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Effects of Fire Severity on Nitrate Mobilization in Watersheds Subject to Chronic Atmospheric Deposition

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Severe fires in chaparral watersheds subject to air pollution from metropolitan Los Angeles mobilized accumulated nitrogen and caused streamwater to be polluted with nitrate at concentrations exceeding the Federal Water Quality Standard. Streamwater NO_3^- concentrations were elevated during peak flows, the largest of which was a debris flow that transported NO_3^- at concentrations as high as 1.12 mequiv/L. Annual NO_3^- loss from severely burned watersheds, averaging 1.2 kequiv/ha, was 40 times greater than that from areas that remained unburned. Fires of moderate intensity produced a more subdued response in stream discharge and soil nitrification and less than one-seventh the NO_3^- loss observed after severe burning. We infer that the combination of atmospheric deposition with severe wildfires provides a strong and recurrent source of nitrate that could contribute to existing groundwater pollution in parts of eastern Los Angeles County. Moderating the fire regime by prescribed burning could provide substantial mitigation.

Introduction

Atmospheric deposition of nitrogen oxides and ammonium is the most extreme yet recorded in the United States where the chronic urban air pollution of the South Coast Air Basin meets the San Gabriel Mountains in southern California (*1*). There the deposition and flux of nitrogen through the vegetation canopy, which amounts to 1.7 kequiv $ha^{-1}\,yr^{-1}$, induces streamwater NO_3^- concentrations that are as much as 3 orders of magnitude greater than where air pollution is minimal (*1*). Yet less than 10% of the deposited nitrogen appears in streamflow (*1*). In the absence of fire, this accumulation substantially enriches the ecosystem. However, extensive and destructive wildfires of 10^3–10^5 ha recurrently sweep the native chaparral in the San Gabriels (*2, 3*). Subsequent soil erosion (*4, 5*), which is massive, combined with potentially rapid nitrification in soils and sediments (*1, 6, 7*) may largely mobilize the accumulated nitrogen. Debris-laden flows produced by even modest storms could be heavily polluted with NO_3^- and contribute to the existing NO_3^- pollution in the aquifer of the main San Gabriel Basin, an important local source of water for Los Angeles County (*8*). If rates of soil organic matter oxidation and erosion are related to the intensity or duration of heating, then intervention in the wildfire regime by prescribed burning when weather and fuel moisture are moderate could reduce water quality impacts by limiting the severity of subsequent wildfires.

Catastrophic fire is a common feature that punctuates the environment of many Mediterranean, seasonally dry tropical, and boreal ecosystems. Although the case we describe for southern California may be extreme, the occurrence here of frequent, severe fires in an ecosystem enriched by nitrogen deposition is instructive of the wider global situation where fire sometimes dominates the long-range balance of biogeochemical cycling.

In this cooperative project involving the USDA Forest Service, the California Department of Forestry and Fire Protection, and the Los Angeles County Fire Department, we ignited especially severe fires in two 16-ha watersheds after felling the *Ceanothus* chaparral. We then compared the subsequent NO_3^- and NH_4^+ fluxes in streamwater to those from two watersheds burned by more moderate fires in undisturbed vegetation and to those from two unburned watersheds. We also examined the effect of fire severity on temporal trends in soil inorganic nitrogen.

