Available online at www.sciencedirect.com
SCIENCE DIRECT®

Forest Ecology and Management 218 (2005) 37–59

Forest Ecology and Management

www.elsevier.com/locate/foreco

Effects of prescribed fire on a Sierra Nevada (California, USA) stream and its riparian zone

Leah A. Bêche *, Scott L. Stephens, Vincent H. Resh

Department of Environmental Science, Policy and Management, 137 Mulford Hall, University of California, Berkeley, CA 94720-3114, USA

Received 9 March 2005; received in revised form 17 June 2005; accepted 17 June 2005

Abstract

Concerns about the effects of fire on ecologically sensitive habitats have limited the use of prescribed fire in the management of forest riparian areas. Using a beyond-BACI (Before-After-Control-Impact) experimental design, we examined the effects of a 26-ha prescribed fire that burned upland and riparian areas of a first-order watershed, and compared this to five unburned sites examined from 1 to 7 years pre-fire and 1 year post-fire. We monitored pre- and post-fire riparian vegetation, large woody debris, sediment, water chemistry, periphyton, and benthic macroinvertebrates. The prescribed fire in the riparian zone was patchy in terms of intensity, consumption, and severity; it consumed 79% of the pre-fire fuel in the riparian zone, 34% of the total surface fuel, and 90% of the total ground fuel. The prescribed fire significantly reduced percent cover of surface vegetation and plant taxa richness in comparison to unburned sites but not plant diversity (Simpson's D). Community composition of understory riparian vegetation changed post-fire, most likely as a result of the reduction in taxa richness and cover. Riparian tree mortality (>11.5 cm DBH) was only 4.4% post-fire. Similarly, there was no post-fire change in large woody debris volume and recruitment, or fine sediment in pools (V^*). Post-fire, there were increases in some water chemistry parameters (SO_4^-, total P, Ca^{2+}, and Mg^{2+}) and a decrease in periphyton biomass; however, these changes were short-term, and recovery occurred in ≤1 year. Macroinvertebrate community composition but not density, richness, or diversity was affected 10–19 d post-fire; composition recovered within 1 year. The trends observed in this study examining multiple abiotic and biotic parameters suggest that this prescribed fire either had no or short-lasting (≤1 year) impacts on Dark Canyon Creek and its riparian zone. The limited observed impacts are at least partially a result of the small portion (<20%) of the watershed area burned, moderate topography, the low- to moderate-severity of the fire, and the relatively low precipitation (and thus, stream flow) that occurred post-fire.

Keywords: Prescribed fire; Stream; Mixed-conifer; Riparian; Macroinvertebrate; Periphyton; Large woody debris; Sierra Nevada; California

1. Introduction

Fire is one of the most important natural disturbances influencing the heterogeneity and diversity of

* Corresponding author. Tel.: +1 510 642 5913; fax: +1 510 642 7428.
E-mail address: leah@nature.berkeley.edu (L.A. Bêche).

0378-1127/$ – see front matter
doi:10.1016/j.foreco.2005.06.010

terrestrial landscapes (e.g., Romme, 1982; Agee, 1993) and aquatic systems (e.g., Resh et al., 1988). Fire can directly and indirectly affect aquatic and riparian communities at spatial scales ranging from microhabitats to entire watersheds, and temporal scales ranging from days to decades (e.g., Minshall et al., 1989; Minshall, 2003). Because of the role that wildfire plays in terrestrial and aquatic ecosystem dynamics, there is an increasing interest in restoring fire to degraded forested landscapes through the use of prescribed burning (e.g., Agee, 1993; Stephens, 1998; Miller and Urban, 2000). For example, prescribed burning has been used successfully to restore mixed-conifer terrestrial habitats in the Sierra Nevada mountains of California (e.g., Keifer et al., 2000), where suppression of frequent, low-intensity surface fires led to severe forest degradation (e.g., Kilgore and Taylor, 1979; Stephens and Collins, 2004).

Research on fire history suggests that fire played an important role in structuring riparian communities (e.g., Everett et al., 2003; Skinner, 2003). Although the importance of fire to riparian ecosystem dynamics is not always clear (e.g., Russell and McBride, 2001), recent research indicates that fire usually occurred less frequently in riparian zones than in adjacent upland areas in California and the western U.S. (Arno, 1996; Russell and McBride, 2001; Everett et al., 2003; Skinner, 2003). However, moisture regimes affect the frequency of riparian fires. For example, in dry climates such as the inland forests of Oregon, fires occurred with similar frequency in both riparian and adjacent upland areas (Olson and Agee, 2005). Similarly, on north-facing aspects (with higher moisture and cooler temperatures), the fire-return interval (FRI) is more similar between upland and riparian forests than on south-facing slopes (Everett et al., 2003).

The use of prescribed fire in riparian zones poses a complex management problem (e.g., Kattelmann and Embury, 1996). Current management practices avoid the use of prescribed burning near aquatic ecosystems (e.g., DeBano and Neary, 1996; Erman, 1996) with few exceptions (e.g., Chan, 1998; Huntzinger, 2003). These practices are designed to prevent the potentially negative impacts (e.g., increased erosion or altered hydrographs) that have been found to occur after wildfires (e.g., Roby and Azuma, 1995; Rinne, 1996; Minshall et al., 1995, 1997, 2001a,b,c).

Prescribed fires may have less severe impacts on aquatic ecosystems than wildfires because they usually are of low- to moderate-intensity, which may cause little mortality to mature trees (e.g., Gresswell, 1999). However, few studies have been conducted on the effects of prescribed fire on riparian and aquatic communities in the western U.S. Furthermore, the applicability of research conducted in other areas and different ecosystems (e.g., Elliott et al., 1999; Townsend and Douglas, 2000; Britton, 1990, 1991a,b) to fire management and the potential responses of stream and riparian communities to fire in western conifer forests is not clear. For example, Britton (1991b) found no effect of prescribed fire on macroinvertebrate communities in South Africa, whereas Chan (1998) found decreases in the diversity and abundance of benthic macroinvertebrates post-fire in mixed-conifer forests of Sequoia National Park, CA, USA, possibly because of an increase in fine sediment deposition.

In this study, a prescribed fire treatment was implemented that was consistent with protocols of actual management burns (i.e., size of the fire, burning conditions, and burn objectives). The exception was that active ignition occurred in the riparian zone. The objectives of this study were to determine: (1) the immediate and direct effects of the fire on riparian plant communities, water chemistry, and physical water quality; (2) the short-term ($\leq$1 year) impacts on the physical stream habitat (sediment composition, channel morphology, hydrology, large woody debris); (3) the $\leq$1 year effects of habitat changes on periphyton and benthic macroinvertebrates; and (4) the time to recovery for physical habitat, periphyton and benthic macroinvertebrates.

2. Methods

2.1. Study site

Blodgett Forest Research Station (BFRS) is located in El Dorado Co., CA, USA (38°55′N, 120°40′W). The six study sites were located in four first-order streams and one second-order stream. For the former, two sites were separated by 400 m in Bacon Creek (B1 and B2), and one site was located in Dark Canyon (D1), Deep Canyon (D2), and Mutton Creeks (M1);

Table 1
Description of prescribed fire and unburned watersheds used in this study

	Prescribed fire	Unburned watersheds			
	Dark Canyon	Bacon (1 and 2)	Deep Canyon	Mutton	Gaddis
Elevation (m)	1452	1380	1280	1310	1243
Stream order	1	1	1	1	2
Watershed area (ha)	129	81	139	244	486
Wetted stream width (m)	1.13 ± 0.25	1.22 ± 0.40	1.93 ± 0.50	1.70 ± 0.18	2.43 ± 0.34
Discharge (1 October; m^3/s)	0.025 ± 0.018	0.011 ± 0.007	0.041 ± 0.019	0.021 ± 0.012	0.057 ± 0.027
Stream gradient (%)	1–3	1–3	4–10	1–3	2–4
Percent of watershed in BFRS	98	100	51	80	80
Dominant tree species (percent of total basal area)	IC (26)	IC (42)	MA (62)	IC and DF (32)	IC (27) and PP (26)
Current land-use	Ecological reserve	Light cattle grazing, housing, timber harvest	Light timber harvest	Light timber harvest, ecological reserve	Light timber harvest

Means are presented ± standard deviation. Tree codes are: DF: Douglas-fir (*Pseudotsuga menziesii*), IC: incense-cedar (*Calocedrus decurrens*), MA: bigleaf maple (*Acer macrophyllum*), PP: ponderosa pine (*Pinus ponderosa*).

the second-order site was located in Gaddis Creek (G2) (Table 1, Fig. 1). Each stream drains watersheds of mixed-conifer forests with large buffer zones between the stream and management activities (at least 50 m). The average annual precipitation (1961–2003) is 155.6 ± 53.8 cm (mean ± standard deviation, S.D.), 15–20% of which occurs as snow. Furthermore, there is high seasonal and annual variation in precipitation at the study sites, which can be seen in the hydrograph at D1 (Fig. 2a). The study period (1995–2003) encompasses an above-average wet period (1995–1998) and a drought period (2001–2002) (Fig. 2b). The last fire occurred in this watershed in 1905, although the median historical fire-return interval was 5 years and the mean FRI was 6.8 years from 1649 to 1921 (based on a 15-ha sample; Stephens and Collins, 2004).

Tree species in the mixed-conifer forest of BFRS include sugar pine (*Pinus lambertiana* Dougl.), ponderosa pine (*Pinus ponderosa* P. & C. Lawson), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr), incense-cedar (*Calocedrus decurrens* (Torr.) Florin), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), California black oak (*Quercus kelloggii* Newb.), tan oak (*Lithocarpus densiflorus* (Hook. & Arn.) Rehder), bush chinquapin (*Chrysolepis sempervirens* (Kell.) Hjelmg.), and Pacific madrone (*Arbutus menziezii* Pursh). The riparian forests also include white alder (*Alnus rhombifolia* Nutt.), Pacific yew (*Taxus brevifolia* Nutt.), Pacific dogwood (*Cornus nuttallii* Audubon ex Torr. & Gray), bigleaf maple (*Acer macrophyllum* Pursh), and willows (*Salix* L. sp.). All of the riparian areas studied are dominated by incense-cedar, with the exception of D2, which is dominated by bigleaf maple (Table 1). For example, at D1 incense-cedar makes up 52.1% of all trees and 26.0% of the basal area. This differs from the upland vegetation in the watershed, where white fir is dominant (Stephens and Collins, 2004), making up 32% of the basal area (incense-cedar comprises 17% of the upland basal area). The common understory plants in the riparian areas include western azalea (*Rhododendron occidentalis* (Torr. & Gray ex Torr.) Gray), twin flower (*Linnaea borealis* L.), sierra-laurel (*Leucothoe davisiae* Torr. ex Gray), western bracken fern (*Pteridium aquilinum* (L.) Kuhn), star flower (*Trientalis latifolia* Hook), false Solomon's seal (*Smilacina racemosa* (L.) Desf.), western prince's pine (*Chimaphila umbellata* (L.) W. Bart), blue elderberry (*Sambucus nigra* L. ssp. *cerulea* (Raf.) R. Bolli), smooth yellow violet (*Viola glabella* Nutt.), and drops of gold (*Disporum hookeri* (Torr.) Nichols. var. *trachyandrum* (Torr.) Q. Jones). All scientific names and authorities were obtained from the plants database (USDA and NRCS, 2004).

