Intense forest wildfire sharply reduces mineral soil C and N: the first direct evidence

Bernard T. Bormann, Peter S. Homann, Robyn L. Darbyshire, and Brett A. Morrissette

Abstract: Direct evidence of the effects of intense wildfire on forest soil is rare because reliable prefire data are lacking. By chance, an established large-scale experiment was partially burned in the 2002 Biscuit fire in southwestern Oregon. About 200 grid points were sampled across seven burned and seven unburned stands before and after the fire. Fire-related soil changes — including losses of soil organic and inorganic matter — were so large that they became complicated to measure. The 51 Mg·ha^{-1} of loose rocks on the soil surface after fire suggests erosion of 127 Mg·ha^{-1} of fine mineral soil, some of which likely left in the fire plume. After accounting for structural changes and erosion with a comparable-layers approach, combined losses from the O horizon and mineral soil totaled 23 Mg C·ha^{-1} and 690 kg N·ha^{-1}, of which 60% (C) and 57% (N) were lost from mineral horizons. Applying a fixed-depth calculation — commonly used in previous fire studies — that disregards structural changes and erosion led to underestimates of loss of nearly 50% for C and 25% for N. Although recent debate has centered on the effects of postwildfire forest management on wood, wildlife habitat, and fuels, this study indicates that more consideration should be given to the possible release of greenhouse gases and reduction of future forest productivity and CO_2 uptake.

Résumé : Il est rare d'avoir une preuve directe des effets d'un incendie de forêt intense sur le sol forestier parce qu'on manque de données fiables antérieures au feu. Par chance, une expérience en cours à grande échelle a été partiellement incendiée lors de l'incendie de Biscuit en 2002 dans le sud-ouest de l'Oregon. Environ 200 points de grille ont été échantillonnés dans sept peuplements incendiés et sept peuplements épargnés par le feu avant et après l'incendie. Les changements dans le sol reliés au feu, incluant les pertes de matière organique et inorganique, étaient tellement importants qu'ils sont devenus difficiles à mesurer. Les 51 Mg·a^{-1} de roches libres à la surface du sol après le feu indiquent que 127 Mg·ha^{-1} de sol minéral à grains fins a été érodé dont une partie a probablement été emportée dans la colonne du fumée. Après avoir tenu compte des changements structuraux et de l'érosion avec une approche d'horizons comparables, les pertes dans l'horizon O et le sol minéral totalisaient 23 Mg·ha^{-1} de C et 690 kg·ha^{-1} de N, dont 60 % (C) et 57 % (N) provenaient des horizons minéraux. Les pertes de C et N ont été sous-estimées de respectivement 50 et 25 % en appliquant un calcul à profondeur fixe, communément utilisé dans les études précédentes sur le feu, qui ne tient pas compte des changements structuraux et de l'érosion. La discussion traite surtout des effets potentiels de l'aménagement après feu sur la matière ligneuse, les habitats fauniques et les combustibles. Il est justifié de se préoccuper davantage de l'émission potentielle de gaz à effet de serre et de la réduction possible de la productivité future de la forêt ainsi que de l'absorption de CO_2.

[Traduit par la Rédaction]

Introduction

Forest ecologists think of wildfire as an important natural process that regulates fuel accumulation and successional patterns across most western US forests (DeBano et al. 1998). Forest wildfires also have great societal consequences. Rural communities and firefighters are well aware of the dangers of high-intensity (high-temperature) large-scale fires. Given dry conditions and sufficient fuels, these fires can make their own weather, spread at alarming rates, and often become nearly unstoppable. The monetary and human costs of fighting such fires — loss of property, timber, wildlife habitat, water quality, C stocks, and other resource values, and remediation expenses — can be substantial (Neuenschwander et al. 2000; Dombeck 2001). The direct cost of fighting wildfires nationally in 2002 was $1.6 billion,

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Fig. 1. Changes in wildfire extent from 1915 to 2005 on federal lands in Oregon and Washington (Bormann et al. 2006). Oregon is additive to Washington (area graph). Data complied by David L. Peterson, Pacific Wildland Fire Sciences Laboratory, 400 N 34th Street, Suite 201, Seattle, WA 98103, USA.
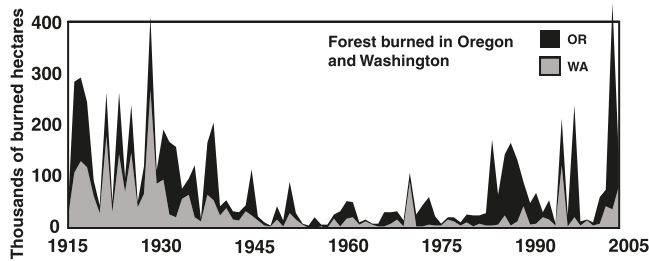
with nearly a tenth of this budget spent on a single fire, reported on here, called the Biscuit complex fire, in southwestern Oregon, USA (Government Accounting Office 2004).

Wildfires in US Pacific Northwest forests are common (Agee 1993), and the area burned annually has varied through the last 90 years (Fig. 1). According to the modern record, the area burned by wildfires declined around 1940 and remained low for 40 years. This decline corresponded to the emergence of more effective fire fighting that was focused on extinguishing fires before they could reach large size and high intensity. The rise in fires after 1980 is often attributed to an increase in the amount of fuels as a result of fire exclusion, but a complex mix of other factors is likely involved, including changes in fire-fighting methods (Government Accounting Office 2004) and climate (McKenzie et al. 2004; Gedalof et al. 2005; Westerling et al. 2006). Regardless of the specific causes, an increase in the area and (or) intensity of wildfire raises many concerns.

Current understanding of the ecological effects of intense wildfire is severely limited by the lack of detailed knowledge of soils before wildfire. To date, only one study has reported effects of wildfire based on before and after soil sampling, but fire intensity was unknown (Murphy et al. 2006; Johnson et al. 2007). In that study, conducted in the Lake Tahoe basin, USA, the fire decreased C and N pools in the O horizon plus woody debris by 90%; no changes in mineral soil pools were observed, but about 1 cm of mineral soil was removed by water erosion immediately after the fire. Other wildfire studies have been conducted retrospectively, in which controls were subjectively selected in nearby unburned forests. In these studies, the assumption that unburned forest patches represent the prefire conditions in burned stands is being questioned by recent findings (Thompson et al. 2007). Indirect evidence of the effects of intense fires on soils can also be acquired by extrapolating the results of laboratory, slash-pile, and prescribed-fire studies of lower intensity (Neary et al. 1999; Johnson et al. 2004; Certini 2005). However, because forest wildfire is a poorly understood and highly chaotic process, laboratory and low-intensity fires are not likely to produce reliable extrapolations.

When the 2002 Biscuit fire serendipitously burned through part of a 150 ha Long-Term Ecosystem Productivity (LTEP) experiment (Bormann et al. 1994; Homann et al.

2008), we were given an opportunity to examine soil changes in paired pre- and post-wildfire samples. Because we had clear evidence that these plots had burned at high intensity, our objective was to determine the effects of high-intensity fire on the loss of soil C and N mass. Volumetric soil sampling, which has not been conducted in previous wildfire studies, allowed us to rigorously evaluate changes on a per-area basis. The parallel data from unburned stands provided important evidence on soil development that was not attributable to the fire. Having archived prefire samples also allowed us to reduce analytical biases. By combining these data, we provide an assessment of the potential effects of wildfire that is the most rigorous to date. Our assessment of soil physical and chemical changes provides unique information about the effects of high-intensity forest wildfire on soils and greenhouse-gas emissions, along with important implications for long-term productivity and future C sequestration.

