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TECHNICAL ARTICLES

PREFIRE AND POSTFIRE EROSION OF SOIL NUTRIENTS WITHIN A CHAPARRAL WATERSHED

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Prescribed burns are an effective and increasingly popular strategy for inhibiting wildfires. The goal of this study was to characterize soil nutrient loss after a prescribed fire within a chaparral watershed in southern California. The study compared hillslope sediments for approximately 1 year before the fire with those during the year after the fire. Samples were collected in 11 sediment traps located randomly along fall line transects throughout a steep, 0.65-ha watershed (~65% average slope). Concentrations of soluble ammonium nitrogen, nitrate nitrogen, phosphate phosphorus, Ca, Mg, K, and Na, exchangeable ammonium nitrogen, and total C and N were analyzed.

Sediment production averaged approximately 1600 g per sediment trap for the entire year before the fire but increased to approximately 10,700 g for the year after the fire. The average sedimentation rates for the less than 2-mm fraction doubled from a prefire rate of 2.7 to 7.5 g d^{-1} after the fire and increased to 86 g d^{-1} after the first major rain event after the fire. The greatest concentrations of soluble and exchangeable base cations (Ca, Mg, K, Na) in the eroded sediment were measured at the first sampling date after the fire. Input and microbial mineralization of fresh organic matter approximately 4 months after the fire is believed to be the source of increases in the concentrations of soluble nitrate nitrogen and phosphate phosphorus and exchangeable ammonium nitrogen. Our findings indicate that prescribed fire may cause less loss of N and P than wildfires. (Soil Science 2006;171:915–928)

Key words: Prescribed burn, hillslope sediment production, chaparral watershed, nutrient erosion, prescribed fire.

CHAPARRAL makes up more than 7% of the native vegetation in California (Hanes, 1976) and is particularly abundant in the southern part of the state. Chaparral ecosystems, often dominated by species of ceanothus (*Ceanothus Spp.*) and chamise (*Adenostoma fasciculatum*) shrubs, have a Mediterranean climate with rainfall mostly between October and March. The absence of rain at a time of year when temperatures are at their greatest causes concern relative to fire management. High shrub densities, the buildup of dead plant material, and drought conditions—all help to promote a 10- to 40-year fire frequency (Muller et al., 1968).

Aside from the obvious damage to property that wildfires may cause, the intense heat given off by these fires can also impact the soil and its nutrients. Severe fires are prevalent in areas where there is a buildup of dead organic debris. These fires can destroy not only vegetation and litter but also the organic matter in the upper horizons of the soil (McNabb and Cromack, 1990). In chaparral, this leads to volatilization and loss of valuable nutrients, such as N and P,

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which are typically the limiting nutrients in these rocky soils. Nutrients are also lost through soil erosion, which is prevalent immediately after the fire and may remain high for a number of years (DeBano and Conrad, 1976; Robichaud and Waldrop, 1994; Cannon and Reneau, 2000).

Soil erosion not only removes nutrients from ecosystems but it can also be a nonpoint source of pollutants. The Los Angeles Basin currently experiences exceptionally high levels of nitrogen deposition. This is primarily dry deposition and can occur in the oxidized form of N from automobile exhaust or the reduced form from dairy operations in the nearby Chino-Norco area (Russell and Cass, 1986). The San Gabriel Mountains have some of the highest total N deposition rates in North America, with dry deposition rates alone of 35 to 45 kg N ha^{-1} yr^{-1} (Fenn, 1991; Grosjean and Bytnerowicz, 1993; Bytnerowicz and Fenn, 1996). High erosion rates brought about by wildfires in these areas under high rates of N deposition lead to the elevated nitrate levels (140–500 μmol/L) observed in stream discharge (Riggan et al., 1985).

Prescribed fires have been used by the United States Forest Service in California and Arizona (Hibbert et al., 1974; Green, 1981) to restore forest ecosystems that have been impacted by fire suppression, management, or insect outbreaks, and this use has grown in recent years (Hardy and Arno, 1996). Prescribed burns are usually conducted when soils are moist. This lowers the fire severity and volatilization rates of soil nutrients, thereby preserving these nutrients better than wildfire (Dunn et al., 1979).

Fires, in general, release organically bound nutrients and increase their availability within the soil (Christensen and Muller, 1975; DeBano and Conrad, 1974; Dunn et al., 1979). Although prescribed fires help to conserve nutrients by reducing volatilization losses relative to wildfires, they may expose readily available nutrients to erosion. Soil that has been covered by vegetation or litter for decades has been sheltered from erosional processes and is particularly vulnerable when the cover is removed (Johansen et al., 2001). When fire destroys the vegetation and litter, the soil is susceptible to dry ravel (Anderson et al., 1959; Doehring, 1968; DeBano and Conrad, 1976) and to water erosion during the first major rain event (Rice, 1974; Dunne and Leopold, 1978; Bryan, 2000).

Dry ravel is the movement of particles by rolling, sliding, and bouncing downslope under the influence of gravity (Krammes, 1965; Rice, 1974; 1982). Under normal conditions, dry ravel is the dominant process of hillslope erosion on steep, arid chaparral landscapes like those found in the San Gabriel Mountains and generally only involves the upper few centimeters of soil (Anderson et al., 1959; Krammes, 1965; Wohlgemuth, 1986). The process is increased after a fire because the shrub stems and branches that hold the litter and colluvial material in place on these steep slopes (50% gradient and higher) (Crawford, 1962; Dunn et al., 1988) are removed. This dry ravel is rich in nutrients from the abundant ash produced by the fire and generally accumulates on lower hillslopes and in drainages until it is removed by rainstorms (Krammes, 1965; Rice, 1974; Scott and Williams, 1978).

