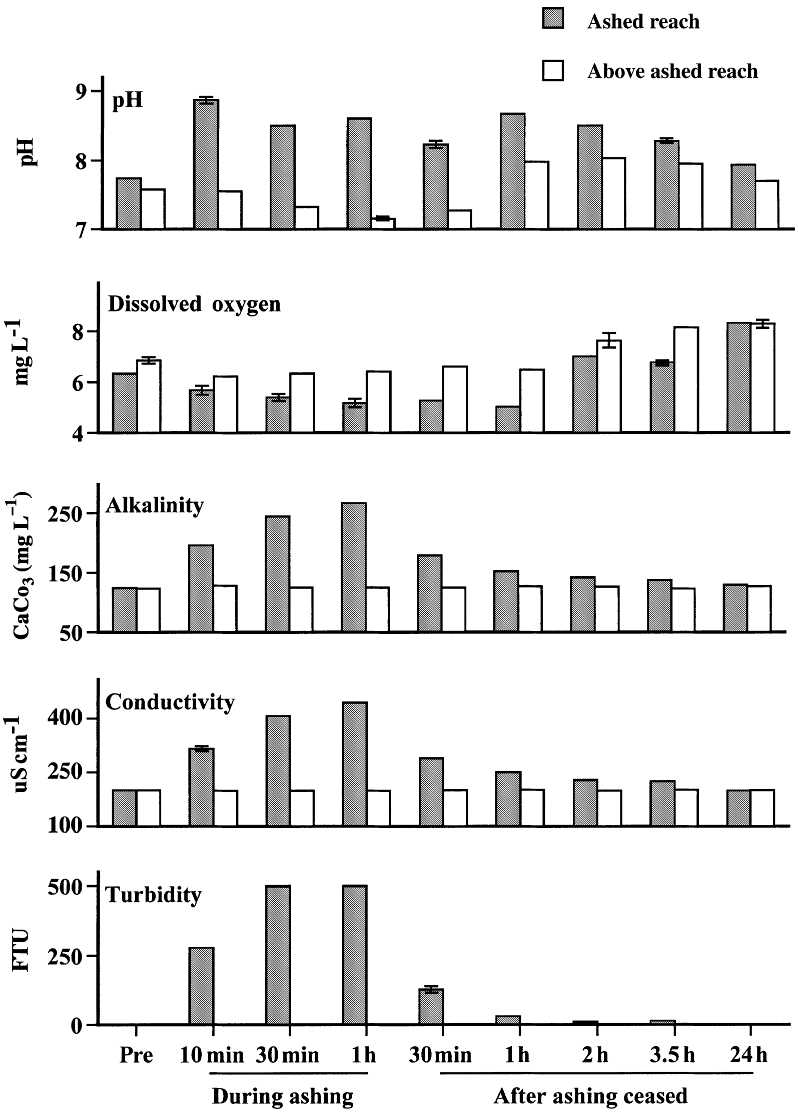

Fig. 2 Mean values (±SE) for selected water quality parameters measured in the ashed reach and above the ashed reach of Meadow Creek at selected periods during the experiment.

the sampling locations. Similarly, periphyton biomass was not significantly different between riffles ($P = 0.61$) and pools ($P = 0.56$) of ashed and reference reaches in Meadow Creek on any collection date.

Although no significant response was noted for periphyton biomass in monitored streams and Meadow Creek, changes in diatom assemblages were apparent in streams with ash input. In both riffles and pools in East and West Fork, there was a shift in the relative abundances of diatom taxa from larger, prostrate species (*Cymbella affinis* Kütz., *Epithemia sorex* Kütz., *Fragilaria ulna* (Nitz.) Lange-Bertalot, *Gomphonema clavatum* (Ehr.) and *Navicula cryptocephala* Kütz.) to smaller, more adnate species (*Cocconeis placentula* Ehr., *Nitzschia frustulum* (Kütz.) Grun., *Achnanthes linearis* (W. Sm.) Grun. and *Reimeria sinuata* (Greg.) Kociolek & Stoermer following ash entry. Riffles showed the greatest decrease in relative proportions of larger diatoms [65 (±13%) to 3 (± 8%) in East Fork; 63 (±8%) to 13 (±7%) in West Fork]. Reductions of larger diatoms in pools were 35 (±4%) to 12 (±8%) in East Fork and 41 (±2%) to 37 (±7%) in West Fork. *Cocconeis placentula* was particularly responsive, exhibiting a five-fold increase in West Fork and a two-fold increase in East Fork within 2 months of ash entry. Similar patterns were noted at Mogollon and GBA. The relative composition of larger taxa decreased by 20 and 13% in riffles and pools, respectively, at Mogollon, and 22 and 9% in riffles and pools, respectively, at GBA.