Materials and methods

Study site

The LTEP program was started by the Pacific Northwest Research Station, the National Forest System, and the Bureau of Land Management in 1989 to study long-term management influences on ecosystem productivity and biodiversity (Bormann et al. 1994, http://www.fsl.orst.edu/ltep/). Four linked operational-scale LTEP experiments were established across the Pacific Northwest, including the experiment we focus on in the Rogue River – Siskiyou National Forest, about 25 km southeast of Gold Beach, Oregon, USA, between 750 and 900 m elevation. Before assigning experimental treatments, homogeneous stands and soils were chosen from an area that had regenerated naturally after an 1890 wildfire (Little et al. 1995). Pretreatment forest composition was 80- to 100-year-old Douglas-fir (*Pseudotsuga menziesii* var. *menziesii* (Mirb.) Franco) with knobcone pine (*Pinus attenuata* Lemmon), some sugar pine (*Pinus lambertiana* Dougl.), and a second story of hardwoods (tanoak, *Lithocarpus densiflorus* (Hook. & Arn.) Rehd.; giant chinquapin, *Chrysolepis chrysophylla* (Dougl. ex Hook.) Hjelmqvist var. *chrysophylla*; and madrone, *Arbutus menziesii* Pursh). The soils are mapped as the Saddlepeak–Threetrees complex, which are loamy-skeletal, mixed, superactive, frigid Typic Dystrupdepts developed on a parent material of weathered sandstone and schist–phyllite. They had been previously identified as Typic Dystrochrept in the area that burned (see below) and Typic Hapludult in the unburned area (Homann et al. 2001). Mean depth of soils to a stony C horizon is about 35 cm. Based on measurements of our samples, of the total soil inorganic material, 56% is <2 mm, 7% is 2–4 mm, and 37% is >4 mm. Of the <2 mm inorganic mass, 26% is clay, 37% is silt, and 36% is sand.

Three experimental blocks were delineated based on within-block homogeneity of initial forest and soil conditions. Within each block, seven LTEP treatments were applied to 7 ha experimental units in 1997 (Table 1). Five years later, the Biscuit fire completely consumed one block, left one unburned, and resulted in partial burning of the

Table 1. Silvicultural treatments in the Long-Term Ecosystem Productivity experiment used in this analysis.

LTEP treatment	Description
Control	Unmanaged mature stands, now 110- to 120 year-old Douglas-fir, with tanoak understory, and some knobcone pine (represents initial conditions for all treatments)
Thinned, low debris	Thinned mature stands with a relative density of 0.4 (150–200 trees·ha^{-1}), underplanted in 1997 to move towards late-successional conditions with little debris left
Thinned, high debris	Same as thinned low debris, with 15% (10–15 trees·ha^{-1}) of harvest left on the ground as woody debris to move towards late-successional conditions
Young Douglas-fir, low debris	A regeneration harvest in 1997 with Douglas-fir planted closely, competing vegetation controlled, and little woody debris left (emulating intensive management)
Young Douglas-fir, high debris	Same as Douglas-fir low debris, with 15% of the harvest left on the ground (30–40 trees·ha^{-1}) for long-term productivity
Young mixed, low debris	A regeneration harvest in 1997 followed by a mix of planted, early-successional knobcone pine, Douglas-fir, and sprouted evergreen hardwoods, with little debris left
Young mixed, high debris	Same as mixed low debris, with 15% of the harvest left on the ground (30–40 trees·ha^{-1}) to enhance long-term productivity

third. Here we report results from all seven treatments in both the completely consumed block and the unburned block.

Sampling and chemical analysis

Before LTEP treatments were implemented, a 1.5 ha measurement plot, with a permanent 25 m grid system and subsampling plots, was established in the center of each stand (Homann et al. 2001; Raymond and Peterson 2005; Fig. 2). For many variables, measurements were made more than once before the fire. Overstory tree position, diameter, height, live or dead status, and fire damage were assessed in 18 m × 18 m tree plots in 1992 before, and in 1998 after, the LTEP treatment and again in 2003 and 2004 after the fire. Understory plot data were collected in 3 m × 3 m plot frames when tree plots were sampled and in 2000 as well. Percent cover was estimated for each understory species to the nearest 1%. Three fine woody debris diameter and burn-duration classes (0 to 0.6 cm, 1 h; 0.6 to 2.5 cm, 10 h; and 2.5 to 7.6 cm, 100 h) were measured in 1 m × 1 m subplots in 1999 and 2003. Coarse woody debris (>7.6 cm diameter) was estimated with the planar intersect technique (Brown et al. 1982). Grid intersections were marked with heavy aluminum tags attached about 20 cm above the ground to steel re-bar posts. A total of 17 erosion boxes, 1.8 m × 3.6 m, were fixed onto slopes up to 35° on burned and unburned treatments in late 2003. Sides were spray painted to delineate initial surfaces, and elevations were measured across at 0.6 m grid inside the box. The volume (cubic metres) displaced downhill was used to describe water-borne erosion.

Soils were sampled during July to October in 1992 and 2003. Sampling points were proximal to each of the 15 or 16 grid intersections in each measurement plot, with 2003 points about 3 m from 1992 points. At several sampling points trees, logs, roots, or rock outcrops prohibited sampler access. In these cases, up to two additional attempts were made at distances of 1 m from the initial attempt. Overall, sampling could not be conducted at an average of 2.0 points per plot in 1992 and 1.5 points in 2003 in the burned plots, and 2.6 points in 1992 and 1.6 points in 2003 in the unburned plots. Thus, the mean number of samples per measurement plot was about 14 in each period.

Soil samples were taken perpendicular to the slope. The layer above the mineral soil surface was collected with a 30 cm diameter ring in 1992 and 21 cm diameter ring in 2003. On unburned plots, this layer was the O horizon, excluding wood greater than 2.5 cm diameter. On burned plots, this layer consisted of loose rocks (Fig. 3) and, in burned control and thinned plots, ash mixed with freshly fallen needles from fire-killed trees. After this top layer was removed, a steel sampler 35 cm long and 10 cm × 15 cm in cross-section was driven into the mineral soil to a depth of 35 cm, excavated, and removed. From the open side of the sampler, three layers of mineral soil were extracted: the A horizon (0 to a mean depth of 2.5 ± 0.3 cm, 95% confidence interval (CI) in 1992; and 0 to a fixed depth of 3.0 cm in 2003), the B1 horizon (bottom of A horizon to 15 cm), and the B2 horizon (15–30 cm). The fixed depth was used for the A horizon in 2003 because no color shift was discernable after the fire.

In the laboratory, O-horizon samples were hand sorted into coarse fragments >4 mm, (rocks), wood fragments with diameters >6.4 mm, and the remaining soil material, and then dried at 70 °C and weighed. Mineral soil samples were sieved with a 4 mm sieve to yield <4 mm soil, >4 mm rocks, and >4 mm wood, which included roots and woody debris. A 2 mm sieve was not used because aggregates >2 to <4 mm are difficult to break up and contain up to 20% of whole-soil C, even after treatment with sodium hexametaphosphate (Homann et al. 2004).