Water erosion usually takes the form of increased splash erosion, rill formation, and overland flow after a fire. Many of the soluble nutrients not taken up by resprouting vegetation or annuals are lost downslope with the flow of water. An important factor in the postfire increase in runoff and sediment removal is the development of water-repellent soils. Water repellency forms when nonpolar organic compounds, vaporized during combustion at the soil surface, condense and concentrate on grains in the cooler soil material at a relatively shallow depth (~5 cm) (DeBano et al., 1970; Wright and Bailey, 1982). This condition is primarily produced by low and moderate intensity burns because the coatings are destroyed at temperatures greater than approximately 288 °C (DeBano et al., 1976; DeBano, 1981). The repellent zone impedes water infiltration so that the overlying layer becomes saturated during rainstorms, loses cohesive strength, and erodes readily.

Despite universal recognition of erosion as a prevalent postfire process, little is known concerning nutrient losses associated with the eroded material. The objective of this study was to determine the changes in both sedimentation rates and nutrient contents of erosional debris in a chaparral ecosystem after a prescribed fire.

MATERIALS AND METHODS

Study Area Description

The study area is on the San Dimas Experimental Forest in the young, geologically active San Gabriel Mountains, approximately 58 km northeast of Los Angeles, California

(34°12'45''N, 117°45'30''W). The elevation of the watershed ranges from 820 to 990 m. The climate is Mediterranean, with hot, dry summers and cool, wet winters. Mean annual precipitation is approximately 680 mm. Precipitation mainly occurs as rain between November and April, with little or no rain falling from May to October.

The natural vegetation is chaparral, characterized by sclerophyllous leaves, 1 to 3 meters of plant height, and dense canopies. Common species are chamise, hoaryleaf ceanothus (*Ceanothus crassifolius*), bigberry manzanita (*Arctostaphylos glauca*), sugar bush (*Rhus ovata*), scrub oak (*Quercus dumosa*), black sage (*Salvia mellifera*), and wild buckwheat (*Eriogonum fasciculatum*). In 2001, the stand age of the chaparral was 41 years, because the watershed had last burned during a 1960 wildfire that consumed 88% of the 7727-ha forest.

The bedrock consists of strongly banded and foliated gneisses and schists, which have been subjected to a high degree of alteration, faulting, folding, fracturing, and weathering. As a result, the rocks are poorly consolidated and very unstable (Sinclair, 1953). Slopes are mostly linear with very narrow shoulder and footslope zones and an average gradient of 65% on both upper and lower slope positions. This corresponds to very steep slopes that are close to the limiting angles for ground cover to remain in place on its own (Young, 1972).

Soils are shallow and coarse-textured, with rock fragments throughout. Movement of soil on these steep slopes is too frequent to allow appreciable development of soil horizons. Thus, the soil character is determined largely by the composition of underlying rock and by its susceptibility to weathering (Graham et al., 1988). The soils are generally loose with low bulk densities, and shrub roots anchor in the bedrock fractures and not in the soil. Rock outcrops are common. Soils mapped in the general area of the study site are in the Trigo series (loamy, mixed, nonacid, thermic, shallow Typic Xerorthents) (Ryan, 1991).

This research focused on a 0.65-ha watershed that was burned on May 15, 2001 as a part of a larger prescribed burn conducted for 2 days in the San Dimas Experimental Forest. A helitorch dripped the flaming material (a gasoline and catalyst thickener combination that creates a flaming gel) onto the chaparral vegetation to ignite it. Fuel conditions, topography, and fire behavior dictated the pattern in which the flaming gel was applied. In areas that were not conducive to burning, extra passes were made with the helitorch. The fire burned for 1 to 3 h in the target watershed. Burn severity was low, as estimated qualitatively from the color of the ash (black) and low degree of litter and duff consumption (~60%) (Wells et al., 1979; Robichaud and Waldrop, 1994).

Field Methods

Sediment traps, in place in this watershed since 1994, were sampled before and after the prescribed burn at regular intervals. The traps were 30.5 × 19 × 15.25-cm covered metal boxes that were installed flush with the ground. The 30.5-cm end was placed perpendicular to the fall line and had a 7- to 8-cm wide metal apron attached to its top to provide a stable connection between the hillslope and the box that would not erode or become undermined. The opening to collect sediment faced upslope and allowed materials smaller than 1 to 2 cm to enter. The traps were designed with a 0.6-cm-diameter hole drilled in each corner to allow captured rainfall to drain out. The effect of leaching soluble nutrients from sediment traps was considered insignificant because of the low rainfall and steep slopes in this area. Sediment traps were all randomly located along fall line transects that were established on uniform slope facets or at the slope/channel interfaces. Each trap collected from similar cross sections, a 30.5-cm swath of the hillslope contour where it was located. Samples collected from sediment traps at 8 different periods were used for analyses. The first two collections represented prefire conditions for a period of 1 year. The remaining six collections represented postfire conditions for a total duration of 11 months. Prefire sampling occurred on January 9, 2001 and on April 30, 2001. Postfire sampling was done on the following dates: May 18, 2001, July 19, 2001, September 5, 2001, November 20, 2001, December 12, 2001, and March 21, 2002. The time interval that these samples represent can be found in Table 1. Only samples that contained more than 20 g of the less than 2-mm fraction of sediment at a given sampling date were analyzed chemically because of the minimum amount of sample needed for full analysis. This led to the use of samples from 11 total sediment traps (Fig. 1). Samples collected in the traps are representative subsamples of the sediment fluxes across hillslopes and being

