Prescribed Fire Alters the Impact of Wildfire on Soil Biochemical Properties in a Ponderosa Pine Forest

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ABSTRACT

Although studies have addressed the influence of fire on soil biochemical processes, there have been no reports on how prescribed fire followed by wildfire influences microbial activity and nutrient cycling. Over a 21-mo period we monitored changes in soil nitrogen (N) and carbon (C) of a ponderosa pine (*Pinus ponderosa* P.&C. Lawson) and Douglas-fir [*Pseudotsuga menziesii* var. *glauca* (Beissn.) Franco] forest (both O horizon and 0–10 cm of mineral soil) that had been exposed either to prescribed fire (PB), wildfire (WF), prescribed fire three months prior to wildfire (PBWF), or no fire as an unburned control. Total N, potentially mineralizable N (PMN), NH_4^+–N and NO_3^-–N concentrations in surface (0–10 cm) mineral soils were significantly increased immediately after WF. Soils exposed to prescribed fire prior to wildfire also had elevated concentrations of total N, PMN and NH_4^+–N, but were significantly lower than in WF alone. Potentially mineralizable N was significantly reduced on all fire-exposed sites from 9 mo to the end of the study period. Although mineral soil NO_3^-–N concentrations in fire-exposed soils were similar to the unburned control 12 mo after fire, resin sorbed NO_3^-–N was 88 μg capsule^{-1} in WF soils vs. 24 μg capsule^{-1} in PBWF soils, and 1.3 μg capsule^{-1} in the unburned control. Microbial biomass in the WF mineral soils was as low as 52 μg g^{-1} 21 mo after fire while microbial biomass in PBWF soils remained above 100 μg g^{-1} throughout the study. It appears that prescribed fire prior to wildfire may attenuate the effects of wildfire on soil and may have predisposed the microbial community to the effects of heating.

Land managers often propose the use of prescribed fire to mitigate potential impacts of wildfire and as a tool in forest restoration, however, actual mitigating effects of prescribed fire on ecological functions of soils subsequently exposed to wildfire are not well documented. Such research could provide insight into the value of prescribed fire in restoring ecosystems and protecting fire-suppressed forests from catastrophic wildfires.

Temperate forests have long been shaped by dynamics of natural disturbances and human activity, and are tolerant of a wide range of physical perturbations such as windthrow or fire (Likens and Borman, 1995). For example, prior to 1900 ponderosa pine forests of western Montana experienced nonlethal fire at intervals of 13 to 50 yr resulting in what has been described as open, park-like, uneven aged stands (Arno et al., 1995). Today, largely as a result of reduced fire frequency, many ponderosa pine stands are characterized by a dense interior Douglas-fir understory, which has gradually led to the Douglas-fir becoming a dominant species.

With recognition of the role of fire in maintaining forest health and plans for widespread prescribed fire to mimic natural processes (Arno et al., 1995), there is a growing need to better understand the effect of fire on nutrient availability, particularly N. Biochemical processes in forest soils are highly sensitive to environmental change, and therefore, can provide important information about early ecosystem response to management activities (Diaz Ravina et al., 1996; Trasar-Cepeda et al., 1998). Prior research has shown that both low-intensity prescribed fires and high-intensity wildfires increase short-term N availability in mineral soils (Giovannini et al., 1990; Fritze et al., 1992; Prieto-Fernandez et al., 1998); however, studies also indicate that inorganic and labile N concentrations will remain reduced for an indefinite period of time, depending on burning intensity, soil water content, climate, and other ecological drivers that impact postdisturbance recovery (Prieto-Fernandez et al., 1998; Neary et al., 1999).

The ecological importance of frequent, low-intensity fires lies not only in fire-adapted plant communities, but also in fire-adapted soil microbial communities, which bolster long-term resilience to natural and anthropogenic disturbances. As microbial colonies acclimate to the recently disturbed environment, populations capable of utilizing the short-term release of inorganic nutrients and simple organic compounds proliferate (Diaz-Ravina et al., 1996). Also, vegetation surviving low-intensity fires has the potential to intercept the short-term pulse of available inorganic N (Arianoutsou-Faraggitaki and Margaris, 1982).

Prolonged periods of fire exclusion commonly result in development of forests with N-limiting conditions (Kimmins, 1996; Wardle et al., 1997). The majority of soil N is found in an organic form and accumulates either in the organic horizon or mixed with the uppermost mineral horizon. Only a small portion of organic N undergoes microbial decomposition, the majority of which is rapidly immobilized by soil microbes, leaving as little as 0.01% of total N for plant uptake (Keenan et al., 1995).