2.2. Study design

We used a beyond-BACI (Before-After-Control-Impact) experimental design (Underwood, 1993, 1994) to examine the effects of a prescribed burn on stream structure and biota at BFRS. Beyond-BACI

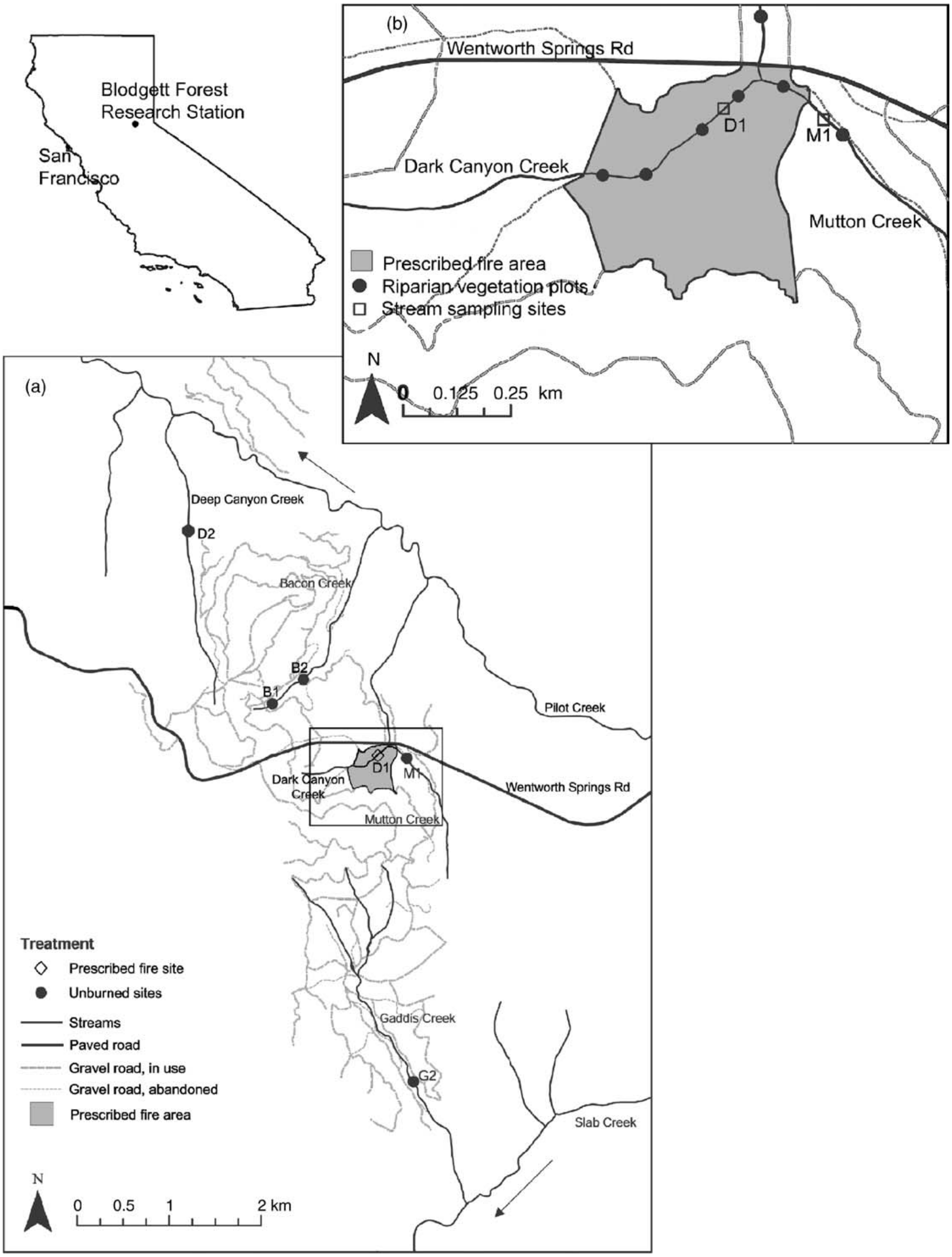

Fig. 1. Map of study sites at BFRS. All sites are on first-order streams and are tributaries of Pilot Creek, except G2 on Gaddis Creek, which is a second-order tributary to Slab Creek. The direction of flow is indicated by the arrows. Gravel roads are shown on BFRS property. Abandoned roads have been naturally revegetated and have not been used for 10 years.

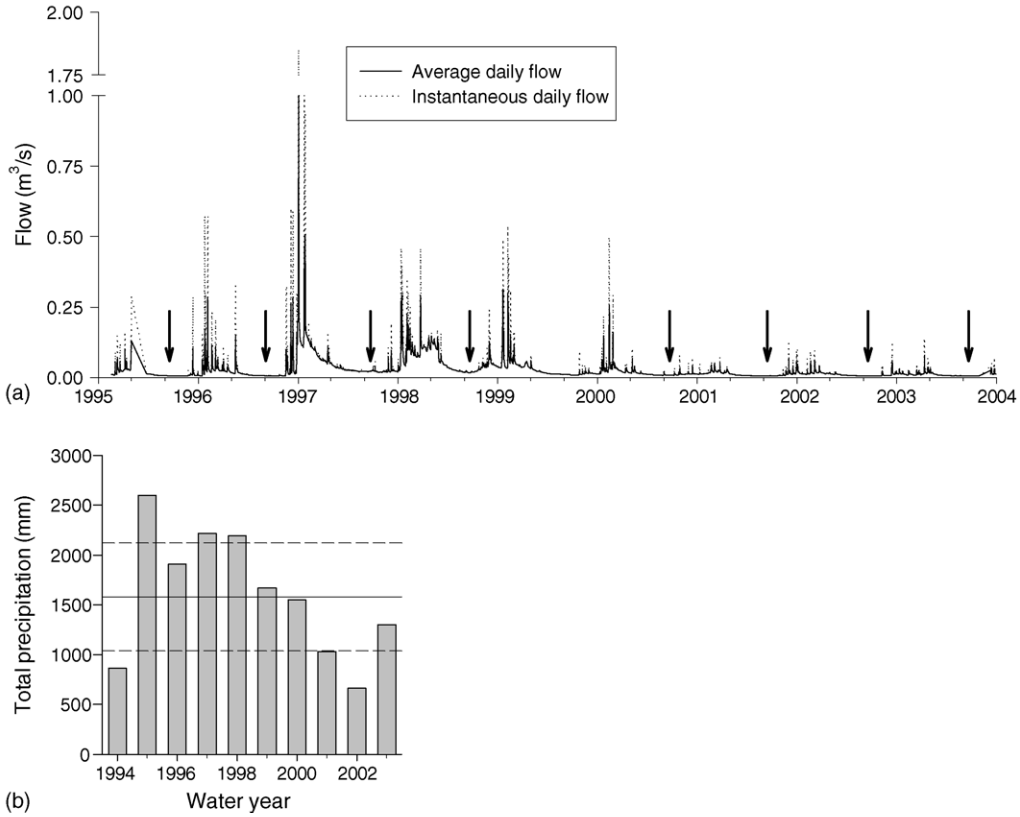

Fig. 2. (a) Hydrograph for Dark Canyon Creek from 1995 to 2003. Sampling periods are indicated by the arrows. (b) Total annual precipitation (1 October–30 September) from 1994 to 2003.

experimental designs are asymmetrical, with multiple control sites and one or more impacted sites that are monitored before and after the impact. Thus, we sampled before (1–7 years) and after (1 year) treatment at one burned and five unburned sites, all located within four watersheds (Fig. 1). Changes in (1) riparian forest communities; (2) large woody debris; (3) sediment; (4) water chemistry; (5) periphyton biomass; and (6) benthic macroinvertebrate communities were documented in both unburned and prescribed fire streams.

2.3. Prescribed fire treatment

On 21–23 October 2002, a prescribed fire was conducted in a 26-ha plot in the Dark Canyon Creek watershed (Fig. 1). Ignition was conducted in the evening, with patterns manipulated to control fireline intensity in the riparian areas using strip-headfires (Martin and Dell, 1978). Average flame lengths were visually estimated during the fire. Fireline intensity, the heat produced by flaming combustion (kW/m), was estimated using the Byram fireline intensity equation and average flame lengths (Byram, 1959). There was active ignition in the riparian zone.

2.4. Data collection

2.4.1. Riparian vegetation

We surveyed permanent riparian vegetation plots to determine the amount of fuel consumption, and the direct (e.g., mortality) and indirect (e.g., changes in community composition) effects of the prescribed fire on riparian vegetation. Permanent riparian vegetation sites were surveyed in the dry-season (July–August) pre-fire (1997, 2001) and post-fire (2003) in both the burned ($n = 5$ sites) and unburned ($n = 6$) watersheds. Specifically, there were five sites established on Dark

Canyon Creek, and two sites established on each of Mutton, Bacon, and Gaddis Creeks (total of six sites in unburned streams). On a particular stream, the riparian vegetation plots were separated by 200 m of stream length. At each site, one large plot (circular, 400 m^2) and two nested sub-plots (circular, 40 m^2) were established (therefore, 5–6 large plots and 10–12 sub-plots were surveyed in the burned and unburned streams, respectively). The center of each large plot was located in the stream thalweg and the plot was a circle around that center. Sub-plot measurements were done consistently on either the right or left bank, with the edge of the first sub-plot being tangential to the stream bank, and the edge of the second sub-plot was tangential to the first sub-plot. In some cases, the riparian zone was >30 m wide (as determined by the presence of riparian indicator species, e.g., alder); however, only 14 m from the bank was surveyed at each site.

Within each large (400 m^2) plot, trees with a diameter at breast height (DBH) >11.5 cm were identified and measured (DBH, height, and height to live-crown base). In both large and nested sub-plots, the relative cover of shrubs and herbaceous plants was determined by visual estimation of percent cover for each plant species; however, only vegetation cover data from the small plots (40 m^2 sub-plots) were used in subsequent analyses. To increase accuracy and to minimize missing species, each sub-plot was divided into quadrants to determine vegetative cover. The amount of cover was determined in increments of 5%, and if a species was present but accounted for <5% cover, it was recorded as 0. Taxa that were difficult to identify (such as many grasses) were recorded at the genus or family level, which likely reduced overall taxa richness estimates. When layered, vegetation cover was sometimes >100%. Furthermore, the amount of bare ground, and ground covered by litter and debris (and not vegetated) was also recorded. Canopy cover was measured in all plots using a sighting tube on a set of grid points.

Fuel characteristics (duff and litter depth, distribution of 1, 10, 100, and 1000 h fuel classes) were measured according to the procedures in Brown (1974) in each large plot (two transects per plot) at each sampling date (therefore, $n = 10$ transects in burned and $n = 12$ for unburned streams per sample date). In addition, fuel was surveyed 2 weeks post-fire (11 November 2002) at a sub-sample of the plots in only the burned watershed ($n = 5$ transects). Fuel consumption was calculated by subtracting post-burn fuel loads from pre-burn fuel loads. We used regression equations for Sierra Nevada forests (from van Wagtendonk et al., 1998) to calculate duff and litter fuel loads, and we modified Brown's (1974) equations with parameters from van Wagtendonk et al. (1996) to calculate surface fuel loads. Coefficients required to calculate surface and ground fuel loads were arithmetically weighted by basal area fraction to produce accurate and precise estimates of fuel loads (Stephens, 2001).

Summary statistics and coefficient of variations (CV = standard deviation/mean) were calculated for the percent cover of understory vegetation for each sample period, including total percent cover, taxa richness (number of taxa), and Simpson's diversity (D). For multivariate analyses, percent cover of ground cover for each taxon was converted to a rank because of the potential difficulties of accurately determining percent cover over a large area (40 m^2). Ranks were determined by grouping percent cover as follows: not present = 0, 0–5% cover = 1, 5–15% = 2, 15–30% = 3, 30–50% = 4, and >50% = 5.

2.4.2. Large woody debris (LWD)

To determine whether the prescribed fire resulted in an increase in LWD recruitment and movement (e.g., Young, 1994), we used standard methods for the inventory and monitoring of in-stream LWD (e.g., Lienkaemper and Swanson, 1987; Platts et al., 1987). Pre-burn surveys of LWD were conducted in 2001 and 2002. The LWD survey was conducted in a reach of variable length that extended between two riparian vegetation plots. During each initial survey, 50 pieces of LWD were measured, tagged, and mapped at each site to determine LWD volume and longitudinal position in the stream. LWD was defined by having a length ≥1 m or a diameter ≥10 cm. Upon subsequent survey in 2003, we documented the movement of tagged wood, and measured new pieces of LWD. The volume of each piece of wood was calculated using the formula from Lienkaemper and Swanson (1987).

2.4.3. Fine sediment

To detect changes in fine sediment in pools as a result of the fire, we measured V^*, which is the residual volume of a pool comprised of fine sediment (Hilton and Lisle, 1993). The fraction of pool volume made up of fine sediment can indicate the supply of mobile sediment, and is equal to the ratio of fine sediment volume to pool water volume plus fine sediment volume (Hilton and Lisle, 1993). V^* was measured in every pool within a 30 m length of stream at five sites in Dark Canyon Creek, and two sites each on Bacon, Gaddis, and Mutton Creeks; these reaches correspond to the location of the riparian vegetation site locations. Therefore, the number of pools sampled varies among sites. Measurements were made by dividing the pool into six transects and taking six measurements along each transect of water depth (depth to the top of fine sediment) and depth-to-coarse sediment (gravel or cobble). Using the methods proposed by Hilton and Lisle (1993), we calculated the amount of the pool volume that is comprised of fine sediment. We measured V^* during low flow conditions (dry-season, July–August) pre-fire in 1997 and 2001, and post-fire in 2003.

