Managing Ponderosa Pine Forests in Central Oregon: Who Will Speak for the Soil?[1]

Matt D. Busse[2] and Gregg M. Riegel[3]

Abstract

The soils of the central Oregon pumice plateau are relatively young and infertile, yet support an array of plant diversity and growth in the region's pine forests. Whether these coarse-textured, pumice and ash soils are resilient to forest disturbance is not well understood. We present results from a long-term experiment that examined changes in soil quality in response to combinations of thinning and repeated prescribed fire. Soil quality was generally unaffected in fifteen years following pre-commercial thinning. The soils were also resilient to fire with the exception of the loss of nearly 25 percent of the ecosystem's total nitrogen due to burning in 1991 and 2002. Natural replacement of N by N-fixing shrubs such as snowbrush ceanothus (*Ceanothus velutinus* Dougl. ex Hook.) and antelope bitterbrush (*Purshia tridentata* (Pursh.) DC.) is an important means to offset N losses, along with careful planning of burn prescriptions to limit forest floor N consumption.

Introduction

How can the
non-farmer   jet-set   bureaucrats

Speak for the green of the leaf? Speak for the soil?

Gary Synder, 1974

Soil is cursed. This complex medium that covers much of the surface of our planet and helps ensure the survival of the human race suffers from an unavoidable fact: it is terminally boring. It just sits there with little aesthetic value, unable to attract attention by changing colors, growing, or communicating. Adding insult to injury, soil is known synonymously as dirt, the common irritant found under fingernails and on pant legs. This is not to suggest that the value of soil is lost upon society, however. Far from it. Both historic and contemporary examples of degraded soils and the human hardships that often follow (Lal and others 2004) provide a constant reminder that soil is (1) an indispensable natural resource, (2) generally nonrenewable in our lifetime, and (3) susceptible to disturbance and mismanagement. Thus, preventing soil degradation is a common-sense requirement for managing forest lands, regardless of the desired social or economic product, a requirement of both legal and ethical proportions for federal land managers as directed by the United States Congress.

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[2] USDA Forest Service, Pacific Southwest Research Station, 3644 Avtech Parkway, Redding CA 96002. mbusse@fs.fed.us

[3]USDA Forest Service, Pacific Northwest Region, Area Ecology Program, Bend, OR, 97701. griegel@fs.fed.us

Have the soils of central Oregon avoided productivity losses during the past century of unprecedented changes in forestry practices (fire suppression, railroad logging, and aggressive fuel reduction)? Or, are there clear examples of irreversible soil damage? Unfortunately, no archived soil collection exists from the pre-fire suppression era, and, thus, a true evaluation of soil changes is not possible. As a substitute, we present results from a forest manipulation experiment (the Black Bark Study) that tracks recent trends in soil quality associated with forest management practices. Changes in soil quality and plant growth in 15 years following a combination of prescribed fire, thinning, and nutrient additions are explored. In particular, we discuss the potential loss of soil quality associated with repeated prescribed burning.

Soil Quality

Soil quality is a fairly new term to the discipline of soil science. Thus, a brief discussion of its origin, definition, and application to forestry is appropriate before examining possible implications for central Oregon soils.

Coined by agricultural soil scientists in the early 1990s (Doran and Parkin 1994), soil quality is a holistic, feel-good concept derived from its cousin and predecessor, soil productivity. Consider the definition of these two terms from the Soil Science Society of America (http://www.soils.org/sssagloss):

Soil productivity: the capacity of a soil to produce a certain yield of crops or other plants with a specified system of management.

Soil quality: the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health.

The language used in defining soil quality is key; terms like function, ecosystem, sustain, environmental quality, and health are pleasing to our 21st-century ears. As a result, soil quality is politically correct, in vogue. Soil productivity, although often used synonymously with soil quality, is a plant-centric term perhaps more fitting for agricultural soils, where farmers can track annual crop yields and examine short-term (5 to 10 year) productivity trends. Translating the concept of soil productivity to forestry is troublesome, however, given the dynamics of stand development, changing management systems within a forest rotation, and the Herculean effort required to measure total vegetative growth. Equally telling, the term productivity is stigmatized in many public circles with respect to managing public lands.