Little is known about the relative impact of prescribed fire in actually mitigating effects of high intensity wildfire on soils. In the fall of 1996, a high-intensity wildfire in the Bitterroot National Forest burned across a forest stand previously not exposed to wildfire for 80 yr, and partially burned across an adjacent stand that had been treated with prescribed fire the previous spring. The objective of this paper is to assess the effect of prescribed fire followed by high intensity wildfire on mineral soil microbial biomass recovery and C and N transformations in a ponderosa pine forest.

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Abbreviations: ARC, anthrone reactive carbon; PB, prescribed fire; PBWF, prescribed fire three months prior to wildfire; PMN, potentially mineralizable N; WF, wildfire.

MATERIALS AND METHODS

Site Description

This study was conducted on ≈16 ha of a ponderosa pine and Douglas-fir forest located in the Bitterroot National Forest, 15 km east of Corvallis, MT. The stand was relatively open with 250-yr-old ponderosa pine trees dominant in the overstory, thickets of smaller Douglas-fir trees in the understory, and chokecherry (*Prunus ceracifera*), common snowberry [*Symphoricarpos albus* (L.) Blake (Caprifoliaceae)], ninebark [*Physocarpus malvaceous* (Green)], and pinegrass (*Calamagrostis rubescens* Buckl.) dominant in the shrub and grass layers. The only previous management activity recorded for this stand was a sanitation salvage harvest in 1978 when dead and diseased trees were removed. In May 1996, a prescribed fire was conducted on ≈40 ha with the purpose of reducing the accumulation of fuels and removing thickets of small Douglas-fir trees and shrubs. Fuel was estimated as 1.5 Mg ha^{-1} of fine material (0–8 cm diam) and 0.2 Mg ha^{-1} of material >8-cm diam. The burn was completed successfully resulting in 42% fine fuel (Oi horizon) consumption, and no mortality reported for the overstory trees. In August of the same year, a human-caused fire started in a portion of the drainage basin that had not been under the spring burn plan. The fire quickly spread through the stand until it reached the previously underburned portion, where intensities were reduced to a low intensity surface fire and it was slowly extinguished before it covered the entire prescribed fire area. The wildfire-only affected portion experienced complete stand mortality and 100% fine fuel consumption, while the portion that had been prescribed burned followed by a wildfire experienced 50% stand mortality and 70% fuel consumption.

As a result, site selection for this research represented three combinations of wildfire and prescribed burning that were compared with a site with no apparent exposure to fire in the past 80 yr. The sites (≈4 ha plots) were as follows: (i) exposed only to an August 1996 stand replacing WF, (ii) exposed to a prescribed fire in May 1996 and subsequently burned by the 1996 PBWF, (iii) exposed only to May 1996 PB, and (iv) an unburned control.

Soil Characteristics

All sites were in an area with similar topography and climate characteristics; slope of 35 to 40%, northeasterly aspect, ≈1500 m elevation, mean annual temperature of 7°C, and an annual precipitation of 44 cm. Soils were predominantly loamy-skeletal, mixed, frigid Lithic Ustochrepts formed on granite residuum and composed of 27% sand and 10% clay. The O horizon was ≈11 cm deep and was composed of 3 cm of Oi, 5 cm of Oe, and ≈3 cm of Oa. The average thickness of surface mineral horizon (ochric epipedon) was 15 cm, and depth to decomposed granite parent material was 24 cm. There was no evidence of aerial fire retardant use within boundaries of the study site. The no-fire, unburned control was located at 1 km from the burned area to assure no exposure to fire or fire retardant. These unburned control soils were somewhat finer in texture (5% sand and 24% clay). The average thickness of surface mineral horizon of the unburned control soils was 15 cm and depth to parent material was 48 cm.

Soil samples were collected every spring (late May) and fall (mid September) for two consecutive years. Data collection started within the first 2 wk after the wildfire and continued until spring 1998. Eight 20-m transects 20 m apart were laid out along the slope gradient in each fire scenario. Eight composite samples consisting of 10 subsamples each of surface mineral horizon (0–10 cm) were collected at random intervals along the eight transects. Bulk samples were mixed and all visible debris and coarse fragments were removed before the single sample was drawn. The same procedure and transects were used in fall 1996 to obtain Oe and Oa (organic) horizon samples. Additional estimates of organic horizon volume and density were obtained in the spring of 1997 by driving a 20-cm diam PVC ring down to the surface of mineral soil along the transects to remove the organic horizon. In spring 1997, we buried four PST-2 ionic resin capsules (Unibest, Bozeman, MT) 8 cm deep in each of eight transects. The capsules remained in the soil for 2 mo before they were removed and brought to the lab where they were stored at −5°C until analysis.