2.4.4. *Water chemistry*

To determine if ash deposition or runoff in the burned area resulted in changes in water chemistry, water was sampled at each site monthly from June 2001 to October 2003; plus, immediately after the fire, samples were taken every 24 h for 1 week. We measured in-stream temperature, pH, dissolved oxygen, and conductivity using portable meters in the field (five measurements per site per month); nitrate (NO_3^-), ammonia (NH_4^+), total nitrogen (TKN), total phosphorous (tot P), soluble phosphorous (sol P), calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+), and sulfate (SO_4^-) concentrations were determined from at least one water sample from each site each month. Water samples were stored on ice (4 °C), and processed in accordance with APHA standard methods for the analysis of water (Clesceri et al., 1998) by the University of California Division of Agriculture and Natural Resources Analytical Laboratory in Davis, CA.

2.4.5. *Periphyton*

We sampled periphyton biomass to determine if an increase in nutrients, reduced shading, or increases in fine sediment deposition caused by the prescribed fire affected periphyton. Standing crop of benthic periphyton was estimated by determining the ash-free dry mass (AFDM). We began sampling periphyton in September 2001, and continued to take samples approximately monthly through October 2003.

Periphyton was collected from colonized artificial substrates (i.e., clay tiles), because only one site (Gaddis Creek) had large enough mineral substrate to sample periphyton. Clay tiles have been shown to be indistinguishable from natural substrate in terms of periphyton biomass and community composition (Lamberti and Resh, 1985). Monthly, two samples were taken from five tiles at each site (10 samples per site) from an 8-cm^2 area. The AFDM was then determined using the methods described by Steinman and Lamberti (1996).

Biomass data often contain many small decimal fractions, and a $\log(x + 1)$ transformation would mask differences between small values. Instead, we used a $[\log(x + d) - c]$ transformation for the periphyton biomass data, where c = an order of magnitude constant = integer value of: log(smallest non-zero value in data), and $d = \log^{-1}(c)$, resulting in a transformation equation of: $\log(x + 0.01) - (-2)$. This transformation tends to preserve the original order of magnitudes in the data and preserves zero values (McCune et al., 2002).

2.4.6. *Macroinvertebrates*

To determine the indirect effects prescribed fire has on benthic macroinvertebrates at the community level, samples were collected on 1 October (±3 d) of each year. Samples were taken in the same location during base-flow conditions at all six sites from 1995–1998 (del Rosario, 2000) and 2000–2003. At each site, five samples were collected in riffle and run habitats within a 50 m reach at each site (each sample separated by 10 m). For each sample, a 1 m^2 area of the streambed was disturbed, including stream banks, and invertebrates were collected with a D-frame net (30 cm wide, 1 mm mesh).

Macroinvertebrates were sorted without sub-sampling and identified to genus, family (for most Diptera), or order (for some non-insect taxa). Additionally, using the same collecting methods, five samples were taken from D1 2 d post-fire (25 October 2002), and three samples from both D1 and M1 were taken 10 and 19 d post-fire (2 and 11 November 2002). Samples from 11 November 2002 (19 d post-fire), were taken 12 h after a 4 d storm that resulted in 18.7 cm of rain, and an increase in discharge of 0.038 m^3/s (from 0.007 m^3/s to a peak of 0.045 m^3/s, discharge at time of sampling was 0.01 m^3/s).

Summary statistics were calculated using raw abundance data for each site and season, including density (numbers/m^2) and taxa richness (number of taxa). Simpson's diversity (D) was calculated using $\log_{10}$(abundance + 1) transformed data. The CV of each statistic was also calculated.

2.5. Data analysis

Because this study is based on a beyond-BACI experimental design (Underwood, 1994), a modified version of the asymmetrical ANOVA proposed by Underwood (1993, 1994) was used to analyse the effects of prescribed burning on periphyton biomass and most water chemistry variables. This type of analysis accounts for multiple sampling events at the same sites over time. The following parameters were included in the ANOVA model: Site, Period (i.e., Before/After), Site by Period, Sample Date within Period, and Site by Sample Date within Period. The variance attributed to the unburned sites was repartitioned from the variance attributed to the burned site, as described in Underwood (1993). Fire effects were determined by examining the F-ratio of: (1) Period by Burned Site/Period by Unburned Sites and (2) Sample Date within Period by Burned Site/Sample Date within Period by Unburned Sites.

All other parameters (riparian vegetation summary measures, LWD, sediment, water chemistry, and benthic macroinvertebrate summary measures) were examined by comparing the difference between the average pre-fire and post-fire value for the unburned sites and the burned site (mean difference approach). A fire effect was concluded to have occurred if the pre-/post-fire difference for the burned site fell outside of the 95% confidence interval for the average difference among the unburned sites. This method was used when there was only one post-fire sampling period or when only one sample was taken on most dates.

To examine changes in riparian and macroinvertebrate community composition between sampling periods (Before/After) and among sites (Unburned/Burned), we used Multi-response Permutation Procedure (MRPP) and non-metric multidimensional scaling (NMS) ordinations (Kruskal, 1964) using PC-ORD 4.27 (McCune and Mefford, 1999). The $\log_{10}(x + 1)$-transformed macroinvertebrate data and ranked riparian vegetation cover data were used in MRPP and NMS ordinations.

MRPP is a non-parametric procedure for testing for multivariate differences among pre-defined groups (i.e., Before-Unburned, Before-Burned, After-Unburned, After-Burned), providing an A-statistic and a p-value based on 999 Monte-Carlo simulations. Bray-Curtis distance was used for the MRPP. For the riparian vegetation and macroinvertebrate community data, four comparisons were also made using MRPP to determine the difference between before and after for burned and unburned sites separately, and the difference between burned and unburned sites both before and after the prescribed fire. Furthermore, to control the error rate for multiple comparisons (i.e., four or six), a Bonferroni correction was applied.

NMS was chosen as the ordination method because it does not make assumptions about the type of data response, and accurately represents the distance between communities in ordination space (Legendre and Legendre, 1998; McCune et al., 2002). The following NMS analyses were performed: (1) riparian vegetation communities at all sites; (2) macroinvertebrate communities at all sites; and (3) macroinvertebrate communities at only M1 (unburned) and D1 (burned) sites. M1 and D1 are adjacent watersheds, with similar vegetation, morphology, taxonomic composition, and sediment composition (Bêche, 2005).

Each NMS was performed using Bray-Curtis distance. An initial ordination was conducted to determine the appropriate number of dimensions using step-down in dimensionality (from six dimensions) and randomized starting coordinates. The final ordination was conducted using the chosen number of dimensions, and the final starting coordinates from the previous ordination. Both initial and final ordinations were conducted with 2000 runs of real data and a Monte-Carlo test using 999 runs of randomized data. Significance was assessed using the 999 Monte-Carlo tests of randomized data and comparing the stress on each axis in real versus randomized data: $p = (1 + n)/(1 + N)$, where n = number of randomized runs with final stress $\leq$ the observed minimum stress, and N is the number of randomized runs. Stress is measured as the departure from monotonicity in the ordination.

3. Results

3.1. Prescribed fire behavior and fuels inventory

Strip-headfires produced flame lengths of approximately 0.75–2 m; average flame lengths were approximately 1.5 m. Fireline intensity varied from 138 to 1165 kW/m in the riparian area (average was approximately 625 kW/m). Higher intensity fire occurred in areas of high localized fuel loads, where slopes were steeper, and fuels were drier.

Pre-fire fuel loads in the riparian area were high (13.79 kg/m^2, 137.9 Mg/ha). Ground fuels, which include only the duff and litter layers, contributed 57% of the total load. Large surface fuels (large woody debris >7.6 cm diameter) accounted for 33% of the total fuel load. Surface and ground fuel consumption ranged from 6 to 100%, depending on the fuel size, condition, and location within the riparian zone (Table 2). Wind and rain caused branches, litter, and several trees to fall in the riparian zone within 1 year of the fire, which increased the fuel load (Table 2).

3.2. Riparian vegetation

Because both the intensity of the fire and the burn patterns were patchy along the stream, the effects on riparian vegetation were heterogeneous. Litter and woody debris covered a 43.4 ± 25.1% (mean ± standard deviation) of the area surveyed prior to the fire, and was reduced to 20.8 ± 25.4% 1 year after the fire. Within the measured plots, the amount of bare ground increased almost 10-fold (pre: 3.5 ± 8.2%, post-fire: 34.2 ± 21.8%). Understory vegetation cover decreased significantly after the fire relative to unburned sites (Table 3). For example, vegetation cover decreased from 54.0 ± 32.7% prior to the fire (2001) to 23.8 ± 31.6% after the fire (2003), with a smaller decrease in the unburned plots (Fig. 3a). In contrast, riparian canopy cover did not measurably change (pre-fire: 83.1 ± 9.5%, post-fire: 80.8 ± 18.6%, unburned: 78.8 ± 14.0%) as a result of the prescribed fire (Table 3). Ground-cover taxa richness decreased significantly more in the burn plots than in the unburned plots (Table 3, Fig. 3b). In contrast, diversity (Simpson's D) decreased post-fire in both the burned and unburned plots (Table 3, Fig. 3c).

The fire resulted in the mortality of only eight trees (4.4% of trees, $n = 5$ plots), ranging in size from 11.7 to 40.4 cm DBH (22.6 ± 11.8 cm). Ten snags fell over in the riparian area after being partially burnt, while four snags were fully consumed during the fire (over 0.16 ha). In the burn unit, 49.4% of tagged trees exhibited scorch of 0.10 m or higher, while 47.6% were not scorched. The average scorch height on tagged trees was 2.5 ± 2.3 m (mean ± S.D.), with a maximum scorch height of 7.6 m. Snags and live trees were scorched in approximately equal proportions (45.6 and 54.4%, respectively).

MRPP showed that community composition of riparian vegetation (ground cover) differed between the four groups (Before-Unburned, Before-Burned, After-Unburned, After-Burned) (Table 4a). Based on

Table 2
Fuel loads (mean ± standard deviation) in the prescribed fire site pre-fire (2001) and post-fire (November 2002 and October 2003)

Fuel type (timelag)	Pre-fire fuel load (kg/m^2), $n = 10$	Two week post-fire (kg/m^2), $n = 5$	Percent consumed (2 week post-fire)	One year post-fire (kg/m^2), $n = 10$	Percent consumed (1 year post-fire)
1 h	0.13 ± 0.08	0.02 ± 0.02	89	0.04 ± 0.03	72
10 h	0.63 ± 0.37	0.20 ± 0.19	68	0.11 ± 0.11	83
100 h	0.70 ± 0.51	0.28 ± 0.32	60	0.23 ± 0.27	67
1000 h					
Sound	2.48 ± 3.92	2.33 ± 3.37	6	2.71 ± 5.71	−2
Rotten	2.13 ± 4.45	0 ± 0	100	2.17 ± 4.50	−9
Duff	4.88 ± 2.79	0.15 ± 0.22	92	1.03 ± 1.68	79
Litter	3.16 ± 1.48	0.39 ± 0.20	88	0.63 ± 0.66	80
Total surface	6.07 ± 5.10	2.82 ± 3.27	34	5.27 ± 6.70	13
Total fuel load	14.11 ± 4.93	2.86 ± 3.51	79	6.92 ± 6.80	51

The number of transects surveyed (n) is indicated for each sampling date.

Table 3
Difference between pre- and post-fire values for several response variables averaged over unburned and burned sites

Response	Unburned	Prescribed fire	Prescribed-fire effect
Riparian vegetation			
Percent ground cover	−8.3 ± 17.8	28.1 ± 19.3	Reduction
Percent canopy cover	7.1 ± 9.8	1.0 ± 8.4	No
Taxa richness	0.8 ± 0.9	3.7 ± 1.1	Reduction
Diversity (D)	0.10 ± 0.08	0.06 ± 0.05	No
LWD volume (m^3)	−0.12 ± 0.17	0.13	No
V^* (fine sediment)	0.07 ± 0.09	−0.02 ± 0.11	No
Macroinvertebrates			
Total abundance	−316.8 ± 308.9	−337.5	No
Taxa richness	−7.1 ± 5.6	−7.7	No
Diversity (D)	0.06 ± 0.08	0.07	No

Values are presented as mean ± 95% confidence interval. A prescribed-fire effect was concluded to have occurred if the burned site difference fell outside of the 95% CI for the unburned sites.

multiple MRPP comparisons, the greatest difference in community composition resulted from differences between unburned and burned sites before the fire treatment and from differences between before and after the treatment in the burned sites (Table 4a). In contrast, NMS ordination showed that there was little difference in riparian community composition between the treatments, but there were differences between periods of sampling. The three-axis NMS solution (stress = 21.4, $p = 0.001$) represented 64.0% of the total variation, with 15.5% on axis 1, 20.1% on axis 2, and 28.3% on axis 3 (Fig. 4a). A dummy variable representing Period by Treatment was significantly correlated with axes 2 (Spearman rank correlation, $r_s = 0.31$, $p = 0.01$) and 3 ($r_s = 0.25$, $p = 0.25$). Year of sampling was correlated with axis 3 ($r_s = 0.29$, $p = 0.02$).

