

RESEARCH ARTICLE

THE INFLUENCE OF FIRE ON THE RADIOCARBON SIGNATURE AND CHARACTER OF SOIL ORGANIC MATTER IN THE SISKIYOU NATIONAL FOREST, OREGON, USA

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ABSTRACT

Forest fires contribute a significant amount of CO₂ to the atmosphere each year, and CO₂ emissions from fires are likely to increase under projected conditions of global climate change. In addition to volatilizing aboveground biomass and litter layers, forest fires have a profound effect on belowground carbon (C) pools and the cycling of soil organic matter as a whole. However, the influence of fire on belowground organic matter cycling is not well defined and varies widely with fire severity. We measured soil organic matter (SOM) characteristics across a range of fire severities two years after the 2002 Biscuit Fire in southwest Oregon, USA, to address the following questions: (1) Which C pools are preferentially volatilized or transformed to charcoal under low-se-

RESUMEN

Los incendios forestales contribuyen significativamente al incremento anual de CO₂ en la atmósfera, y es probable que las emisiones de CO₂ provenientes de estos incendios incrementen en las condiciones proyectadas de cambio climático global. Además de volatilizar la biomasa superficial y la hojarasca, los incendios forestales tienen un profundo efecto sobre los depósitos de carbono (C) subterráneo y en el ciclo de la materia orgánica. Sin embargo, la influencia de los incendios en el ciclo de la materia orgánica subterránea no ha sido totalmente comprendida y varía mucho en función de la severidad de cada incendio. En este estudio medimos las características de la materia orgánica del suelo (MOS) en un rango diverso de severidades de fuego dos años después del incendio Biscuit, ocurrido en 2002 en el sureste de Oregon, en los EUA y formulamos las siguientes preguntas: (1)

verity and high-severity fire? (2) How does fire change the distribution of SOM among density fractions and depths? (3) How does fire affect the general character of SOM including such variables as abundance, C:N ratio, ^{13}C abundance, and ^{14}C abundance? We examined soils from a mixed hardwood-evergreen forest across a range of burn severities: unburned, low severity, mixed severity, and high severity. Results indicated that increasing burn severity led to progressive loss of forest floor mass, but not to progressive loss of C from the mineral soil. Although fire significantly increased the charcoal content of the soils, fire did not significantly change the distribution of soil organic matter between heavy and free or light fractions. Other significant changes in soil organic matter characteristics included a progressive increase in nitrogen (N) with increasing burn severity, possibly due to the encroachment of N-fixing shrubs following the loss of native vegetation. Although qualitative changes in total root abundance following fire were noted, differences among burn severity treatments were not statistically significant. Increased concentrations of rock fragments in burned areas may be suggestive of erosion in these areas, consistent with previous studies documenting varying degrees of soil erosion following fire. In addition, although ^{13}C abundances were similar among severely burned and unburned plots, soils from severely burned plots were significantly depleted in ^{14}C in comparison to soils from unburned plots. This ^{14}C depletion is most likely the combined result of erosion and preferential combustion of organics enriched in ^{14}C relative to the bulk soil, perhaps reflecting a historical pattern of fire occurrence and severity across the landscape.

¿Cuáles depósitos de C son volatilizados o transformados en carbono vegetal bajo incendios de alta y baja severidad? (2) ¿A raíz del incendio, cómo cambia la distribución de la SOM en sus distintas fracciones en cuanto a densidad y profundidad del suelo? (3) ¿Cómo afecta el incendio el carácter general de la SOM incluyendo aquellas variables como abundancia, relación C:N, y abundancia de ^{13}C y ^{14}C ? Examinamos suelos provenientes de bosques perennifolios mixtos de crecimiento lento a lo largo del siguiente rango de severidades de fuego: sin quemar, baja, mixta y alta. Los resultados indicaron que el incremento en la severidad del incendio lleva a la pérdida progresiva de la masa del suelo forestal, pero no a pérdidas progresivas de carbono en el suelo mineral. Aunque el incendio incrementó significativamente el contenido de carbono en el suelo, no produjo cambios significativos en la MOS en sus distintas fracciones. Otros cambios significativos en las características de la MOS incluyeron un incremento progresivo en el nitrógeno (N) a medida que aumentó la severidad, debido posiblemente al incremento en arbustos fijadores de nitrógeno después de la pérdida de vegetación nativa. Aunque los cambios cualitativos en la abundancia total de raíces fueron notables después del incendio, éstos no fueron significativos entre los distintos niveles de severidad. Los incrementos observados en la concentración de fragmentos de rocas en áreas quemadas podrían sugerir procesos erosivos, lo que ha sido documentado anteriormente por estudios previos. Aunque la abundancia de ^{13}C fue similar en parcelas quemadas y no quemadas, el ^{14}C en las parcelas quemadas severamente disminuyó significativamente en relación a las parcelas no quemadas. Este agotamiento en el ^{14}C es probablemente el resultado de los efectos combinados de la erosión y la combustión de compuestos orgánicos enriquecidos con ^{14}C y relativos a la masa total del suelo, que probablemente reflejen el patrón histórico de incidencia y la intensidad de los incendios en el paisaje.