Laboratory Analyses

Bulk samples of fresh soil were sorted to remove all visible root mass and coarse fragments (>2 mm). A subsample was drawn for gravimetric water content (Gardner, 1986). Fresh soil and organic horizon samples (25 g oven dry equivalent) were shaken with 50 mL of 2 *M* KCl for 30 min and filtered through Whatman no. 42 filter paper. Extracts were analyzed for NH_4^+–N with the Berthelot reaction (Willis et al., 1993), and NO_3^-–N by nitration of salicylate (Yang et al., 1998). Microbial biomass was determined with the fumigation extraction, ninhydrin-reactive N method as described by Joergensen and Brookes (1990), and as modified by DeLuca and Keeney (1993). Biomass C was calculated as difference between ninhydrin reactive N in fumigated samples and the control multiplied by a factor of 21, while biomass N was calculated as the difference between ninhydrin reactive N in fumigated samples and the control, multiplied by a factor of 3.1 (Joergensen and Brookes, 1990).

PMN was determined by a 14-d anaerobic incubation in which 5 g of moist soil or organic horizon was placed in a centrifuge tube with 12.5 mL of nanopure water and the air space displaced by N_2 gas (Hart et al., 1994). After a 2-wk incubation at 25°C, 12.5 mL of 4 *M* KCl was added to each tube to create 2 *M* KCl extracts. Soils were shaken for 30 min, filtered and analyzed for NH_4^+–N, as described above. Soluble hexose sugar concentrations measured as anthrone reactive carbon (ARC) were determined on 0.5 *M* K_2SO_4 extracts obtained from 25 g of fresh soil (soil/extract ratio of 1:2) and analyzed within 48 h post sampling (DeLuca, 1998). Microbial respiration was measured during a 3-d aerobic incubation with alkaline traps (Zibilske, 1994). Fifty grams of fresh soil was brought to 60% water-holding capacity and placed in a mason jar, and an open vial containing 20 mL of 1 *M* NaOH was carefully placed atop the soil, then sealed with a screw top and incubated at 25°C. The CO_2 traps containing NaOH were then quantitatively transferred to 200-mL flasks, 24 mL of 4 *M* $BaCl_2$ and five drops of phenylthaleine indicator solution added, and the solution titrated with 1.0 *M* HCl to a clear end point. Inorganic N was extracted from ionic resin capsules by shaking them in three successive 25-mL volumes of 2 *M* HCl (creating a 75-mL sample). Extracts were analyzed for resin extractable NO_3^-–N following the method as described above.

The remaining fresh soil material was air dried, subsamples were fine ground to pass through a 76-μm sieve and analyzed for total C and N with a Fissions EA1100 dry combustion CNSHO analyzer (Fissions Inst. Inc., Milan, Italy). Organic horizon samples obtained in the spring 1997 were oven dried at 102°C, and the horizon density estimated based on mass by horizon depth by PVC cylinder area.

Statistical Analysis

Field replication had to be performed within treatment units because the study was implemented ex post facto on a wildfire site and because of physical space limitations within the area exposed to the fire, thereby resulting in our use of pseudoreplication; however, with a large number of replicates and a random pooling of transect subsamples, we hoped to avoid statistical incoherence. Data were analyzed with the Statistical Analysis System (SAS Institute, 1997) by a *t* test to individually perform a pairwise comparison between the fire exposed soils and the unburned control while avoiding pooling of variance across treatment units. Homogeneity of variance was tested with a folded *F* statistical test, and degrees of freedom were computed by Satterthwaite's approximation in case of unequal variances. Pearson's correlation was run on resin NO_3^-–N and PMN ($n = 32$) and between biomass C, ARC and CO_2 evolution ($n = 8$). We calculated the following ratios for each treatment separately; biomass N/total N, PMN/total N, PMN/biomass N, NH_4^+–N/PMN, biomass C/total C, ARC/biomass C, CO_2/biomass C (qCO_2), and CO_2/ARC (CO_2 coefficient).