Soil quality is admittedly in its infancy as a practicable and reliable indicator of the soil condition. Lacking is a scientifically-accepted set of indices that reflect disturbance-related changes in soil properties. Proposed indices encompass a combination of soil physical (e.g. water infiltration, erosion, soil strength, aeration), chemical (e.g. pH, organic matter, total C, total N, salinity, cation exchange capacity), and biological (e.g. microbial biomass, respiration, N mineralization) properties (Doran and Parkin 1994). Difficulty in selecting indices that are both insightful and affordable is a stumbling block and a common theme discussed in several review articles (Burger and Kelting 1999; Page-Dumroese and others 2000; Powers and others 1998). As Page-Dumroese and others (2000) state, “efforts to construct definitive soil quality/sustainability standards and guidelines are still in their infancy..... because of the diversity of soil properties to be measured, appraisal techniques, and soil uses.”

Regardless of any technical ambiguities, soil quality has gained strong support among land resource managers and the scientific community as a conceptual and communicative tool. If told there is a substantial decline in soil quality due to forestry practices, resource managers will likely listen (even if they don’t know exactly what it means). From there, the door is open for professional soil scientists to identify the soil properties that fail to meet forest guidelines, and to propose ameliorative treatments when needed.

Soils of Central Oregon

Central Oregon offers a beautiful landscape, starting with the high mountain peaks of the Cascades to the west, and traversing across mixed-conifer forests, to ponderosa pine (*Pinus ponderosa* P. & C. Lawson) and lodgepole pine (*Pinus contorta* Dougl. ex Loud.) forests at lower elevations, and finally to the high desert plateau to the east. The soils are diverse and reflect a geomorphically-complex landscape, which includes 5 of the 8 geomorphic provinces found in Oregon. Soils supporting ponderosa pine growth, in contrast, are fairly uniform, and are primarily derived from wind deposits of pumice and ash from the eruption of Mt. Mazama (Crater Lake). Smaller volcanic activity scattered across the plateau has added subordinate parent material diversity.

In general, the soils are poorly developed Andisols with loamy sand and sandy loam textures. The benchmark soil (Lapine series), for example, has a 6 to 10 cm (2.4 to 4 inch) surface horizon (A horizon) above a 12 to 24 cm (5 to 10 inch) transition horizon (AC) and an undeveloped C horizon (see http://www.nrcs.usda.gov for details). The depth to the buried soil (itself a poorly developed, infertile soil) is 60 to 100 cm (24 to 39 inch), and generally declines with increasing distance from Crater Lake. Clay contents are 8 percent or less throughout the profile. Other dominant soil characteristics include (1) low fertility, and (2) unusually high porosity and water-holding capacity for a sandy soil. Poor fertility is a function both of the young parent material and the semi-arid conditions on the pumice plateau (30 to 75 cm annual precipitation), which limits plant growth and, ultimately, soil organic matter input. The soils have impressively low N concentrations beneath the A horizon (Dyrness and Youngberg 1966; Youngberg and Dyrness 1963).

The Black Bark Study

The Black Bark study began in the 1980s as a collaborative project between the Deschutes National Forest and the Pacific Northwest Research Station (Bend Silviculture Laboratory) to better understand the ecology of central Oregon ponderosa pine forests. With a wide array of silvicultural treatments replicated across a productivity gradient of pine forests, the study provides an interesting insight to the resilience of the area’s soils.

The study concept was prompted by a large-scale mountain pine beetle infestation in the lodgepole pine forests south of Bend, OR. Whether the insect population would migrate into ponderosa pine stands was unclear and, consequently, of immense concern to the Deschutes National Forest. Prior evidence suggested that thinning of overstocked pine stands was the most successful means to prevent further insect damage (Mitchell and Preisler 1991). As a result, the Deschutes National Forest adopted an aggressive thinning program in its second-growth ponderosa pine stands. The targeted area included approximately 100,000 hectares (247,000 acres) of "black bark" ponderosa pine, considered at the time the most productive second-growth pine forests in central Oregon.

The management objective of the study (as conceived by Pat Cochran, retired PNW Soil Scientist, Bill Hopkins, retired Area Ecologist, and Don Peterson, retired Timber Staff Officer) was to evaluate ecological changes following partial overstory harvesting, and thus contribute to the monitoring needs of the Forest's thinning program. From a scientific standpoint, the objective was expanded greatly to evaluate soil and plant responses to manipulation of site organic matter. Sixteen treatments were selected which created a gradient of surface organic matter retention following combinations of thinning and prescribed fire (*table 1*).