Keywords: Biscuit Fire, charcoal, density separation, radiocarbon, soil organic matter, wildfire

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INTRODUCTION

Forest fires play a substantial role in the global carbon (C) cycle as a whole, with forest loss and degradation due to fire accounting for an estimated 12% of total anthropogenic CO₂ emissions (van der Werf *et al.* 2009, Halofsky *et al.* 2010). Both the frequency and severity of forest fires are likely to increase under scenarios of future climate conditions (Dale *et al.* 2001, Westerling *et al.* 2006, IPCC 2007, Littell *et al.* 2009, Westerling *et al.* 2011, Moritz *et al.* 2012), highlighting the importance of assessing the impacts of forest fire on the terrestrial C cycle.

Frequently, estimates of C loss and ecosystem disturbance due to fire only account for aboveground losses and disturbances. Given that, globally, the soil organic C pool is nearly three times the size of the aboveground biomass C pool (Lal 2004), the effects of fire on the soil organic C cycle should be included in assessments of overall ecosystem disturbance and C emissions. Though more recent investigations (Yermakov and Rothstein 2006, Campbell *et al.* 2007, Bormann *et al.* 2008, Mack *et al.* 2011, Nave *et al.* 2011, Kashian *et al.* 2013) have begun to examine the effects of fires on soil organic C stocks as well as aboveground C stocks, few studies have examined the effects of fire on the nature and properties of the soil organic matter cycle as a whole. Additionally, there is an inherently large amount of uncertainty associated with soil organic C stock stability under changing climate conditions. Taking into consideration the effects of major disturbance events such as fire on the belowground C cycle only increases these uncertainties, offering further proof of the need for more intense research into the effects of forest fires on soil organic matter cycling.

Previous studies have explored the effects of fire on soil organic matter properties from the ecosystem scale to the molecular level. The effect of fire on soil organic matter stocks is highly variable, and depends on ecosystem type, fire severity, landscape position, and soil properties. Changes in soil organic matter molecular structure depend strongly on fire temperature and intensity. When soils reach temperatures above ~50 °C, microbial mortality, root mortality, and soil organic matter structural alteration are initiated. As temperatures rise and the time of exposure to intense heat increases, soil organic matter goes through progressive molecular alteration including dehydration, followed by dehydrogenation, volatilization of nitrogenous compounds, loss of functional groups (i.e., decarboxylation and demethylization), and finally formation of polycyclic aromatic hydrocarbons through condensation reactions (Hernández *et al.* 1997, Knicker 2007, Ciampa *et al.* 2009). These molecular scale changes can lead to increases in soil organic matter hydrophobicity and decreases in biodegradability, resulting in increased mean residence times (MRT) for pyrogenic materials (commonly referred to as charcoal, char, or black carbon). Soil organic C mineralization rates may be additionally suppressed following intense fires due to loss of microbial biomass. Severe fires may sterilize the upper centimeters of mineral soil, and microbial communities may take anywhere from months up to 10 years to fully recover to previous activity levels (Fritz *et al.* 1993, Dumontet *et al.* 1996, Hamman *et al.* 2007, Dooley and Tresender 2012, Ginzburg and Steinberger 2012). Other commonly reported changes in soil parameters following fire include increases in pH, hydrophobicity, exchangeable cation concentration, and soil temperature. The ef-

fect of fire on large scale estimates of soil organic carbon (SOC) stocks is highly variable. Many studies report an increase in soil organic C and N (nitrogen) stocks following fire, while other studies find the opposite (Johnson and Curtis 2001, Wan *et al.* 2001, Hart *et al.* 2005, Johnson *et al.* 2007, Smithwick *et al.* 2009, Nave *et al.* 2011).

In the current study, we took a comprehensive approach to assessing the influence of fire on forest soil organic matter cycling. Soils subjected to a range of fire severities were examined with the goal of assessing: (1) Which C pools are preferentially volatilized or transformed to charcoal under low and high severity fire? (2) How does fire change the distribution of soil organic matter (SOM) among density fractions and depths? (3) How does fire affect the general character of SOM including such variables as abundance, C:N ratio, ^{13}C abundance, and ^{14}C abundance?

METHODS

Changes in root abundance, soil C and N concentrations, and forest floor characteristics were evaluated across a range of fire severities. Soil erosion, which is often a consequence of vegetation and litter layer loss following fire, was evaluated by comparing rock fragment content among soils subjected to different burn severities. Additionally, we used natural abundance radiocarbon as a conservative tracer to characterize changes in soil organic C distribution among density fractions as the result of high severity stand-replacing forest fire, and to gain insight into the suitability of adjacent unburned plots for use as controls when studying the effects of fire on forested ecosystems

Sample Sites and Soil Sampling

The Biscuit Fire was a mosaic fire that burned over 200 000 hectares of mixed conifer forest (overstory dominated by *Pseudotsuga menziesii* [Mirb.] Franco, *Pinus lambertiana* Douglas, *Pinus ponderosa* Lawson and C.

Lawson, *Abies concolor* [Gord. and Glend.] Lindl. ex Hildebr., and *Quercus* spp.) in Siskiyou National Forest, Oregon, USA, in 2002. The extent and effects of the Biscuit Fire are well documented (Harma and Morrison 2003a, b; Campbell *et al.* 2007; Bormann *et al.* 2008; Donato *et al.* 2009a, b). The mosaic pattern of the fire yielded patches of forest impacted by different burn intensities. Areas that showed no evidence of fire were designated as control plots. Burned areas in which at least 75% of the aboveground dominant vegetation survived were designated as low severity plots (Hann *et al.* 2004). Burned areas that showed local spatial variation in burn severity, alternating between low and high severity, were designated as mixed severity plots (Smith 2000). Areas in which at least 75% of the aboveground dominant vegetation was consumed or died as a result of fire were designated as high severity plots (Hann *et al.* 2004).