RESULTS AND DISCUSSION

Fire consumed 100% of the organic horizon in WF, 65% in PBWF, and 42% in PB (Table 1). Chemical analysis of remaining fire-exposed organic horizons in fall 1996 showed significant increases in NH_4^+–N and total N but reduced labile N (PMN) per unit volume. PMN concentrations in PBWF and PB organic horizons were significantly lower than in the unburned control, suggesting that the majority of labile N originally present prior to prescribed fire was lost from the organic horizon through erosion, translocation to the subsurface mineral soil, or mineralization. These results are similar to those reported by Jurgensen et al. (1981) who observed significant losses of labile N from organic horizons following fire.

Soil Nitrogen Transformations

Mineral soil analysis indicated that in fall 1996, total N, PMN, NH_4^+–N, and NO_3^-–N concentrations in fire-exposed soils were significantly higher than the unburned control with the highest total N contents observed in WF and PBWF soils (Table 2); however, by fall 1997, total N in WF and PBWF were similar to those of the PB. Total N concentrations in the unburned control were consistently the lowest of all soils throughout the study.

PMN concentrations in wildfire-exposed soils were four times (WF) and two times (PBWF) greater than in the unburned control (Table 2). This immediate increase in soil labile N concentrations probably resulted from downward translocation of labile N released from fire-consumed litter and soil organic matter, fire-killed vegetation, and microbial biomass (Fritze et al., 1992; Neary et al., 1999). In contrast, the lowest soil PMN concentrations in PB soils corresponded with enhanced mineralization of simple organic N compounds to NH_4^+–N (DeLuca and Zouhar, 2000) between the spring burn and fall 1996 sampling.

In spring 1997, concentrations of PMN in WF and PBWF soils also declined below that of the unburned control and remained lower until the end of study. Concentrations of PMN in all fire-exposed soils remained in the N deficiency range for ponderosa pine as outlined by Powers (1980); however, by spring 1998, labile N increased 33% in WF and 99% in PBWF soils compared with their average concentrations from spring and fall 1997. Although PMN content in fire-exposed soils represented only a small portion of total N (consistently low ratio of PMN/total N) (Fig. 1), the annual turnover of grass established after fire may rebuild soil PMN structurally and chemically different than that of the prefire forest (Andreu et al., 1996).

Ammonium concentrations in fire-exposed soils remained significantly higher than the unburned control for one year (Table 2). In research by Dunn et al. (1985), the concentration of inorganic N was positively correlated with burning intensity and proportion of total organic horizon consumed. They hypothesized that a portion of the inorganic N released from the overlaying organic layer recondensed and leached down into the mineral soil. The increase in inorganic N after fire has been found in previous studies (Kovacic et al., 1986; Covington and Sackett, 1992). Soils in the PBWF site had the lowest NH_4^+–N concentrations of all fire-exposed soils, possibly a result of combined N mineralization induced by the prescribed burn and subsequent

Table 1. Selected organic (Oe and Oa) horizon characteristics for wildfire (WF), prescribed fire followed by a wildfire (PBWF), prescribed fire (PB) and unburned control in fall 1996.

Characteristic	WF	PBWF	PB	Control
Depth of Oe and Oa (cm)	0.0**†	3.3**	6.3**	10.8
Bulk density (g cm^{-3})	–	0.17	0.24	0.16
PMN (μg cm^{-3})	–	83.6**	190**	354
NH_4^+–N (μg cm^{-3})	–	18.8**	20.1**	9.6
Biomass C (μg cm^{-3})	–	1140**	3650**	5690
ARC (μg cm^{-3})	–	31.7**	32.7**	90.8
Total C (g kg^{-1})	–	256**	216**	139
Total N (g kg^{-1})	–	10.3**	9.2**	5.4

†/− Difference between treatments and control determined by *t* test. Probability of mean separation indicated by ** = $P \leq 0.01$.

Table 2. Soil N (0–10 cm mineral soil) characteristics: potentially mineralizable nitrogen (PMN), ammonium (NH_4^+–N), nitrate (NO_3^-–N), and total N for wildfire (WF), prescribed fire followed by a wildfire (PBWF), prescribed fire (PB) and unburned control measured seasonally.

Time	Treatment	PMN	NH_4^+–N	NO_3^-–N	Total N
		μg g^{-1}			g kg^{-1}
Fall 1996	WF	41.4**†	19.5**	6.6*	3.3**
	PBWF	20.3**	6.3**	3.3	2.9**
	PB	8.5**	10.6**	3.0	2.3**
	Control	11.2	1.9	2.4	1.2
Spring 1997	WF	7.4**	7.5**	2.9**	2.7**
	PBWF	9.1**	4.0*	1.4**	2.6**
	PB	8.0**	1.2	0.4	2.2**
	Control	20.7	1.7	0.4	1.2
Fall 1997	WF	7.6**	6.1**	5.8**	1.9**
	PBWF	4.8**	4.0*	1.6	1.9**
	PB	11.6	1.7	1.6	1.9**
	Control	18.0	2.2	1.3	1.1
Spring 1998	WF	10.0**	0.8	3.0**	2.2**
	PBWF	13.8*	1.0	0.8	2.5**
	PB	9.8**	0.3**	0.7	2.2**
	Control	17.9	0.7	0.6	1.2

†/− Difference between burned soils and unburned control determined by *t* test. Probability of mean separation indicated by ** = $P \leq 0.01$ and * = $P \leq 0.05$.