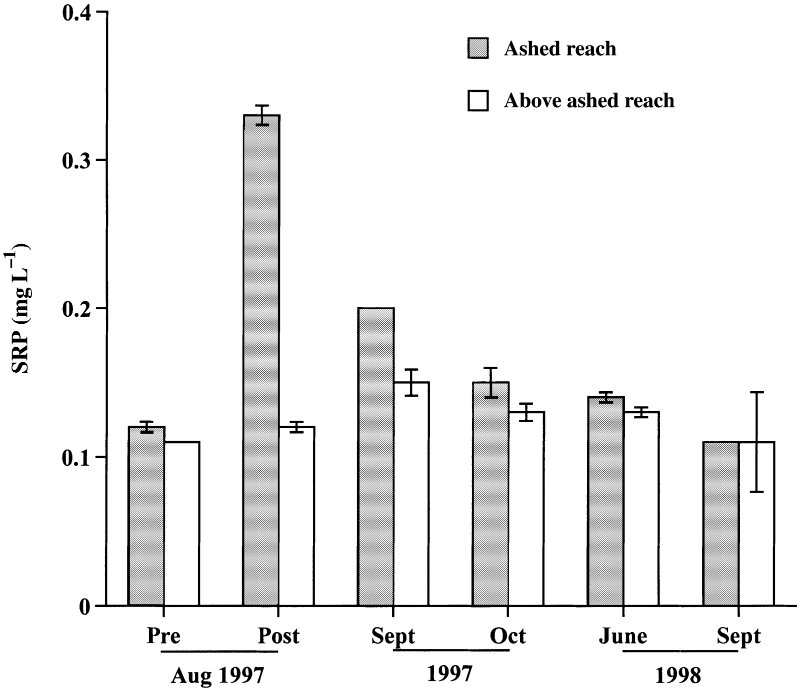

Fig. 3 Mean SRP concentrations (mg L^{-1} ± SE) in the ashed reach and above the ashed reach in Meadow Creek at selected periods during 1997 and 1998. The precollection on 11 August 1997 was immediately before the introduction of ash, whereas the postcollection on 2 August, 1997 was 24 h after the experiment.

In contrast to monitored streams, few clear trends were found in the diatom assemblage following the delivery of ash slurry into Meadow Creek. Analysis of dissimilarity indicated relatively little change in diatom community composition between the ashed and reference reaches in either habitat, except for pools in October 1997, 2 months after the ashing experiment (Fig. 4). The difference in composition was due to the greater relative abundance of the small, adnate species, *Fragilaria famelica* Kütz. in the ashed reach (33%) than to the reference reach (13%). The abundance of *F. famelica* contributed to an overall disparity in physiognomy between the two reaches, with larger, prostrate taxa more abundant in the reference reach (41%) than to the ashed reach (27%).

Because of the small sample size, collection dates were pooled in order to perform the minimum number of permutations necessary to test the significance of whether between group dissimilarity was greater than within group dissimilarity. Dissimilarity between diatom assemblages in the ashed and reference reaches was not significantly ($P = 0.43$) greater than dissimilarity among diatom assemblages within each reach when all postash collections were included in the analyses.

Macroinvertebrate response

No trend was apparent for either total macroinvertebrate density or density of individual taxa in East Fork two months after the Bonner fire. Furthermore, total macroinvertebrate density in both riffle and pool habitats in East Fork was not significantly different between like seasons of different years.