The 1992 O-horizon sample (ground to <0.8 mm, 20 mesh) was analyzed for Kjeldahl N (Bremner and Mulvaney 1982) and for loss on ignition by heating at 550 °C for >12 h. Mineral soil samples <4 mm from 1992 (ground to <0.25 mm, 60 mesh) were analyzed for total C and N with a Carlo-Erba NA 1500 Series 2 analyzer (Carlo Erba Strumentazione, Rodano, Italy). The O-horizon and mineral soil samples from 2003 were measured with a Thermo NC 1112 Analyzer (CE Elantech, Inc., Lakewood, New Jersey). To compare values from the different instruments, we converted 1992 values to equivalent Thermo NC values with equations developed from a subset of 1992 samples analyzed on the Thermo NC analyzer (Table 2). The inorganic matter concentration of the <4 mm soil samples was calculated as (100% – %C × 1.89), where 1.89 is the mean ratio of loss on ignition to Thermo C, based on 28 samples.

Fig. 2. Schematic of the measurement plot with a 5 ha buffer, 15 grid points, 5 tree plots, 16 understory plots, and 1992 and 2003 soil-sampling points. We analyzed 7 burned and 7 unburned plots in this paper.
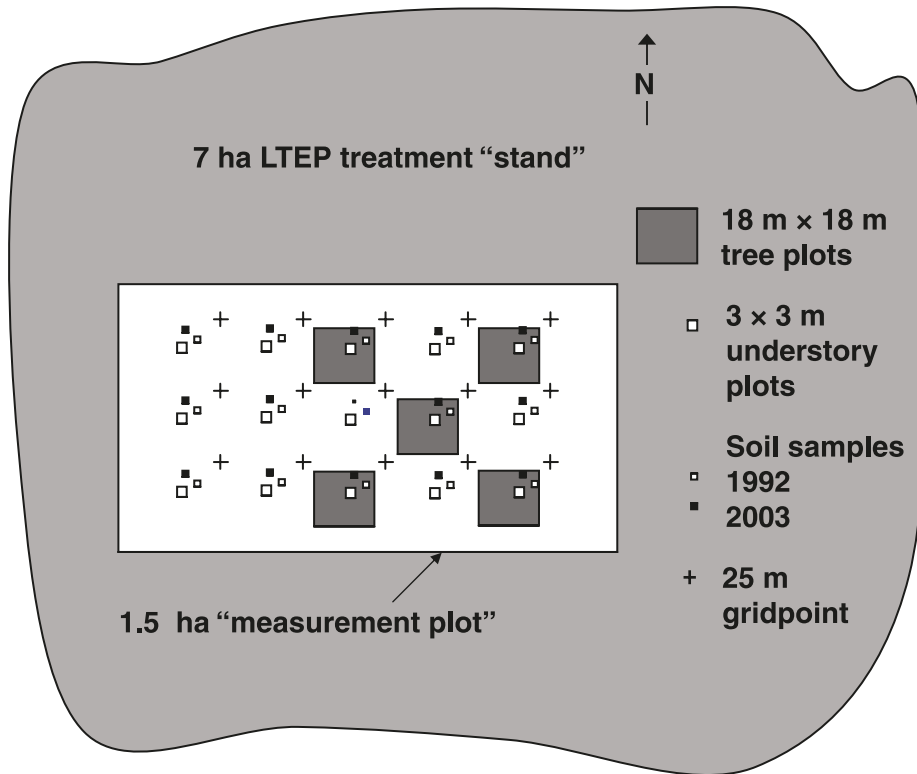

Fig. 3. Surface rocks were found consolidated in mineral soil horizons but were rarely found in the litter layer before the fire (left). Postfire surfaces nearly always had loose surface-rock accumulations (right).

Data analysis

Soil mass and rock mass per hectare were calculated for each sample layer of each sampling point of each sample year by dividing corresponding sample mass by slope-corrected cross-sectional area. For each layer, point, and year, the mass per hectare of C, N, and <4 mm inorganic matter was calculated by multiplying their respective concentration within the soil by the soil mass per hectare.

The presence of postfire surface rocks, i.e, rocks that had not been observed before the fire (Fig. 3), and an analysis of changes in rock distributions indicated potential fire-related loss of surface mineral soil and change in soil volume. The traditional approach of comparing soil in fixed-depth layers does not take these factors into account and thus may bias estimates of soil C and N change. To minimize bias, we used a modified comparable-layers approach. The

Table 2. Regression equations used to convert analyses of 1992 samples to Thermo C and N equivalent values, based on analysis of archived soil samples.

Sample type	y	x	Intercept	Slope	r^2	n
O horizon	Thermo N (%)	Kjeldahl N (%)	0.24	1.071	0.94	14
O horizon	Thermo C (%)	Loss on ignition (%)	1.56	0.512	0.91	14
Mineral soil	Thermo N (%)	Carlo Erba N (%)	0.01	1.206	0.99	32
Mineral soil	Thermo C (%)	Carlo Erba C (%)	0.08	0.978	0.99	32

comparable-layers approach builds on and expands methods pioneered in the long-term Rothamsted studies (Jenkinson and Rayner 1977) and methods developed by others (Bormann et al. 1995; Homann et al. 2001; Ellert et al. 2002). This analysis mathematically assembles sampled-soil values to create new layers based on fixed masses of inorganic soil (<4 mm) rather than fixed depths, so that soil layers are better matched before and after the fire.

We applied the comparable-layers approach to each sampled grid point by calculating five successively deeper comparable layers defined by fixed masses of <4 mm inorganic matter. Comparable layers 1 and 2 each had a fixed inorganic mass of 100 Mg·ha^{-1} (1: 0 to 100 Mg·ha^{-1} and 2: 100 to 200 Mg·ha^{-1}). Comparable layers 3 to 5 each had a fixed inorganic mass of 400 Mg·ha^{-1} (3: 200 to 600 Mg·ha^{-1}, 4: 600 to 1000 Mg·ha^{-1}, and 5: 1000 to 1400 Mg·ha^{-1}). Masses of soil C and N for the comparable layers at each sampling point were calculated from the proportions of the sampled layers that fell within each comparable layer. We modified this method by accounting for fine soil lost during and after the fire that had previously been associated with postfire surface rocks (Fig. 4). The missing fine soil was calculated by multiplying surface rock mass by plot-specific <4 mm inorganic soil:rock ratios of the prefire O plus A horizons.

To assess differences between the comparable-layer and fixed-depth approaches, we calculated soil C and N masses in five successively deeper layers corresponding to the following mean prefire comparable-layer depths: O horizon plus 0–3.7 cm mineral soil, 3.7–5.2, 5.2–10.9, 10.9–16.6, and 16.6–22.2 cm.

Sampling points were averaged to yield a plot value. Change in a plot was calculated as the difference between 2003 and 1992 plot values. To account for background changes, we subtracted the 1992 to 2003 change in a corresponding unburned plot from change in the burned plot. Mean plot change and its 95% confidence interval were calculated from the individual plot changes.