Soils were sampled in August 2004, approximately two years post fire, from areas of different burn severity: unburned control stands (not burned in the past 50 years), low severity stands, mixed severity stands, and high severity stands. Sites were selected so that variation in stand age, slope, aspect, and elevation were minimized (Table 1). Samples were taken at four 1 ha sites in each burn severity class for root, rock fragment, C and N concentration measurements, and soil sampling sites were distributed across each plot area. An additional two plots were sampled at high-severity burn sites and unburned control sites. Ten to twenty cores were composited at each site, mixed, and stored frozen until they were processed. Soils used for root, rock fragment, and C and N concentration measurements were collected from one depth, 0 cm to 5 cm, using a large-diameter (6 cm) auger. Soils used for density separations were collected at depth increments of 0 cm to 5 cm and 5 cm to 15 cm from the high severity sites and unburned control sites using a small-diameter (2 cm) auger.

Table 1. Sample site characteristics.

Burn severity	Elevation (m)	Aspect (°)	Slope (%)
Unburned control	495 to 1199	NW to NE	23 to 46
Low severity	830 to 1145	NW to N	35 to 51
Mixed severity	939 to 1187	varied	41 to 51
High severity	924 to 1316	varied	22 to 34

Soil Analysis

Volumetric rock fragment content of the soils was estimated in the lab from soil cores of known volume as the volume of the rocks (>2 mm) divided by the total volume of core sample. Organic C and total N concentrations were measured on the fine earth fraction on a Leco CNS-2000 Macro Analyzer (LECO Corporation, St. Joseph, Michigan, USA) at the Central Analytical Laboratory at Oregon State University. Separate cores were used for root abundance measurements. Roots were separated from the mineral soil through hydro pneumatic elutriation (Caldwell and Virginia 1989) and classified according to diameter: fine roots of <2 mm diameter, and medium roots of 2 mm to 10 mm diameter. Larger roots were not found in the examined soil samples. Forest floor samples, measuring 180 cm², were collected from 12 locations randomly distributed throughout each study plot.

Density separation. Soils were sieved at 2 mm (#10 sieve), and all other analyses were performed on the fine earth fraction. Roots were removed, and soils were dried at 65 °C for three days after sieving. Soils were density fractionated densimetrically using a method modified from Strickland and Sollins (1987) and Swanston *et al.* (2005). Soil subsamples (50 g) of the soils were weighed into 250 ml Nalgene® centrifuge bottles. One hundred millimeters of sodium polytungstate at a density of 1.65 g cm⁻³ were added to each bottle. Bottles were capped and shaken at low speed on a shaker table for one hour. Samples were then

allowed to settle overnight to permit ample time for separation of organic and mineral fractions. The supernatant was then aspirated, and the aspirated fraction was rinsed with double deionized water over a 0.8 µm polycarbonate filter before being oven-dried at 65 °C. Materials that floated at a density of 1.65 g cm⁻³, and were subsequently aspirated, were classified as the light fraction. Light fraction removal was repeated until no organics were visible in the supernatant solution. The remaining soil pellet (considered the heavy fraction) was rinsed three times with double deionized water through repeated suspension in water, centrifugation at 2800 × g, and aspiration of supernatant. The rinsed heavy fractions were dried at 105 °C. Coarse charcoal particles, operationally defined as charcoal particles large enough to be visible at 40× magnification, were picked out of the light fractions using tweezers and a dissecting microscope. Carbon concentrations of fractions were estimated manometrically during the graphitization process by combusting an aliquot of each fraction and measuring the volume of CO₂ produced. This method of percentage C estimation and the process of density separation itself introduce a nontrivial amount of error into calculations of C recovery, as can be seen in Table 2. Although mass recoveries after density separation were very close to 100% with small standard errors, C recoveries were systematically skewed to values >100%.

δ¹³C and radiocarbon analysis. After density separation, subsamples of each density fraction were sent to the Stable Isotope Ratio

Table 2. Mass and C recovery from density separations. Standard errors are given in parentheses.

	Depth	Recovery (%)	
		Mass	C
Unburned control	0 cm to 5 cm	99 (1)	120 (15)
	5 cm to 15 cm	101 (<1)	128 (20)
High severity	0 cm to 5 cm	100 (1)	130 (12)
	5 cm to 15 cm	101 (<1)	109 (11)

Facility for Environmental Research at the University of Utah, Salt Lake City, for ^{13}C analysis. The $\delta^{13}\text{C}$ values are reported in reference to the international Pee Dee Belemnite standard. Density fractions and bulk soils were measured for radiocarbon abundance in 2004. Coarse charcoal fragments were measured for radiocarbon in 2011. Samples were combusted with CuO and Ag in sealed glass test tubes to form CO_2 gas. The CO_2 was then reduced to graphite through heating in the presence of H_2 gas and a Fe catalyst (Vogel *et al.* 1987). The graphitized samples were measured for radiocarbon ($\Delta^{14}\text{C}$) at the Center for Accelerator Mass Spectrometry, Lawrence Livermore National Lab, California (Davis *et al.* 1990). Radiocarbon data is reported as $F^{14}\text{C}$ (Reimer *et al.* 2004) and $\Delta^{14}\text{C}$. Reported radiocarbon values included a background subtraction determined from ^{14}C -free coal, a $\delta^{13}\text{C}$ correction for isotopic fractionation, and a decay correction to account for decay of the primary standard (Oxalic acid 1) since 1950 (Stuiver and Polach 1977). Analytical error on average was $\pm 4.4\%$.