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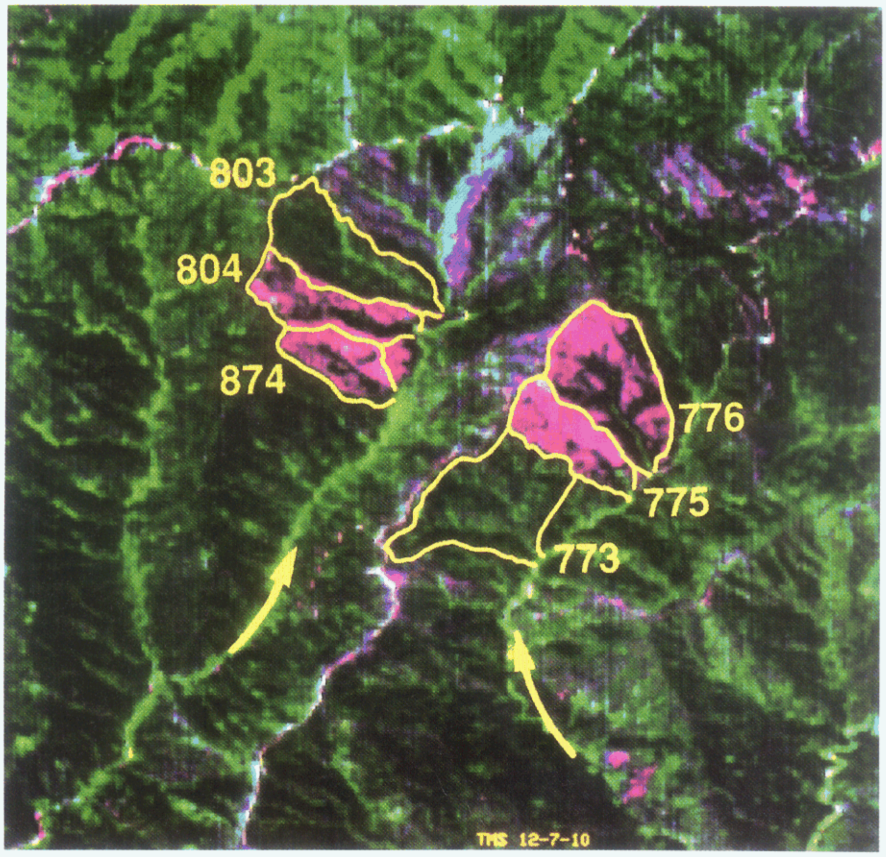

Figure 1. Experimental watersheds at the San Dimas Experimental Forest in the Angeles National Forest, California. Severe fires were generated after felling *Ceanothus* chaparral in watersheds no. 874 in Bell Canyon and no. 775 in the West Fork of San Dimas Canyon. Adjacent watersheds (nos. 804 and 776) were burned by more moderate fires in standing vegetation. Watersheds 803 and 773 remained unburned. In this image produced from thematic mapper simulator data, differences between watershed treatments are apparent in the infrared reflectance of postfire soils. The 0.76–0.9-μm wavelength band is depicted in red, 0.63–0.69-μm wavelength is in green, and 2.08–2.35-μm wavelength is in blue. Arrows depict the direction of on-shore air flow from the South Coast Air Basin.

Experimental Methods

Site and Treatment. Treatments were assigned in blocks to watersheds in Bell and West Fork of San Dimas Canyons at the San Dimas Experimental Forest (34° 12′ N, 117° 40′ W), 10 km northeast of Glendora, CA (Figure 1). Soil there is a loamy, mixed, thermic, shallow, Typic Xerorthents (*9*), less than 1 m deep, that has developed from a complex of igneous and metamorphic rocks.

Prescribed fires were ignited by helitorch on October 29 and 30, 1984. They burned 24-year-old *Ceanothus crassifolius–Adenostoma fasciculatum* chaparral with runs of 100–500 m length up the slope. Weather and fuel conditions were as follows: fuel-stick moisture—a measure of the water content of deadwood, 10%; *A. fasciculatum* live-fuel moisture, 70%; relative humidity, 30–60%; air temperature, 21–28 °C; winds, 5–10 km/h from the south with gusts to 15 km/h. Fire coverage was 80–90% in all watersheds. Aboveground biomass, averaging 35 Mg/ha (SE = 5.2, n = 12) at age 21 years, had previously been measured in 32-m^2 plots in watershed 804. Dead biomass constituted 12% of live *C. crassifolius* shrubs and 32% of the live *A. fasciculatum* (*10*).

Radiant plus sensible energy release during the burning of 24-year-old stands of the dominant *Ceanothus* was estimated as the energy content of consumed biomass minus the heat of vaporization of water in the fuels. In standing chaparral of this composition, we have observed fire to consume foliage, deadwood, and 9% of the live wood (unpublished Forest Service records); biomass distributions and moisture contents of these components at age 21 are given by Riggan et al. (*10*). Basal area growth of 0.06/yr (*10*) and a heat of combustion of 2.1×10^7 J/kg (*11*) were assumed in the estimate of energy release.

Temperatures were monitored on a ridge in the northeast quadrant of watershed 874 with 10 chromel/alumel thermocouples, five at the surface and five at 2 cm depth. Data were recorded at intervals of 5 s by a Campbell CR-21x digital recorder.