Three study sites were selected along a west-to-east transect on the Bend-Fort Rock District. Swede Ridge, the most productive site, is proximal to the Cascade crest, and thus receives considerably more annual precipitation than the other sites; Sugar Cast, the middle site, is near the heart of the pumice plateau on the Lava Cast forest near Sunriver, OR; and East Fort Rock, the poorest site, is on the eastern edge of the ponderosa pine forests, within 5 km of the desert fringe. Additional site characteristics are presented in *table 2*. All treatments were replicated at each site (block) on 0.4 ha plots in a randomized complete block experimental design.

Thinning to a target basal area of 13.7 m^2 ha^{-1} (60 ft^2 ac^{-1}) was completed in 1989, and the prescribed burns were conducted in spring 1991 and repeated in spring 2002. The burns were generally low to moderate intensity (0.5 to 1.2 m (1.6 to 4 ft) average flame lengths) with 45 to 50 percent duff reduction. Details of the 1991 burns, including fuel loading, consumption, and soil heating, are presented by Shea (1993). Fertilizer (224 kg ha^{-1} N; 112 kg ha^{-1} P; 37 kg ha^{-1} S, or the equivalent of 200 lb ac^{-1} N; 100 lb ac^{-1} P; 33 lb ac^{-1} S) was applied in the fall of 1991 and again in 1996.

Table 1—*Treatment design of the Black Bark study. Treatments are ranked generally from the lowest to highest level of organic matter retention.*

Treatment rank	Thinning method	Additional treatment
1	Whole-tree harvest	Fire
2	Whole-tree harvest	Fire + fertilize
3	Whole-tree harvest	Fertilize
4	Whole-tree harvest	None
5	Bole-only harvest	Fire
6	Bole-only harvest	Fire + fertilize
7	Bole-only harvest	Fertilize
8	Bole-only harvest	None
9	Thin, leave trees on site	Fire
10	Thin, leave trees on site	Fire + fertilize
11	Thin, leave trees on site	Fertilize
12	Thin, leave trees on site	None
13	No thin	Fire
14	No thin	Fire + fertilize
15	No thin	Fertilize
16	No thin	None

Table 2—*Characteristics of the three Black Bark study sites prior to treatment in 1988.*

Site characteristic	Swede Ridge	Sugar Cast	East Fort Rock
Relative site quality	High	Medium	Low
Site index (m; Barrett 1978)	35	31	25
Age (yr)	40	49	56
Density (trees ha^{-1})	783	707	495
Basal area (m^2 ha^{-1})	39	33	28
Dominant understory vegetation	Snowbrush ceanothus, antelope bitterbrush	Antelope bitterbrush, greenleaf manzanita	Antelope bitterbrush, greenleaf manzanita
Elevation (m)	1,520	1,398	1,554
Precipitation (cm)	65	50	38
Mean July temperature (^oC)	15	18	18

Soil nutrients, wood decay, tree height and volume, diameter at breast height, and fuel loading were measured on a 5-year cycle. Herbaceous production was measured annually by clipping all vegetation within ten 0.25 m^2 (2.7 ft^2) frames in each plot. Shrub cover was determined every third year by measuring the canopy size of all plants within three 100 m^2 (1076 ft^2) belt transects per plot. Percent shrub cover was converted to a dry weight basis using site-specific biomass equations. Additional soil quality measurements (microbial biomass, respiration, surface CO_2 efflux, C utilization, litter decay, phospholipid fatty acid content) were made periodically between 1989 and 2005.

Results

No trends in chemical or biological indices of soil quality were detected in the initial 15 years following thinning. In contrast, substantial increases in tree vigor and production of wildlife browse were evident. The periodic annual increment for diameter growth increased 60 percent, while shrub cover increased more than 200 percent due to thinning (*fig. 1*). Antelope bitterbrush (*Purshia tridentata* (Pursh.) DC.; hereafter bitterbrush), in particular, responded to thinning: average cover was 7 percent on unthinned plots compared to 25 percent on thinned plots by 2002. Maximum bitterbrush cover on thinned plots reached 50 percent. Herbaceous production also increased due to thinning, although this result is tempered by the fact that biomass production was extremely small regardless of treatment (14 kg ha^{-1} for thinned plots versus 4 kg ha^{-1} for unthinned plots). Plant diversity (number of species) was low regardless of treatment, and declined slightly due to thinning. Whether thinning has improved forest health and resistance to insect attack can only be assumed, since no insect-related mortality has been identified on thinned or unthinned plots.