Modeling and Statistical Analysis

Mean residence times (MRT) of bulk soils and density fractions were calculated using a time-dependent steady-state model (Trumbore 1993, Torn *et al.* 2009). This model assumes that all organic inputs to each modeled fraction bear the atmospheric radiocarbon signature of the previous year, and does not account for lag

times associated with the transfer of organics among fractions. These calculations are thus likely to overestimate MRTs. Thermonuclear testing in the 1950s and 1960s doubled the amount of radiocarbon in the atmosphere. Due to the incorporation of this “bomb” radiocarbon into soil fractions at different rates, some fractions that cycle carbon more rapidly have two possible MRTs. In all cases where two MRT solutions were possible, the shorter MRT was eliminated as a possible solution due to the unreasonably large predicted yearly output of C from the fraction ($>30\%$ of total pool size per year).

Each sample site was treated as an independent statistical replicate when evaluating the influence of burn intensity on soil characteristics. For data associated with percent C, percent N, litter, roots, and rock fragments, $n = 4$ for all burn severities. Only control and high-severity burn soils were used for density separation and radiocarbon analysis, and although 6 plots were sampled for each, $n = 5$ or $n = 6$ due to the loss of some samples during laboratory processing. Significant differences in soil characteristics among treatments were determined by either one-way ANOVA by burn treatment or two-way ANOVA by burn treatment and soil horizon depth ($\alpha = 0.05$). The ANOVA analysis was followed by a Tukey-Kramer *post hoc* test at a 95% confidence limit.

RESULTS

Forest Floor

Forest floor mass was significantly reduced in all burned stands, and reductions were highest in the high-severity burn areas. Areas that experienced high-severity burning also showed changes in forest floor composition, with evidence that woody debris in the litter layer was nearly double that in the unburned control stands (Table 3).

Table 3. Forest floor and root data. Fine roots are <2 mm; medium roots are 2 mm to 10 mm. Parenthetical values are the standard error of *n* replicate plots. Within each column, values with the same letter are not significantly different at a $\alpha = 0.05$ level.

	Forest floor			Roots			
	<i>n</i>	Mass (kg m ⁻²)	Woody debris* (%)	Depth (cm)	<i>n</i>	Fine root mass (g m ⁻²)	Medium root mass (g m ⁻²)
Unburned control	12	3.3 (±0.5) ^A	20 (±4) ^A	0 to 15	4	346 (±12) ^A	199 (±67) ^A
Low severity	12	2.1 (±0.5) ^B	19 (±4) ^A	0 to 15	4	479 (±149) ^A	147 (±24) ^A
Mixed severity	12	2.0 (±0.2) ^{AB}	20 (±4) ^A	0 to 15	4	501 (±86) ^A	84 (±26) ^A
High severity	12	0.9 (±0.3) ^B	38 (±8) ^A	0 to 15	4	616 (±111) ^A	67 (±34) ^A
ANOVA <i>P</i> -value		0.0003	0.0536			0.6658	0.1135

* Percent of total forest floor mass.

Mineral Soil

Soil organic C concentrations did not vary significantly with burn severity. Soil N concentrations in the soils from burned stands were higher than N concentrations in the unburned control sites. Soil N concentrations in the high-severity burn sites were more than twice that of the control sites (0.18% versus 0.40%), while the mixed-severity sites and low-severity underburn sites had intermediate N concentrations (0.21% and 0.29%). No significant differences were noted in $\delta^{13}\text{C}$ signatures between unburned and severely burned soils. Light fractions had high average C contents of 36.7 ± 1.0% and were composed of organic matter in various forms and stages of decay, including coarse charcoal. Coarse charcoal comprised a significant portion of the total C pool in soils from severely burned sites (40.7 ± 4.6%); whereas soils from the control plots contained very little coarse charcoal (3.6 ± 0.4% of total C). Coarse charcoal was found in light fractions from both sampled depth increments (0 cm to 5 cm and 5 cm to 15 cm). Heavy fractions were mostly mineral material and had low C concentrations of approximately 3.3% (±0.2%). No coarse charcoal particles were observed in the heavy fractions from either the control or high-severity burn soils. In both sampled depth increments, the heavy

fractions from the high-severity burn plots appeared to have higher average C concentrations than the equivalent fraction from the control plots (average of 3.5% C versus 3.0% C respectively), but due to the large within-treatment variation, these differences were not statistically significant (Figure 1). Variation in mineral soil C:N ratios followed variation in N concentrations, with soils from control plots having significantly higher C:N ratios than soils from severely burned plots.

The skeletal nature of the soils examined in this study prevented any accurate measurements of bulk density, but estimates of soil erosion were attempted by comparing coarse fragment contents of the soils from different burn severity classes, following Bormann *et al.* (2008). This method of estimating erosion assumes that, as soil fines are removed from the soil surface by water or wind, the coarse rock fragments are left behind, leaving eroded soils with qualitatively higher rock fragment contents than their uneroded counterparts. Although differences in rock fragment content among soils were not statistically significant due to high variability within treatment groups, soils from high-severity burn plots were 4% greater than the control soils; percent rock fragment content of low- and mixed-severity plots were 3% greater than control soils (Table 4).