TABLE 1

Rainfall accumulation and rates of intensity for San Dimas Experimental Station, California (Western Regional Climate Center)

Sampling period	Rainfall		
	Total accumulation	Average intensity	Maximum intensity
	mm	-mm/h-	
April 26, 2000–January 9, 2001	65	2	4
January 9, 2001–April 30, 2001	553	5	11
April 30, 2001–May 18, 2001	0	0	0
Fire event			
May 18, 2001–July 19, 2001	8	2	3
July 19, 2001–September 5, 2001	0	0	0
September 5, 2001–November 20, 2001	17	5	13
November 20, 2001–December 12, 2001	52	8	17
December 12, 2001–March 21, 2002	149	6	11

delivered to stream channels. These sites, when compared with other sites in the same area, were not the highest or the lowest sediment producers and may provide a good representation of the area especially because the area is small (0.65 ha), the sites are randomly located, and statistical analyses were used. When data for total sediment production rate for these 11 traps were compared across the dates with the remaining 64 traps, they exhibited the same trends, and the values were not significantly different ($\alpha = 0.05$). Originally, sample sites had been separated on the basis of upper and lower hillslope positions but because of the continuous steepness of the entire slope, results between the two groups were not significant ($\alpha = 0.05$). This agrees with physical evidence by Wohlgemuth (1986) who found no difference in surface transport between upper and lower hillslope positions. The groups were merged, and the analyses presented represent comparison among all 11 sites by date.

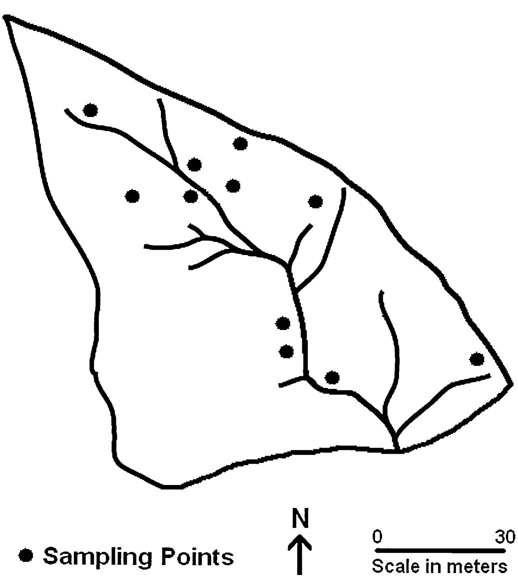


Fig. 1. Location of sample points in the study watershed in San Dimas Experimental Forest, California.

Laboratory Methods

All samples used were air-dried and sieved to yield the less than 2-mm fraction of sediment. Concentrations were corrected for moisture content and expressed on a dry-weight basis. Total C and N were determined using the dry combustion method (Nelson and Sommers, 1986) with ground subsamples and a C/N/S autoanalyzer (Carlo Erba Strumentazione, Milan, Italy). Soil pH was measured on a 1:1 soil/water mixture using a glass electrode and pH meter (McLean, 1982). A 1:5 soil/water suspension was used to extract soluble ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), phosphate phosphorus (PO_4-P), Ca, Mg, K, and Na (Rhoades, 1986). Exchangeable NH_4-N values were obtained using a 2 *M* KCl extractant (Keeney and Nelson, 1986), and exchangeable Ca, Mg, K, and Na were extracted using 1 *M* NH_4OAc at pH 7 (Thomas, 1986). Samples from each site studied were extracted and analyzed in duplicate. Exchangeable NH_4-N and soluble NH_4-N and PO_4-P were analyzed with colorimetric methods on a segmented flow autoanalyzer (Alpkem Corp, Clackamas, OR). Soluble NO_3^- was analyzed using colorimetric methods on an ion chromatograph (Dionex Corp, Sunnyvale, CA). Exchangeable and soluble Ca, Mg, K, and Na analyses were done using ICP-OES (Perkin Elmer, Wellesley, MA).

Values given for exchangeable cation concentrations are the extracted concentrations with the soluble concentrations subtracted from them.

Statistical Analysis

Comparisons were made between dates for each of the quantities analyzed. A Box-Cox linearity plot was used which determined that a log transformation was suitable for all data except NH_4-N (soluble, exchangeable) and total C and N, which were not transformed. An analysis of variance test was used for all data sets, and the least significant difference between each mean for each treatment was found. Data presented in the text represent the backtransformed means. The SAS computer program was used for all of the previous analysis (SAS Institute Inc, 1999).

RESULTS

Sediment Production

Precipitation data for Tanbark Flats, California during our study period are provided in Table 1 (Western Regional Climate Center). The year before the fire (2000–2001) had more total rainfall, and the year after the fire (2001–2002) had more intense storms with greater average precipitation rates. The prefire year was more indicative of a normal wet season in terms of total rainfall accumulation, whereas the second year had a much drier winter.