3.3. Large woody debris

The prescribed fire did not change the amount (Table 5) or movement (Bêche, 2005) of LWD in D1 relative to unburned streams. The LWD characteristics (mean diameter, length, function, and total volume) were similar across all streams (Table 5). There were few new pieces of LWD in subsequent surveys (2002, 2003), with the exception of site M1, where several rotten and decaying snags fell over the stream in 2002. LWD did not result in an increase in LWD volume in the prescribed fire stream compared to the unburned streams (Tables 3 and 5). However, LWD increased in other stream reaches along D1 where less intensive sampling was conducted (i.e., at riparian vegetation sites). For example, seven snags fell into that stream channel in a 400-m reach (Bêche, 2005).

Table 4
Results from MRPP analysis of (a) riparian vegetation (based on ranked percent cover data); (b) all macroinvertebrate abundance data ($\log_{10}$-transformed); and (c) macroinvertebrate abundance data without samples taken 10 and 19 d post-fire (i.e., 2 and 11 November 2002 samples omitted)

	(a) Riparian vegetation		(b) Macroinvertebrates: all data		(c) Macroinvertebrates: w/o immediate post-fire	
	A	*p*-value	*A*	*p*-value	*A*	*p*-value
Period × Treatment	0.132	<0.00001***	0.066	<0.00001***	0.048	<0.00001***
Before cf. After (Unburned)	0.030	0.0642	0.037	<0.00001***	0.038	<0.00001***
Before cf. After (Burned)	0.067	0.0114*	0.098	0.00002***	0.025	0.0904
Unburned cf. Burned (Before)	0.123	<0.00001***	0.012	0.0064*	0.012	0.0064*
Unburned vs. Burned (After)	0.022	0.1826	0.057	0.0031*	0.023	0.1765

Significance is indicated as $^{*}p < 0.05$, $^{***}p < 0.001$ after Bonferroni correction for multiple comparisons. Period: Before or After, Treatment: Unburned or Burned (prescribed fire).

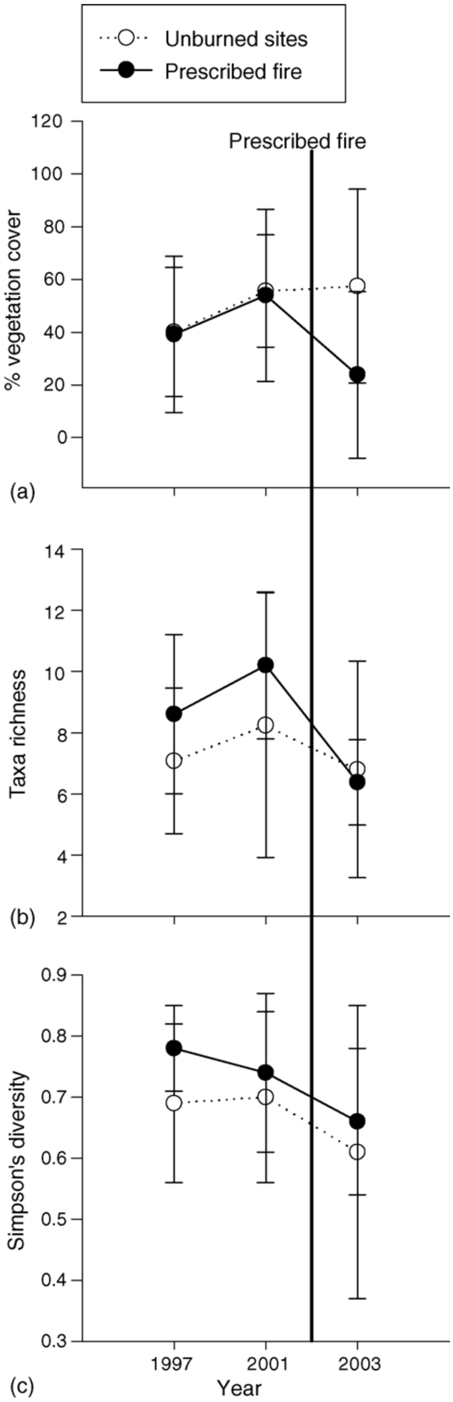

Fig. 3. Riparian vegetation (ground cover) summary measures (mean ± standard deviation) before (1997, 2001) and 1 year after (2003) prescribed fire (2002) for three unburned streams (Bacon, Gaddis, and Mutton Creeks, $n = 12$ plots) and the prescribed fire site (Dark Canyon Creek, $n = 10$ plots): (a) percent vegetation cover; (b) taxa richness; (c) diversity (Simpson's D).

3.4. Fine sediment

Fine sediment in pools, as measured by V^* (the average residual pool volume of fine sediment), did not significantly change post-fire. For example, post-fire changes in V^* at D1 were not significantly different from post-fire changes in the unburned streams (Table 3). Furthermore, V^* did not significantly change in D1 based on comparisons of pre-fire (1997 and 2001) and post-fire (2003) measurements ($p = 0.89$, Mann–Whitney U-test, d.f. = 25) (Table 6).

3.5. Water chemistry

There is no evidence of post-fire change in dissolved oxygen, conductivity, or NO_3^- at D1 relative to the unburned sites (Table 7). Before the prescribed fire, NH_4^+ was mostly below the detection limit of 0.05 mg/L. Post-fire, all streams (unburned and prescribed fire) exhibited a periodic increase in NH_4^+ that was reduced to below detection limits by October 2003 (Bêche, 2005). At D1, TKN increased 1.35 mg/L in D1, but no change was observed in the unburned streams (Bêche, 2005). Post-fire TKN concentrations returned to pre-fire levels within 19 d post-fire (Table 7).

Sulfate concentration was low at all sites until the end of the first post-fire rainstorm (19 d post-fire), when SO_4^- increased in D1 to 0.30 ± 0.12 mg/L (Table 7); however, SO_4^- returned to ≤ 0.10 mg/L within 1 month after the post-fire flood (Bêche, 2005). After the first post-fire rain event, soluble P increased to 0.08 mg/L, and returned to below the detection limit within 2 months (Bêche, 2005). In contrast, total P increased 1 week post-fire in D1 to 0.20 mg/L (unburned streams remained ≤ 0.10 mg/L), and returned to pre-fire levels within 2 weeks, perhaps as a result of the post-fire rain event (Table 7). Soluble Ca^{2+} and Mg^{2+} concentrations were low at all sites prior to the fire (≤ 0.10 μeq/L, the detection limit), and increased after the post-fire spring snowmelt in D1 relative to the unburned streams. Soluble K^+ concentrations, however, were similar between unburned sites and D1 at all sample periods (Table 7).

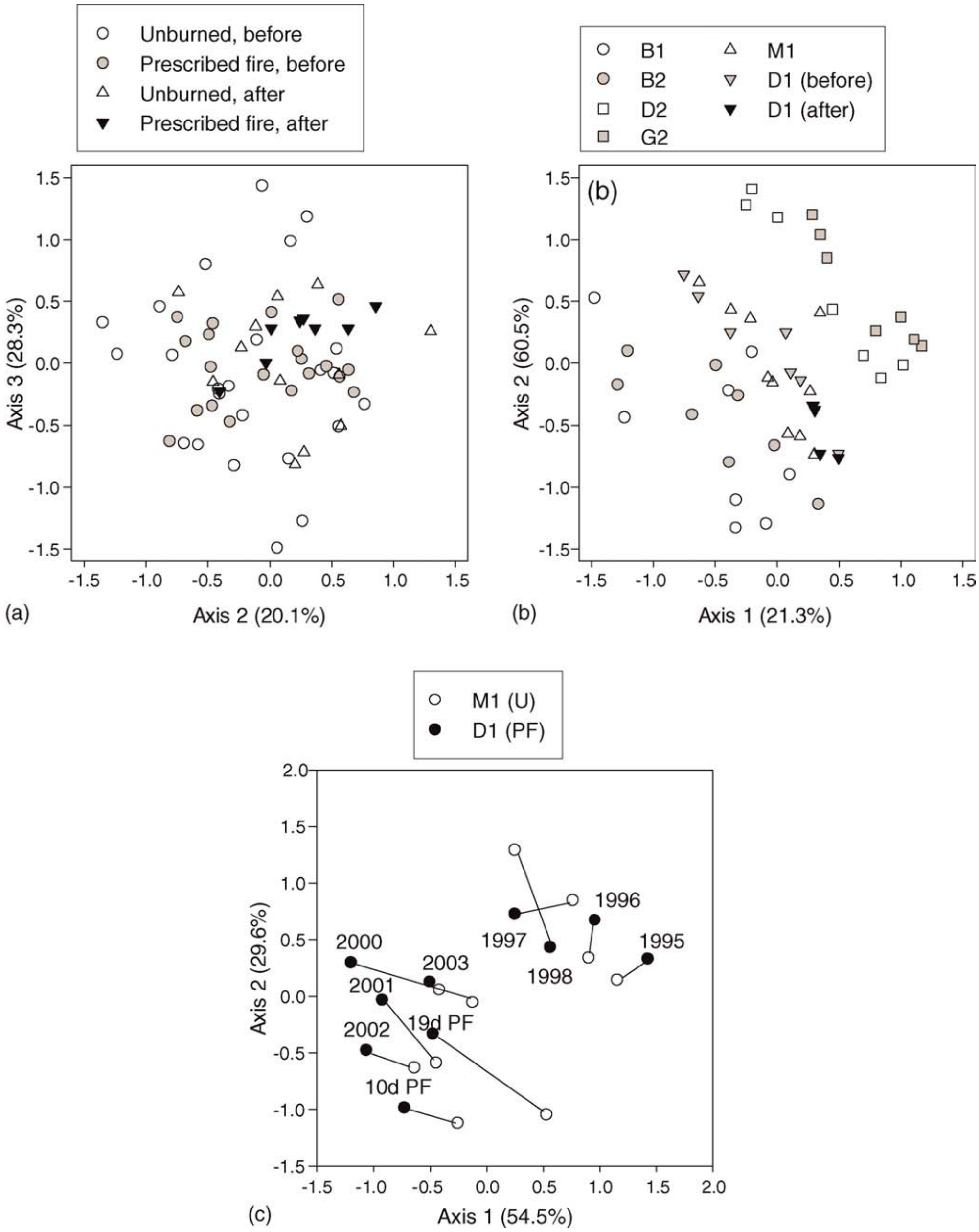

Fig. 4. NMS ordination of: (a) riparian vegetation (ranked ground cover); (b) macroinvertebrate abundance (log_{10}-transformed) from unburned sites and the prescribed fire site (D1); (c) macroinvertebrate abundance from the adjacent Mutton (M1, unburned) and Dark Canyon (D1, prescribed fire) creeks; data points from the same year are connected by a line. The amount of variance represented by each axis is indicated in parentheses.

To test for significant trends following the prescribed fire, the mean difference approach or a modified asymmetrical ANOVA was used for all water chemistry variables. A significant effect of the fire was found only for NH_4^+ for the Sample Date within Period by Treatment interaction (F-ratio of Sample Date within Period by Burned site/Sample Date within Period by Unburned sites = 6.73, d.f. = 6, 12;

Table 5
Characteristics of LWD and changes in LWD volume before (2001, 2002) and after (2003) the prescribed fire

	Mean diameter (m)	Mean length (m)	Stream length (m) with 50 pieces LWD	2001 volume (m^3)	2002 volume (m^3)	2003 volume (m^3)
Bacon 1 (U)	0.13	1.92	68.8	2.39	2.42 (2)	2.42 (0)
Bacon 2 (U)	0.16	3.53	58.6	9.55	9.88 (4)	9.88 (0)
Deep Cyn (U)	0.13	2.92	86.5	9.01	9.97 (4)	9.97 (0)
Gaddis 1 (U)	0.14	3.28	92.0	5.46	5.48 (1)	5.50 (2)
Gaddis 2 (U)	0.17	3.60	105.5	7.98	7.98 (1)	8.52 (3)
Mutton (U)	0.19	2.80	59.0	3.63	16.79 (7)	16.95 (7)
Dark Cyn (PF)	0.16	3.59	67.3	9.09	9.12 (1)	9.25 (5)
Mean ± S.D. (CV)	0.15 ± 0.14 (0.90)	3.09 ± 2.48 (0.80)	76.8 ± 18.0 (0.23)	6.73 ± 2.89 (0.43)	8.81 ± 4.45 (0.51)	8.90 ± 4.53 (0.51)

U: unburned, PF: prescribed fire site. Volume of LWD is presented as total volume (number of new pieces of wood that year).