In contrast, ash input to West Fork from the 1996 Langstroth fire had a negative impact on macroinvertebrate density. Total macroinvertebrate densities (100 ± 50 animals m^{-2} in riffles and 230 ± 140 in pools) on 7 August 1996 following ash input were significantly lower in both riffles (3850 ± 1750 animals m^{-2}; $P = 0.005$) and pools (4450 ± 3060 animals m^{-2}; $P = 0.03$) than 1 year earlier on 12 August 1995. Total macroinvertebrate density was also lower although not significantly ($P = 0.42$ in riffles; $P = 0.22$ in pools), than densities 2 years earlier on 6 August 1994 (440 ± 130 animal m^{-2} in riffles; 1170 ± 230 animals m^{-2} in pools). No prefire collections were made in 1996 (Table 1). Diptera, Ephemeroptera (*Epeorus* sp., *Traverella* sp., *Choroterpes* sp., *Tricorythodes* sp. and Baetidae), and Trichoptera (*Polycentropus* sp., *Cheumatopsyche* sp. and Hydroptilidae) were most sensitive to the ash disturbance.

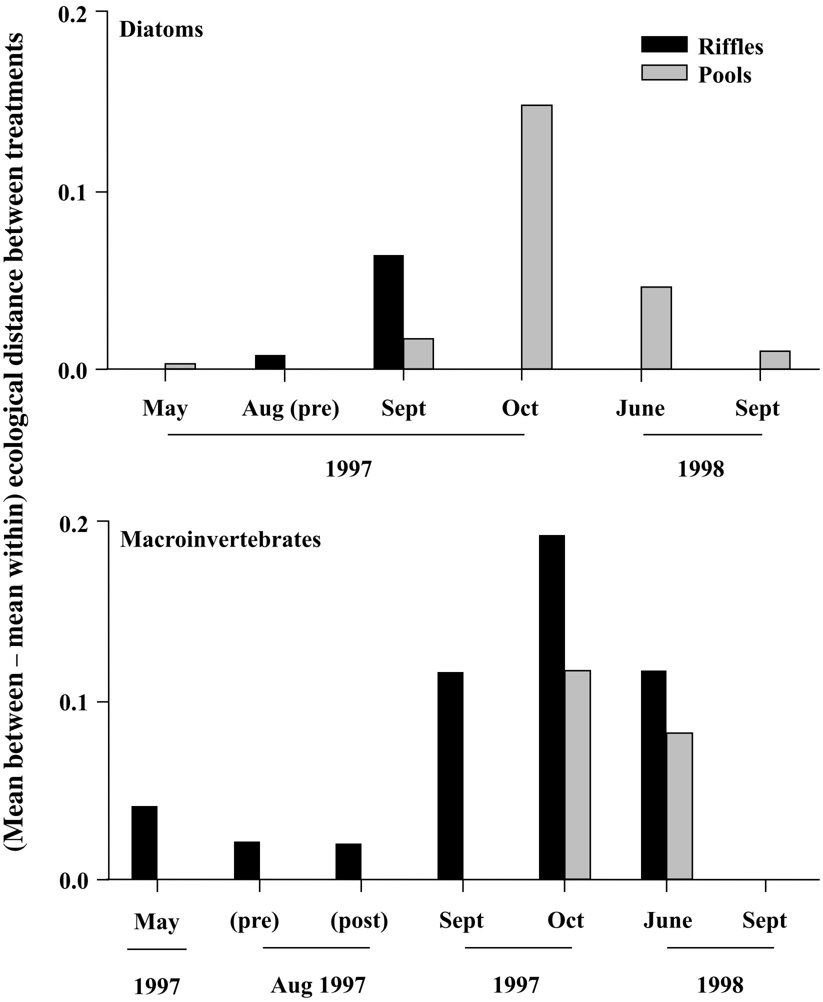

Fig. 4 Mean between group ecological distance minus mean within group ecological distance of diatom and macroinvertebrate assemblages in riffle and pool habitats of Meadow Creek from 19 May, 1997 until 12 September, 1998. Ecological distances were computed from Bray–Curtis distance metrics on percent abundance of diatom taxa and abundance of macroinvertebrates. Values >0 represent greater ecological distance between the ashed reach and the reach above the ashed zone than within reaches (ashed versus ashed and reference versus reference), thus indicating an ash effect. The August precollection is immediately before the introduction of ash and the postcollection is 24 h after ashing ceased.