The year before the fire averaged approximately 1600 g of total sediment for each sediment trap, whereas the year after the fire produced 10,700 g (total sediment data not shown). The sampling periods, excluding November 20 to December 12, 2001, had average sedimentation rates of 11 g/d for the total sediment and 3 g/d for the less than 2-mm fraction (Fig. 2). There was an increase in the average sedimentation rate for the total sediment (18 g/d) and the less than 2-mm fraction (7.5 g/d) directly after the fire (Fig. 2). These new rates decreased to prefire rates 2 months later (July 19, 2001). The period between November 20, 2001 and December 12, 2001 had the first major winter storm and the greatest rates of precipitation. This, in conjunction with the loss of the protective vegetation and litter layer and formation of water-repellent soils because of the fire (DeBano and Letey, 1969), contributed to the greatest average sedimentation rate observed during the study (328 g/d for total sediment, 86 g/d for the less than 2-mm fraction) for all of the 11 sampling points shown in Figure 2.

Carbon and Nitrogen

Average sediment total C concentrations ranged from 6% to 11% (Table 2). There was an increase in C concentration after the fire (although not significantly different from prefire levels) from 8% to 11% followed by a significant decrease from 11.4% to 7.40%. The average total N concentrations ranged from 0.3% to

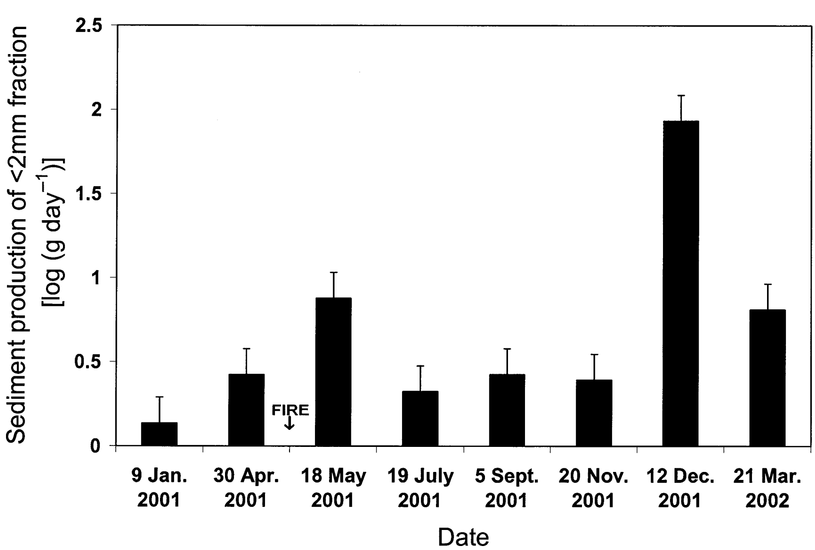


Fig. 2. Log-transformed values of average sediment production rates for the less than 2-mm fraction of sediment. Error bars reflect the least significant difference between the means for each sampling date where $\alpha < 0.05$.

TABLE 2

Averages for percentage of total N and C and C/N ratio for each sampling date

Sampling date	Total N	Total C	C/N
	- - - - -%- - - - -		
January 9, 2001	0.372	9.04	24.3
April 30, 2001	0.332	7.96	23.7
Fire event			
May 18, 2001	0.390	11.4	28.9*
July 19, 2001	0.382	7.40	18.7*
September 5, 2001	0.377	6.39	17.2*
November 20, 2001	0.315	5.87	18.5*
December 12, 2001	0.272	5.74	21.3
March 21, 2002	0.400	8.46	21.1

*Significant from prefire values at 0.05 probability level.

0.4%, with no significant change throughout the entire study period (Table 2). The C/N ratios increased significantly immediately after the fire to approximately 29 from prefire levels of approximately 24 (Table 2). Two months after the fire (July 19, 2001), this had dropped significantly to approximately 19. By December 12, 2001, the C/N ratio had increased to approximately 21 and was not significantly different from prefire levels.

Immediately after the fire (May 18, 2001), the concentration of soluble NH_4-N (Fig. 3A) had increased significantly to 29.0 mg kg^{-1}. This was followed 2 months later by a significant decrease to prefire levels (6.59–16.4 mg kg^{-1}) for the remainder of the study period.

Immediately after the fire, the exchangeable NH_4-N (Fig. 3B) concentration was very similar to prefire conditions (38.2 mg kg^{-1}). By the next sampling date, July 19, 2001, the exchangeable NH_4-N increased to 56.4 mg kg^{-1}. The sediment concentration remained steady until November 20, 2001 where it returned to prefire values.

Soluble NO_3-N concentrations (Fig. 3C) followed the trend same as that of exchangeable NH_4-N, with no significant difference between immediate postfire and prefire levels (1.0-1.7 mg kg^{-1}) and with values increased significantly on the next sampling dates of July 19, 2001 and September 5, 2001 to 11 and 23 mg kg^{-1}, respectively. The concentration then decreased to approximately 7 mg kg^{-1} for the remainder of the study period. Although this continued to follow the trend same as that of exchangeable NH_4-N, these last values were significantly higher than what the prefire levels had been.

Phosphorus

Soluble inorganic PO_4-P (Fig. 3D) differed from all the other nutrients analyzed in that the concentration decreased significantly after the fire, dropping from a prefire level of approximately 6 to 2.47 mg kg $^{-1}$. On both the sampling dates of July 19, 2001 and September 5, 2001, this increased significantly to 4.58 and 9.45 mg kg^{-1}, respectively. Concentrations of PO_4-P then decreased to prefire levels for the rest of the study period.