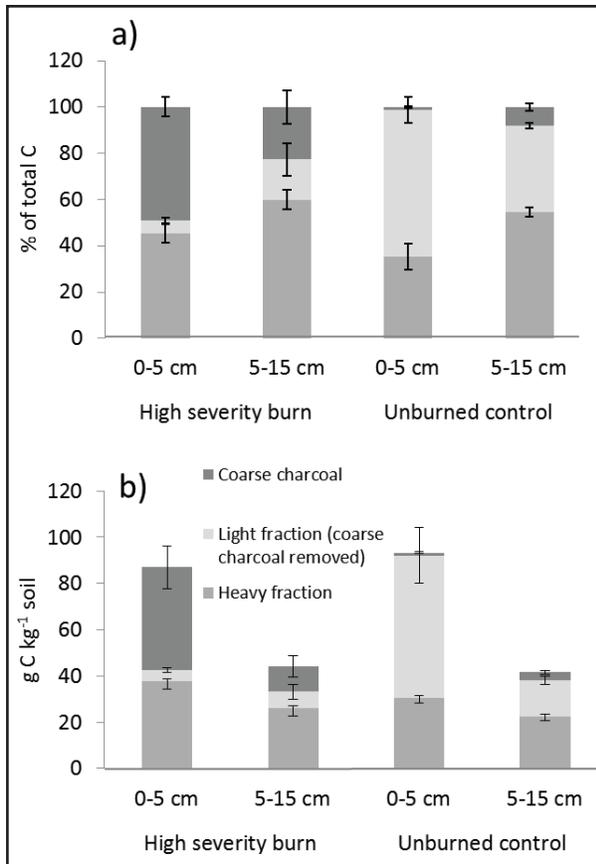


Figure 1. Distribution of organic C among coarse charcoal, light fraction (with coarse charcoal removed), and heavy fraction in both severely burnt and unburned soils: a) distribution as a percent of total recovered C; b) C abundance of each fraction in grams of C per kg of soil. Bars are the standard error of either 5 or 6 experimental replicates.

Radiocarbon Analysis

Density fractions and coarse charcoal particles from unburned and high-severity burn plots were measured for radiocarbon abundance. Light fractions were measured both with coarse charcoal included and with coarse charcoal particles removed. In all cases, heavy and light fractions from the high-severity burn soils had longer MRT than their counterparts from the unburned control soils (Table 5). Differences in MRT were significant only in the 0 cm to 5 cm depth increment ($P = 0.050$ for light fractions, $P = 0.041$ for heavy fractions; Figure 2). In the 0 cm to 5 cm depth increment, $\Delta^{14}\text{C}$ of charcoal fragments taken from the burned soils were variable, but averaged $\sim 1\text{‰}$ (parts per thousand), indicating that these charcoals were a mix of contemporary C as well as charcoal formed prior to 1950. In contrast, the average $\Delta^{14}\text{C}$ of charcoal from the 0 cm to 5 cm layer of the unburned soils was -45‰ . In the 5 cm to 15 cm depth, the reverse is true; charcoals from the severely burned soils were radiocarbon depleted in comparison to charcoal in the control soils (Table 5, Figure 2c).

Roots

Qualitatively, fine root abundance seemed to increase with increasing burn severity, and medium root abundance seemed to decrease with increasing burn severity. However, these

Table 4. Mineral soil data by burn severity. Parenthetical values are the standard error of n replicate plots. Within each column, values with the same letter are not significantly different at a $\alpha = 0.05$ level.

	Depth (cm)	n	Rock fragments* (%)	C (%)	N (%)	C:N
Unburned control	0 to 15	4	26 (± 2) ^A	8.3 (± 1.6) ^A	0.18 (± 0.03) ^B	46 (± 4) ^A
Low severity	0 to 15	4	29 (± 2) ^A	6.8 (± 1.2) ^A	0.21 (± 0.03) ^B	34 (± 3) ^B
Mixed severity	0 to 15	4	29 (± 2) ^A	10.8 (± 0.13) ^A	0.29 (± 0.01) ^{AB}	37 (± 2) ^{AB}
High severity	0 to 15	4	30 (± 3) ^A	9.8 (± 1.6) ^A	0.40 (± 0.09) ^A	26 (± 2) ^B
ANOVA P -value			0.2882	0.1283	0.005	0.0003

* >2 mm, by volume.

Table 5. Radiocarbon data for the control and high severity plots. Values in parentheses represent the experimental standard error of 5 or 6 replicates.