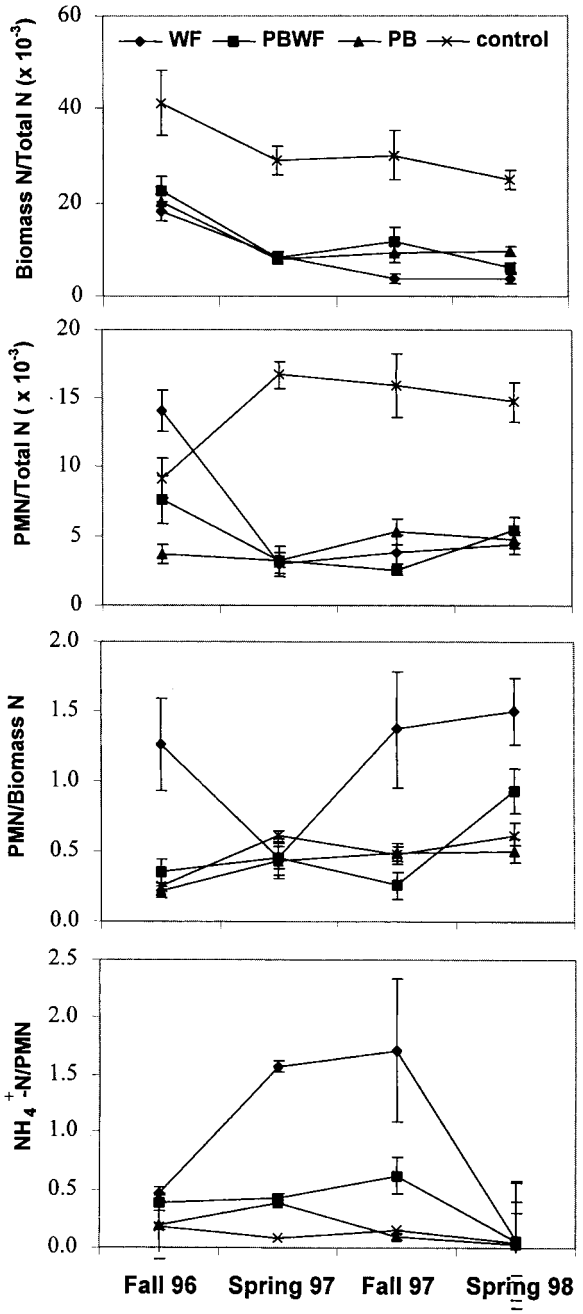

Fig. 1. Seasonal changes in biomass N/total N, PMN/total N, PMN/biomass N, and NH_4^+–N/PMN (±1 SE) calculated for wildfire (WF), prescribed burn followed by a wildfire (PBWF), prescribed fire (PB) and unburned control.

chemical oxidation by the wildfire. Because the prescribed burn consumed a significant amount of the insulating organic horizon, surface temperatures may have become high enough to volatilize significant quantities of N during the wildfire. For example, Giovannini et al. (1990) demonstrated that NH_4^+–N concentrations in mineral soil decreased rapidly after soil samples were exposed to temperatures exceeding 220°C.

The NH_4^+–N concentrations in WF and PBWF soils remained significantly higher than the no-fire unburned control until spring 1998. In the PB soils NH_4^+–N concentrations decreased to levels similar to the unburned control by spring 1997, and were significantly lower than the unburned control by spring 1998.

Table 3. Resin sorbed nitrate (resin NO_3^-–N) and correlation between resin NO_3^-–N and soil potentially mineralizable nitrogen (PMN) for wildfire (WF), prescribed fire followed by a wildfire (PBWF), prescribed fire (PB) and unburned control measured in Summer 1997.

Treatment	Resin NO_3^-–N	Resin NO_3^-–N:PMN correlation
	μg capsule^{-1}	
WF	88.4**†	0.75**
PBWF	23.5**	0.71**
PB	10.4**	−0.01
Control	1.3	−0.87

†/− Difference between treatments and controls determined by *t* test. Probability of mean separation indicated by ** = $P \leq 0.01$.