$p = 0.003$) based on ANOVA. Using the mean difference approach, significant effects of the fire were found for only TKN and Mg^{2+} (Table 7). However, the latter test does not reflect the recovery of the TKN and Mg^{2+} to normal levels within 19 d to 1 year (Table 7).

3.6. Periphyton

The prescribed fire had a short-term impact on periphyton; biomass decreased within 2 months post-fire, but recovered within 1 year post-fire. Periphyton biomass in D1 prior to the fire was generally equal to or higher than biomass in the unburned streams during most sample periods (Fig. 5). Three days post-fire, periphyton biomass was not substantially different in D1 relative to samples taken 4 d pre-fire (3.26 ± 3.55 mg/cm^2 cf. 1.20 ± 0.57 mg/cm^2, mean ± S.D.). Within 1 week post-fire, periphyton biomass from D1 was equal to biomass in unburned streams (1.50 ± 1.16 mg/cm^2 cf. 1.03 ± 0.95 mg/cm^2). Periphyton biomass was lower in the burn stream than in the unburned streams from 7 weeks post-fire until June 2003 and, within 1 year post-fire, periphyton biomass in D1 was higher than in the unburned streams, which is consistent with pre-fire patterns (Fig. 5). A modified asymmetrical ANOVA revealed a significant Period by Treatment interaction (F-ratio of Period by Burned Site/Period by Unburned Sites = 242.5; d.f. = 1, 2; $p = 0.008$), which indicates an overall effect of the prescribed fire on periphyton biomass. Furthermore, the Sample Date within Period by Treatment interaction was also significant (F-ratio of Sample Date within Period by Burned Site/Sample Date within Period by Unburned Sites = 7.56; d.f. = 7, 14; $p = 0.001$), which indicates that following the prescribed fire, the temporal profile before and after the fire was different in the burned site than in the unburned sites.

Table 6
Percentage of pool volume comprised of fine sediment (V^*) before (1997, 2001) and after (2003) the prescribed fire in the unburned streams (B2, G2, M1) and the prescribed-fire stream (D1)

	Unburned	Prescribed fire
1997	76.0 ± 9.3 (9)	82.3 ± 8.1 (10)
2001	81.8 ± 11.4 (6)	73.0 ± 12.4 (10)
2003	71.3 ± 14.1 (8)	80.7 ± 9.0 (7)

Data are presented as mean ± standard deviation (the number of pools surveyed).

3.7. Macroinvertebrates

Little to no response to the prescribed fire was observed in the macroinvertebrate communities. For example, relative to the unburned sites and pre-fire data, there was no immediate (10 and 19 d post-fire) or delayed (1 year post-fire) effect of the fire on macroinvertebrate abundance, taxa richness, or Simpson's diversity (Fig. 6), despite the large rainfall event 15 d post-fire. The difference between pre- and post-fire abundance, taxa richness, and diversity in the prescribed fire site fell within the 95% confidence interval for the difference in the unburned sites (Table 5).

MRPP results, in contrast to the summary measures above, indicate that there was a significant change in overall macroinvertebrate community composition

Table 7
Summary of water chemistry before and after the prescribed fire in unburned (U, $n = 5$ streams) and prescribed fire (PF, $n = 1$ stream) sites

		Pre-fire		Post-fire (all)		Effect	1–3 d post-fire		1 week post-fire		2 week post-fire		30 week post-fire		1 year post-fire	
		U	PF	U	PF		U	PF	U	PF	U	PF	U	PF	U	PF
Cond.	6/01	18.1	21.0	21.6	25.2	N	21.4	25.8	21.4	25.3	24.7	29.6	18.8	20.5	15.1	16.9
D. O_2	9/00	9.4	9.5	10.1	11.1	N	10.7	10.4	11.0	11.8	10.1	10.4	8.5	8.5	8.8	8.9
NO_3^-	6/01	0.07	b	b	b	N	0.06	0.05	0.06	0.05	b	b	0.06	b	0.06	b
NH_4^+	6/01	b	b	0.12	0.05	N	b	b	b	0.05	b	b	0.07	0.07	b	b
TKN	6/01	1.0	1.0	0.41	0.82	I	0.4	*1.2*	0.8	*1.1*	0.9	0.7	0.8	1.0	b	b
SO_4^-	6/02	0.10	0.10	0.13	0.13	N	b	0.10	0.10	0.10	0.15	*0.30*	0.10	0.10	b	0.10
Sol P	11/01	0.05	0.06	b	b	N	b	0.05	b	0.06	0.07	0.07	b	b	b	b
Tot P	6/01	b	b	b	b	N	b	0.10	0.10	0.11	0.10	0.10	b	b	b	b
Ca^{2+}	4/02	0.10	0.10	b	0.11	N	0.10	0.10	0.10	0.10	0.10	0.10	0.10	*0.20*	0.10	0.10
Mg^{2+}	4/02	b	b	b	b	I	b	0.10	0.10	0.10	0.10	0.10	b	*0.85*	b	b
K^+	4/02	1.09	1.21	0.93	1.20	N	1.39	1.39	0.94	0.97	1.35	1.35	0.52	0.67	0.55	0.58

Values are means of all samples; 1–10 samples were taken per site and sample period. All units are in mg/L except for conductivity, which is in $\mu s\ cm^{-1}$. Numbers in italics represent post-fire changes in the prescribed fire treatment; b indicates that value was below the detection limit of 0.05 mg/L (NO_3^-, NH_4^+, and soluble P) or 0.10 mg/L (TKN, SO_4^-, total P, Ca^{2+}, Mg^{2+}, K^+). Significant effects of the prescribed fire are indicated as I (increase) or N (none), based on comparisons between pre- and post-fire means at the unburned and prescribed fire sites.

between sample periods (before and after fire) and between treatments (prescribed fire and unburned) (Table 4b). However, when pair-wise comparisons were made, all combinations were significant, which indicates that the differences found in the overall MRPP analysis cannot be attributed to the prescribed fire treatment. Furthermore, the observed significant effects may be a result of the sampling that occurred immediately after the fire. For example, when samples taken 10 and 19 d post-fire were excluded from the analysis (i.e., comparing only 1 October samples), the overall MRPP (Period by Treatment) was significant, but multiple tests showed that the comparisons of Period (i.e., Before cf. After) by Prescribed Fire and of Treatment (i.e., Unburned cf. Burned) within the after period were not significant (Table 4c).

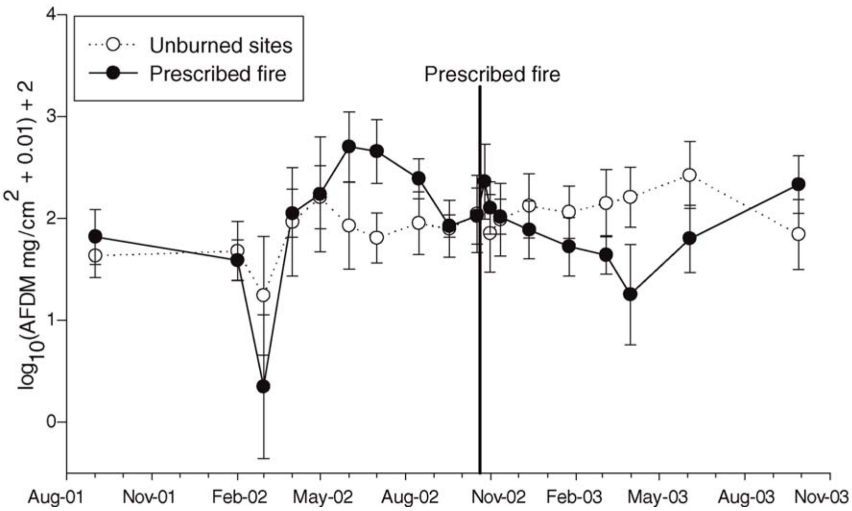

Fig. 5. Periphyton biomass before and after the prescribed fire in Dark Canyon Creek and all unburned streams. Biomass is presented as mean ± standard deviation. The solid line indicates the time of the prescribed fire (21–23 October 2002).

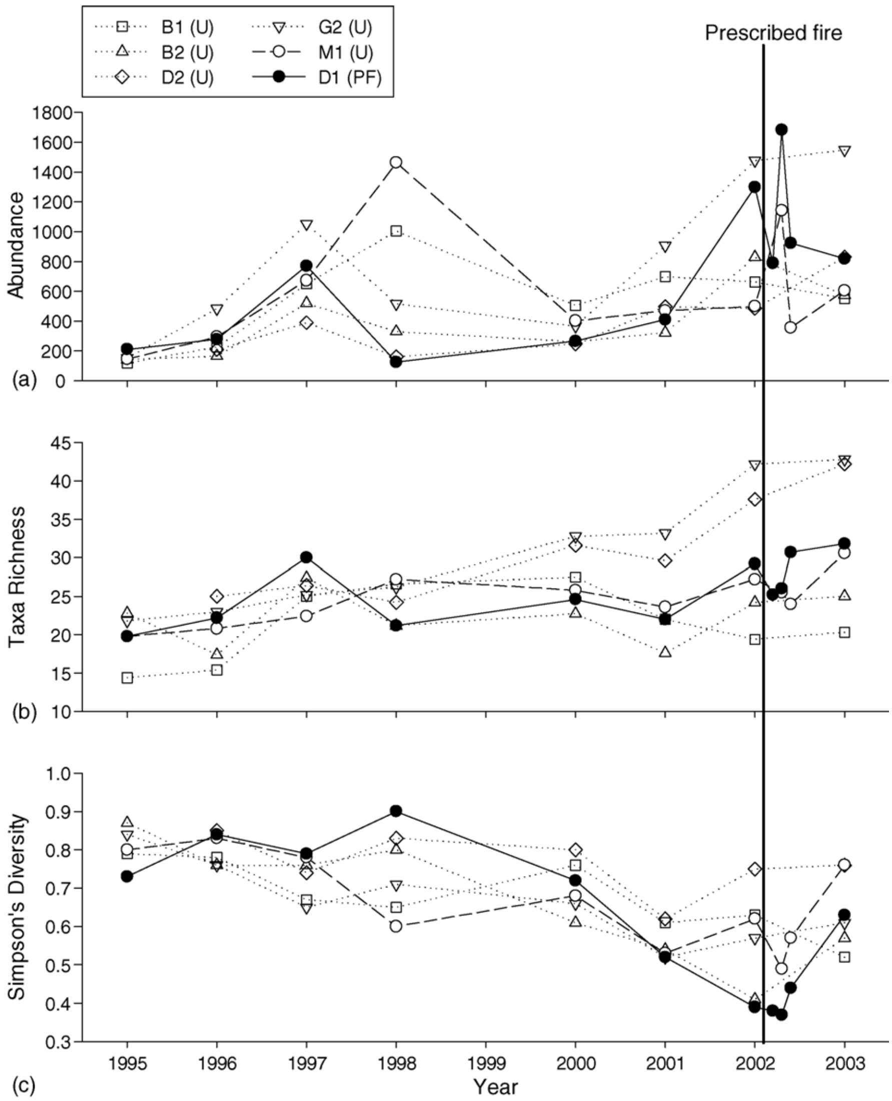

Fig. 6. Aquatic macroinvertebrate community measures before (1 October 1995–2002) and after the prescribed fire (10 d, 19 d; 1 year, 1 October 2003): (a) total abundance (per m^2); (b) taxa richness (number of taxa); (c) Simpson's diversity (D). U: unburned streams, PF: prescribed fire stream. The streams are abbreviated as B1: Bacon Ck. 1, B2: Bacon Ck. 2, D2: Deep Canyon Ck., G2: Gaddis Ck., D1: Dark Canyon Ck.