Differences in soil quality or vegetation growth due to the method of thinning (whole-tree harvest; bole-only harvest; thin, leave trees on site) were slight. Thus, the gradient of organic matter retention associated with pre-commercial thinning appears to have little short-term effect in these pine ecosystems. Whether this will change with time as the larger organics decompose and their products become incorporated in the soil is unclear.

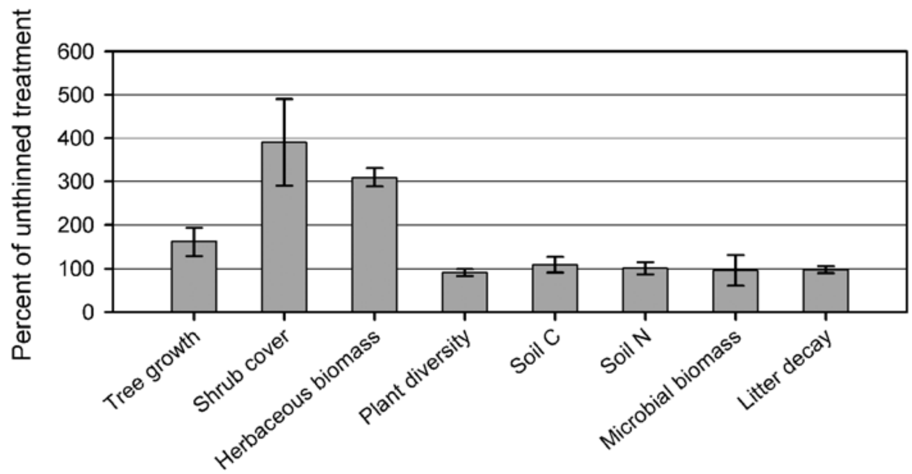

Figure 1—Thinning effects in central Oregon pine stands. Bars represent mean percentage plus standard error (n=3) of Whole-tree harvest (treatment 4) plots relative to No thin (treatment 16) controls (e.g. thinned and unthinned treatments are equivalent at 100 percent). Tree growth is the 10-year diameter increment. Soil C and N values include forest floor plus mineral soil to a depth of 60 cm.

The strongest ecosystem-level response to date has been to nutrient additions. Soil C and N, microbial biomass and activity, tree growth, and herbaceous production and diversity are among the numerous measures that increased substantially following fertilizer applications in 1991 and 1996 (*fig. 2*). Only shrub growth failed to respond positively, indicative of the poor competitive ability of shrubs relative to other vegetative lifeforms in acquiring and utilizing available nutrients. Our results not only support previous findings from central Oregon of soil nutrient limitations to tree growth (Cochran 1978; Cochran 1979; Youngberg and Dyrness 1965), but extend the results to include other important lifeforms and processes. Does this mean that the Deschutes National Forest should consider applying fertilizers in its second-growth pine forests? There is an important message from our finding, but it has little to do with the economic benefits of applying fertilizer. The results show soil-plant processes in central Oregon are not just controlled by moisture limitations; they are also nutrient limited. Forest practices that conserve (or even increase) total nutrient pools while meeting the primary ecological, social, or economic objectives should be emphasized.

The effects of prescribed fire have generally been benign to date. Strong declines in soil N content and shrub cover are the only exceptions. Soil N content was more than 20 percent lower on burn plots compared to controls as a direct result of consumption of forest floor organics (*fig. 3*). No changes in mineral soil N (total or available) were detected following either burn, nor were changes evident in mineral soil C, pH, extractable phosphorus, cation exchange capacity, C utilization or phospholipid fatty acid composition. Only an inconsistent decline in microbial biomass was found between 1991 and 2003 in the surface mineral soil of burned plots.