Results

Melted heavy-duty aluminum tags, on steel grid posts, indicated that the Biscuit fire burned at high intensity (high temperature) rather consistently across most of the burned LTEP plots (Table 3). Melting these tags in an oven, to look like tags melted in the fire, took 100 s at 780 °C or 1000 s at 720 °C. These temperatures are far higher than the low- to medium-intensity fires (<500 °C) on which most soils literature has focused (Certini 2005). The area burned at >700 °C and amount of fuel consumed varied somewhat

across the diverse stand structures in the LTEP plots. Tree mortality was positively related to both the percent area burned at >700 °C ($r^2 = 0.78$, p <0.01) and the amount of fine fuels consumed ($r^2 = 0.70$, $p = 0.01$). The role of fuels in tree mortality in these plots is discussed in Raymond and Peterson (2005).

The clearest effect of intense wildfire on our plots, which has also been widely noted across the Biscuit fire,[2] was a substantial increase in the amount of near-surface rocks on the burned plots — there was 51 ± 8 Mg·ha^{-1} (mean ± 95% CI) of loose rocks on the soil surface after the fire. (Fig. 5). One source of loose postfire surface rocks was the prefire O layer, whose combustion could have contributed the 9 ± 3 Mg·ha^{-1} of rocks contained in the layer before the fire. To produce the total amount of surface rocks observed, another 42 Mg·ha^{-1} of rocks must have come from the underlying mineral soil, which contained about 35 Mg·ha^{-1} of rocks for each centimetre of soil depth (Homann et al. 2001). To expose all the rocks in the mineral soil, we calculated that 127 ± 20 Mg·ha^{-1} of fine mineral soil would have had to been removed by combustion, fire-driven convective erosion, and postfire wind and water erosion (Fig. 6). An alternative mechanism that could account for the increased amount of rocks on the soil surface is fire-induced resorting, a process through which rocks from deeper soil layers move to the surface; however, resorting would have resulted in a decrease in the amount of rocks deeper in the soil, which was not observed (Fig. 5). The accumulation of rocks near the surface in prefire and unburned plots suggests that the soils in this area have been strongly influenced by previous fires, perhaps mainly by the stand-replacing fire in 1890. Regardless, the Biscuit fire further increased these near-surface rock concentrations.

The second notable effect of high-intensity fire was the major loss of soil organic matter at the soil surface that extended into the mineral soil, and corresponding losses of soil C and N. In the uppermost comparable layer 1 (the O horizon and mineral soil to 3.7 cm), soil C decreased by 19 ± 2 Mg·ha^{-1} from the prefire sampling value in 1992 (Fig. 7 left side, Table 4 method 1). A small but significant (p <0.05) amount of C (2.5 Mg·ha^{-1}) was lost from the two deepest layers combined (4 and 5). If all C in the prefire O horizon (9 ± 1 Mg·ha^{-1}) was combusted, then 60% of the soil C loss came from mineral soil layers.

Soil N losses were also large, 547 ± 79 kg·ha^{-1}. No significant subsurface soil N losses were seen, but an increase of 40 ± 32 kg·ha^{-1} was observed in layer 3. If all N in the prefire O horizon (226 ± 21 kg·ha^{-1}) was volatilized, then 57% of the soil N loss came from mineral soil layers.

The 11 year soil C and N changes in unburned soils were

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[2] Melody Culp, personal communication, Illinois Ranger District, Rogue River Siskiyou National Forest.

Fig. 4. Concepts behind the comparable-layers and the fixed-depth approaches to estimating fire-induced changes in soil C and N. For comparable layers (A), fire causes loss of surface fine mineral soil — both organic and inorganic components — leaving a layer of surface rocks (C). Fire causes additional loss of organic matter from the remaining fine surface soil, causing decrease in soil volume (D). Comparable layers accounts for processes C and D by using the masses of rocks and fine inorganics as references. In contrast, the fixed-depth approach (B) compares prefire and postfire layers that are inherently different (F) because the elevation of the mineral soil surface, from which fixed depths are measured, is lowered by the fire (G). The fixed-depth surface layer underestimates fire-induced losses because postfire measurement is inflated by soil that was not part of the prefire surface layer. The fixed-depth subsurface layer overestimates fire-induced losses because the postfire layer starts and ends deeper in the profile, where organic matter and N are lower (H).
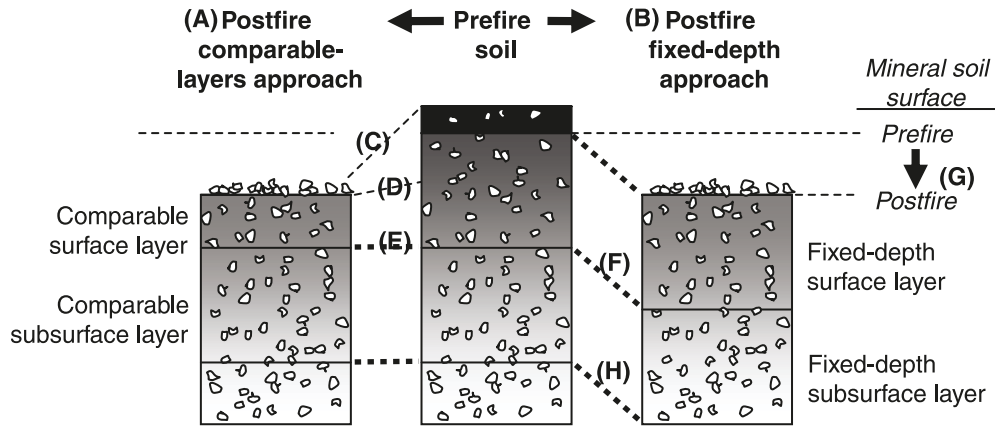

Table 3. Fire effects suggesting variable fire intensity on seven Long-Term Ecosystem Productivity experimental treatments burned in the 2002 Biscuit complex wildfire.

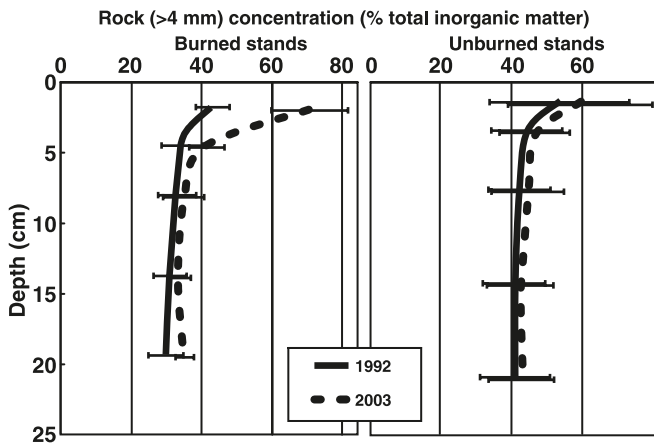| LTEP treatments		Fire effects			
Prefire stand structure	Added woody debris	Area burned at >700 °C (%)	Fine woody fuels consumed (Mg·ha^{-1})*	Coarse woody fuels consumed (Mg·ha^{-1})†	Tree mortality (%)
Medium-intensity burn					
Mature forest, control	Low	62	12	6	77
High-intensity burn					
Mature forest, thinned	Low	88	18	7	91
	High	100	24	12	100
Young forest, Douglas-fir	Low	87	34	9	100
	High	87	33	28	100
Young forest, mixed species	Low	94	26	8	100
	High	94	34	35	100

*Fine fuels are 0–7.6 cm.
†Coarse fuels are >7.6 cm.

generally smaller than those in the burned soils (Fig. 7 right side), although some were significant ($p < 0.05$) — raising questions about whether burned soils underwent changes before the Biscuit fire in 2002. Gains of soil N were partly attributable to residue left on the ground after harvesting, especially on high-debris treatments, or after vegetation control. Soil N additions from leguminous herbs, other known N$_2$-fixers like ceanothus, and precipitation appear to have been small. Across the unburned plots, in the years before the fire, cover of known N$_2$-fixing species was 0.00% (1992), <0.01% (1998), and 0.18% (2000). Across the burned plots, in the years before the fire, cover of known N$_2$-fixing species was 0.00% (1992), <0.01% (1998), and 0.30% (2000). Precipitation and fog-interception inputs would have been less than 2 kg N·ha^{-1}·year^{-1} (Bormann et al. 1989).