Analytical Methods. Clear streamwater was sampled four times daily by ISCO wastewater samplers; debris-laden flows were manually sampled, and the samples were filtered. Concentrations of NO_3^-, NH_4^+, and SO_4^{2-} were determined with a Technicon AutoAnalyzer II (Industrial Methods No. 100-70W/B, January 1978; No. 334-74W/B, revised March 1977; and No. 329-74W/B, revised March 1977). Means and confidence intervals were computed by principal components analysis for the ratios between treatments of daily-mean volume-weighted NO_3^- concentrations. Annual yield of NO_3^- was estimated by numerically integrating the product of $[NO_3^-]$ and streamwater discharge (corrected for sediment content). Streamwater and sediment discharge are reported elsewhere (*12*). A linear interpolation of concentration on discharge was used to estimate concentrations at the more frequent times of sampling for stream discharge, with the limitation that the estimates not exceed measured concentrations.

Figure 2. Fire spread in standing chaparral of Bell Canyon. Rates of spread were estimated at 1.5 m/s with flame lengths reaching 30 m. Total elapsed times from the left are 18 (**center**) and 61 s (**right**).

Samples of soils from 0 to 5 cm depth were collected along approximately 100 m transects in Bell Canyon and composited with 12 cores per transect from each of three elevations in each experimental watershed, with four transects per elevation in the severe burn and eight composites per elevation elsewhere. Soil NH_4^+ and NO_3^- were extracted with 2N KCl and analyzed on a Technicon AutoAnalyzer II. All comparisons of soil nitrogen concentrations as discussed in this text are based on significant hypothesis tests made with a general linear model and linear contrasts with $P \leq 0.05$. Comparisons across watersheds are assumed to reflect the treatments.

Results and Discussion

Experimental Fires. The experimental fires were explosive (Figure 2); flames 15–30 m in length burned in all four canyons, and radiant heat could be felt by observers at a distance of over 0.5 km. Radiant and sensible heat flux from the fires in standing vegetation was estimated to be 5.1×10^{11} J/ha. That from the fires in felled vegetation was approximately three times greater, 1.6×10^{12} J/ha. The felled vegetation burned for a noticeably longer duration and was completely incinerated with less latent heat loss. Soil surface temperatures under the felled chaparral peaked at 780 °C 3 min after combustion began; a peak of 265 °C was reached at 2 cm depth in the soil after 7 min.

Sediment Flux and Stream Flow. Even small storms during the succeeding winter produced debris-laden flows in the burned canyons, entraining sediment at up to 59% by weight (*12*). Sedimentation following the moderate-intensity fires was three-eighths as great as that from the severely burned watersheds (*12*). Sedimentation from the unburned watersheds was negligible. The burned watersheds were coursed by powerful debris flows during a thunderstorm on December 19, 1984. Those flows in the severely burned canyons peaked at more than 330 L s^{-1} ha^{-1}; concurrent flow in the unburned watersheds peaked at less than 1.1 L s^{-1} ha^{-1} (*12*). Annual water yield from the severely burned watersheds was on average four times greater than after the moderate-intensity fires and 14 times greater than from the unburned watersheds (Table 1) (*12*).

Streamwater NH_4^+ and NO_3^-. Flux of dissolved NH_4^+ in streamwater from burned canyons was dominated by ash-laden flows initiated by an early storm on November 13. Peak NH_4^+ concentrations at San Dimas Canyon were observed during that event: 0.25 mequiv/L from the moderate-intensity fire and 0.75 mequiv/L from the severe fire. Subsequent NH_4^+ concentrations were generally 1–2 orders of magnitude lower. In all cases NH_4^+ comprised less than one-eighth the annual flux of dissolved inorganic nitrogen (Table 1).

Streamwater NO_3^- concentrations responded to the passage of even small storms, rising rapidly with increasing discharge and declining slowly during recessional flows (Figure 3). $[NO_3^-]$ in the large debris flows from the severely burned watersheds reached 1.12 mequiv/L, exceeding the Federal Water Quality Standard (0.73 mequiv/L). The effect on NO_3^- yield of the intense rain on December 19 was qualitatively different from other storm periods in that it caused large quantities of soil to be suspended and removed rather than leached.