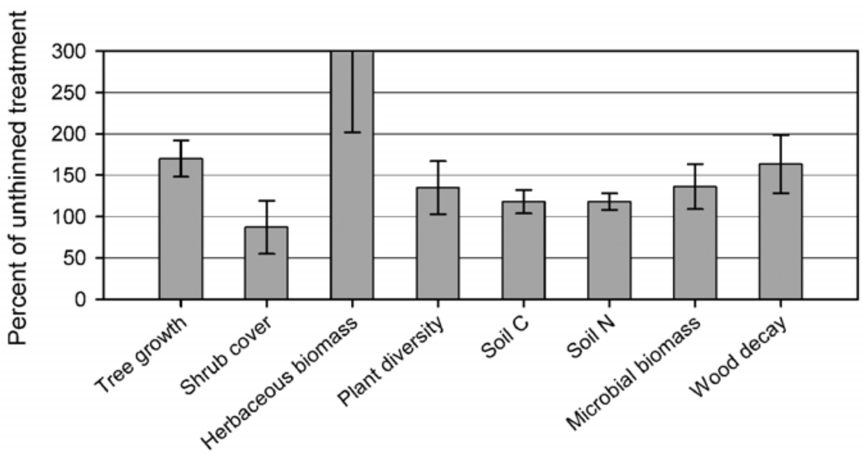

Figure 2—Nutrient limitations in central Oregon pine stands. Bars represent mean plus standard error (n=3) of fertilized plots (treatment 3) relative to unfertilized controls (treatment 4). Mean value for herbaceous biomass is 845 percent.

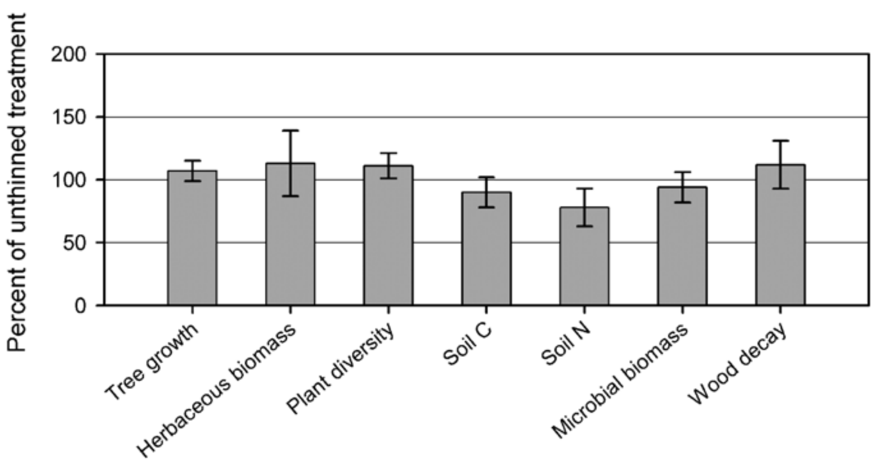

Figure 3—Fire effects in central Oregon pine stands. Bars represent mean values plus standard error (n=3) of burned plots (treatment 1) relative to unburned controls (treatment 4).

Bitterbrush cover was severely reduced after the initial burn, although post-fire seed germination led to partial recovery in the following eleven years. By 2001, bitterbrush cover on burned plots averaged 13 percent compared to 25 percent on unburned plots. After the second round of burns, however, bitterbrush was virtually eliminated from the stands, with little sign of basal stem sprouting or seed germination noted by the end of the second growing season.

Tree growth between 1991 and 2001 was slightly higher on burned plots compared to controls, which contradicts previous studies in central Oregon showing tree growth declines associated with burning (Busse and others 2000; Cochran and

Hopkins 1991; Landsberg 1993). We suspect the increase in tree growth was a result of a small pulse of available N following burning in combination with reduced understory competition for water and nutrients. It remains to be seen whether these 5 to 10-year changes in tree growth have long-term implications to stand growth.

Fire and Soil Nitrogen

Our planet and atmosphere are loaded with N. It's in the air, in the soil and ground water, and rising in exhaust emissions. Nitrogen is everywhere. So, there should be more than enough for every living organism to thrive, right? Then why is N generally considered the most limiting of all nutrients for plant growth on a global scale? And what role does fire play in altering plant-available N?