As at West Fork, there was a substantial reduction in total macroinvertebrate density at GBA following ash entry into the Gila River. Total macroinvertebrate density at GBA was reduced from 8870 (±1409) to 44 (±23) animals m^{-2} in riffles and from 4263 (±1430) to 56 (±36) animals m^{-2} in pools 6 weeks after ash input from the Sprint and Langstroth fires. Although macroinvertebrates were not collected at the Mogollon site prior to ash entry into the Gila River, total macroinvertebrate density in both riffles and pools at both Mogollon and GBA were over three-fold higher in summer collections made on 30 June 1997 and 5 June 1998 than in the summer following ash entry (1996) into the Gila River. All taxa exhibited a reduction in density following the event; however, Arachnida, Diptera, the ephemeropteran, *Seratella* sp. and particularly, the trichopterans, *Cheumatopsyche* sp., *Hydropsyche* sp. and Hydroptilidae, exhibited the greatest decreases in density.

Total macroinvertebrate density in Meadow Creek was measurably lower in both riffle and pool habitats of the ashed reach than the reference reach 2 months after the introduction of ash (Fig. 5). However, total densities in the ashed and reference reaches were not significantly different in either riffle ($P = 0.10$) or pool ($P = 0.35$) habitats on any collection date. Macroinvertebrate densities in the ashed and reference reaches were comparable 1 year after the ashing (Fig. 5).

The ash input also increased the density of drifting organisms. Thirty minutes after ashing, drift density was 8.0 (±2.7) animals m^{-3} min^{-1} in the ashed reach compared with 0.6 (±0.1) animals m^{-3} min^{-1} in the above reference reach. Species of Diptera, Ephemeroptera and Trichoptera comprised >85% of the drifting organisms.

Collection dates on which macroinvertebrate density in the reference reach exceeded density in the ashed reach (October 1997 and June 1998) coincided