When the background changes estimated from the unburned plot were accounted for in the burned plots, the esti-

mates of fire-related soil N loss were substantially increased (Table 4, method 2). Losses in the upper layers increased by about 100 kg N·ha^{-1} and total losses rose by 170 kg N·ha^{-1}. Considering background changes implies that prior to the fire, the burned plots underwent changes similar to those we observed on the unburned plots.

We also calculated soil C and N change using a more traditional analysis method based on fixed depths before and after fire (Table 4, third method); no significant background changes were seen with this approach. Estimated soil C and N losses shifted radically with depth relative to corresponding comparable layers. Fixed-depth C losses were lower in the top layer (about 50%) but much greater (400%) in the two deepest layers combined than the losses in the equivalent comparable layers. The fixed-depth estimate of N loss was lower for the top layer (25%), but much higher for the two deepest layers combined (600%) than the estimates obtained for the corresponding comparable layers. Although

Fig. 5. Curves of changes in rock (>4 mm) concentrations with depth in six stands burned with high-intensity fire (left) and seven unburned stands (right). The results are based on comparable layers before accounting for fire-induced fine-soil loss (with 95% confidence intervals at layer means).

Fig. 6. The mass of comparable-layer soil components and mean layer thickness before and after high-intensity fire. The missing soil was calculated by using the postfire surface rocks as a standard and applying ratios of amount of rocks to amount of <4 mm inorganic matter. The comparable layers are based on fixed masses of inorganic matter (100 or 400 Mg·ha^{-1}) as the internal standard.

fixed-depth changes in the summed profile — 21 ± 3.5 Mg·ha^{-1} for soil C and 517 ± 197 kg·ha^{-1} for soil N — are surprisingly similar to comparable-layer sums, this is not because the fixed-depth approach lacked biases. The fixed-depth calculation shifts losses downward because the 1992 and 2003 layers represent different strata. The comparable-layer calculation avoids this shift by allowing variable thickness, but constant <4 mm inorganic mass (Fig. 5). The lower upper-layer losses in the fixed-depth analysis are partly compensated for by greater losses in the deeper sampled layer, but only if sufficient depth is sampled. When the fixed-depth analysis was used only for surface layers, major underestimates of soil C and N losses occurred.

Losses of soil C and N appear to be related to the range of fire intensity. When the plot that burned at moderate intensity (with 62% of the area burned >700 °C; Table 3) is considered along with stands that burned at high intensity (Fig. 8), soil C change is negatively related to fire intensity:

$$[1] \qquad \text{C change (Mg} \cdot \text{ha}^{-1}) = 0.88 - 0.030 \times \text{Area\%}$$
$$r^2 = 0.804, p < 0.01$$

as is soil N change:

$$[2] \qquad \text{N change (kg} \cdot \text{ha}^{-1}) = 242 - 8.65 \times \text{Area\%}$$
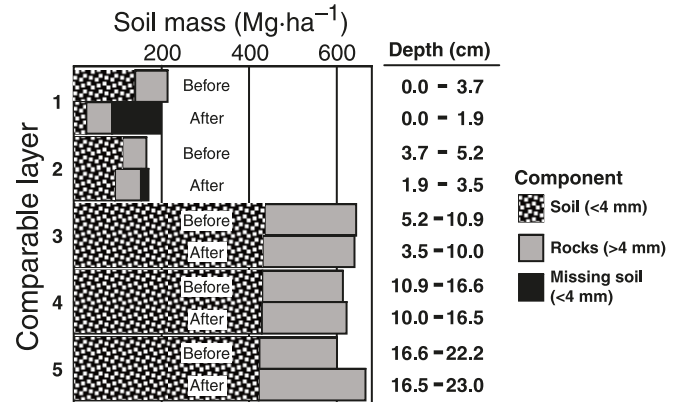$$r^2 = 0.658, p < 0.03$$

Analysis of other LTEP plots, most of which burned at low to moderate intensity, may help refine these relationships.

Discussion

Comparison to other studies

Our estimated loss of 23 Mg C·ha^{-1} from organic and minerals soil layers is higher than most previous estimates. Our losses of 500 to 700 kg N·ha^{-1} fell in the upper range of reported values. Comparing our results with others is challenging, however, given the variety of assumptions, sampling methods and depths, and analyses used. Uncertainties with these studies come from multiple sources and cloud

our knowledge of the effects of intense fire on soils. Because of the lack of opportunities to directly measure soil changes before and after intense wildfire, researchers have had to rely on estimates obtained in retrospective studies or extrapolated from laboratory and lower-intensity prescribed-fire studies. Reviews of these studies suggest highly variable soil C and N changes after wildfire — burned areas lost 6 to 14 Mg C·ha^{-1} and <10 to 855 kg N·ha^{-1} compared with paired unburned areas (Grier 1975; Dyrness et al. 1989; Baird et al. 1999). Other retrospective studies based on concentration data alone suggest an even wider range of effects (Prieto-Fernandez et al. 1993; Fernandez et al. 1997). Perhaps the best local estimate comes from a recent retrospective study of the entire Biscuit fire (Campbell et al. 2007). The authors of that study estimated losses of 8 to 15 Mg C·ha^{-1} from litter, duff, and upper mineral soil in the portion of the Biscuit fire that most resembled our plots based on vegetation damage classes (fire severity as determined by remote sensing; Environmental Impact Statement 2004).

These discrepancies could have resulted from a higher intensity of the Biscuit fire, although temperatures are rarely known in wildfires. We cannot rule out a bias in retrospective studies because they assume that unburned areas can be used to represent the preburn conditions (Baird et al. 1999). Inherent differences between burned and unburned areas with respect to moisture, site conditions, and burn history can influence soil properties, as has been demonstrated for part of the Biscuit fire (Thompson et al. 2007). Different site histories confound interpretations and may lead to incorrect conclusions about soil dynamics (Yanai et al. 2003).

In the only other forest wildfire study with pre- and postfire soil sampling, Murphy et al. (2006) and Johnson et al. (2007) evaluated soil change resulting from the Gondola fire in a Sierra-Nevada pine forest and estimated postfire erosion with off-site measurements (Carroll et al. 2007). Their estimates of combined O horizon and wood C loss (10 Mg C·ha^{-1}) compare well with our estimated O-horizon loss of 9 Mg C·ha^{-1}. In contrast, they did not observe a decrease in mineral soil C mass, as determined by a fixed-depth approach to 100 cm depth, whereas we observed a C

Fig. 7. Mean changes in soil C and N from 1992 to 2003, in relation to comparable-layer or prefire depth of sampling. The values for the six stands intensely burned in the 2002 Biscuit fire are on the left; the values for the seven stands not affected by the fire are on the right.