At the heart of the N conundrum is the fact that most organisms primarily assimilate inorganic N (ammonium and nitrate), which, in turn, comprises a miniscule fraction of the total N pool (Paul and Clark 1996). The great majority of soil N, for example, is found in organic complexes, and is released as inorganic forms in a slow bottleneck process by soil microorganisms. Nitrogen-fixing legumes and actinorhizal plants (e.g., snowbrush ceanothus (*Ceanothus velutinus* Dougl. ex Hook.; hereafter snowbrush) and bitterbrush) that convert atmospheric N directly into plant-available N are the main exceptions to the rule. Accentuating this problem is the fact that N is not distributed evenly in terrestrial ecosystems either at micro- or macro-scales. Dry environments, such as those found in central Oregon, are characteristically on the low end of the N distribution scale. Comparing the N content of forest ecosystems in central Oregon versus those in moist western Oregon provides a good example of this inequity. As much as three to ten times more total ecosystem N is found west of the Cascade crest than in central Oregon (*fig. 4*). Nitrogen conservation, therefore, is vital to the integrity of central Oregon forests. Any N disruptions, additions, or subtractions will have a much greater ecological effect in these poorly-buffered forests than in N "rich" forests west of the Cascade crest. As an example, a loss of 350 kg N ha^{-1} would result in a 15 percent reduction of total N in central Oregon, but only a 3 percent reduction of site N in western Oregon.

Fire plays a major role in determining total and available N in forests of the interior west by consuming organic matter, altering rates of soil N processes, and modifying post-fire plant community composition and growth. Total ecosystem N is reduced during burning primarily as a result of volatilization losses from forest floor material. Temperatures in the mineral soil, in contrast, rarely exceed the critical range for N volatilization of 300 to 500^{o}C (Hungerford and others 1991), and thus N loss from the mineral fraction is generally of little concern. The absolute quantity of N lost during burning is difficult to predict, and depends on factors such as pre-burn fuel loading, fuel moisture, fuel continuity, and fire severity. Estimates of N loss range from 20 to 400 kg ha^{-1} in central Oregon studies (*table 3*). On the positive side, fire typically results in an abundant release of plant-available N, as volatilization losses to the atmosphere are usually incomplete. This often-reported flush of inorganic N (Covington and Sackett 1986, 1992) can result in short-term increases in plant growth. Therefore, fire wields a double-edged sword: providing a short-term flush of plant-available N, while reducing the total ecosystem N pool.

An impressive quantity of forest floor N was lost during repeated burning in thinned and unthinned plots on the Black Bark study: the combined loss for the 1991

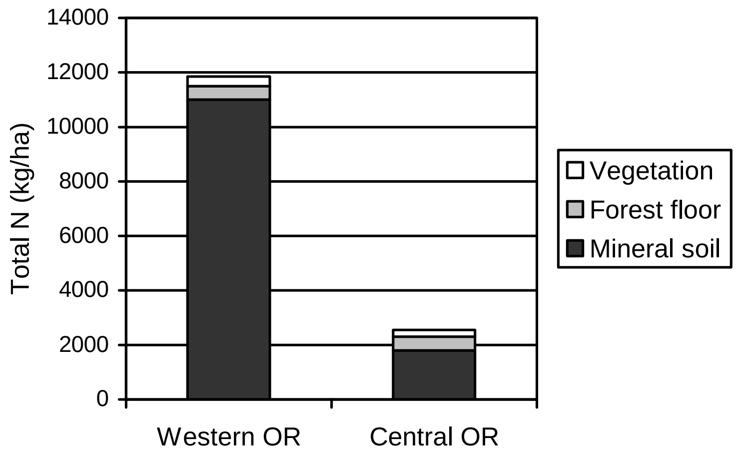

Figure 4—Comparison of total ecosystem N in a mesic-climate forest of western Oregon and a semi-arid ponderosa pine forest in central Oregon. Data for western Oregon is from R. Zasoski[1] and for central Oregon is from the Black Bark study[2].