Water movement through previously leached soils and denitrification or biological uptake in the streams probably account for the reduced NO_3^- concentrations observed in peak flows later in winter and at low flow. Active uptake has been documented for streams in unburned watersheds at San Dimas (*1*). Furthermore, during stream recession in spring, water flowing at low rates through a small pond in the debris basin of watershed 804 (burned by moderate-intensity fire) was largely renovated as the $[NO_3^-]$ fell from 0.2 mequiv/L at the inflow to 0.0014 or less at the dam outlet. Strong reducing conditions in the basin were also indicated by sulfate concentrations that concurrently declined from 1.0 to 0.69 mequiv/L as H_2S evolved.

Both streamwater NO_3^- concentration and flux reflected the severity of burning. The daily-mean volume-weighted NO_3^- concentrations after severe burning were 1.7 times those after the moderate-intensity fires (with 95% confidence interval for that ratio of 1.3, 2.3). Moderate fires produced concentrations 3.0 times those of the unburned controls (confidence interval of 2.0, 5.2). The response of NO_3^- flux to fire severity was even more striking: annual NO_3^- loss from the severely burned watersheds, averaging 1.2 kequiv/ha, was approximately 7 times greater than after moderate-intensity burning and 40 times greater than from the unburned watersheds (Table 1).

Soil Nitrogen Response. Concentrations of mineral nitrogen in soils were also altered according to the severity of the fire treatments. The $[NH_4^+]$ in surface soils rose during the severe fire but not as markedly as during the

Table 1. Fluxes of Water, Nitrate, and Ammonium and Volume-Weighted [NO_3^-] in Streamwater during the Hydrologic Year Beginning October 1, 1984

watershed no.	treatment	area (ha)	water yield (cm)	NO_3^- flux[a] (equiv/ha)		NH_4^+ flux[a] (equiv/ha)		[NO_3^-] (mequiv/L)
				annual	high flow[b]	annual	high flow	
			West Fork of San Dimas Canyon					
773	unburned	40.5	2.1	38[d]	0	1.0	0	0.18[d]
776	moderate fire	34.9	4.4[c]	108	39	7.3	4.9	0.25
775	severe fire	14.9	18[c]	870	510	124	113	0.48
			Bell Canyon					
803	unburned	25.1	1.6	23	0	1.4	0	0.14
804	moderate fire	15.9	7.6	230	190	3.9	2.1	0.30
874	severe fire	16.2	32	1590	770	143	105	0.50

[a] Estimates reported here for annual flux in each watershed reflect a 1 L/s minimum detection limit set by performance of the trapezoidal flumes calibrated for low flow. For comparison, computed NO_3^- flux at flow rates to a 0.02 L/s minimum was 26 equiv/ha from watershed no. 803 and 240 equiv/ha from no. 804. NH_4^+ flux at all flow rates was 2.7 equiv/ha from no. 803 and 5.1 equiv/ha from no. 804. [b] Discharge during debris-laden flows, estimated as flows at rates >40 L/s; no flows of this magnitude occurred on the unburned watersheds. [c] Approximate standard errors for these annual water yield estimates, computed by a Monte Carlo simulation based on errors in the high- and low-flow calibration functions, duration of the December 19 debris flow, and peak-flow sediment concentration, were 5.4% of the point estimate for watershed no. 775 and 13% of the estimate for no. 776. [d] Annual NO_3^- flux and concentration estimates shown here for watershed no. 773 were weighted by a ratio (2.6:1) of mean prefire concentrations between watersheds 773 and 776 as measured during peak and recessional flows of 1983. This provides an estimate of performance by watershed no. 776 during 1984 had it not burned. Concentrations in no. 773 were consistently higher than those from watersheds further east in West Fork San Dimas Canyon, possibly as a result of greater exposure to onshore winds and elevated deposition of air pollutants rising from the Los Angeles Basin. The watersheds in Bell Canyon did not substantially differ prior to treatment.