Change in soil C (Mg· ha⁻¹)

Change in soil N (kg· ha⁻¹)

* Significant at α = 0.05
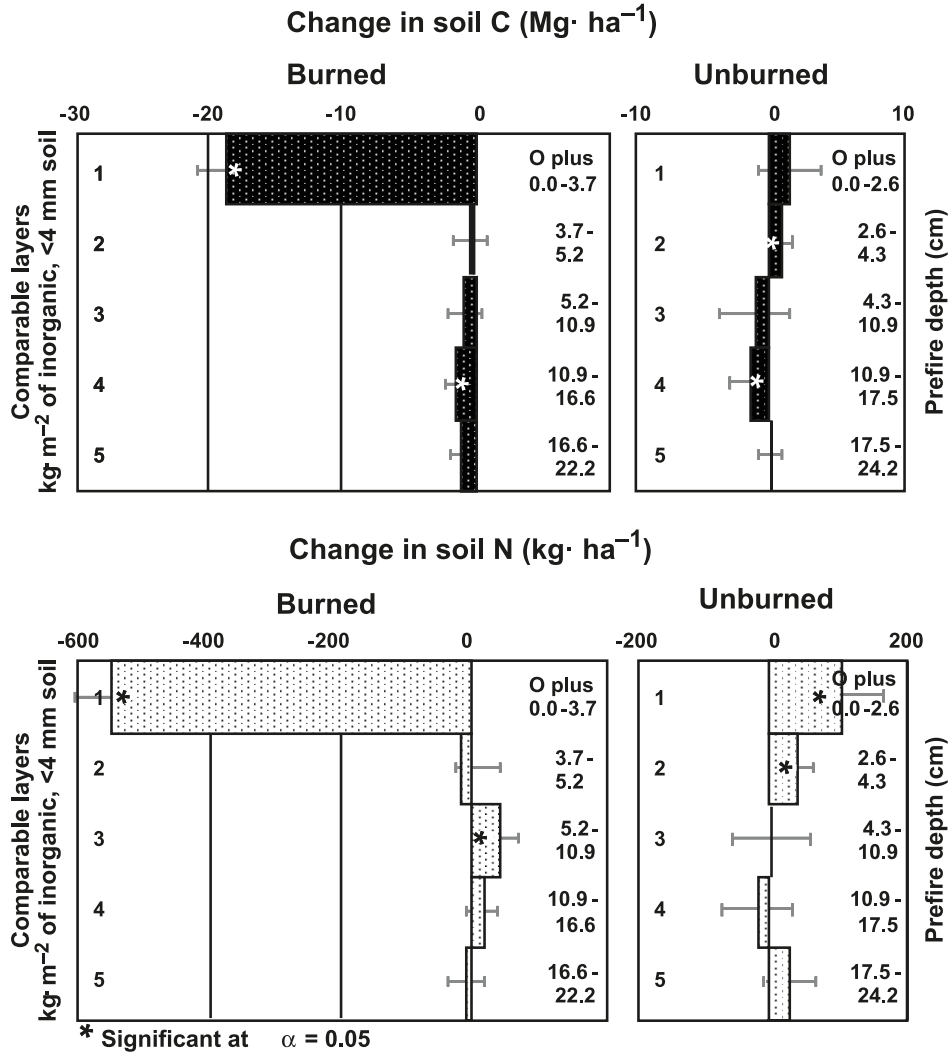
loss from mineral soil of 14 Mg C·ha⁻¹ regardless of measurement approach. The combined N loss from the O horizon and wood losses in the Gondola fire (142 kg N·ha⁻¹) were somewhat lower than our estimates of O-horizon loss (226 kg N·ha⁻¹). The Gondola fire had no significant N loss from the mineral soil, as determined by the fixed-depth approach, but had a N loss of 25 to 110 kg N·ha⁻¹ from soil erosion soon after the fire (Murphy et al. 2006; Johnson et al. 2007). In contrast, we observed 290 to 370 kg N·ha⁻¹ loss of mineral soil N associated with the Biscuit fire. We speculate that the greater losses associated with the Biscuit fire are due to the higher intensity of the fire, which generated more combustion and convection losses.

Evaluating soil changes in any fire study, even with before and after samples, is fraught with additional uncertainties. Bias associated with fixed-depth analyses deserves special attention because this method was used in nearly all previous wildfire studies of soil effects. The fixed-depth studies that only examined changes in the upper mineral soil are likely to have substantially underestimated soil C and N loss (Table 4, Fig. 4).

Ignoring the background changes in unburned plots, i.e, the changes that occurred from the first sampling in 1992 to the second sampling in 2003, affects estimates as well (Table 4). In the unburned stand, significant change in soil N was observed in just 11 years, raising the possibility that similar change was occurring in the stands that eventually burned. Also, our estimates of soil C and N loss would have increased if we could have determined the extent to which high fire temperatures fractured rocks into <4 mm particles (Blackwelder 1927).

Several mechanisms may explain the loose surface rocks after fire (Fig. 3): postfire erosion of fines, small-scale re-sorting of soil constituents, and atmospheric losses during the fire. Most of the soil organic matter in the O horizon was burned, and the products of combustion, including CO_2 and volatilized nutrients, were exported as gas or smoke particles, leaving behind over seven times more rocks above the mineral soil surface. Losses of fine mineral soil from upper mineral soil layers are usually attributed to postfire water-driven erosion, and our erosion-box estimates support this explanation to a point. Water-driven erosion for the 2003–2004 water year on burned soils in erosion boxes, placed

Table 4. Mean changes (±95% confidence intervals) in fine-soil (<4 mm) C and N between 1992 and 2003 of the six Long-Term Ecosystem Productivity plots burned at high intensity in the 2002 Biscuit fire.

(A) Mean changes.

			Method		
			Comparable layers		
Layer	Mass of inorganic matter (Mg·ha^{-1})	Mean prefire depth (cm)	1. Without back-ground change	2. With back-ground change	3. Fixed depth*
C (Mg·ha^{-1})					
1	0–100	O + 0–3.7	–18.9±2.2	–20.2±3.0	–10.5±2.1
2	100–200	3.7–5.2	–0.6 ns[†]	–1.6±1.4	–1.1±0.2
3	200–600	5.2–10.9	–0.9 ns	0.3 ns	–4.2±0.9
4	600–1000	10.9–16.6	–1.4±0.9	–0.1 ns	–3.5±0.7
5	1000–1400	16.6–22.2	–1.1 ns	–1.1±0.9	–1.7±1.1
	Sum	O + 0–22.2	–22.9±3.0	–22.7±5.4	–21.0±3.5
N (kg·ha^{-1})					
1	0–100	O + 0–3.7	–547±79	–653±115	–166±67
2	100–200	3.7–5.2	–18 ns	–60 ns	–32±16
3	200–600	5.2–10.9	40±32	31 ns	–123 ns
4	600–1000	10.9–16.6	15 ns	28 ns	–112±53
5	1000–1400	16.6–22.2	–10 ns	–40 ns	–84±59
	Sum	O + 0–22.2	–520±94	–693±187	–517±197

(B) Calculation differences.