NMS ordination of all sites also showed that there was no difference in community composition between Period or Treatment, based on a three-axis solution that represented 92.0% of the variation (Fig. 4b). Axis 1 represents 21.3% of the variation, axis 2 represents 60.5%, and axis 3 represents 10.2%. Year of sampling is positively correlated with axis 1 ($r_s = 0.60, p < 0.0001$) and is negatively correlated with axis 2 ($r_s = -0.63$, $p < 0.0001$) and axis 3 ($r_s = -0.35$, $p = 0.01$). Site was positively correlated with axis 1 ($r_s = 0.44$, $p = 0.001$) and axis 2 ($r_s = 0.44$, $p = 0.001$). A dummy variable representing Period by Treatment was negatively correlated with axis 2 ($r_s = -0.29$, $p = 0.04$), but not axis 1 or 3 ($r_s < 0.25$, $p > 0.05$). However, the data points from post-fire sampling in D1 fall within the cloud of points from pre-fire and unburned samples (Fig. 4b). Furthermore, a comparison of the prescribed burned D1 and the adjacent unburned M1 using NMS showed that there was no difference in community composition after the fire, based on a two-axis solution representing 84.0% of the variance (Fig. 4c). Axis 1 represents 54.5% of the variation and axis 2 represents 29.6%. In this ordination, year is significantly correlated with axis 1 ($r_s = 0.68$, $p = 0.0009$) and axis 2 ($r_s = -0.66$, $p = 0.002$). A dummy variable representing Period by Treatment was not significantly correlated with either axis ($r_s < 0.40$, $p > 0.05$).

Community composition differed the most between D1 and M1 in 1998 after the El Niño Southern Oscillation (ENSO) winter (Bray-Curtis distance, $d = 0.35$) and 19 d post-fire ($d = 0.32$). Despite these two sampling dates, the distance between D1 and M1 is relatively constant ($d = 0.24 \pm 0.03$, mean $\pm$ S.D.); additionally, D1 and M1 communities are the most similar in 2003, 1 year post-fire ($d = 0.21$, Fig. 4c).

4. Discussion

In this study, we have demonstrated that a low- to moderate-intensity prescribed surface fire that was actively ignited in the riparian zone had minimal effects on a small stream and its riparian zone during the first year post-fire. Although multiple unburned sites were used in this study, it was not possible to replicate the prescribed fire treatment because of logistical constraints (Bêche, 2005). Nevertheless, evidence from both abiotic and biotic components suggest that there were few effects of the prescribed fire on Dark Canyon Creek, and the few features that were affected quickly recovered during the study period (e.g., periphyton). In contrast to the effects on the stream, there were marked effects of the fire on the riparian zone (e.g., crown scorch, fuel consumption). Here, we consider each line of evidence separately, in the context of the direct and indirect effects of the fire, the immediate and delayed impacts, and the patterns described in the literature. Finally, we discuss the factors that may have contributed to the effects of the prescribed fire in the 1-year period following the fire.

4.1. Prescribed fire in the riparian zone: fuels and vegetation

The prescribed fire in the riparian zone was patchy in terms of intensity, consumption, and severity. Fuel consumption was similar to other upland fall prescribed burns in the Sierra Nevada (e.g., Stephens and Finney, 2002; Stephens et al., 2004). The fire was most severe in those areas with large accumulations of conifer litter and debris, and usually self-extinguished when it came in contact with moist soil and characteristic riparian vegetation. As expected, high soil and fuel moisture, and high relative humidity can reduce fire intensity and retard fire spread in riparian zones (e.g., Dwire and Kauffman, 2003).

Understory vegetation was directly affected by the prescribed fire, reducing the amount of cover, taxa richness, and Simpson's diversity (Fig. 3). However, the prescribed fire did not result in substantial riparian tree mortality. For example, even though 49.4% of all tagged trees and snags were scorched by the prescribed fire, only 4.4% of all tagged trees died between the 2001 and 2003 surveys. The trees that were killed by the prescribed fire were generally small (mean $\pm$ S.D. = 22.2 ± 12.1 cm DBH, range = 11.4–40.4 cm) incense-cedar (75%), which were near areas of high litter accumulation. In contrast, upland-only prescribed fires (i.e., those prevented from entering the riparian zone) generally do not affect riparian vegetation. For example, abundance and distribution of surface vegetation, and percent cover and species composition were not affected by upland prescribed fires (Elliott et al., 1999; Lamb et al., 2003, respectively). Similarly, Elliott et al. (1999) found no mortality in near-riparian overstory trees following an upland prescribed fire in the Appalachian mountains.

Community composition of understory vegetation changed following the prescribed fire in the burned sites (Fig. 4a, Table 4a). However, much of this change may be result of the overall reduction in total vegetation cover and richness, rather than a shift in the community composition. For example, no new species were found in the burned sites following the fire (although a few species were lost in some plots). There is a large source of potential colonizers as a result of the patchiness of the fire and the small area burnt, which may lead to rapid recovery of the riparian vegetation.

In general, the riparian zone acted as a buffer between the moderate severity burned-portions of the upland zone and the stream, as has been hypothesized by several others (e.g., Timoney et al., 1997; Dwire and Kauffman, 2003; Lamb et al., 2003). There are two main factors that may have contributed to this buffering capacity in our study. First, the riparian zone experienced a range of fire intensities (mostly low to moderate), with many areas not burning at all as a result of high moisture content of riparian vegetation. Second, the relatively flat slopes of the majority of the riparian zone (riparian zone slopes range from 6.2 to 30.6%, mean ± S.D. = 19.9 ± 6.9%) and low post-fire precipitation (Fig. 2b) may have increased this buffering capacity by keeping erosion minimal, and increasing infiltration of precipitation (e.g., Townsend and Douglas, 2000).

4.2. Large woody debris

Even though several snags fell over as a result of the fire, either in the riparian zone or over the stream channel, there was no overall increase in the amount or movement of LWD in D1 relative to the unburned streams. This pattern most likely resulted because (1) most of the trees that were killed were small and (2) the trees and snags that fell over in the riparian zone had not yet moved into the stream channel. In comparison, Chan (1998) found great increases in LWD in two first-order streams following prescribed fire in mixed-conifer forests of Sequoia National Park (California, USA). Furthermore, prescribed fire often increases the number of standing snags and surface woody debris (e.g., Waltz et al., 2003). Thus, fire may contribute to increased in-stream LWD in the long-term, as compared to unburned watersheds (e.g., Bragg, 2000). For example, in simulations of LWD recruitment, Bragg (2000) found that moderately severe fire increased the amount of in-stream LWD over the long-term (300+ year) in headwater streams draining watersheds of old-growth mixed-conifer forest.

In contrast to prescribed burns, wildfires often have dramatic and immediate effects on LWD (e.g., Young, 1994; Minshall et al., 1997). For example, severe wildfires can consume in-stream LWD and post-fire floods can remove the remaining LWD. Dramatic increase in in-channel LWD can occur as fire-felled trees enter the channel (e.g., Minshall et al., 1997). LWD movement can also increase in burned streams as a result of higher magnitude floods and decreased bank stability (Young, 1994; Minshall et al., 1997). In general, the effects of wildfire on LWD are often more pronounced in smaller streams (first to third order) than in larger streams (fourth+ order) (Minshall, 2003). Furthermore, LWD is a major structural element in streams that forms pools and traps fine sediment (Keller and Swanson, 1979), and increases in post-fire LWD may buffer changes in channel morphology and fine sediment deposition (but see Berg et al., 1998).

4.3. Fine sediment

The residual volume of fine sediment in pools did not change as a result of the fire (Table 6). The study streams are sandy-bottomed, with a high percentage of fine sediment in both riffles and pools (except G2) (Bêche, 2005). The naturally high amount of fine sediment in pools may make it more difficult to detect small changes in V^*. Pebble counts in riffles also revealed little to no change in sediment composition 1 year post-fire; 86–100% of bed sediments were finer than 11.3 mm before and after the fire (Bêche, 2005). Similarly, little to no change was observed in channel morphology post-fire, based on longitudinal and cross-section surveys (Bêche, 2005). These results indicate that there was no substantial change in erosion or deposition in the surveyed reaches (Bêche, 2005). In comparison, in the steep canyons of Sequoia National Park (mixed-conifer forest), fine sediment increased 1 year following fire (Chan, 1998), which is likely a result of post-fire increases in hillside erosion.

Whereas severe wildfires often denude the entire watershed and leave few living trees, this prescribed fire only removed surface vegetation from 70% of the total area burned, representing only 14% (18 ha) of the total watershed area (129 ha). Wondzell and King (2003) also demonstrated that the risk of hillslope erosion is lessened with increasing vegetation, litter, and debris cover, which stabilize and protect forest soils. For example, there was no increase in surface runoff or erosion following a prescribed fire in the coast ranges of California (a ponderosa pine forest), and this was attributed to the preservation of partially decomposed duff and litter (Biswell and Schultz, 1957). Thus, post-fire erosion is potentially reduced by the unburned areas in the watershed.

Other studies have shown that fire severity may have a large effect on erosion, sediment yields, and time to recovery after a fire (e.g., Swanson, 1981; DeBano et al., 1996; Wondzell and King, 2003). For example, areas of low-severity wildfire in eastern Oregon resulted in small increases in sediment yield, which recovered within 3 years post-fire; in contrast, moderately and severely burnt areas took much longer to recover (7 and 14 years, respectively) (DeBano et al., 1996). Considering that DeBano et al.'s (1996) results were based on wildfires where the entire watershed was affected, it is not surprising that the low- to moderate-severity prescribed fire (affecting only a small portion of the watershed) in this study resulted in minimal hillside erosion and geomorphic change, or changes in sediment composition (Bêche, 2005), and fine sediment in pools (Table 6).

4.4. Water chemistry

Ash deposition from the prescribed fire appeared to have a minimal impact on stream water chemistry. Only NO_3^-, TKN, and total P increased immediately post-fire (1 d and 1 week post-fire, respectively) relative to unburned streams, and returned to pre-fire levels after the 19 d post-fire flood. Although the post-fire increase in NO_3^-, TKN, and total P was not seen in the unburned streams, the measured post-fire concentration was within the pre-fire range of concentrations (Table 7; Bêche, 2005). In general, the direct effects of most fires (prescribed or wildfire) on streams (ash deposition and fire-induced temperature increases) are usually negligible (e.g., Minshall et al., 2001a; Minshall, 2003). However, in the case of some severe wildfires, ash deposition and diffusion of smoke into the water can dramatically increase phosphorous and nitrogen (e.g., Spencer and Hauer, 1991; Minshall et al., 1997).

Other chemical components either increased after the first major post-fire flood (19 d post-fire; SO_4^-), after the spring snowmelt (Ca^{2+}, Mg^{2+}), or did not exhibit any change relative to unburned sites (K^+, NH_4^+, soluble P). The increase in SO_4^- after the first post-fire flood was not long-lasting, and returned to post-fire concentrations within 2 months. Similarly, Stephens et al. (2004) found that after a high-consumption prescribed fire with moderate intensity, SO_4^-, Ca^{2+}, and Mg^{2+} increased for up to 4 months post-fire in intermittent streams in the Lake Tahoe Basin (northern Sierra Nevada). In contrast, SO_4^-, NO_3^-, K^+, and Ca^{2+} increased post prescribed-fire in Sequoia National Park (a southern Sierra Nevada, mixed-conifer forest; Williams and Melack, 1997), and slowly decreased for 3 years post-fire.

Previous studies have found that hillslope steepness, fuel consumption, and the pattern of post-fire precipitation influence changes in stream water chemistry following prescribed fire. For example, Townsend and Douglas (2000) found that low- to moderate-intensity prescribed fires do not cause an increase in sediment delivery to streams (and thus, many chemical components), particularly in areas with gentle slopes. D1 has low- to moderate-gradient hillslopes (mean 20%); thus, accelerated erosion and sediment and ash delivery would not be expected. Similarly, Stephens et al. (2004) also report minimal effects of prescribed fire on water chemistry in areas of gentle hillside slopes (<20%). Furthermore, precipitation in the year following the prescribed fire was below average (Fig. 2b), which may have contributed to the minor effects of the prescribed fire on water chemistry and sediment deposition into the streams.