The ratio of NH_4^+–N/PMN as an index of change in these pools after fire had elevated values in WF soils until fall 1997, suggested enhanced labile N mineralization (Fig. 1). In contrast, the lower NH_4^+–N/PMN ratio in PBWF may indicate slower PMN turnover with greater retention of labile organic N compared with WF.

Fire immediately increased soil NO_3^-–N concentrations in the WF soils, where on average, it was nearly three times greater than the unburned control ($P \leq 0.05$). Nitrate increase after fire probably resulted from nitrification of fire-released NH_4^+–N after the soil was recolonized by population of heat-sensitive nitrifiers (Fritze et al., 1994; Neary et al., 1999). Soil NO_3^-–N concentrations remained elevated in the WF soils until the study was terminated; however, the cumulative amount of NO_3^-–N sorbed to mixed bed ionic resins (2-mo incubation) during Summer 1997 demonstrated accelerated NO_3^-–N accumulation in all fire-exposed soils (Table 3). The greatest quantity of NO_3^-–N sorbed to the resins was in WF soils (88.4 μg capsule^{-1}), followed by PBWF soils (23.5 μg capsule^{-1}). A similar increase in net NO_3^-–N accumulation observed by Kaye and Hart (1998) was attributed to reduced rates of NO_3^-–N immobilization.

Additionally, a strong positive correlation between resin-sorbed NO_3^-–N and PMN concentrations for WF and PBWF soils in spring 1997 suggests enhanced chemoautotrophic nitrification. Conversely, a weak correlation between resin NO_3^-–N and PMN in the unburned control, in combination with small net NO_3^-–N accumulation (1.29 μg capsule^{-1}) and the lowest net soil NO_3^-–N concentrations in spring and fall 1997 suggest either: (i) restricted nitrification due to competitive NH_4^+–N uptake by heterotrophic microorganisms (Bauhus et al., 1993); (ii) rapid NO_3^-–N immobilization (Stark and Hart, 1997); (iii) altered pathway of NO_3^-–N production not linked to the autotrophic NH_4^+–N oxidation (Barraclough and Puri, 1995; Barraclough, 1997); or (iv) a combination of these explanations.

Changes in Microbial Biomass

Fire significantly reduced the amount of microbial biomass C in both mineral soils and organic horizons in fall 1996 (Tables 1 and 4). Soil microbial biomass C concentrations remained low in all fire-exposed sites

Table 4. Soil C (0–10 cm mineral soil) characteristics: microbial biomass C (Biomass C), hexose sugars (ARC), basal respiration (CO_2 evolution), and total C for wildfire (WF), prescribed fire followed by a wildfire (PBWF), prescribed fire (PB) and unburned control.

Time	Treatment	Biomass C	ARC	CO_2 evol.	Total C
		—— μg g^{-1} ——		mg g^{-1} d^{-1}	g kg^{-1}
Fall 1996	WF	249	12.7**†	0.33**	67**
	PBWF	342	3.6	0.22	61**
	PB	290	2.2	0.17**	48**
	Control	327	4.7	0.20	24
Spring 1997	WF	134**	3.6	0.15**	54**
	PBWF	141**	2.1**	0.18**	59**
	PB	117**	2.3*	0.13**	46**
	Control	237	3.9	0.21	25
Fall 1997	WF	54**	4.1**	0.09	34**
	PBWF	188**	2.9**	0.13	40**
	PB	132**	4.1*	0.10	36**
	Control	273	5.5	0.11	19
Spring 1998	WF	52**	1.9**	0.08**	41**
	PBWF	104**	1.5**	0.13	53**
	PB	140**	2.5*	0.15	43**
	Control	202	3.1	0.12	20

†/− Difference between treatments and controls determined by T-test. Probability of mean separation indicated by ** = $P \leq 0.01$ and by * = $P \leq 0.05$.

until the study ended. Suppressed microbial biomass recovery has been reported elsewhere (Vazquez et al., 1993; Prieto-Fernandez et al., 1998), resulting from release of chemicals inhibiting fungal growth (Diaz-Ravina et al., 1996) and a limited supply of available C (Bauhus et al., 1993). For example, Pietikainen and Fritze (1993) reported that fungal growth did not reach the levels in the unburned control within the first 3 yr following fire. Interestingly, starting from spring 1997, microbial biomass C concentrations in PBWF soils were always higher than WF soils, suggesting their faster recovery in PBWF (Table 4).