	1. Without back-ground change	2. With back-ground change	3. Fixed depth*
Accounts for changing soil depth?	Yes	Yes	No
Accounts for export of fine-soil mass?	Yes	Yes	No
Accounts for background soil changes, not caused by wildfire?	No	Yes	Yes

*There were no significant ($p > 0.05$) background changes, so no background adjustments were made in the fixed-depth analysis.

[†]ns, nonsignificant ($p > 0.05$).

across a range of slopes, averaged 57 m³·ha^{-1} compared with 0 m³·ha^{-1} on unburned soils. Below a 15 ha catchment with two burned LTEP stands, only a tiny fraction of the estimated 850 m³ of moving sediment (extrapolated from the erosion-box data) appeared in the ditches along logging roads. The export of the 127 Mg·ha^{-1} of missing soil, estimated by the difference between pre- and post-fire soil sampling (Fig. 6) and extrapolated to this catchment, would be about 1900 m³ (assuming a sediment bulk density of 1 Mg·m^{-3}). The complex microtopography — partly created by windthrows, downed logs, tree trunks, and needles cast off after the fire — appeared to capture much of the moving soil. The fire also created soil voids — where decayed stumps and roots burned deeply into the soil — that filled over time. This vertical sorting mechanism does not appear to be responsible for the increase in the amount of surface rocks because we observed no corresponding drop in the rock concentration at deeper soil layers (Fig. 5).

An intriguing alternative explanation for most of the missing fine soil is transport via the massive smoke plume. The elevation of the smoke column and the spread of the plume provide a plausible convective erosion process for off-site transport of substantial material. Large plumes of smoke, some more than 1500 km long, were visible most days during the months of the fire from the NASA MODIS satellite (Fig. 9). Fine soil particles have been detected in smoke (Palmer 1981; Samsonov et al. 2005), and wind speeds near the soil surface — driven by extremely strong vortices resulting from fire-driven atmospheric convection (Palmer 1981; Banta et al. 1992) — can carry smoke to the lower stratosphere (Trentmann et al. 2006). The possibility that a substantial mass of fine particles, including mineral soil, was transported high into the atmosphere raises questions about the effects of intense fire on radiation interception, water-droplet nuclei, and off-site terrestrial and ocean fertilization.

Implications of intense-fire-induced soil changes on climate, forest productivity, and management decisions

Many previous estimates of fire contributions to greenhouse gasses (e.g., Crutzen and Andreae 1990) are based on biomass combustion alone and fail to consider mineral soil losses. Although Campbell et al. (2007) considered soil C losses from the entire Biscuit fire, a concern about the lack of prefire soils data in their estimates is expressed in the range of their C-emission estimates, 0.7 to 1.2 Tg C for the portion of the fire with vegetation damage classes similar to those of our plots. If we extrapolate our results to this area of the Biscuit fire, the resulting soil C loss would be about 1.6 Tg and N loss about 45 Gg. Mineral soil (<4 mm) particulate losses (Fig. 6), extrapolated to the same area, sum to nearly 9 Tg.

Fig. 8. Soil C (top) and N (bottom) changes in seven individual Long-Term Ecosystem Productivity (LTEP) experimental plots displayed from least to most intensively burned (left side). Corresponding unburned LTEP plots are on the right side.
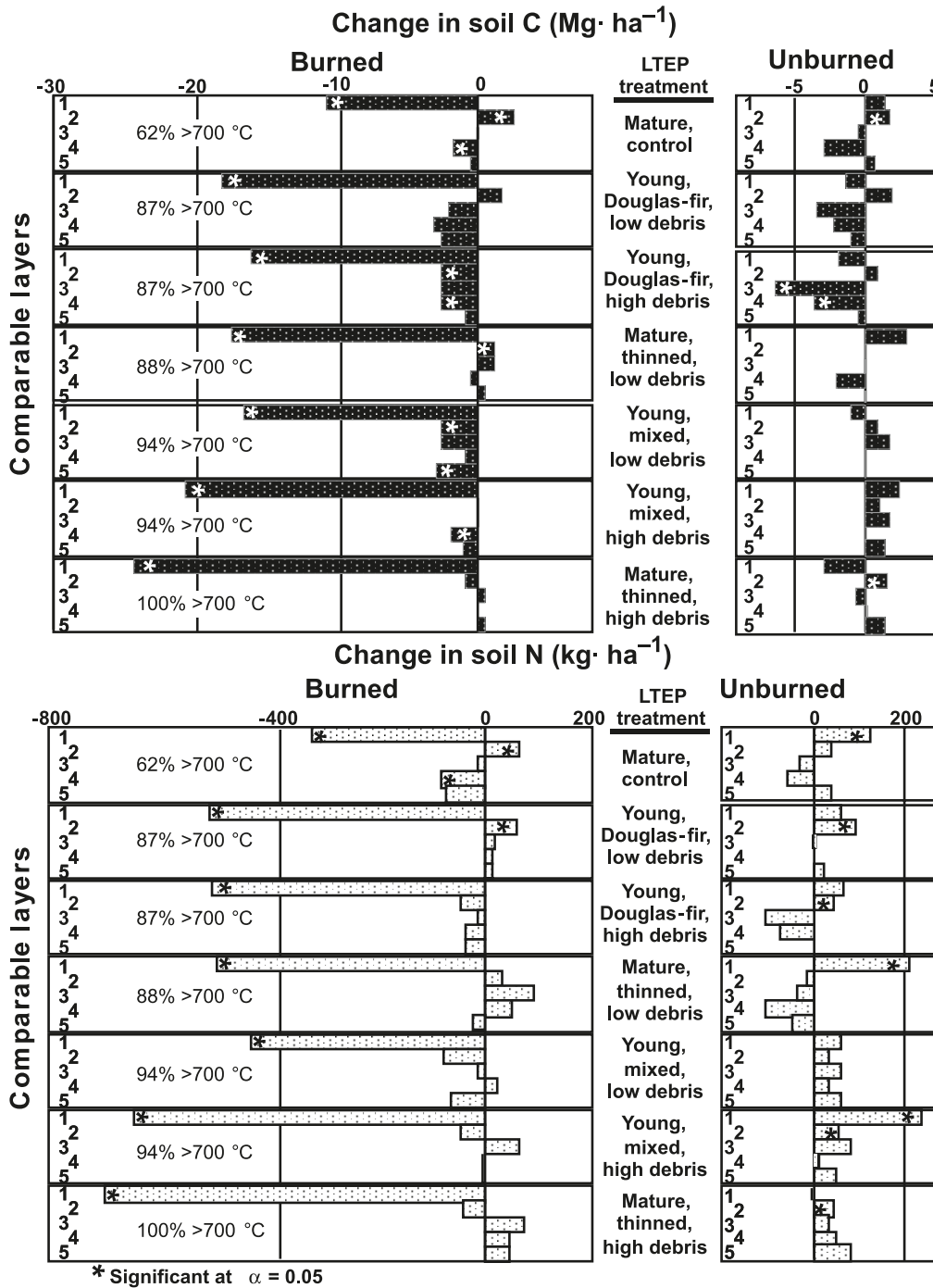

Our soil C loss is greater than the high end of the estimates of Campbell et al. (2007); this discrepancy may be related to bias from their unburned controls or to our small sample of the Biscuit fire area. To the extent that our estimates might apply more broadly to other intense fires, climate models may need to be recalibrated to account for effects of intense fire, including fire-induced greenhouse gases and emissions of particulates.