	Depth	Fraction	n	$\delta^{13}\text{C}$ (‰)	F^{14}C	$\Delta^{14}\text{C}$ (‰)	MRT (yr)
Unburned control		Bulk soil	6	-25*	1.082 (± 0.007)	75 (± 7)	
	0 cm to 5 cm	Light (no char)	6	-26.3 (± 0.2)	1.114 (± 0.011)	107 (± 11)	85 (± 9)
		Charcoal	6	-25.9	0.962 (± 0.007)	-45 (± 7)	
		Heavy	5	-24.2 (± 0.2)	1.076 (± 0.009)	69 (± 9)	137 (± 12)
	5 cm to 15 cm	Bulk soil	6	-25*	1.052 (± 0.006)	45 (± 6)	
		Light (no char)	5	-26.1 (± 0.2)	1.078 (± 0.013)	71 (± 13)	135 (± 19)
		Charcoal	6	-25.6	0.917 (± 0.023)	-88 (± 22)	
		Heavy	6	-24.2 (± 0.2)	1.020 (± 0.012)	13 (± 12)	310 (± 56)
High severity		Bulk soil	6	-25*	1.057 (± 0.009)	50 (± 9)	
	0 cm to 5 cm	Light (no char)	6	-25.7 (± 0.2)	1.071 (± 0.012)	64 (± 12)	146 (± 23)
		Charcoal	5	-25.6	1.008 (± 0.030)	1 (± 30)	
		Heavy	6	-24.2 (± 0.2)	1.042 (± 0.007)	35 (± 7)	192 (± 19)
	5 cm to 15 cm	Bulk soil	6	-25*	1.005 (± 0.024)	-2 (± 24)	
		Light (no char)	5	-25.5 (± 0.2)	1.062 (± 0.013)	55 (± 12)	153 (± 27)
		Charcoal	6	-25.2	0.901 (± 0.026)	-127 (± 26)	
		Heavy	5	-24.2 (± 0.1)	1.008 (± 0.014)	1 (± 14)	358 (± 76)

* Estimated value.

trends were not statistically significant ($P > 0.05$; Table 3).

DISCUSSION

Despite decades of intense study, the net effect of fire on forest soil C and N storage remains unresolved. This is partially due to the highly variable nature of fires, as well as the inherent heterogeneity of soils. However, whether mineral soils appear to gain or lose C and N as a result of fire also depends to a great degree on whether soil erosion and concurrent changes in soil bulk density are factored into loss estimates (Bormann *et al.* 2008, Nave *et al.* 2011). If the possibility of erosion and subsequent changes in soil bulk density are discounted, fire did not significantly reduce C concentrations in any of the soils examined in this study, regardless of burn severity. The main effect of high-severity fire seemed to be

a change in the character of the organics through conversion of the low density organic matter into charcoal (Figure 1).

The difference in radiocarbon abundance between unburned and high severity burn soils also offers compelling evidence that fire had a significant impact on the character of the remaining soil organic matter. In all density fractions and at both examined depths, soils from severely burned plots were depleted in ^{14}C in comparison to soils from unburned plots. The lower $\Delta^{14}\text{C}$ values yielded longer modeled MRTs for organics in burned soils, giving them the appearance of increased SOM stability and turnover time.

The magnitude of ^{14}C depletion observed in the SOM from the high-severity burn plots suggests that the 2002 Biscuit Fire was not solely responsible for the observed SOM transformations observed at these plots. Most likely, the depletion of ^{14}C and resultant increase

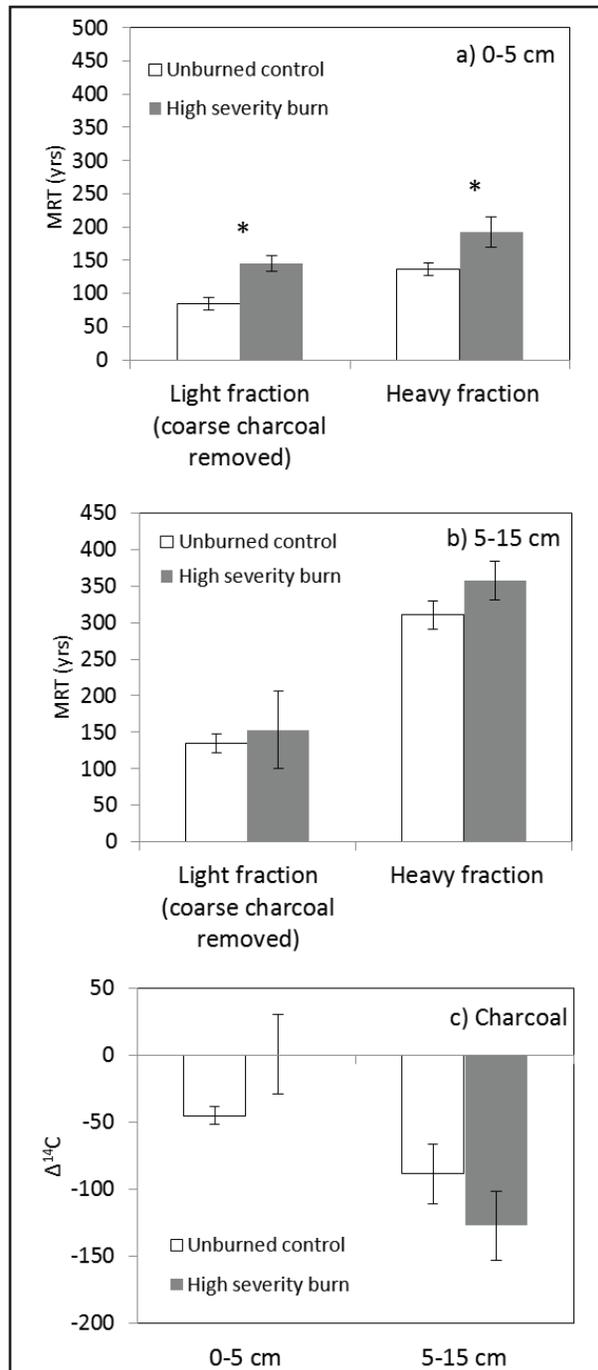


Figure 2. Modeled mean residence times (MRT) of density fractions from high-severity burn and unburned control soils at two depths: a) 0 cm to 5 cm, and b) 5 cm to 15 cm. c) $\Delta^{14}\text{C}$ (‰) of coarse charcoal removed from the 0 cm to 5 cm and 5 cm to 15 cm depths of high-severity burn and unburned soils. Error bars are the standard error of either 5 or 6 experimental replicates. Asterisks (*) indicate significant differences between control and burnt soils at $\alpha = 0.05$.