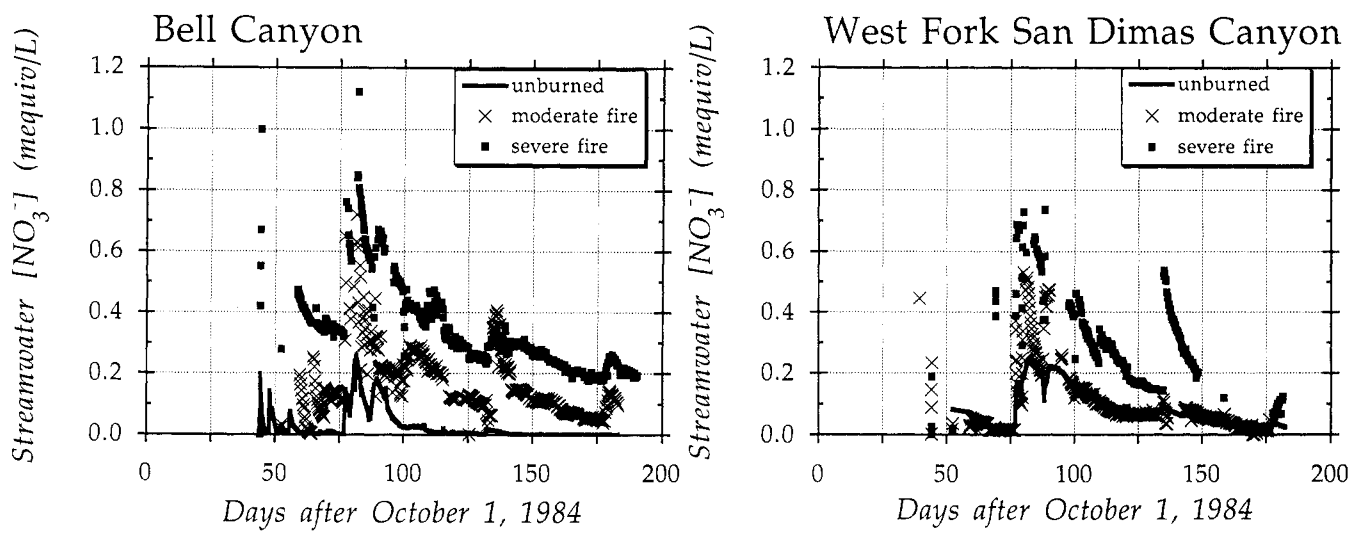

Figure 3. Streamwater NO_3^- concentrations (mequiv/L) in experimental watersheds in Bell and West Fork of San Dimas Canyons. Peak concentrations were associated with flows during storms. Irregular data during early storms reflect intermittent streamflow. Data for the unburned watershed in the West Fork of San Dimas Canyon (no. 773) have been weighted using prefire data to provide a control for the moderate-intensity fire (see Table 1).

more moderate heating of the fires in standing vegetation (Table 2). Because of the 3-fold greater energy release of our severe fires, soils in that treatment were undoubtedly subjected to the greatest heating below the immediate soil surface and the greatest rates of volatilization, which would explain their lower [NH_4^+]. This effect is consistent with laboratory observations that show a maximum [NH_4^+] in soil heated to 300 °C (*13*); at lower temperatures, pyrolytic mineralization of organic nitrogen is less complete and at greater temperatures, soil NH_4^+ is more completely volatilized. We also observed that soil NO_3^- concentrations declined by an average of 40% during burning. Since our sample extended to 5 cm depth, which is below the zone of substantial heating, it is likely that most NO_3^- was lost in the upper 1–2 cm.

During the ensuing winter and spring (December 12–June 4), NH_4^+ concentrations were consistently highest in soils that had been subject to severe heating and lowest in soils of the unburned watershed. Differences in accumulated soil NO_3^- with fire severity were also apparent after the succession of winter storms in November and December; the average soil NO_3^- concentration at that time (January 31–June 4) was greatest in soils subjected to severe fire (1.20 mequiv/kg of soil), intermediate in those from the moderate-intensity fire (0.79), and least in the unburned soils (0.49) (Table 2). The burned watersheds remained barren at least through January, so a relative acceleration of nitrogen mineralization and nitrification in the severely-burned soils, and not differences in plant uptake, is a likely cause of early differences in mineral nitrogen between the fire treatments. Nitrification is also strong in the unburned watersheds as has been previously demonstrated by rapid NO_3^- production after application of ammonium to both soils and waters (*1, 6*).