Base Cations

The soluble (Fig. 4A) and exchangeable (Fig. 4B) base cations exhibited similar trends with respect to the prefire and postfire conditions. All concentrations of soluble and exchangeable base cations significantly increased directly after the fire, on the sampling date of May 18, 2001, when compared with prefire levels. Soluble Ca, Mg, and Na concentrations increased 4-fold from a range of 6.42 to 44.8 mg kg^{-1} to a range of 29.1 to 168 mg kg^{-1}. Soluble K had an even greater increase, a factor of 40, from a prefire concentration of 19.0 to 774 mg kg^{-1} on May 18, 2001. All soluble base cations resumed prefire levels by the sampling date of December 12, 2001. Exchangeable Ca, Mg, and Na concentrations also increased 3- or 4-fold immediately after the fire, from a range of 24.5 to 2269 mg kg^{-1} to a range of 78.9 to 9114 mg kg^{-1}. Exchangeable K increased by a factor of 18 from a prefire concentration of 90.3 to 1619 mg kg^{-1} on May 18, 2001. All exchangeable base cations resumed prefire levels by December 12, 2001 (i.e., within 7 months).

DISCUSSION

Sediment Production Rates

The trend in sedimentation rates (Fig. 2) after the prescribed fire was similar to the trends found by other authors. These trends included significant increases directly after the fire and during the first major rain, mostly because of the losses in the vegetative and litter cover and the influence of dry ravel (Anderson et al., 1959; Wells et al., 1979; Soto et al., 1994) and water erosion (DeBano and Conrad, 1976), respectively. The winter or rainy season after the fire was relatively dry compared with the average, so erosion was likely less than what might have occurred in an average to above average rainfall in a year.

Carbon

Carbon concentrations (Table 2) increased immediately after the fire probably because of a C-rich ash (Christensen and Muller, 1975; Herman and Rundel, 1989; Monleon et al., 1997). Close inspection of the sediment from both hillslope positions using a dissecting microscope revealed that all of the recognizable organic matter remaining was charred. Christensen and Muller (1975) found a C concentration of approximately 17% associated with ash after burning of chaparral litter and vegetation. We found lower C concentrations possibly because ash had mixed with sediments, and the lower burn severity produced lower inorganic C concentrations (Ulery et al., 1993). Inorganic C was not assumed to be a factor because few samples effervesced when 1 *N* HCl was applied and all samples consisted of blackened soil and organic matter rather than white, completely ashed plant material with high concentrations of $CaCO_3$. The increase in C concentration of sediments after the fire was most likely the result of the initial erosion of burned organic matter from the litter layers (Doehring, 1968), the organics involved in

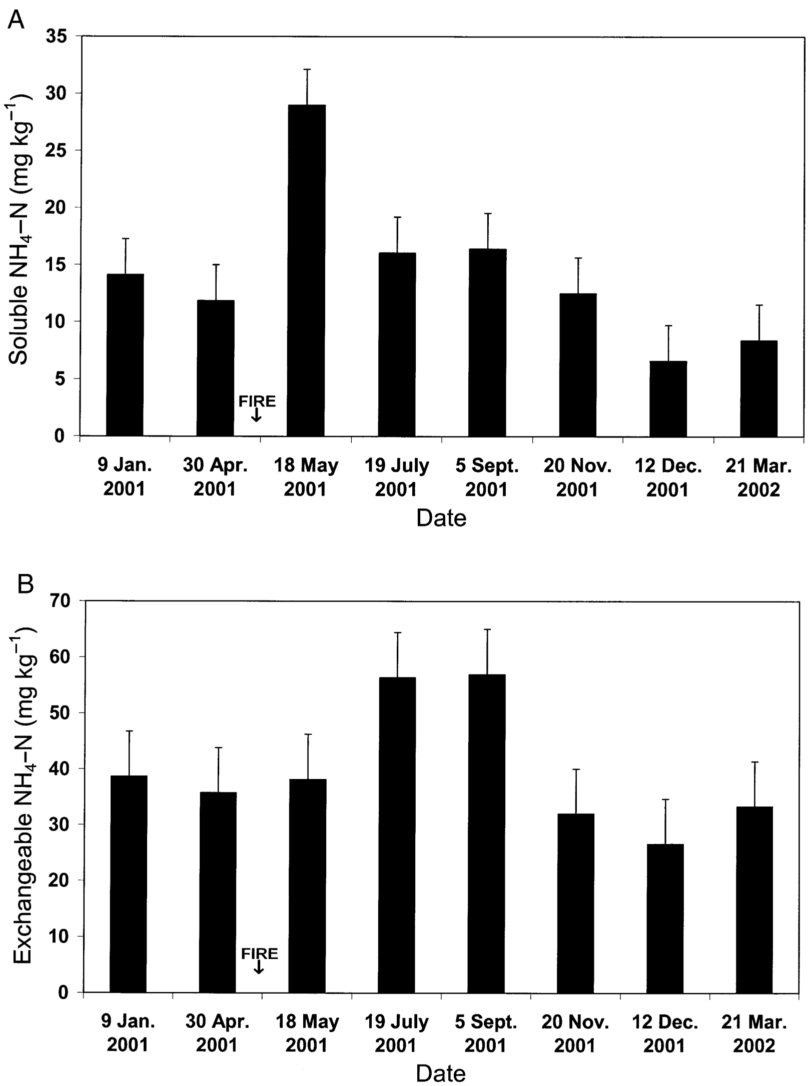
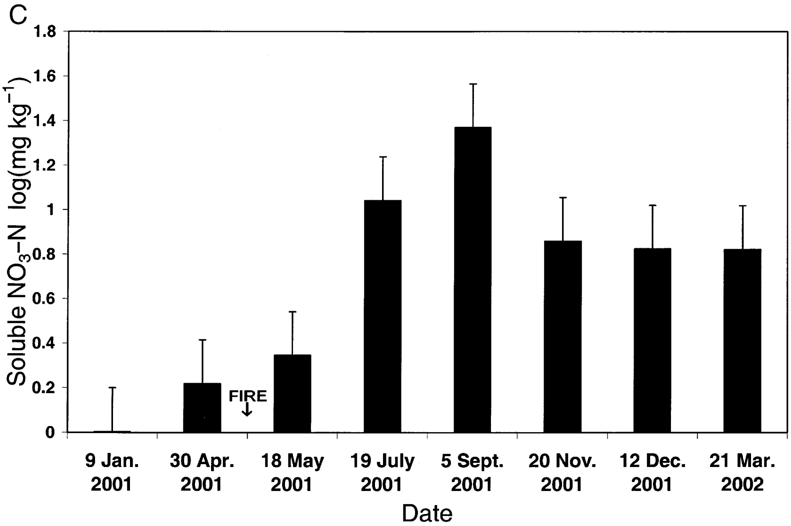