Low-severity fires that result in partial fuel consumption of the duff and litter layer may be effective in maintaining high infiltration and preventing surface runoff (Robichaud, 2000), which in turn may mediate the effects of fire on water chemistry. Biswell and Schultz (1957) found that following a prescribed fire in the coast ranges of California (a ponderosa pine forest), there was no indication of fire-induced surface runoff or erosion. Similarly, Richter and Ralston (1982) found that there were no changes in water chemistry following multiple prescribed fires in a southeastern USA watershed. Unlike low- to moderate-intensity prescribed fires, large wildfires often result in dramatic increases in stream solutes, which may last for years post-fire (e.g., Tiedemann et al., 1979; Chorover et al., 1994; Hauer and Spencer, 1998; Spencer et al., 2003). In most of these studies, post-fire increases in nutrient concentrations that occur after rainfall or snowmelt have been attributed to overland flow or subsurface transport, except in cases where there were high amounts of ash deposition and/or smoke diffusion into the streams (e.g., Spencer and Hauer, 1991; Roby and Azuma, 1995; Earl and Blinn, 2003).

4.5. Periphyton

Despite the reduction of periphyton biomass as a result of scouring from floods, the prescribed fire resulted in a statistically significant decrease in periphyton biomass in D1 relative to unburned streams, starting 7 weeks post-fire and recovering within 1 year (Fig. 5). Periphyton biomass was lowest post-fire during the April sampling period, which coincides with peak snowmelt when flows averaged 0.07 m^3/s (baseflow prior to the snowmelt averaged 0.01 m^3/s). The pre-fire reduction in periphyton biomass was also associated with high flows (Figs. 2 and 4).

The decrease in periphyton could be a result of the increase in fine sediment in riffle areas, which was found after the spring snowmelt, but which recovered to pre-fire values within 1 year post-fire (Bêche, 2005). Although we have found no published studies examining the effects of prescribed fire on periphyton biomass, wildfire has resulted in decreases in periphyton biomass, which was attributed to increases in sediment (Robinson et al., 1994; Minshall et al., 1995). In contrast, Earl and Blinn (2003) found no change in periphyton biomass following wildfire; however, they only focused on the immediate impacts of fire as a result of smoke and ash deposition.

4.6. Macroinvertebrates

Prior to the first post-fire flood, there were no direct effects of the prescribed fire on benthic macroinvertebrates in D1 (Figs. 4 and 6), based on sampling 2 and 11 d post-fire. Generally, the direct effects of fire on benthic macroinvertebrates are usually minimal or non-existent, as the heating and ash deposition that may be associated with some fires rarely exceeds the lethal limit for macroinvertebrates (Rinne, 1996; Minshall et al., 1997; Minshall, 2003).

The prescribed fire and post-fire flood also had no effect on benthic macroinvertebrate communities as measured by abundance, taxa richness, and Simpson's diversity relative to unburned streams 1 year post-fire (Table 3, Fig. 6). The lack of response by the macroinvertebrate communities to the prescribed fire likely occurs because the fire resulted in no changes in hydrology and fine sediment in pools (Table 6), minimal changes in channel morphology (Bêche, 2005), and the below average post-fire precipitation (Fig. 2b). Similarly, in other studies, low- to moderate-intensity prescribed fire has been shown in other studies to have a subtle or negligible effect on benthic macroinvertebrate communities (Britton, 1991b; Chan, 1998). For example, Britton (1991b) found no change in abundance, dominance, diversity, or functional feeding group composition in response to a prescribed fire in a South African watershed dominated by fynbos (chaparral-like) vegetation. This prescribed fire resulted in very little mortality of riparian vegetation and did not affect sediment inputs (Britton, 1991b). In contrast, Chan (1998) found that prescribed fires in Sequoia National Park resulted in a decrease in benthic macroinvertebrate diversity 1 year post-fire, although no changes in taxa richness or abundance were found. He attributed this response to increases in in-stream fine sediment; however, sampling was not continued after the first year post-fire, so the recovery interval could not be determined (Chan, 1998).

Multivariate analyses of all of the study sites also indicated that the prescribed fire had little (Table 7) to no effect (Fig. 4) on benthic macroinvertebrate communities relative to unburned sites 1 year post-fire. However, when only the adjacent stream communities (D1, prescribed fire treatment, and M1, unburned, Fig. 1) were ordinated together, it appears that the 19 d post-fire rain event may have had different impacts on stream communities in D1 than in M1 (Fig. 6). The Bray-Curtis distance between the prescribed burned community and the adjacent unburned stream was greatest in October 1998 ($d = 0.35$), followed by the sample period 19 d post-fire (11 November 2002; $d = 0.32$). Differences in community composition in 1998 are likely the result of the climatic and hydrological effects of the ENSO winter of 1997–1998 (Bêche, 2005). The differences between the communities following a single flood and an abnormally wet year suggest that the two streams respond differently to flooding events, irrespective of whether they follow a fire. Furthermore, the relatively dry conditions (below average precipitation) in the year following the fire (Fig. 2b) reduced the number of flooding events and may have contributed to the lack of response of the macroinvertebrate community to the prescribed fire.

Because the effects of fire on macroinvertebrates are indirect, and occur as a result of changes in the physical stream habitat, the recovery of communities is dependent on the recovery of the riparian zone and stream channel (Minshall et al., 2001a,c). Following wildfire, stream communities may return to pre-fire conditions in as little as 1–2 years (taxa richness and abundance), or as long as 5–10 years (e.g., predominance of disturbance-adapted species) (Minshall, 2003; Vieira et al., 2004). Stream recovery following large and severe wildfires is rapid on an ecological time scale and even in the midst of recovery, streams maintain the ability to support benthic macroinvertebrates and other biota (Gresswell, 1999; Minshall, 2003).

5. Conclusion

In this study, the prescribed fire treatment was not replicated because of logistical constraints, which is a common problem in fire-effects monitoring (e.g., Schindler, 1998; van Mantgem et al., 2001). However, the trends observed in examining multiple abiotic and biotic parameters suggest that the prescribed fire had no or short-lasting (≤1 year) impacts on Dark Canyon Creek and its riparian zone.

Several factors may have contributed to the lack of effects that this fire had on riparian and stream habitat and communities. For example, a small portion of the watershed (14%) was burned, leaving most of the area intact. Furthermore, the fire was low- to moderate-severity in the riparian zone, leaving most trees intact. The topography of the watershed (low- to moderate-gradient stream) probably contributed to imperceptible changes in fine sediment in pools. Finally, the relatively low amount of precipitation in the year following the fire likely reduced potential inputs of sediment and nutrients to the stream, and resulted in few post-fire flooding events that could have adversely affected riparian and macroinvertebrate communities.

We postulate that the relatively intact riparian zone (14% of the watershed area was patchily burned) acted as a filter between the upland area and the stream (e.g., Naiman and Décamps, 1997), thereby buffering the effects of the prescribed fire from the stream (also reported by Britton, 1991b). Similarly, Minshall (2003) noted that the recovery of streams post-fire is largely dependent on the recovery of the riparian zone, which is usually more rapid than upland recovery. Thus, the extent of the upland area that is burned may be of equal, or even greater, importance than the amount of riparian burn.

In summary, the exclusion of fire from riparian zones may be contributing to the accumulation of fuels and increasing the risk of an uncharacteristically large and severe wildfire (e.g., Ellis, 2001; Skinner, 2003), particularly in areas characterized by frequent, low- to moderate-severity fires (e.g., mixed-conifer zone of the Sierra Nevada, California, USA). Nevertheless, prescribed fire is generally not allowed to burn into riparian zones (e.g., Arno, 1996). In this study, a small prescribed fire that was actively ignited in the riparian zone of Dark Canyon Creek reduced fuel loads in the riparian zone up to 80% and resulted in few short-term effects. Ultimately, fire history, watershed condition, and management goals must be carefully considered for each watershed in determining the appropriateness of implementing prescribed fire as a management strategy.

Acknowledgements

We are greatly indebted to J. Moghaddas for planning, obtaining permits for, and managing the prescribed fire, and to Blodgett Forest Research Station (BFRS) for allowing us to conduct this study. Without the help of many volunteers, BFRS staff, and the U.S. Forest Service (El Dorado Co., CA), and the California Department of Forestry and Fire Protection (CDF), this field experiment would not have been possible. We thank J.F. Bêche, V. Chan, A. Corr, N. Hamey, J. Prentiss, and L. Schweitzer for assistance in fieldwork, L. Aguas, K. Farrar, L. Kitchin, B. Martin, and J. Oh for laboratory assistance, and T. Minear for valuable comments on the manuscript. This research was supported in part by the U.S. Environmental Protection Agency STAR graduate fellowship U-91572101 to L. Bêche.

References

Agee, J.K., 1993. Fire Ecology of the Pacific Northwest Forests. Island Press, Washington, DC.

Arno, M.K., 1996. Reestablishing fire-adapted communities to riparian forests in the ponderosa pine zone. In: Arno, S.F., Hardy, C.C. (Eds.), The Use of Fire in Forest Restoration. USDA Forest Service Intermountain Research Station, Ogden, UT, INT-GTR-341, pp. 42–43.

Bêche, L.A., 2005. Long-term variability, disturbance, and biological traits of aquatic invertebrates in mediterranean-climate streams. Ph.D. dissertation, University of California, Berkeley, CA, USA, 321 pp.

Berg, N., Carlson, A., Azuma, D., 1998. Function and dynamics of woody debris in stream reaches in the central Sierra Nevada California. Can. J. Fish. Aquat. Sci. 55, 1807–1820.

Biswell, H.H., Schultz, A.M., 1957. Surface runoff and erosion as related to prescribed burning. J. Forest. 55, 372–374.

Bragg, D.C., 2000. Simulating catastrophic and individualistic large woody debris recruitment for a small riparian system. Ecology 81, 1383–1394.

Britton, D.L., 1990. Fire and the dynamics of allocthonous detritus in a South African mountain stream. Freshwater Biol. 24, 360.

Britton, D.L., 1991a. Fire and the chemistry of a South African mountain stream. Hydrobiologia 218, 117–192.

Britton, D.L., 1991b. The benthic macroinvertebrate fauna of a South African mountain stream and its response to fire. S. Afr. J. Aquat. Sci. 17, 51–64.

Brown, J.K., 1974. Handbook for inventorying downed wood material. USDA Forest Service Intermountain Forest and Range Experimental Station, Ogden, UT. General Technical Report INT-16, pp. 1–24.

Byram, G.M., 1959. Combustion of forest fuels. In: Davis, K.P. (Ed.), Forest Fire: Control and Use. McGraw-Hill, New York, pp. 113–126.

Chan, I.A., 1998. Aquatic macroinvertebrates of small streams in the Mineral King region of Sequoia National Park: baseline communities and response to prescribed fire. M.S. thesis, University of California, Davis, CA, USA, 109 pp.

Chorover, J., Vitousek, P.M., Everson, D.A., Esperanza, A.M., Turner, D., 1994. Solution chemistry profiles of mixed-conifer forests before and after fire. Biogeochemistry 26, 115–144.

Clesceri, L.S., Greenberg, A.E., Eaton, A.D., 1998. Standard Methods for the Examination of Water and Wastewater. American Public Health Association, Washington, DC.

DeBano, L.F., Neary, D.G., 1996. Effects of fire on riparian systems. In: DeBano, L.F., Baker Jr., M.B., Gottfried, G.J., Solis-Garza, G., Edminster, C.B., Neary, D.G., Allen, L.S., Hamre, R.H. (Eds.), Effects of Fire on Madrean Province Ecosystems: A Symposium Proceedings. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. General Technical Report RM-GTR-289, pp. 69–76.

DeBano, L.F., Ffolliott, P.F., Baker, M.B., 1996. Fire severity effects on water resources. In: DeBano, L.F., Baker Jr., M.B., Gottfried, G.J., Solis-Garza, G., Edminster, C.B., Neary, D.G., Allen, L.S., Hamre, R.H. (Eds.), Effects of Fire on Madrean Province Ecosystems: A Symposium Proceedings. Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, CO. General Technical Report RM-GTR-289, pp. 77–84.

del Rosario, R.B., 2000. Aquatic insect responses to hydrologic intermittency and organic enrichment in headwater streams. Ph.D. dissertation, University of California, Berkeley, CA, USA, 140 pp.

Dwire, K.A., Kauffman, J.B., 2003. Fire and riparian ecosystems in landscapes of the western USA. For. Ecol. Manage. 178, 61–74.

Earl, S.R., Blinn, D.W., 2003. Effects of wildfire ash on water chemistry and biota in South-Western U.S.A. streams. Freshwater Biol. 48, 1015–1030.

Elliott, K.J., Hendrick, R.L., Major, A.E., Vose, J.M., Swank, W.T., 1999. Vegetation dynamics after a prescribed fire in the southern Appalachians. For. Ecol. Manage. 114, 199–213.