Watershed NO_3^- loss must have been partly mediated by the soil nitrification response to fire severity and not just the response to water yield and sedimentation alone. The severe fire treatment produced on average a 4-fold

Table 2. Soil NO_3^- and NH_4^+ Concentrations (mequiv/kg of Soil), Average Soil Water Content (g/g of Soil) at Surface (0–5 cm Depth), and Cumulative Precipitation (cm) during the Hydrologic Year Beginning October 1, 1984

watershed no.	treatment	Oct 25[a]	Nov 7[b]	Dec 12	Jan 3	Jan 31	Mar 5	Apr 24	Jun 4
				$[NO_3^-]$[c]					
803	unburned	2.16	2.39	0.67	0.50	0.66	0.44	0.29	0.56
804	moderate fire	2.07	1.19	0.91	0.84	0.96	0.91	0.51	0.77
874	severe fire	2.43	1.38	0.81	0.83	1.31	1.19	0.67	1.63
				$[NH_4^+]$[c]					
803	unburned	2.48	1.74	0.74	0.87	0.79	0.49	0.46	0.37
804	moderate fire	2.06	4.31	2.94	1.96	1.31	0.79	0.71	0.61
874	severe fire	2.03	3.14	4.90	3.46	3.09	1.56	2.15	2.11
				Soil Water					
all watersheds		0.024	0.015	0.186	0.112	0.130	0.070	0.057	0.019
				Cumulative Precipitation					
		0.6	0.6	16.8[d]	39.8	43.4	48.9	55.0	55.6

[a] Prior to burning. [b] After burning. [c] Standard errors of least-square means for $[NH_4^+]$ are 0.15 for watersheds 803 and 804 and 0.22 for no. 874. For $[NO_3^-]$ they are 0.11 for nos. 803 and 804 and 0.15 for no. 874. [d] The first postfire precipitation occurred November 8, 1984.

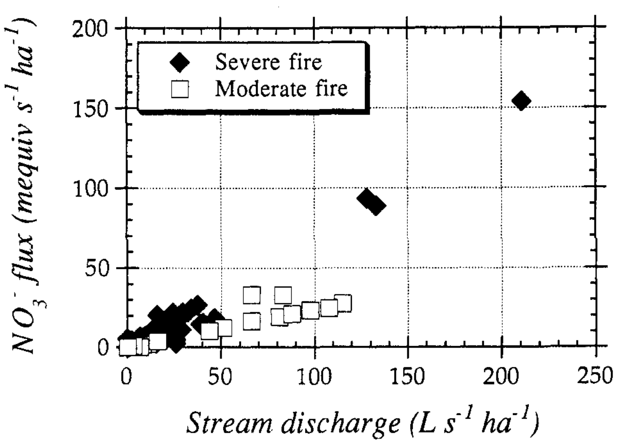

Figure 4. NO_3^- yield as a function of stream discharge rate in the experimental watersheds of the West Fork of San Dimas Canyon. At comparable streamflow rates, NO_3^- yield was consistently higher from the severely burned watershed than from the watershed burned by fire of more moderate intensity. A portion of the variance for a given flow rate is due to hysteresis in the relationship between the rising and falling limbs of the stream hydrograph.

increase in water yield over the moderate-intensity fires yet caused a 7-fold increase in NO_3^- yield due to concurrent increases in stream $[NO_3^-]$. These concentration increases did not result from the higher water yields even though high concentrations are generally correlated with high flows; at comparable flow rates NO_3^- yield was consistently higher from the severely burned watersheds than from the more moderate treatment (Figure 4). Furthermore, the correlation of concentration with stream discharge probably results from greater biological uptake at low flow, and the severe fires yielded several-fold more NO_3^- even at low flows (Table 1).

Role of Atmospheric Deposition. It remains to ask whether the high rates of NO_3^- flux in streamwater were a result of high rates of atmospheric deposition. A direct test is not available to us since our experimental design was not replicated in regions of low deposition. But we can make some inference regarding the high NO_3^- flux by examining three alternative explanations: that it results from the postfire environment alone; that symbiotic N_2 fixation associated with *Ceanothus* provides a primary source of nitrogen for nitrification and NO_3^- loss; and that the intense storm in late December entrained a quantity of NO_3^- that was unusually large for postfire situations.

High NO_3^- concentrations and rates of flux are apparently not a universal characteristic of burned chaparral watersheds. Peak concentrations after burning in Arizona chaparral, 0.14 mequiv/L, were only one-tenth as great as measured here (*14*). The Arizona watersheds studied were remote from any urban air basin and unlikely to have had chronically high rates of dry deposition as occur at San Dimas.

NO_3^- flux at San Dimas was also unusually high in comparison with that after fires in some coniferous forests, where nitrification is typically suppressed (*15, 16*) and recorded NO_3^- concentrations in streamwater have remained below 0.04 mequiv/L (*17, 18*). One exception was a prescribed fire in logging slash from a forest comprised of 80% *Pseudotsuga menziesii* and 20% *Alnus rubra*, which hosts symbiotic N_2 fixation, where peak NO_3^- concentrations remained below 0.15 mequiv/L but NO_3^- flux reached 1120 equiv/ha (*19*).