Fig. 3. Average (A) soluble NH_4-N, (B) exchangeable NH_4-N, (C) soluble NO_3-N, and (D) soluble PO_4-P concentrations for the eroded sediment at each sampling date. Error bars reflect the least significant difference between the means for each sampling date where $\alpha < 0.05$. Soluble NO_3-N and PO_4-P are log-transformed values.



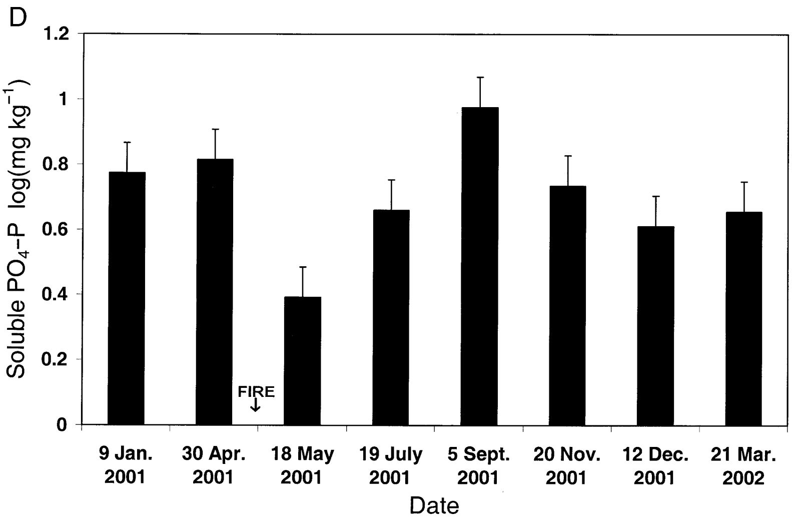


Fig. 3. (continued)

hydrophobicity (DeBano et al., 1970), and soluble sugars from the lysing of plant and microbial tissues (Hungerford et al., 1991; Diaz-Ravina et al., 1996; DeLuca and Zouhar, 2000).

By the next sampling period (July 19, 2001), the C concentrations had decreased significantly because the carbon-rich charred material and ash had already been eroded away most likely resulting from decreases in sediment production. Charred material and ash were readily eroded because the protective cover of vegetation and litter was largely removed. The increase in C concentration between the sampling periods December 12, 2001 and March 21, 2002 was most likely caused by the commencement of warmer temperatures and a rejuvenation of plant growth and organic matter turnover (St. John and Rundel, 1976).

The C/N ratio (Table 2) after the fire increased significantly (from 24 to 29) because of the increase in %C immediately after the fire. Two months after the fire, this ratio had decreased significantly to 19 and was significantly below prefire levels as well. Between the sampling periods May 18, 2001 to July 19, 2001, there was a small amount of rainfall (~7.5 mm). Although not much, it promoted the resprouting of some shrubs and germination of annuals that were observed soon after the fire. Fire has been found to promote the germination of different species of herbs and shrubs in chaparral (Quick, 1935; Sampson, 1944; Hadley, 1961). Such sprouts have been found to absorb a high amount of nutrients from the ash that is then incorporated into their biomass (Radtke, 1984; Rundel and Parsons, 1984). During the summer, this new vegetation died off most likely because of low soil moisture contents caused by low rainfall and increased soil insolation after the fire (Sampson, 1944; McPherson and Muller, 1969; Christensen and Muller, 1975). This pulse of fresh organic matter (sprouts and roots) and its corresponding microbial decomposition were the probable cause of the decreases in C/N ratio identified during this period.

Nitrogen

Soluble NH_4-N (Fig. 3A) increased significantly after the fire from 11.9 to 29.0 mg kg^{-1}. Ash is generally found to be rich in soluble nutrients, including ammonium (Marion et al., 1991). By July 19, 2001, concentrations of soluble NH_4-N had decreased to prefire levels.