Table 3—*Estimated N losses from prescribed fire studies conducted in central Oregon ponderosa pine stands.*

Source	**Fire intensity**	**N loss (kg ha^{-1})**
Landsberg (1993)	Severe	280
Shea (1993)	Moderate (1991)	366 to 397
Black Bark study	Moderate (2002)	126 to 183
Simon[1]	Low to moderate	100
Nissley and others (1980)	Low	20 to 260

[1] Data on file, Deschutes National Forest, Bend, OR

Table 4—*Forest floor nitrogen (kg ha^{-1}) before and after repeated prescribed fires in second-growth ponderosa pine stands in central Oregon. N=6 in 1991, n=3 in 2002.*

	Initial burns in 1991[1]			Repeated burns in 2002			
Treatment	Preburn	Postburn	N loss	Preburn	Postburn	N loss	Total N loss
Thinned	756	359	397	287	161	126	523
Unthinned	764	398	366	301	118	183	549

[1] Data for 1991 burns are from Shea (1994).

and 2002 burns was more than 500 kg ha^{-1} (*table 4*). Unfortunately, the post-fire flush of inorganic N was not quantified, although circumstantial evidence suggests that the amount released was inconsequential. Specifically, measurements taken 18 months after burning showed no differences in available soil N between burned and unburned plots. Also, herbaceous production did not increase following fire as might be expected with a flush of N, as was seen on fertilized plots.

[1] Personal communication, data on file, Deschutes National Forest, Bend, OR
[2] Data on file, Deschutes National Forest, Bend, OR

Table 5—*Estimated N losses due to silvicultural practices in central Oregon pine stands from the Black Bark study. Total ecosystem N includes all vegetation, downed woody material, forest floor, and mineral soil through the C1 horizon, and averaged 2467 kg ha^{-1} for the three study sites (Little and Shainsky 1995).*

Stand treatment	Estimated N loss (kg ha^{-1})	Percentage of ecosystem N
Prescribed fire (1991+2002)		
Thinned	523	21
Unthinned	549	22
Pre-commercial thinning		
Bole-only	36	1
Whole tree	94	4

Just how important is the loss of 500 kg N ha^{-1} to the sustainability of these pine forests? Certainly no detrimental effects to vegetation growth have been expressed so far in the study, indicative of a nutritionally resilient system. Forest productivity, therefore, has not declined. On the other hand, 500 kg N ha^{-1} represents a loss of nearly one-fourth of the total ecosystem N pool (*table 5*), indicating that soil quality was compromised. As a comparison of alternative methods to reduce fire risk, pre-commercial thinning had a relatively innocuous effect on ecosystem N, while mowing of shrubs, although not measured, would have minimal affect on total N. Interestingly, the practice of slash retention (bole-only harvest) as a means to conserve site organic matter resulted in little improvement in site N compared to whole-tree harvest.

Taking the concept of repeated fire in eastside pine forests one step further, Johnson and others (1998) estimated total N losses during a 100-year period for a range of fire-return intervals (1, 3, 5, 10, or 20 burns in 100 years). Assuming that each fire consumed 50 percent of the forest floor N, and given the natural rate of litter input and decay for their site, they found that burning every five years resulted in a loss of nearly 1500 kg ha^{-1} of N (*fig. 5*). Burning every 10 years resulted in a 50 percent reduction in total N by the end of 100 years. Admittedly their model is simplistic and does not account for changes in stand structure, litterfall, or burn prescriptions during the 100-year rotation. Nevertheless, the potential loss of soil quality is daunting.

This exercise in predicting N losses during 100 years brings up an interesting question. If repeated burning results in near-catastrophic losses of soil N, and if the pine forests in central Oregon burned on a 4 to 24 year interval prior to the era of fire suppression (Bork 1984), then why didn't the forests run completely out of N a few centuries ago? Several possible explanations come to mind: (1) the forests actually ran out of N; (2) N input from atmospheric deposition and N-fixing plants was sufficient to offset N losses; (3) the model of Johnson and others (1998) and the results from the Black Bark study over-predict N loss in forests prior to fire suppression. Of course, explanation (1) is nonsensical, and explanation (2) is highly unlikely since fewer N-fixing shrubs were present in the understory of ponderosa pine forests prior to the 20th century (Cochran and Hopkins 1990). The third explanation is intuitively acceptable (although unproven) if we acknowledge that the earlier forests had open, park-like structure with less litterfall and forest floor accumulation, which led to lighter burns and, consequently, less N volatilized per burn.