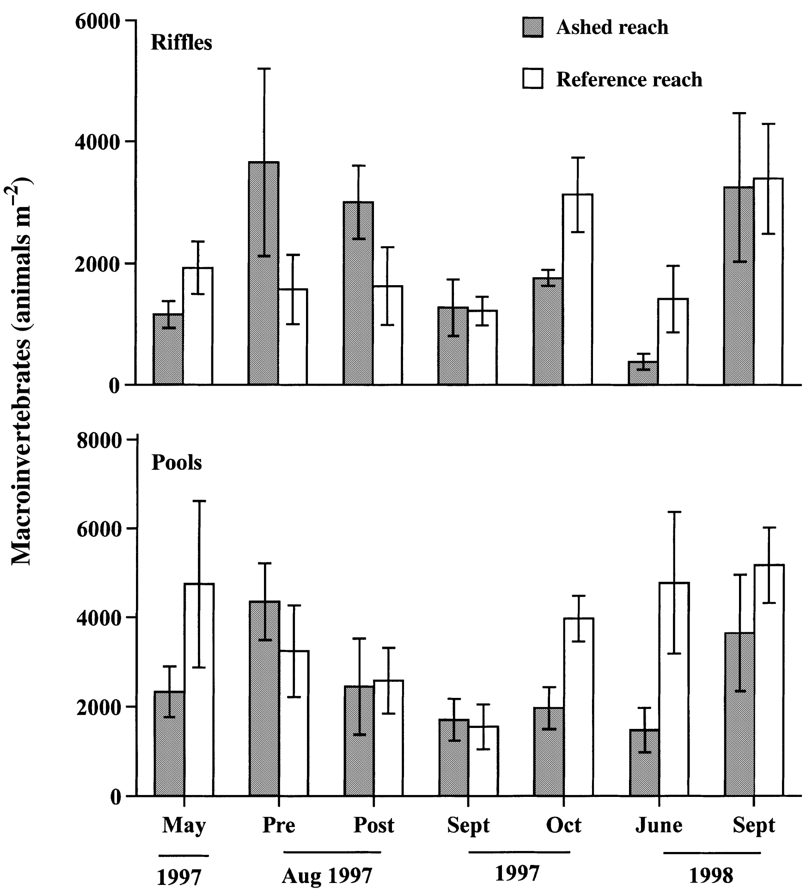

Fig. 5 Mean macroinvertebrate density (animals m^{-2} ± SE) in riffle and pool habitats of the ashed reach and the above reference reach in Meadow Creek from 19 May 1997 until 12 September 1998. The precollection is immediately prior to the ash, and the postcollection is 24 h after ashing ceased.

with these on which the largest discrepancies in macroinvertebrate community structure were found between the two reaches (Fig. 4). Dissimilarity between macroinvertebrate communities in the reference and ashed reaches was significantly greater than variation within reaches in postash collections for nearly 1 year in riffle habitats ($P = 0.03$), but not for pools ($P = 0.17$). While densities of all macroinvertebrate taxa were lower in the ashed reach than the reference reach in October 1997 and June 1998, the greatest discrepancies were for members of Diptera, Ephemeroptera (*Choroterpes* sp., *Tricorythodes* sp.), Trichoptera (*Lepidostoma* sp. and *Gumaga* sp.), Coenagrionidae (Odonata) and Psephenidae (Coleoptera).

Discussion

Selected physicochemical changes were observed after ash entered the streams following wildfires in the Gila National Forest, NM. These findings concur with those other studies in which increases in selected ion concentrations in lakes and streams have been reported following wildfires (Tiedemann, Helvey & Anderson, 1978; Spencer & Hauer, 1991; Brass *et al.*, 1996; Minshall *et al.*, 1997; Hauer & Spencer, 1998; Scrimgeour *et al.*, 2001).

Water quality parameters responded immediately to the experimental introduction of fire-ash slurry into Meadow Creek, although these changes were generally short-lived since most parameters returned to control levels within 24 h of ash input. Cushing & Olson (1963) reported a similar increase in chemical conditions followed by a rapid recovery after a controlled burn near a small stream in Washington, and Spencer & Hauer (1991) reported a return of SRP to background within 2 weeks following a wildfire in Montana. Furthermore, Hauer & Spencer (1998) not only found an immediate response during and following a wildfire in Montana, but also periodic increases in subsequent years associated with spring runoff. These findings indicate that water quality in streams responds instantaneously to the entry of ash following wildfires. The potential rapid recovery of ions to preash conditions indicates that even dramatic changes in stream water chemistry may go undetected if sampling does not coincide with ash delivery, especially with limited ash flow. These dramatic short-term changes in water quality may cause subtle disturbances to stream communities and need further investigation. In general, water quality in the monitored stream sites returned to prefire conditions within 4 months of the respective fires. Although the impact of ash on water quality was generally similar at monitored sites, observed discrepancies among them may be due, in part, to the time of collection relative to ash delivery. Further, the delayed delivery of ash into streams, as noted in the Mogollon catchment, may subject ash to different stages of microbial conditioning prior to delivery.

Climate and weather patterns are among the many factors that contribute to the impact of wildfires on stream conditions (Gresswell, 1999; Minshall *et al*., 2001a). The timing, duration and amount of precipitation influence the quantity and quality of ash delivered to streams. Therefore, streams in different regions (e.g. arid versus mesic), or even in nearby catchments, may exhibit different responses. For example, the Bonner fire in the East Fork catchment burned an area nearly five times that in the West Fork catchment, yet macroinvertebrates were more impacted in West Fork. This may have resulted from several weeks of rain in the West Fork catchment after the wildfire compared with only 2–3 days of rain in the East Fork catchment. Following wildfires in central Idaho, Minshall *et al*. (2001a) also found differences in response by streams following wildfires in central Idaho depending upon the amount and duration of precipitation.