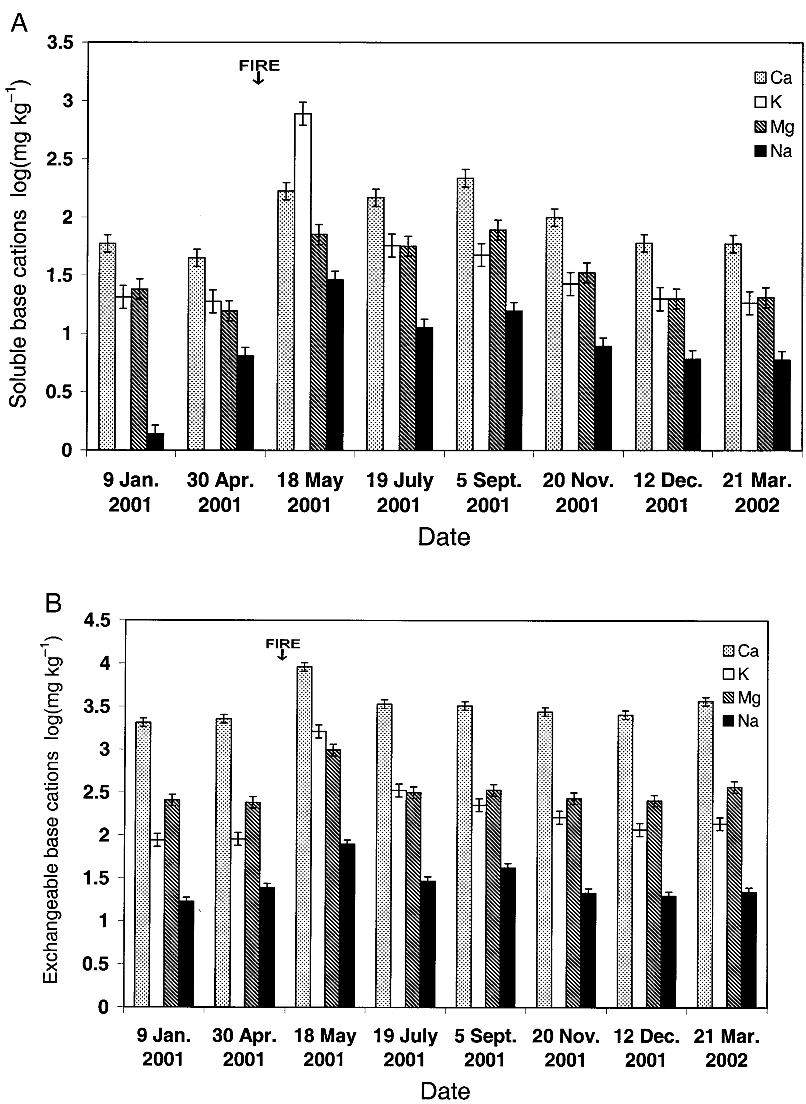


Fig. 4. Log-transformed average (A) soluble and (B) exchangeable base cation concentrations for the eroded sediment at each sampling date. Error bars reflect the least significant difference between the means for each sampling date where $\alpha < 0.05$.

This was caused by the combination of ash loss by dry ravel erosion (St. John and Rundel, 1976), uptake by newly sprouted vegetation (Radtke, 1984), and the nitrification of NH_4^+ to NO_3^- as heat-sensitive nitrifying bacteria recolonized the area (Fritze et al., 1994; Neary et al., 1999).

Exchangeable NH_4-N concentrations (Fig. 3B) did not change significantly immediately after the fire. Some authors have found an increase in extractable NH_4-N (Christensen and Muller, 1975; Riggan et al., 1994; DeLuca and Zouhar, 2000) in ash or the upper mineral soil probably because of soluble NH_4-N increases in ash mentioned previously. This would be the case in our data if the soluble NH_4-N was not subtracted from the extractable. Other authors have found a decrease in extractable NH_4-N (Wienhold and Klemmedson, 1992; Choromanska and DeLuca, 2001) that can be attributed to translocation and condensation in lower regions of the soil. The lack of change found in exchangeable NH_4-N right after the fire followed by significant increases (from 38.2 to 56.4 mg kg^{-1}) in exchangeable NH_4-N at 2 and 4 months afterward gives evidence that may be consistent with the translocation and condensation mechanism for NH_4-N. Erosion of surficial soil material may have accessed this zone of exchangeable NH_4-N concentration at a later time, depending upon how deep within the profile it was translocated. Another probable cause for this increase is the onset of microbial activity and the mineralization of N from the flush of fresh organic matter after the fire. Excess NH_4^+ would have been immobilized by microbes, converted to NO_3^-, or bound to exchange sites, which is why there is no corresponding increase in soluble NH_4-N. The ammonium bound to exchange sites would be less available to microbes than the soluble form, which is why the soluble form decreases even as the exchangeable form increases between May 18, 2001 and July 19, 2001. After reaching a maximum on the sampling dates July 19, 2001 and September 5, 2001, the concentration decreased quickly to prefire conditions by November 20, 2001 because there was no more readily available organic matter for decomposition and microbes began scavenging the exchangeable forms of N. This decrease with time after the fire has been observed by other authors (Carreira and Niell, 1995; Choromanska and DeLuca, 2001).

Soluble NO_3-N concentrations (Fig. 3C) showed little change immediately after the fire (compared with prefire conditions), as previously reported for soils in California chaparral because of low overall organic matter contents in the upper horizons (Christensen, 1973; Dunn et al., 1979). Instead, the NO_3-N concentrations increased significantly from prefire levels at 2 and 4 months after the fire to 11.0 and 23.4 mg kg^{-1}, respectively. Microbial activity is important in the production of nitrate and generally proceeds when both N and moisture contents within the soil are sufficient to support it. Although fires, especially wildfires, are known to sterilize the soil or delay microbial growth and activity (Vasquez et al., 1993; Prieto-Fernandez et al., 1998), less intense fires (e.g., prescribed fires) cause a steady increase in heterotrophic microbial ammonification afterward (Dunn et al., 1979; Overby and Perry, 1996). Within approximately 4 months after the fire, heterotrophic microbial activity is generally found to be the same as in unburned soil, and nitrification is intensified (Arianoutsou-Faraggitaki and Margaris, 1982; Riggan et al., 1985; Frazer et al., 1990). The significant decrease in soluble NO_3-N on the sampling date November 20, 2001 is attributed to reduced microbial activity similar to that causing lower exchangeable ammonium concentrations.