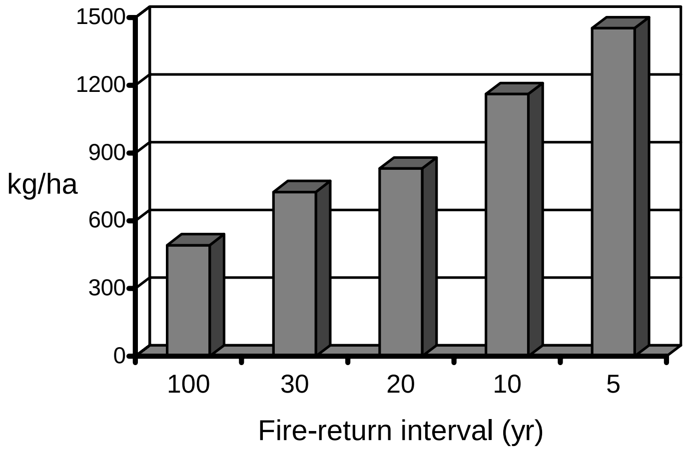

Figure 5—Predicted N losses during a 100-year rotation due to repeated prescribed burning at differing fire intervals. Data are from Johnson and others (1998) for stands of Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.).

Can N Be Replaced By Natural Processes?

Maybe so. Nitrogen fixation–the conversion of atmospheric N into biologically-available forms via atmospheric deposition, lightning, symbiotic N fixation, and non-symbiotic N fixation–can partially or completely counter N losses due to fire. Symbiotic N fixation, in particular, is an energy efficient and productive means of replacing N losses.

Bitterbrush and ceanothus are the most common N-fixing plants in central Oregon. Bitterbrush is found on mid- to low-elevation pine sites, while snowbrush is found at comparatively higher elevations. N-fixation rates range from 0.5 to 2 kg ha^{-1} yr^{-1} for bitterbrush and from 4 to 16 kg ha^{-1} yr^{-1} for snowbrush in central Oregon (Busse 2000). These are not large quantities of fixed N, like is common in pure stands of agricultural legumes. Thus, replacing N losses after fire is a long-term venture (*table 6*), and, in the case of bitterbrush, will never be complete. Snowbrush, by comparison, can replenish the total N pool in 25 years following a single prescribed fire.

Table 6—*Estimated time require to replace N losses (kg $ha^{-1)}$ from prescribed burning.*

		Years required to replenish lost N[3]	
Fire	N loss	Snowbrush	Bitterbrush
Single	280[1]	25	140
Repeated	523[2]	48	262

[1] From Landsberg (1993)
[2] Black Bark study, data on file, Deschutes National Forest, Bend, OR
[3] Nitrogen-fixation rates, based on the results of Busse (2000), are 10 kg ha^{-1} yr^{-1} for snowbrush; 1 kg ha^{-1} yr^{-1} for bitterbrush; and 1 kg ha^{-1} yr^{-1} for atmospheric deposition

Management Considerations

Let's face it, soils are boring to the masses. If soil was only impervious to disturbance, we wouldn't have to give it much thought. Then we would accept Kimmins (1994) statement "it is neither possible nor is it necessary to consider all of the processes and components in order to make useful predictions about the long-term consequences of forest management" as appropriate for forest soils. Alas, the misfortune of past civilizations that failed to respect their soils or those exposed to catastrophic events leaves a lasting imprint of the need to revere soil.

Obviously, the results from the Black Bark study in central Oregon pine forests paint a far less stunning picture of soil devastation compared to images from the Dust Bowl or from hydraulic mining in the Sierra Nevada mountains. Most soil characteristics and processes, in fact, were unaffected by combinations of thinning and fire. Repeated burning, however, resulted in a loss of soil quality due to reduced total N. Continued burning on the plots may only accentuate this problem.

We offer the following suggestions to avoid similar fire-induced losses in soil quality in central Oregon ponderosa pine forests:

- Prescribe low severity burns, particularly in stands lacking previous prescribed fire entry. Prescriptions that leave 75 percent or more of the duff layer intact will help conserve N.
- Reduce fire frequency. Soil N loss increases with increasing fire frequency.
- Incorporate mechanical treatments (thinning and mowing) to reduce fire risk. Little N is removed by these methods compared to burning.
- Value N-fixing understory plants. Nitrogen can be restored on burned sites, particularly at higher elevations where snowbrush is present.

Forest management does not revolve around protecting soil N pools. Instead, this is the era of wildfire risk reduction, threatened and endangered species protection, wildlife habitat restoration, and recreation use. Long-term changes in soil quality, however, need not occur in central Oregon if simple precautions are taken such as conserving soil N. Most importantly, they will be avoided if all that work (and play) in the forests speak for the soil.

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