All fire-exposed sites demonstrated continuously low biomass N/total N and biomass C/total C ratio throughout the study (Fig. 1 and Fig. 2) similar to the findings of Prieto-Fernandez et al. (1998). Conversely, the same ratios were high in the unburned control, which is related to the relatively small total soil C and N pools in the unburned control and the relatively large portion of C and N accounted for by the microbial biomass.

The relationship between biomass N and PMN may provide a sound estimate of microbial turnover of N (Hart et al., 1986; Myrold, 1987) (Fig. 1). In our study the PMN/biomass N ratio was consistent and low for PB, PBWF and the unburned control, but dropped markedly in WF soils in spring 1997, and recovered by fall 1997. This could indicate fire-induced temporary depletion of labile N supplies below microbial demand. The subsequent recovery of the PMN/biomass N ratio in fall 1997 may come from PMN from senescing perennial vegetation and annual turnover of grasses.

Soil Carbon Transformations

In fall 1996, concentrations of hexose sugars (soluble ARC) in the organic horizon of PBWF and PB were significantly lower than in the unburned control (Table 1). At the same time, WF soils had significantly higher concentrations of soluble ARC than the unburned control (Table 4); however, during winter, ARC declined to concentrations similar to the unburned control. Interestingly, the ARC concentrations in PBWF soil were consistently lower than the WF soil throughout the study. The initial disturbance-induced increase of hexose sugars was probably rapidly immobilized or mineralized (Fernandez et al., 1997; DeLuca, 1998; DeLuca and Zouhar, 2000).

High concentrations of soluble ARC in WF probably resulted in increased basal soil respiration rates in fall

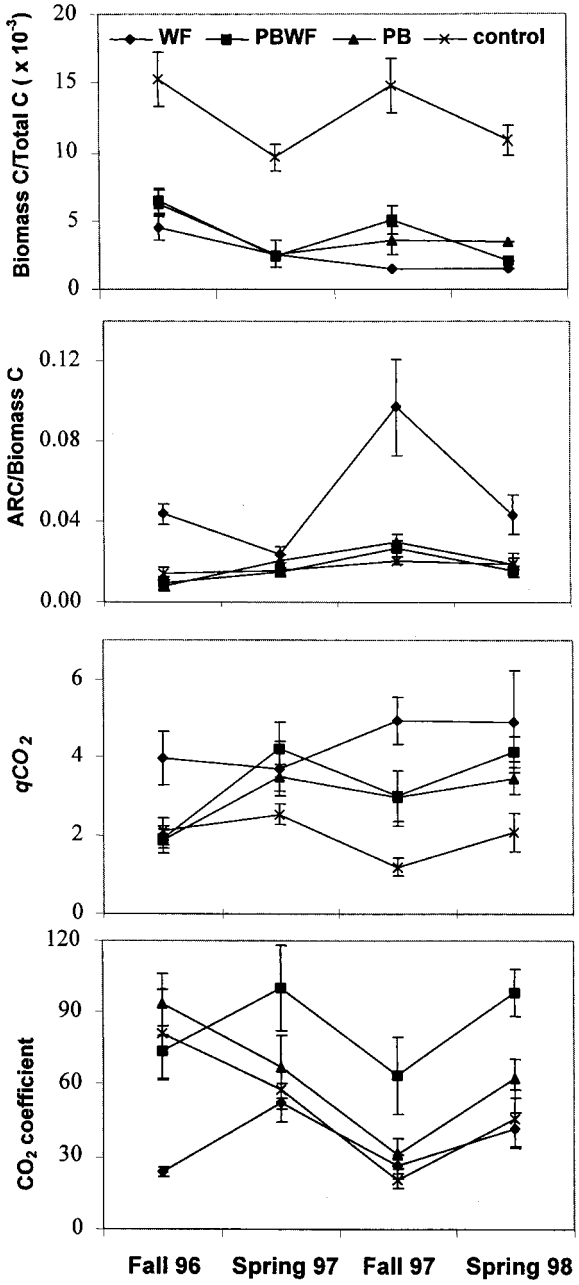

Fig. 2. Seasonal changes in biomass C/total C, ARC/biomass C, qCO_2 (CO_2:biomass C), and CO_2 coefficient (CO_2:ARC) (±1 SE) calculated for wildfire (WF), prescribed burn followed by a wildfire (PBWF), prescribed fire (PB) and unburned control.