Probably any source of nitrogen that is cycled through the ecosystem may equally contribute to nitrogen loss after fire. Unfortunately, there are no reliable estimates of N_2 fixation in these chaparral communities that may be compared to the known rates of deposition. *Ceanothus crassifolius* can produce large nodules, although these have not been observed to be abundant at San Dimas. Yet N_2 fixation is at least seasonally active in those that do occur, as has been shown by acetylene reduction assay with excised nodules (*20*; S. Williams, *personal communication*).

Although fixation may enrich the ecosystem, it is apparently not sufficient to saturate the ecosystem and drive a large NO_3^- flux in streamwater in the absence of fire. This was evident in our earlier survey of unburned watersheds in southern California (*1*), where streamwater $[NO_3^-]$ at high flow was exceptionally low in regions of minimal air pollution, whether or not extensive stands of *Ceanothus* were present (as in portions of the western Santa Monica Mountains). The *uniformly* high $[NO_3^-]$ in waters from unburned watersheds of the front range of the San Gabriel Mountains, as much as 3 orders of magnitude greater than in areas subject to low levels of air pollution, reflects in these ecosystems both a dominance of deposition in the nitrogen cycle and a developing nitrogen saturation, or inability to sequester additional nitrogen. This is not surprising since the throughfall nitrogen flux at San Dimas, which is dominated by dry deposition to the plant canopy, is 2.3 times the rate of nitrogen accretion in the biomass of mature *Ceanothus* stands there (*1*). The deposition alone represents a very high rate of ecosystem loading, regardless of what the nitrogen fixation rates might be.

One possible explanation for an inability to sequester nitrogen is a depletion or limitation of the available carbon in soils, which would limit the immobilization of mineral nitrogen by heterotrophic microorganisms. Such a response to chronic nitrogen loading has been observed in the Pacific Northwest where continuous irrigation of *Pseudotsuga menziesii* plantations by nitrogen-rich wastewater depleted soil carbon and induced high nitrification rates and NO_3^- leaching from soils in which nitrification had been largely absent (*21*, *22*).

We also reject the possibility that debris-laden flows caused by unusually intense precipitation in the year of our experiment caused abnormally high rates of NO_3^- flux. Although debris flows dominated sediment movement and mobilized NO_3^- and sediment from across the burned watersheds, NO_3^- flux was dominated by interstorm flow, and thus NO_3^- and sediment losses were not strongly coupled. The large debris flows that were generated in the burned watersheds of the West Fork of San Dimas Canyon carried 50–90% of the annual sediment production (*12*) but only 15–21% of the annual NO_3^- flux. Furthermore, these flows were produced by a period of intense precipitation (5 mm in 5 min) that is encountered annually on average at San Dimas (*12*).

We do expect that more sustained winter precipitation would have accelerated NO_3^- loss by expanding the volume of soils contributing water to streamflow and by lowering residence time of waters and the chance for biological uptake of NO_3^-. NO_3^- yield was apparently limited by the amount of water percolating through the soil since stream NO_3^- concentrations were consistently high during peak streamflow and soil NO_3^- was by no means exhausted by leaching during winter rains. Precipitation rate during our observations was not unusual: annual rainfall was in the 50th percentile of a 60-year record at San Dimas.

We infer from these arguments that the very large postfire NO_3^- losses in streamwater we observed at San Dimas were most likely a result of the chronic nitrogen enrichment of the ecosystem by atmospheric deposition.

Environmental Implications. By interposing young-age classes in the expanse of older, even-aged chaparral, prescribed fire can interrupt the succession of extensive, high-intensity wildfires that recurrently sweep the San Gabriel Mountains (*23*). Results reported here show that moderation of this fire regime, by reducing the magnitude of postfire debris flows and the concentrations of NH_4^+ and NO_3^- within soils and waters, would partly mitigate water pollution derived from chronic atmospheric deposition of nitrogen oxides and ammonium. Postfire waters, especially those at high flow, should also be managed downstream to avoid additions of nitrate to the aquifer of the eastern Main San Gabriel Basin, where nitrate concentrations now commonly exceed the Federal Standard for drinking water (*8*).

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