The intensity of wildfires and magnitude of losses of fine

soils and soil C and N have additional implications for soil fertility and subsequent rates of plant production and C sequestration. Soil C losses lead to increased bulk density and reduced soil water-holding capacity, cation-exchange capacity, and sources of energy for microbial communities. To the extent that soil N, soil C, and soil structure control productivity, these changes should result in major declines that will last as long as it takes to return to prefire conditions. The growth of many forests in western Oregon is typically limited by low N availability (Edmonds and Hsiang

Fig. 9. Smoke plume traveling southwest of the Biscuit fire and several smaller fires on 29 July 2002 (MODIS image). For scale, the Channel Islands offshore from Los Angeles, California, are seen in the lower right.
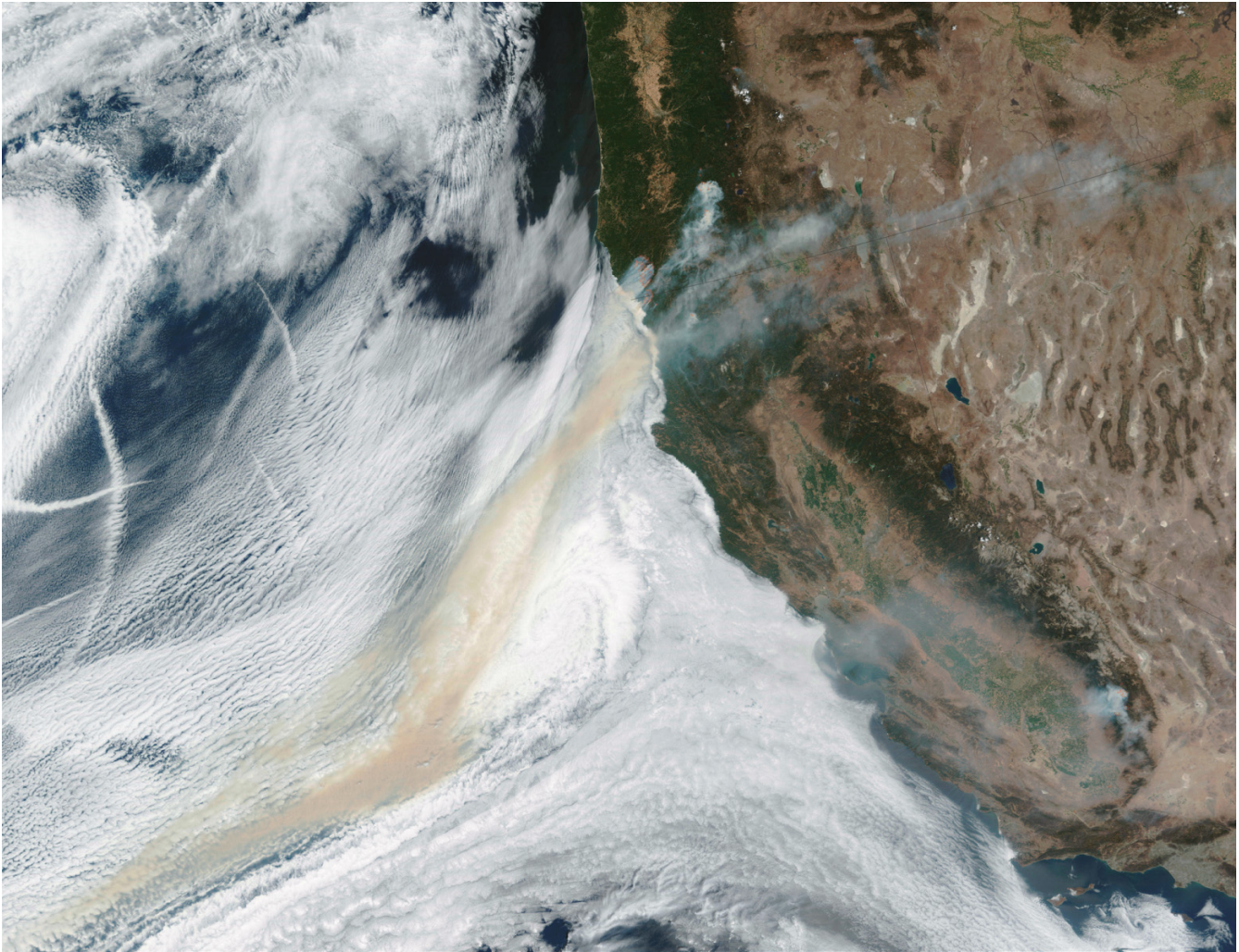
1987; Stegemoeller and Chappell 1990). The high N losses we observed could substantially exacerbate this N limitation.

Any potential loss in productivity is relevant to the US National Forest Management Act of 1976, where the Secretary of Agriculture is required, "through research and continuous monitoring, to ensure that management systems will not produce substantial and permanent impairment of the productivity of the land". The US Endangered Species Act of 1973 is also relevant to the management of high-intensity fires, for example, in the case of the northern spotted owl that nests primarily in stands of large trees averaging only 32 large trees·ha^{-1} (Hershey et al. 1998). When soils can no longer produce such trees, the area of suitable habitat that could redevelop after fire is also lessened.

However, we express uncertainty about whether current knowledge of ecosystem processes can allow reliable predictions of declining productivity, CO_2 uptake, or habitat potential in actuality. Further work is needed to examine the effects of fire on changes in N_2 fixation, N availability, losses of other nutrients (likely at the temperatures experienced), and possible increases in the weatherability of rock particles by rock fragmentation and changes in clay mineralogy and particle surface coatings. Most important is the need to follow soil and vegetation development and productivity for at least several decades after the fire.

Much of the recent debate has centered on the effects of postwildfire management on tree regeneration, wildlife habitat, and future fire risk (Donato et al. 2006; Newton et al. 2006; Shatford et al. 2007; Thompson et al. 2007). In light of the first direct evidence of major effects of intense wildfire on soils — based on extensive and detailed pre- and post-fire soil sampling — we think that soil changes, especially the potential loss of soil productivity and greenhouse-gas additions resulting from intense wildfire, deserve more consideration in this debate. In forests likely to be affected by future intense fire, preemptive reduction of intense-fire risks can be seen as a way to reduce losses of long-term productivity and lower additions of greenhouse gases. Preemptive strategies may include reducing fuels within stands but also improving fire-attack planning and preparation and

changing the distribution of fuels across the landscape to reduce the size of future fires. Practices can include thinning and removing or redistributing residues and underburning. In forests already affected by intense fire, amelioration to increase C sequestration, tree growth, and eventually late-successional habitat should be strongly considered. Amelioration practices might include seeding or planting N_2-fixing and other plants, fertilizing, and managing vegetation and fuels through time. To the extent that receipts from pre- and post-wildfire logging are the only means of paying for these practices, such logging should be balanced against other management objectives and concerns. Harvesting before and after fire to generate revenue, if done improperly, has the potential to harm soils, but this outcome needs to be weighed against the outcomes resulting from increased high-intensity fire and from not ameliorating after soils have been burned intensely.

Acknowledgments

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