1996 (Table 4). Low concentrations of available C may have resulted in lower respiration rates in all three fire-exposed soils starting as early as fall 1996 in PB soils and spring 1997 in WF and PBWF soils; however, of all fire-exposed sites, PBWF soils had the highest levels of basal respiration and remained significantly higher than the WF soil through the spring of 1998.

A strong correlation between CO_2 evolution and soluble ARC was observed in fall 1996 in WF, PBWF and PB soils (strongest in PB soils at $r = 0.98$, $P \leq 0.01$). This, in combination with a weak correlation between ARC and biomass C and lack of correlation between CO_2 evolution and biomass C, suggests slower microbial biomass recovery at this early postfire stage (i.e., CO_2 evolution from direct C utilization during maintenance respiration rather than from rebuilding the microbial biomass). Similarly, Bääth and Arnebrant (1994) observed increased respiration rates without increased growth rates. In contrast, the unburned control lacked a strong CO_2 and ARC correlation, but at the same time demonstrated a strong correlation between ARC and biomass C, and between CO_2 and biomass C, suggesting use of simple hexoses for microbial growth.

The correlation between ARC and biomass C for PBWF soils improved in spring 1997 ($r = 0.73$, $P \leq 0.01$), and became highly significant in WF soils the following fall ($r = 0.68$, $P \leq 0.01$). It is possible that microbial populations became competitive for this limited C source. The prescribed fire may have facilitated development of a stress tolerant microbial population that could more rapidly recover after a subsequent wildfire. Large populations of spore-forming bacteria and algae found after laboratory soil heating simulations by Vazquez et al. (1993) support this suggestion. This was also demonstrated by the consistently high ratio of ARC/biomass C in WF soils throughout the study (Fig. 2). In contrast, this ratio in PBWF and PB soils remained comparable to the unburned control. Additionally, the CO_2 coefficient (CO_2/ARC) that represents the allocation of soluble ARC for microbial respiration (Fig. 2) was the lowest in WF soils while in PBWF soils it demonstrated the highest value, suggesting more efficient biological mineralization of hexose sugars hence the fastest microbial recovery.

Fire also had a significant effect on qCO_2 (CO_2/biomass C) (Fig. 2). This ratio was consistently high in WF soils and low in the unburned control while the PBWF and PB soils had modestly higher qCO_2 throughout the study. According to Wardle and Ghani (1995), disturbance increases soil qCO_2. Low values in the unburned control were probably due to undisturbed conditions, where the environment was dominated by complex detrital food webs of highly competitive fungal populations (Fritze et al., 1994). Rapid change in substrate composition post disturbance and drastically reduced fungal communities (both hyphae and propagules) in fire-exposed sites lead to CO_2 evolved directly from non-structural simple organic substrates and high demand for C during fire recovery (Fritze et al., 1994; Vazquez et al., 1993).

CONCLUSIONS

Our research demonstrated the role of prescribed fire in mitigating the effects of high intensity wildfire on soil processes. Overall, the soils we studied exhibited significant change in microbial activity and C and N transformations for 2 yr following fire, even though concentrations of inorganic N had returned to prefire concentrations within the first year. In spite of being exposed to fire twice in one year, the PBWF soils had significantly lower rates of net N mineralization and higher levels of PMN than in soils exposed to a single intense WF. Potentially mineralizable N on all fire exposed sites declined to concentrations below that of the unburned control within 9 mo after fire and did not recover by the end of the study period.

Microbial activity remained low in all fire-affected soils throughout this study; however, microbial recovery in the PBWF soil progressed at a faster rate than in the WF soil and ultimately biomass C and basal respiration increased on PBWF soils to levels twice that of the WF soils. Slow microbial recovery corresponded with a low availability of labile C and low basal respiration rates. Concentrations of hexose sugars per unit microbial biomass C in PBWF soils was low and comparable to the unburned control; however, microbial C and N demand in PBWF soils appeared to be satisfied (based on PMN/Biomass N and ARC/Biomass C).

Slow microbial biomass recovery during the first year following fire probably created an opportunity for fire-surviving plants and pioneer species to take advantage of the large pool of available NO_3^-–N. The presence of lush grass on WF and PBWF sites may begin the PMN pool recovery, which is probably different compared with the prefire forest (different organic matter quality and turnover rates); however, NO_3^-–N losses to leaching would probably have been greatest on the WF site (based on the high rates of NO_3^-–N sorption to buried resin capsules). Overall, our results indicate that prescribed fire not only reduced wildfire severity, but also lessened the losses of labile C and N and improved the resistance of soil microbial communities to subsequent wildfire thereby affording more rapid recovery following fire on this site.

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