Phosphorus

Soluble PO_4-P concentrations (Fig. 3D) decreased significantly after the fire from prefire levels of 6.54 to 2.47 mg kg^{-1}. The decrease was likely caused by the formation of Ca phosphate compounds (DeBano, 1974; Wright and Bailey, 1982; DeBano and Klopatek, 1988) and the low concentrations of soluble P that occur in ash (Sampson, 1944; Christensen and Muller, 1975; Carreira and Niell, 1995). We speculate that the increase in soluble P concentrations on the sampling dates of July 19, 2001 and September 5, 2001 was caused by the additions of fresh organic matter and subsequent mineralization by microbes. Once this material had been lost from the system via erosion and microbial activity had decreased, concentrations dropped to prefire concentrations. Notably, this is the same type of trend found for soluble nitrate and exchangeable ammonium.

Base Cations

The concentrations of soluble (Fig. 4A) and exchangeable (Fig. 4B) base cations exhibited the same general trends with respect to prefire and postfire conditions. Our finding of increased soluble base cations directly after the fire (sampling date of May 18, 2001) was a normal effect after the fire for this type of vegetation and climate (Carreira and Niell, 1995) and other forest types (White et al., 1973; Grove et al., 1986; Khanna and Raison, 1986). The change in order of magnitude from Ca > K = Mg > Na to K > Ca > Mg > Na was caused by the high soluble K content of the ash (Sampson, 1944; Durgin and Vogelsang, 1984; Ulery et al., 1993). Chaparral vegetation contains higher levels of K than other base cations (DeBano, 1974), and burning converts the K to a soluble form in the ash. Furthermore, common K salts are more soluble than salts formed by the other cations (Lindsay, 1979). In the next sampling period (July 19, 2001), the concentrations for all soluble base cations decreased, especially K and Na because the ash was eroded from the system by dry ravel and wind. During this period, Ca became the dominant soluble cation again. Between the sampling dates of September 9, 2001 and December 12, 2001, all soluble base cations returned to prefire concentrations as a result of the rainfall that marked the onset of the wet season in October and the decrease in the levels of microbial activity mentioned previously.

As with soluble base cations, the exchangeable base cations also exhibited a dramatic increase in concentrations after the fire, as has been observed in other studies of soil (Carreira and Niell, 1995) or litter (St. John and Rundel, 1976; DeBano and Conrad, 1978) after fire. There was one main difference, however, between the soluble and exchangeable values. Calcium was the dominant exchangeable base cation both before and after the fire. The order of magnitude of Ca > Mg > K > Na that was typical for our study was also discussed in other studies of prefire chaparral soils (Ulery et al., 1995; Quideau et al., 1996). The large increase in K after the fire was attributed to the higher K content in the chaparral vegetation mentioned previously (DeBano, 1974). Exchangeable base cations had resumed prefire concentrations on the sampling date of December 12, 2001 because the sediments with elevated levels were flushed out of the watershed by that time.

During a fire, most of the base cations in chaparral litter are liberated as part of the ash which can later be lost by erosion. Under these circumstances, when regrowth occurs, base cations are obtained through mineral weathering to replace those that were lost (Quideau et al., 1999). Although the fine feeder roots of the ceanothus and chamise shrubs are most likely destroyed by fire (Doehring, 1968; DeBano et al., 1977; DeBano and Conrad, 1978), the tap roots that extend deep into the bedrock (Hellmers et al., 1955; Sternberg et al., 1996) generally survive. Base cations from weathered bedrock minerals are taken up by these roots along with plant-available water. Those roots remaining within the bedrock may deliver most of the nutrients because much of the shallow soil mantle, particularly the nutrient-rich upper part, is lost downslope after the fire. Our data indicate that this nutrient loss, in the form of eroded ash and charred debris, occurs rapidly after the fire. In the case of this study, it was flushed from the system before the winter or rainy season. From this, we speculate that continuous and frequent cycles of prescribed burns may increase the pace of mineral weathering in the bedrock.

Implications for Chaparral Ecosystems

The shallow, rocky soils of southern California chaparral watersheds are typically low in nutrients (specifically N and P) and, because of their steepness and the continuous erosional losses, are particularly poor at storing nutrients. Nutrients are lost through volatilization during fire events, and erosional losses are exacerbated in the postfire environment.

Our data suggest that revegetation after the fire is an important way of preventing the erosion of nutrients, specifically N and P, and corroborates the idea proposed by Rundel and Parsons (1984). Revegetation is a naturally occurring process that commences soon after the fire but cannot be sustained throughout the year in southern California because of continuous dry, hot conditions in the late spring through early fall. Although few nutrient analyses have been done on eroded sediment after chaparral fires, the concentrations of soluble NO_3-N and PO_4-P and exchangeable NH_4-N that were measured were similar to those of soil samples taken after prescribed fire in chaparral (Carreira and Niell, 1995; Overby and Perry, 1996). The concentrations were also significantly less than those in soil samples taken after a wildfire in another California chaparral area (Christensen and Muller, 1975). This, in addition to the reestablishment of prefire N and P concentrations within 1 year, indicates that prescribed fire is less disruptive to watershed biogeochemistry compared with wildfires.

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