in apparent MRT in the 0 cm to 5 cm layer of burned soils was due to the combined effects of erosion and combustion, possibly over several fires. In general, soil radiocarbon abundance decreases with depth; therefore, erosion of surface soil could expose organics depleted in radiocarbon relative to the removed soil. Selective combustion of organics enriched in radiocarbon relative to bulk SOM could also leave behind soils depleted in radiocarbon (longer MRT) relative to unburned soils. This selective combustion could be due to both the physical distribution of organics and their molecular characteristics. Molecular structure may play a role in determining which organics are completely combusted and which are converted to charcoal. Laboratory thermal analysis has indicated that when soil organics are burned, carbohydrates and low molecular weight organics are lost prior to the combustion of more complex aromatic compounds (see Plante *et al.* 2009). In such a case, relatively undecomposed organics may combust prior to condensed polyaromatics or charcoals formed in previous fire events. Polyaromatic and pyrogenic compounds are also more persistent in many soils (Skjemstad *et al.* 1996, Knicker 2007), with correspondingly low radiocarbon abundance (Schmidt *et al.* 2002, Gavin *et al.* 2003). The 0 cm to 5 cm light fraction (char removed) was 107 $\Delta^{14}\text{C}$ in the unburned plots, but only 60 $\Delta^{14}\text{C}$ in the burned plots. This difference could reflect selective loss of the light fraction due its relative molecular structural lability as well as its primary location in more aerated macropores within the soil matrix. The addition of charred material to the soil matrix from more contemporary needles and litter is much less likely to have contributed to significant net ^{14}C depletion of the light fraction, since such pools often reflect atmospheric values in the 5 to 10 years prior to the fire, which ranged from 136 $\Delta^{14}\text{C}$ to 75 $\Delta^{14}\text{C}$ (Hua and Barbetti 2004, Levin and Kromer 2004).

The most basic hypothesis was that if the radiocarbon signature of the eroded control

soil matched the radiocarbon signature of the high-severity burn soil, then erosion could be identified as the overriding mechanism responsible for the changes in radiocarbon content observed in the burned soils. To determine whether erosion could account for the differences in apparent MRT between high-severity burn soils and unburned control soils, we calculated the effect of 1.8 cm of erosion on the radiocarbon signature of the 0 cm to 5 cm layer of the unburned control soils (Figure 3). We chose an erosion value of 1.8 cm as the most likely scenario through comparison with published erosion rates from nearby soils (Bormann *et al.* 2008) and documented increases in coarse fragment content of the burned soils. Around 2 cm of erosion is necessary to account for a 4% increase in rock fragment content, which is very close to the 1.8 cm of soil erosion measured by Bormann *et al.* (2008) on similar soils in the area after the Biscuit Fire. Calculation indicated that the apparent MRT of the heavy fraction of the high-severity burn soils was not significantly different than that of

the eroded control soil ($P = 0.698$), indicating that 1.8 cm of erosion could account for the observed radiocarbon abundance and MRT of the high-severity burn soils. However, erosion could not fully explain differences in apparent MRT of the light fraction.

The apparent MRT of the light fraction from the high-severity burn soil was still significantly longer than that of the eroded control soil ($P = 0.053$). Since there was no significant difference in percent C between unburned control soils and high-severity burn soils, differences in radiocarbon abundance of the light fractions can only be explained by replacement of relatively “young” organic matter (^{14}C signature close to modern), with organic matter possessing a lower ^{14}C concentration, either during or after the fire.

High-severity burn soils also contained substantially more coarse charcoal in the 5 cm to 15 cm layer than the unburned control soils. The radiocarbon abundance of these charcoals indicates that in neither case was the charcoal found at depth created in the most recent fire event. Taken together, the differences in $\Delta^{14}\text{C}$ and proportions of charcoal between the two plot types implies that the high-severity burn plots and unburned control plots have significantly different burn histories. It may also indicate that the high-severity plots have historically burned more intensely or frequently than the control plots, a pattern that has been noted before (Thompson *et al.* 2007). The proposition of historically different burn patterns and the fire-induced mechanisms of SOM transformation described above are not mutually exclusive. However, differences in fire history among plots do insert greater uncertainty about the timing of the observed changes in SOM characteristics.