We also observed raised concentrations of nutrients in streams in unburned catchments. Although the Bonner fire was entirely within the East Fork catchment, increased concentrations of ammonium and nitrate in the West Fork tributary corresponded directly to the Bonner fire. This suggests that nutrients can be dispersed aerially onto adjoining catchments. Spencer & Hauer (1991) also reported increased SRP, ammonium and nitrate in streams of north-western Montana, U.S.A., despite minimal overland flow into them following fires. They argued that increases in SRP resulted from the aerial deposition of ash, and that increases in nitrogen were primarily because of the diffusion of smoke across the watershed. In addition, Lathrop (1994) reported that aerial deposition from the Yellowstone National Park (YNP) fire of 1988 had a greater influence on the water chemistry of lakes in the region than ash inputs via overland flow.

Regional soil chemistry is important in determining the chemical influence of ash on streams. Ash flows include a combination of ash *per se* and chemical constituents from surface soils. Under postfire conditions, soil type contributes directly to the chemical loading of streams in subsequent runoff (Gresswell, 1999). For example, drainages with sedimentary rocks, as in the Gila River drainage, may have greater influences on stream alkalinity than those with other rock formations. We noted increased alkalinity in streams under the influence of wildfires in the Gila River drainage and in the Mogollon Creek experiment. Similar changes in stream water chemistry were not observed in these systems during monsoon periods not associated with wildfires (Earl, 1999).

The chemical properties of soil may also determine the mobilisation of certain nutrients, including SRP, as phosphate ions are actively adsorbed to solid surfaces (Stumm & Morgan, 1970). This may explain why SRP remained high in Meadow Creek for up to 1 month, whereas other ions returned to preash concentrations within 24 h.

No significant changes in periphyton biomass were measured at any sites in the monitored Gila River system in response to wildfires, or during the 13-month study at Meadow Creek. Robinson *et al*. (1994) also reported little change in periphyton biomass 2 years after the 1988 fires in YNP. However, they did report a decline of periphyton biomass 3 years later because of increased erosional events in the catchments, particularly in smaller first- and second-order streams. The limited response of periphyton biomass is probably because of the resilience of diatoms to short-term physical–chemical disturbances and to their rapid turnover (Duncan & Blinn, 1989; Steinman & McIntire, 1990).

Changes in diatom community structure after ash entry were more apparent and our findings concur with other studies that report a shift in the relative abundance of taxa following disturbance by wildfire. We observed similar postfire patterns in diatom composition to those reported by Robinson *et al*. (1994, 1996) in streams following the YNP fires and by Rushforth, Squires & Cushing (1986) and Steinman

& Lamberti (1988) in streams impacted by the eruption of Mt St Helens. All these studies noted that smaller, adnate taxa frequently dominated after the disturbances, although changes in composition were short-lived.

Macroinvertebrates showed a greater response to the entry of fire ash than diatoms. However, the high variation in response by macroinvertebrate communities following ash input may be, in part, a function of the characteristics of ash material that entered each system. Ash delivered into the East Fork, where the macroinvertebrate assemblage appeared unaffected, was composed predominately of fine particulate matter and there was little evidence of ash in postfire collections. In contrast, ash input to the Gila River following the Sprite and Lookout fires consisted of coarse burned and unburned detrital material (pers. obs.). The ash flow deposited material up to 40-cm deep in pool habitats at Mogollon and scoured riffles at sites along the Gila River, greatly affecting the macroinvertebrate assemblage at Mogollon and GBA. Evidence of scouring by ash was also noted in Meadow Creek, where macroinvertebrate assemblages were reduced after its introduction. Increased drift rates after ashing provided further evidence of scouring.

Fire ash may alter the detrital food base temporarily as many primary consumers lack the ability to feed on burned organic matter (Mihuc & Minshall, 1995). Therefore, ash may be detrimental to both habitat and food quality in postfire webs. However, we did not observe any influences of ash on macroinvertebrate assemblages during the study at Meadow Creek nor on assemblages in the monitored systems. Furthermore, fire ash affected the entire macroinvertebrate community, and acute responses by individual taxa were not apparent. Rather, the delivery of fire ash appeared to exacerbate the influence of spates through scouring of bed substrata, as evidenced at Mogollon and GBA.