Ellis, L.M., 2001. Short-term response of woody plants to fire in a Rio Grande riparian forest, Central New Mexico, USA. Biol. Conserv. 97, 159–170.

Erman, N.A., 1996. Status of Aquatic Invertebrates, vol. II, Chapter 35. University of California Centers for Water and Wildland Resources, Davis, CA. Sierra Nevada Ecosystem Project, Final Report to Congress.

Everett, R., Schellhass, R., Ohlson, P., Spurbeck, D., Keenum, D., 2003. Continuity in fire disturbance between riparian and adjacent sideslope Douglas-fir forests. For. Ecol. Manage. 175, 31–47.

Gresswell, R.E., 1999. Fire and aquatic ecosystems in forested biomes of North America. Trans. Am. Fish. Soc. 128, 221.

Hauer, F.R., Spencer, C.N., 1998. Phosphorus and nitrogen dynamics in streams associated with wildfire: a study of immediate and longterm effects. Int. J. Wildland Fire 8, 183–198.

Hilton, S., Lisle, T.E., 1993. Measuring the fraction of pool volume filled with fine sediment. Research note, USDA Forest Service, Pacific Southwest Research Station, Albany, CA, PSW-RN-414-WEB.

Huntzinger, M., 2003. Effects of fire management practices on butterfly diversity in the forested western United States. Biol. Conserv. 113, 1–12.

Kattelmann, R., Embury, M., 1996. Riparian Areas and Wetlands, vol. 3. Sierra Nevada Ecosystem Project, Wildland Resources Center, University of California, Davis. Report No. 38, pp. 201–274.

Keifer, M., Stephenson, N.L., Manley, J., 2000. Prescribed fire as the minimum tool for wilderness forest and fire regime restoration: a case study from the Sierra Nevada, CA. In: Cole, D.N., McCool, S.F., Borrie, W.T., O'Loughlin, J. (Eds.), Proceedings of the Wilderness Science in a Time of Change Conference, 23–27 May 1999. USDA Forest Service, Rocky Mountain Research Station, Missoula, MT, RMRS-P-15-VOL-5, pp. 266–269.

Keller, E.A., Swanson, F.J., 1979. Effects of large organic material on channel form and fluvial processes. Earth Surf. Proc. Land. 4, 361–380.

Kilgore, B.M., Taylor, D., 1979. Fire history of a sequoia mixed conifer forest. Ecology 60, 129–142.

Kruskal, J.B., 1964. Nonmetric multidimensional scaling: a numerical method. Psychometrika 29, 115–129.

Lamb, E.G., Mallik, A.U., Mackereth, R.W., 2003. The early impact of adjacent clearcutting and forest fire on riparian zone vegetation in northeastern Ontario. For. Ecol. Manage. 177, 529–538.

Lamberti, G.A., Resh, V.H., 1985. Comparability of introduced tiles and natural substrata for sampling lotic bacteria, algae and macroinvertebrates. Freshwater Biol. 15, 21–30.

Legendre, P., Legendre, L., 1998. Numerical Ecology. Elsevier, Amsterdam.

Lienkaemper, G.W., Swanson, F.J., 1987. Dynamics of large woody debris in streams in old-growth douglas-fir forests. Can. J. For. Res. 17, 150–156.

Martin, R.E., Dell, J.D., 1978. Planning for prescribed burning in the inland northwest. USDA Forest Service, Pacific Southwest

Forest and Range Experiment Station Portland, OR, GTR-PNW-175.

McCune, B., Mefford, M.J., 1999. PC-ORD: multivariate analysis of ecological data. MjM Software, Version 4.27, Gleneden Beach, OR, USA.

McCune, B., Grace, J.B., Urban, D.L., 2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, OR.

Miller, C., Urban, D.L., 2000. Modeling the effects of fire management alternatives on Sierra Nevada mixed-conifer forests. Ecol. Appl. 10, 85–94.

Minshall, G.W., 2003. Responses of stream benthic macroinvertebrates to fire. For. Ecol. Manage. 178, 155–161.

Minshall, G.W., Brock, J.T., Varley, J.D., 1989. Wildfires and Yellowstone's stream ecosystems. BioScience 39, 707–715.

Minshall, G.W., Robinson, C.T., Royer, T.V., Rushforth, S.R., 1995. Benthic community structure in two adjacent streams in Yellowstone National Park five years after the 1988 wildfires. Great Basin Nat. 55, 193–200.

Minshall, G.W., Robinson, C.T., Lawrence, D.E., 1997. Postfire responses of lotic ecosystems in Yellowstone National Park, USA. Can. J. Fish. Aquat. Sci. 54, 2509–2525.

Minshall, G.W., Brock, J.T., Andrews, D.A., Robinson, C.T., 2001a. Water quality, substratum and biotic responses of five central Idaho (USA) streams during the first year following the Mortar Creek fire. Int. J. Wildland Fire 10, 185–199.

Minshall, G.W., Robinson, C.T., Lawrence, D.E., Andrews, D.A., Brock, J.T., 2001b. Benthic macroinvertebrate assemblages in five central Idaho (USA) streams over a 10-year period following disturbance by wildfire. Int. J. Wildland Fire 10, 201–213.

Minshall, G.W., Royer, T.V., Robinson, C.T., 2001c. Response of the Cache Creek macroinvertebrates during the first 10 years following disturbance by the 1988 Yellowstone wildfires. Can. J. Fish. Aquat. Sci. 58, 1077–1088.

Naiman, R.J., Décamps, H., 1997. The ecology of interfaces: riparian zones. Annu. Rev. Ecol. Syst. 28, 621–658.

Olson, D.L., Agee, J.K., 2005. Historical fires in Douglas-fir dominated riparian forests of the southern Cascades, Oregon. Fire Ecol. 1, 50–74.

Platts, W.S., Armour, C., Booth, B.D., Bryant, M., Buford, J.L., Cuplin, P., Jensen, S., Lienkaemper, G.W., Minshall, G.W., Monsen, S.B., Nelson, R.L., Sedell, J.R., Tuhy, J.S., 1987. Methods for evaluating riparian habitats with applications to management. USDA Forest Service, Intermountain Research Station, Ogden, UT, INT-GTR 221.

Resh, V.H., Brown, A.V., Covich, A.P., Gurtz, M.E., Li, H.W., Minshall, G.W., Reice, S.R., Sheldon, A.L., Wallace, J.B., Wissmar, R.C., 1988. The role of disturbance in stream ecology. J. North Am. Benthol. Soc. 7, 433–455.

Richter, D.D., Ralston, C.W., 1982. Prescribed fire: effects on water quality and forest nutrient cycling. Science 215, 661–663.

Rinne, J.N., 1996. Short-term effects of wildfire on fishes and aquatic macroinvertebrates in the southwestern United States. North Am. J. Fish Manage. 16, 653–658.

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

Robinson, C.T., Rushforth, S.R., Minshall, G.W., 1994. Diatom assemblages of streams influenced by wildfire. J. Phycol. 30, 209–216.

Roby, K.B., Azuma, D.L., 1995. Changes in a reach of a northern California stream following wildfire. Environ. Manage. 19, 591–600.

Romme, W.H., 1982. Fire and landscape diversity in subalpine forests of Yellowstone National Park. Ecol. Mon. 52, 199–221.

Russell, W.H., McBride, J.R., 2001. The relative importance of fire and watercourse proximity in determining stand composition in mixed conifer riparian forests. For. Ecol. Manage. 150, 259–265.

Schindler, D.W., 1998. Replication versus realism: the need for ecosystem-scale experiments. Ecosystems 1, 323–334.

Skinner, C.N., 2003. A tree-ring based fire history of riparian reserves in the Klamath mountains. In: Faber, Phyllis, M. (Eds.), California Riparian Systems: Processes and Floodplain Management, Ecology, and Restoration. Pickleweed Press, Mill Valley, CA, pp. 116–119.

Spencer, C.N., Hauer, F.R., 1991. Phosphorous and nitrogen dynamics in streams during a wildfire. J. North Am. Benthol. Soc. 10, 24–30.

Spencer, C.N., Gabel, K.O., Hauer, F.R., 2003. Wildfire effects on stream food webs and nutrient dynamics in Glacier National Park, USA. For. Ecol. Manage. 178, 141–153.

Steinman, A.D., Lamberti, G.A., 1996. In: Hauer, F.R., Lamberti, G.A. (Eds.), Biomass and Pigments of Benthic Algae. Academic Press, San Diego, CA, pp. 295–313.

Stephens, S.L., 1998. Effects of fuels and silvicultural treatments on potential fire behavior in mixed conifer forests of the Sierra Nevada, CA. For. Ecol. Manage. 105, 21–34.

Stephens, S.L., 2001. Fire history differences in adjacent Jeffrey pine and upper montane forests in the eastern Sierra Nevada. Int. J. Wildland Fire 10, 161–167.

Stephens, S.L., Finney, M.A., 2002. Prescribed fire mortality of Sierra Nevada mixed conifer tree species: effects of crown damage and forest floor combustion. For. Ecol. Manage. 162, 261–275.

Stephens, S.L., Collins, B.M., 2004. Fire regimes of mixed conifer forests in the north central Sierra Nevada at multiple spatial scales. Northwest Sci. 78, 12–23.

Stephens, S.L., Meixner, T., Poth, M., McGurk, B., Payne, D.Y., 2004. Prescribed fire, soils, and stream water chemistry in a watershed in the Lake Tahoe Basin, California. Int. J. Wildland Fire 13, 27–35.

Swanson, F.J., 1981. Fire and geomorphic processes. In: Mooney, H.A., Bonnicksen, T.M., Christensen, N.L., Lotan, J.E., Reiners, W.A. (Eds.), Proceedings of the Conference on Fire Regimes and Ecosystem Properties. USDA Forest Service Washington, DC. General Technical Report WO-26, pp. 201–420.

Tiedemann, A.R., Conrad, C.E., Dieterich, J.H., Hornbeck, J.W., Megahan, W.F., Viereck, L.A., Wade, D.D., 1979. Effects of fire on water: a state-of-knowledge review. USDA Forest Service Denver, CO. General Technical Report WO-10.

Timoney, K.P., Peterson, G., Wein, R., 1997. Vegetation development of boreal riparian plant communities after flooding, fire, and logging, Peace River, Canada. For. Ecol. Manage. 93, 101–120.

Townsend, S.A., Douglas, M.M., 2000. The effect of three fire regimes on stream water quality, water yield and export coefficients in a tropical savanna (northern Australia). J. Hydrol. 229, 118–137.

Underwood, A.J., 1993. The mechanics of spatially replicated sampling programmes to detect environmental impacts in a variable world. Aust. J. Ecol. 18, 99–116.

Underwood, A.J., 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. Ecol. Appl. 4, 3–15.

USDA and NRCS, 2004. The PLANTS Database, Version 3.5 (http://plants.usda.gov). National Plant Data Center, Baton Rouge, LA 70874-4490, USA.

van Mantgem, P., Schwartz, M., Keifer, M., 2001. Monitoring fire effects for managed burns and wildfires: coming to terms with pseudoreplication. Nat. Area J. 21, 266–273.

van Wagtendonk, J.W., Benedict, J.M., Sydoriak, W.M., 1996. Physical properties of woody fuel particles of Sierra Nevada conifers. Int. J. Wildland Fire 6, 117–123.

van Wagtendonk, J.W., Benedict, J.M., Sydoriak, W.M., 1998. Fuel bed characteristics of Sierra Nevada conifers. Western J. Appl. For. 13, 73–84.

Vieira, N.K.M., Clements, W.H., Guevara, L.S., Jacobs, B.F., 2004. Resistance and resilience of stream insect communities to repeated hydrologic disturbances after a wildfire. Freshwater Biol. 49, 1243–1259.

Waltz, A.E.M., Fulé, P.Z., Covington, W.W., Moore, M.M., 2003. Diversity in ponderosa pine forest structure following ecological restoration treatments. Forest Sci. 49, 885–900.

Williams, M.R., Melack, J.M., 1997. Effects of prescribed burning and drought on the solute chemistry of mixed-conifer forest streams of the Sierra Nevada, California. Biogeochemistry 39, 225–253.

Wondzell, S.M., King, J.G., 2003. Postfire erosional processes in the Pacific Northwest and Rocky Mountain regions. For. Ecol. Manage. 178, 75–87.

Young, M.K., 1994. Movement and characteristics of stream-borne coarse woody debris in adjacent burned and undisturbed watersheds in Wyoming. Can. J. For. Res. 24, 1933–1938.