Radiocarbon patterns observed in the subsurface layer (5 cm to 15 cm) were different than those of the 0 cm to 5 cm layer. After removal of the coarse charcoal, the estimated MRT of the 5 cm to 15 cm light fraction was equivalent between the control and high-sever-

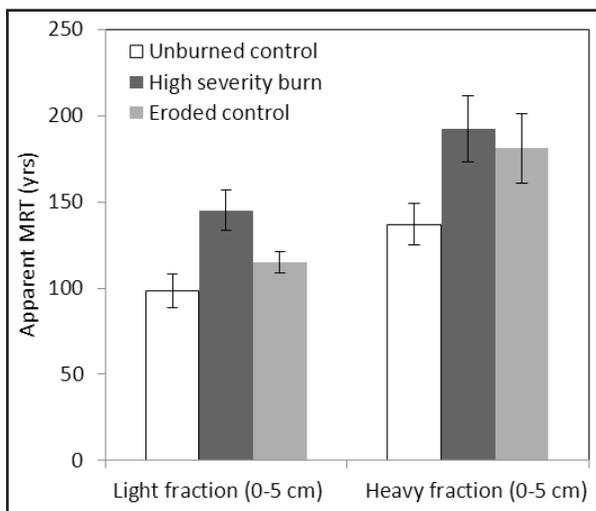


Figure 3. Comparison of modeled mean residence times of light and heavy density fractions for unburned control soils, high severity burn soils, and unburned control soils modeled with 1.8 cm of erosion. 1.8 cm was chosen as the most probable erosion scenario following the work of Bormann *et al.* (2008) on soils in the same area.

ity burn plots, indicating that non-charcoal organics were cycling at roughly the same rate in all soils, and possibly arguing against the occurrence of substantial long-term erosion. The radiocarbon measurements, coupled with other soil measurements, provided compelling evidence that the main long-term effect of fire on SOM characteristics was not necessarily organic matter or soil loss, but organic matter transformation.

Other important changes in SOM characteristics were noted as well. Nitrogen concentrations in the soils seemed to increase with increasing burn severity, and a near doubling of N concentration was observed in soils from the highest burn severity class. Additionally, in a previous study of soils in the Siskiyou National Forest, Bormann *et al.* (2008) found no significant subsurface soil N losses (although losses of N from surface horizons were substantial due to erosion) and one subsurface horizon from burned soils was enriched in N in comparison to preburn N concentrations measured at the same site. Post-fire increases in soil N content are not uncommon (Johnson *et al.* 2001, Wan *et al.* 2001), and have been attributed to a combination of N additions from ash, increases in nitrogen mineralization through stimulation of microbial communities, and post-fire establishment of N-fixing plant species (see Smithwick *et al.* 2005). These increases in soil N concentration may have a significant influence on future rates of both N and C mineralization from these soils, potentially interacting with different SOM pools through priming effects or stabilizing effects (Sollins *et al.* 1984, Homann *et al.* 2001, Neff *et al.* 2002, Hagedorn *et al.* 2003, Swanston *et al.* 2004). Additionally, increases in soil N may have important ramifications for growth rates of post-fire vegetation. Nitrogen is commonly a limiting nutrient in forested ecosystems (LeBauer and Treseder 2008), a fact that is incorporated into many Earth System Models (University Corporation for Atmospheric Research 2013). Increases in N availability following fire may

stimulate growth rates of post-fire vegetation, a phenomenon recognized in models such as the Community Land Model (Oleson *et al.* 2010), and which may significantly affect plant growth rate responses to elevated atmospheric CO₂ concentrations (Reich *et al.* 2006).

Changes in forest floor composition and root composition will have long-lasting effects on soil C cycling in these systems, since roots and the litter layer are sources of soil organic matter inputs. The increasing loss of forest floor material with increasing burn severity is a commonly noted, important consequence of wildfire. Increases in the woody debris content of the forest floor in the severely burned plots may serve as a significant and continuing source of charcoal as the debris is degraded and incorporated into the soil (Schulze *et al.* 1999, Donato *et al.* 2009a). Although differences in root abundance can only be commented on speculatively due to the high within-treatment variability, increases in fine root abundance concurrent with decreases in medium root abundance in burned soils may indicate a shift to early succession species (e.g., grasses, forbs), as found by Irvine *et al.* (2007) on another fire in Oregon, and thus in the amount of C contributed to soils from roots each year. This can ultimately lead to a change in the composition of the organic matter. Taken as a whole, data indicate that soils subjected to fire undergo significant changes in soil organic matter character, with changes increasing in magnitude with increasing burn intensity. These changes in soil organic matter characteristics cause fundamental changes in the soil organic C cycling in these systems.

Radiocarbon data suggest that these observed changes in SOM characteristics are not the product of a single fire event, namely the 2002 Biscuit Fire. Instead, radiocarbon data suggest a long history of differing fire frequencies for the so-called unburned control and high severity plots. The greater abundance of charcoal in the high-severity plots at both sampled depths (Figure 1) suggests that these areas

have burned more frequently (González-Pérez *et al.* 2004). Radiocarbon abundances of the charcoal fragments separated from the unburned control and high severity plots are also indicative of differences in fire history among these plots. The effect of longer-term cycles of repeat burning on variables such as soil N turnover and ^{13}C signatures cannot be surmised from the current dataset. However, the data presented here highlight the usefulness of radiocarbon as a natural tracer in soils that can be exploited to evaluate fire histories in forest soils. Application of radiocarbon analysis to

elucidate relationships between fire frequency, erosion, and accumulation or loss of SOM could additionally be useful in a modeling context. Although fire is often a part of C cycle models, its influence on the SOM pool is generally poorly constrained due to the complexity of the variables influencing SOM dynamics following fire events (Arora and Boer 2005, Kloster *et al.* 2010). Radiocarbon measurements of soil charcoals and mineral soils could offer insights into how to parameterize the effects of fire on forest soils (Sun *et al.* 2004).

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