Features of the catchment and riparian community are critical in determining the degree of impact and recovery in lotic systems. Molles (1982) reported that frequent burns in the Sante Fe National Forest, NM, resulted in long-term changes in riparian vegetation that had tremendous effects on macroinvertebrate community structure and stream channel morphology. He reported that frequent fires created patches of early successional aspen forests that favoured caddisfly grazers in adjacent streams, while streams along old-aged coniferous forests favoured caddisfly shredders. Also, Troendle & Bevenger (1996) reported increases in stream flow and sediment transport in a denuded drainage following the YNP fires, conditions that can disturb systems through scouring and the redistribution of substrata in stream channels. Minshall, Brock & Varley (1989) and Minshall *et al*. (2001a) reported that removal of riparian vegetation, sediment movement and channel restructuring were directly related to the percentage of the catchment burned, and concluded that these factors overrode changes in temperature and nutrients in terms of impacts on stream ecosystems. Richards & Minshall (1992) attributed decreased species richness and greater year-to-year variation of macroinvertebrate assemblages in the Middle Fork of the Salmon River, ID, to decreased catchment stability because of loss of upland and riparian vegetation.

While other studies have reported that wildfire disturbances have long-term effects on channel morphometry and stream biota (e.g. Molles, 1982; Minshall *et al*., 2001b), the drainages, we investigated in the Gila River system were resilient and biotic communities recovered within a year of ash entry. We observed fire ash in streams after the first storm following each wildfire, indicating that fire ash was quickly washed from catchments by heavy summer monsoons with discrete events in which water quality changed for a short period of time and biota was scoured depending on amount and characteristics of ash material. This pattern was noted in the recovery of the macroinvertebrate community in the Gila River sites following ash entry from the Sprite and Lookout fires. Postfire macroinvertebrate densities were three times higher during summers of 1997 and 1998 than those reported following ash entry in the summer of 1996.

The catchments within the drainages we studied were mainly ponderosa pine, which tends to have a lower biomass per unit acre and burn more frequently but at lower intensity than mixed conifer forests (P. Boucher, pers. comm.). The intense periodic summer monsoon rains in the south-west, compared with more frequent and greater annual amounts of precipitation in the north-west, may also have contributed to the short-lived nature of fire disturbance in the Gila National Forest. In addition, gradients within catchments in the Gila River catchment are considerably less than those in much of the north-west and less likely to undergo long-term erosion (P. Boucher, pers. comm., Fire Staff Officer, Gila National Forest, Silver City, NM). Therefore, it appears the Gila drainages were not subjected to the major erosional events after wildfires that have been reported in catchments with mixed conifers in north-western U.S.A. (Hauer & Spencer, 1998; Minshall *et al*., 2001a, b) nor to long-term changes in forest stand structure as reported by Molles (1982).

In summary, our study provided a frame of reference for evaluating the effects of fire ash on streams in south-western U.S.A. Our findings from an *in-situ* ashing study and a 5-year wildfire monitoring programme indicated that ash input resulted in dramatic, but relatively short-lived, effects on water quality, while macroinvertebrates exhibited a greater response to fire ash than periphytic diatoms. Other studies have shown that fire disturbance on riparian forests adjacent to streams and erosion from denuded catchments and stream banks have long-term effects on the community structure in lotic systems. These conditions were not observed in the Gila National Forest in south-western U.S.A. Therefore, comparative studies need to be conducted to determine regional differences in the response of other systems and stream communities to wildfires. Many of the changes noted in the monitoring study occurred kilometres downstream of the actual fires, indicating that even small fires can potentially influence stream chemistry and biota over considerable distances, depending upon the duration of ash flows and the characteristics of the ash. Streams and stream biota in south-western U.S.A. have evolved in a landscape characterised by frequent fire regimes (Covington & Sackett, 1984). Fire suppression may have created a condition of greater potential for severe wildfires (Covington & Sackett, 1984) that could have more dramatic impacts on lotic systems as witnessed at YNP.

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